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

16–20 October 2017

San Sebastian, Spain



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Executive summary

The pan-regional Working Group on Multispecies Assessment Methods (WGSAM) met in San Sebastian, Spain, 16–20 October 2017. In this eleventh report of the group, work focused on three of the multi-annual ToRs (B, C, D).

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.

A Key Run (ToR B) of the North Sea Stochastic Multispecies Model (SMS) 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 a subset of experts post-meeting. The Key Run is documented in detail in Annex for ToR B, with key outputs summarised in Section 5 and data files made available on the WGSAM webpage and the ICES expert group Github (https://github.com/ices-eg/wg_WGSAM). Since the M2 values are used for the assessment of important North Sea stocks, it is recommended to publish the annex also on the official stock annex website. In addition, WGSAM does not recommend updating existing data series of natural mortality by simply adding the latest three new years. The time-series as a whole shows patterns which are not retained by this procedure.

Multispecies model skill assessment (ToR C) and multi-model ensemble methods (ToR D) were emphasized this year. Considerable progress has been made towards advancing both aspects of multispecies modelling. Investigation of skill assessment and ensemble methods and case studies is critical to ensure that outputs of multispecies assessment models are reliable for use in operational assessment and to inform management decisions.

Progress was also made on investigations of top predator impacts on managed fish across several regions (ToR F), including the North Sea where new information was included in the SMS key run. Further progress was also made on multispecies and ecosystem level reference points and harvest control rules in mixed fisheries (ToR G).

1 Administrative details

Working Group name

Working Group on Multispecies Assessment Methods (WGSAM)

Year of Appointment within current cycle

2016

Reporting year within current cycle (1, 2 or 3)

2

Chair(s)

Alexander Kempf, Germany

Sarah Gaichas, USA

Meeting dates

16–20 October 2017

Meeting venue

San Sebastian, Spain

2 Terms of Reference a) – z)

Work on all ToRs. Focus on B, C, D, (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 multi-species and ecosystem models for different ICES regions (North Sea SMS).

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)

3 Summary of Work plan

Year	Work
Year 1	Work on all ToRs. Focus on ToR e, f and g. ToR b: Keyruns (as required)

Year 2	Work on all tors. Focus on ToR c and d. ToR b: Keyruns (North Sea SMS, as required)
Year 3	Work on all tors. Focus on Synthesis ToR c-g. ToR b: Keyruns (as required)

4 List of Outcomes and Achievements of the WG in this delivery period

- Key runs
 - Baltic EwE (see 2016 report)
 - North Sea SMS (2017): A comprehensive description of the key run can be found in the Annex for ToR B and a summary below in section 5.
- Model framework review
 - LeMans Ensemble (see 2016 report)
 - FLBEIA (2017): A subgroup conducted a review of the structure of the FLBEIA MSE tool, to investigate its potential suitability as a basis for developing a multispecies MSEs tool. It was decided that the tool did have potential for such use, and that a more detailed workshop should occur, with visiting scientists and programmers working to investigate extending the model towards multispecies use and/or identifying lessons which could be learned for developing such tools.
 - Multispecies state-space model (2017): The full group reviewed a new state-space modelling framework in development for comparing performance between single and multispecies models. The framework is implemented in TMB and gives the user many options for estimating parameters within or outside the model, for running in single or multispecies mode, and for doing simulation or estimation. There was considerable interest in applying this model to other systems. More details are reported under ToR C.

5 Progress report on ToRs and workplan

Progress by ToR

A summary of progress is given here, with full reports including figures and references available in ToR-specific Annexes to this report

5.1 ToR A: Review further progress and deliver key updates in multi-species and ecosystem modelling throughout the ICES region

Multispecies assessment methods updates were received for eight ICES regions, including Greenland and Iceland Seas, Barents Sea, Norwegian Sea, Celtic Sea, North Sea, Baltic Sea, South European Atlantic Shelf, and US Northwest Atlantic. Note that Baltic progress was reported under ToRs C and G.

We received updates on Atlantis model skill assessment and sensitivity in Iceland, multiple studies using Atlantis in the Norwegian and Barents Seas, and progress on Atlantis model reconfiguration in the US. Ecopath with Ecosim (and other food web modelling) updates for the Southern North Sea, English Channel, Bay of Biscay, and Northeast US were provided. Multispecies models including Gadget, SMS, LeMans, and Mizer were presented for the Celtic Sea, North Sea, Bay of Biscay, Iberian Peninsula, and US Northwest Atlantic.

Progress on frameworks incorporating multispecies and ecosystem models along with single stock models into MSE frameworks (REDUS, Norway and LeMans, North Sea), and the US (New England Atlantic herring) was reviewed. Lastly, ecosystem reporting and general approaches to ecosystem based fishery management were reviewed (Norway, US).

5.2 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 (North Sea SMS)

North Sea SMS model

A key run for the North Sea SMS model was produced using data from the period 1974–2016. This included updates to the input data and some modification to the structure of the model. These are described in detail in the stock annex (see Annex for ToR B) for the SMS North Sea, where also the main results of the 2017 key-run can be found. Model code, input and output can be found at the ICES expert group Github (https://github.com/ices-eg/wg_WGSAM). This section gives only a short overview of the key-run. For further details see Annex for ToR B.

Data input

There have been several changes in input data to the SMS since the last key run in 2014:

- Update of “single species data” (catch at age numbers, mean weights, proportion mature, survey indices *etc.*) with use of the most recent ICES assessment input data. Re-estimation of quarterly mean weight at age in the sea from ICES annual data and quarterly differences from existing SMS da-

ta. Some stocks have been benchmarked since the 2014 key run, giving substantial changes in both the ICES and the SMS assessments

- Inclusion of mackerel as a dynamic species, which replaces the “external predators” North Sea mackerel and Western stock mackerel. With both approaches the proportion of the north Atlantic mackerel within the North Sea needs to be known. In lack of a documented time-series for that, WGSAM made their own estimate of stock distribution, which were used in SMS.
- Re-calculation of “single species data” for the two sandeel stocks, as the present ICES stock areas for sandeel fit poorly into the northern and southern sandeel areas used in SMS.
- Update of consumption estimates (daily ration) of fish predators, particularly mackerel and horse mackerel using updated parameter for the evacuation model.
- Bias correction of diet estimate from observed stomach contents taking variable evacuation rate of prey species, stomach fullness and temperature into account for the fish stocks (cod, whiting, haddock saithe and mackerel) and taking variable evacuation rates of otolith (sizes) into account for harbour porpoise.
- Inclusion of distribution of fish stocks making calculations of M2 based only on the predator and prey stock numbers present within the North Sea area.

Results

The substantial changes of input data to the new key run and ICES benchmarks for some of the stocks since the 2014 key run have produced stock summaries (recruitment, mean F and SSB) from the 2017 key run that is somewhat different from the summaries from the 2014-key run. However, the new estimated predation mortalities (M2) are overall consistent with the M2 values from the previous key run and relatively minor changes occurred. The robustness of the estimate of predation mortality corresponds well to the conclusion made by the long row of ICES working groups using the SMS model or previous model versions like 4M and MSVPA, that the estimate of M2 is robust to e.g. changes in consumption rates and the amount of “other food” in the diet or level of M1.

The following sections describe the changes in the main output variable between the (in 2015 updated) 2014 key run and the new 2017 key run. Figures showing a comparison between the 2014 and 2017 keyrun can be found in the Annex for ToR B.

5.2.1 Cod

The main differences for cod between the two key runs are a somewhat higher recruitment in the last two decades in the 2017 key run. The higher recruitment fits very well to the higher M2 in the new run (Annex 4, Figure 5.2.1, upper panel).

There has been very little change in the predation mortality of cod of age 1 and 2 between the 2014 and 2017 key runs.

Predation mortality of cod age 3 has increased substantially since the last key run. This is a result of the updated time-series for weight at age of cod in the stock which results in lower mean weights which are consistently within the range which can be consumed by both harbour porpoise and grey seals. Harbour porpoise has been observed to eat 3.029 kg cod, grey seal up to 4.066 kg. SMS uses the mean weight within

prey size range group such that these values are translated into 1.773 kg and 2.754 kg respectively. This means that 4-year olds are not consistently included in the diet since they are only occasionally below 4 kg. The high predation mortality of age 3 cod around year 2000 corresponds to the increase in the grey seal population (see section 5.3). When age 4 is predated on, M2 is always low.

5.2.2 Whiting

Recruitment is pretty much the same in the two runs which is in line similar M2 for ages 0–1. Predation mortality of older age groups has decreased in the new run. This is partly a result of the correction of harbour porpoise consumption to account for longer residence times of whiting otoliths in porpoise stomachs than those of e.g. herring and sandeel. The bias correction of diet data for fish predator may also influence.

5.2.3 Haddock

Predation of haddock is largely the same between the two key runs. The two series may be different due to the updated time-series for weight at age of haddock in the stock. SSB is estimated lower in the new run, probably due to the lower mean weight at age used in 2017.

5.2.4 Saithe

The two saithe assessments are quite similar; despite the saithe assessment has been benchmarked since the last key run.

5.2.5 Herring

The two herring assessments are quite similar, however with slightly higher F and lower SSB in the 2017. There has been little change in the predation mortality of herring of age 1 and 2 between the 2014 and 2017 key runs. Predation mortality of age 0 has increased further, while that of age 3 and 4 has decreased. The changes to predation mortality at age 0 seems to be linked to the changes in mackerel and horse mackerel biomass, consumption and diet, as mackerel now feeds less on sandeel and hence more on alternative prey. Predation mortality of age 3 and older herring has decreased compared to earlier key runs as the mean weight and consumption of larger cod and saithe has decreased due to the change of the assumption of constant mean weights and rations at age of the predators.

5.2.6 Northern sandeel

There is a substantial difference in the recruitment, F and SSB for the two assessments. The predation mortalities of older northern sandeel has become more variable as the mean weight of sandeel now varies from year to year and furthermore exhibits trends over the time-series. Values for older age groups have increased somewhat, likely as a result of the lower mean weight at age in the second half of the time period. Re-estimation of single species data (new stock definition) may also influence the results.

5.2.7 Southern sandeel

The predation mortalities of older northern sandeel has become more variable as the mean weight of sandeel now varies from year to year and furthermore exhibits trends over the time-series. Values for older age groups have increased somewhat, likely as

a result of the lower mean weight at age in the second half of the time period which has increased the predation of grey gurnards and whiting on older sandeel. Re-estimation of single species data (new stock definition) may also influence the results.

5.2.8 Norway pout

The assessment of Norway pout has changed considerably between the two key runs, probably linked to the benchmark and inter-benchmark for this stock in the period. Predation mortality of Norway pout is very similar in the 2017 key run to those of the 2014 key run. At the end of the time period, hake becomes an important predator and is responsible for the increase in recent years.

5.2.9 Sprat

The sprat assessment has changed (benchmark) which is also reflected in the stock summary for the two key-runs. M2 from the 2017-run are more variable than in the previous key-run, but the trend in the two time-series is the same. The higher variability in the 2017 key run is probably due the variable mean weight in the sea used in the 2017 run.

Key run summary sheet

Area	North Sea
Model name	SMS
Type of model	Age-length structured statistical estimation model
Run year	2017
Predatory species	Assessed species: Cod, haddock, saithe, whiting, mackerel Species with given input population size: North Sea horse mackerel, western horse mackerel, grey gurnard, starry ray, hake, fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin, razorbill, grey seal, harbour porpoise
Prey species	Cod, haddock, herring, Norway pout, southern North Sea sandeel, northern North Sea sandeel, sprat, whiting,
Time range	1974–2016.
Time step	Quarterly
Area structure	North Sea
Stomach data	Fish species: 1981, 1985, 1986, 1987, 1991, 2005, 2013 Grey seals: 1985, 2002 Harbour porpoise: Decadal 1985, 1995, 2005
Purpose of key run	Making historic data on natural mortality available and multispecies dynamic
Model changes since last key run	All time-series updated. Mackerel included as a modelled stock. Proportion of the stock within the North Sea given as input and used for estimating M2. Daily food ration of changed for the main fish species. Bias correction of diet composition of harbour porpoise and the main predatory fish.
Input and output available at	Sharepoint/data/North_Sea_key_run and from the ICES expert group Github (https://github.com/ices-eg/wg_WGSAM)).
Further details in	Report of the Working Group on Multispecies Assessment Methods 2017

Conclusion, new key run

WGSAM 2017 discussed the changes in input data and the results in detail and concluded that:

- The new time-series is seen as more accurate than the previous time-series as the change in input data is based on the best available knowledge.
- M2 seems consistently estimated between key-runs and shows a very limited retrospective pattern using the last key run and excluding 1–4 years of data. Changes in ration and diet data had also a rather limited effect on M2 values.
- Some ICES assessments make use of the estimated natural mortalities (M1+M2) from SMS and update those in benchmark. If used, WGSAM does not recommend updating existing data series of natural mortality by simply adding the latest three new years. The time-series as a whole shows patterns which are not retained by this procedure. For example, herring shows an increased natural mortality over the past decade, but adding only the latest three years will give the impression that natural mortality has decreased over the last five years.

Identified areas of priority research

WGSAM 2017 considers that the following topics should be priority areas of study prior to the next North Sea key run:

- Estimating the proportion of hake, mackerel and horse mackerel stocks present in the North Sea and their distribution in northern and southern areas for a better estimation of M2 for the two sandeel stocks.
- Estimating distributions of seabirds in southern and northern North Sea.
- Reviewing the method used to estimate grey gurnard and starry ray abundance to identify the reference period and sizes to which the average biomass estimates apply. Consider if the SMS model by its likelihood statistics can estimate a likely mean biomass over a given period.
- Update the number of seabirds, grey seals and harbour porpoise with the most recent information.
- Update the diet and consumption data for grey seal with the most recent data.
- Assigning prey to length groups for the 2013 mackerel stomach data.
- Establishing quarterly catch histories for the all predator species (cod, whiting, haddock, saithe, mackerel) as initiated with data from InterCatch.
- Investigate changes to modelling performance when including overwintering mortality of sandeel (M1, possible condition or weight at age dependent).
- Investigate the most appropriate species and size selection of different predators.

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

Several approaches focussing on different aspects of skill assessments for multi species and ecosystem models were presented during the meeting of WGSAM 2017. The approaches ranged from testing the predictive power of diet selection and consump-

tion sub-models up to performance testing of whole models in an MSE approach. The analyses also included sensitivity tests of models to input data as well as model structure. In some case studies model predictions were challenged with observed hindcast time-series or retrospective patterns in hindcasts and forecasts were analysed. One of the main messages was that the quality of hindcasts does not allow for conclusions on the predictive power of the model.

Key points from the different presentations are available below and more details can be found in the Annex for ToR C.

Predicting the diet of Baltic Sea cod using stomach data and an age-length base model

- A multispecies model for cod, herring and sprat in the Baltic Sea for 1974–2013 was implemented in the Gadget;
- Changes in prey length preference and species composition investigated with respect to ontogenetic groups;
- Ontogenetic shifts in predicted species composition fitted quite well to observations;
- Temporal shifts fitted better to observed composition at the beginning of time-series;
- Length preference for sprat fitted better towards the end of time-series;
- Length preference for herring haven't fitted that well.

Can EwE mimic the Atlantis ecosystem?

- The Atlantis model for Icelandic waters was used as an operating model to test the performance of EwE;
- Balancing and fitting routines were written to make the modelling process more automatic and less subjective;
- The EwE model was able to replicate the Atlantis ecosystem;
- The forecasting ability of the model was however not reliable.

Influences of model structure on estimates of community dynamics and multispecies biomass production

- Estimation model structures influence estimates of species-pair interactions within a complex of ten commercially important species on Georges Bank, USA, as well as model predictive ability.
- We will estimate species-pair interactions in a Bayesian state-space multi-species production model fit to annual catches, abundance indices from bottom-trawl surveys, and diet composition data.
- Model fits will be compared using evaluations of posterior predictive loss and cross-validation error in the final five years of the survey indices.

New state-space multispecies age-structured stock assessment model for Georges Bank finfish community

- Four simulations models of 1000 iterations were run to test their performance (bias in estimated parameters and derived outputs) against the new state-space multispecies stock assessment model created for Georges Bank cod and haddock. The importance of estimating process errors in the mod-

el was also investigated (state-space vs. statistical catch at age (SCAA) models).

- The state-space multispecies model (same configuration than the operating model) has a tendency of underestimating fishing mortality at age and overestimating predation mortality at age and this is accentuated in the multispecies SCAA model.
- As expected, the SCAA models perform less well than the state-space models.
- Overall, the bias in recruitment and SSB is below 20% but the state-space single species model performs better than the state-space multispecies model which is unexpected. This may come from a problem in the simulated diet data since the estimated diet parameters are the only ones showing a large bias. This may also come from the low interactions between the fish species in the models. This will be investigated further in the next months.

Retrospective patterns in SMS hindcasts and forecast

To test the skill of SMS to provide robust estimates of natural mortalities retrospective patterns were analysed. Also the stability of forecasts was tested. Following conclusions could be drawn:

- The SMS keyrun hindcasts from 2015 showed no serious retrospective patterns for cod, whiting, haddock, sprat and Northern sandeel. SMS can provide consistent estimates of natural mortalities over the years for these stocks. For Norway pout and Southern sandeel retrospective patterns occurred.
- Retrospective bias in forecasts occurred for both single species and multi species versions of SMS. Forecasts with more than 4 years provided often considerably different results compared to short term forecasts.
- SMS multi species forecasts tended to be overpessimistic with regard to stock dynamics at low abundances. The Holling type II functional feeding response may be causing this.

Global sensitivity analysis of a multi-species model

- Used global sensitivity analysis to quantify how uncertainties in the inputs contribute to uncertainties in the outputs.
- The aim was to use variance based sensitivity analysis. However, due to the large number of inputs this is computationally infeasible.
- Used derivative based sensitivity analysis to decide which inputs had a negligible effect on the sensitivity of the output. This enables the variance based sensitivity analysis to be more tractable.

Estimating hake natural mortality based on multispecies model results and longevity

- Cetaceans are the main predators of hake in the Iberian peninsula.
- Multispecies models provide a way to estimate preys M-at-age as the sum of two components (M1 and M2). M2 is based on predator diet but M2 can be difficult to estimate if there is not a minimum likelihood at realistic M1 values.

- The use of life history parameters (longevity) was explored as a tool to select and adequate M1.

Hindcast Evaluation of a North Sea Multispecies Model

- A multispecies ensemble model in development was evaluated against single stock assessment models to see how useful it might be in terms of a) forecasting biomass trajectories, and b) predicting whether limit reference points might be breached.
- The model was tuned to stock assessments for the period 1990–2010 and then used to make a “forecast” for the period 2010–2015 using the same level of fishing as in the assessments for that period, with the assessed outcomes assumed to represent “truth”. Model outcomes were compared with simply persisting the 2010 assessment, to judge whether the model was adding useful skill when compared against the best available information at the time.
- We found that the raw model output was less useful than persisting the last available assessment, but once corrected for biases that would have been known at the start of the forecast period, it outperforms persistence, and is able to forecast the recovery of the cod stock.
- The raw model output was more skillful than persistence at predicting whether limit reference points had been breached, but had similar skill following adjustment. The method presented here is potentially useful for evaluating the utility of multispecies models.

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

Plans to use multi model ensembles and lessons from the usage of multi model ensembles were presented during WGSAM 2017. Although the setup of multi model ensembles is extremely time consuming, they get more and more used to improve the robustness of scientific advice but also to provide a full picture of underlying uncertainties. One approach dealing with model output comparisons based on the Jacobian matrix was presented. Another approach presented dealt with the important question how the strength of models in an ensemble can be exploited while discounting the weaknesses of each of the models to get an overall best estimate including estimates of uncertainty.

Key points of each presentation can be found below and more details can be found in the Annex for ToR D.

A multi-model approach to understanding the role of Pacific sardine in the California Current food web

- Multi-model approach is useful, but is only a stepping stone toward true ensembles and more directly comparable models;
- Structural assumptions influence predictions: taxonomic resolution (whether brown pelican modelled explicitly) and age structure and density dependence (Atlantis and MICE vs Ecopath/PREP);
- Monte Carlo approaches (MICE) give probability distributions, but we can also translate to make MICE comparable to Atlantis, Ecosim, etc.;

- Atlantis identifies species MICE might consider: Dolphins, large flatfish (halibut) – both decline at moderate levels of sardine abundance; other birds.

Alaska Climate Change Integrated Modelling (ACLIM) model ensemble

- Five Climate-Enhanced (CE) models (stock, multispecies, ecosystem, fleet and human community) will be used together to evaluate potential responses to projected climate change in the eastern Bering Sea.
- Alternative management strategies will be evaluated under different climate projections.

Comparisons of Models using the Jacobian Matrix

- An interactive multispecies model T-ONS was designed for stakeholders to use by the MareFrame project. It is implemented in a spreadsheet.
- A multispecies Schaefer model fitted to SMS outputs is converted to a Jacobian matrix, which is a linear approximation to the response surface at status quo effort.
- This could be a good way to compare model outputs, could do similar runs with LeMans, Mizer, etc. calculate Jacobian for each and put in this model.

A dynamic multi-model ensemble for marine ecosystem simulators

- A method for combining different ecosystem models with different outputs.
- The ensemble model exploits the strengths and discounts the weaknesses of each of the models.
- A proof of concept example was shown. It demonstrated that the ensemble model is able to predict, with quantifiable measures of uncertainty what would happen in the future under specific scenarios.

5.5 ToR E: Test performance and sensitivity of ecosystem indicators

There were no reports under this ToR in 2017.

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

The importance of top-predators in ICES waters was highlighted for several case studies during WGSAM 2017. Thereby not only their impact on modelling results (see also under ToR B North Sea keyrun) was presented but also the spatial and temporal scales of interactions with prey species and with fisheries (e.g., depredation, bycatch) were analysed.

Keypoints of each presentation are provided below and further details can be found in the Annex for ToR F.

Modifying longlining operational techniques to limit fish depredation by marine mammals

- Longline fisheries are the most impacted fisheries by mammal depredation.

- Recent technological approaches implemented proved to be less if not at all effective to limiting depredation impact to fisheries
- Changes in operational techniques, specifically on fishing behaviours, have been tested taking into account the case study of longline fisheries in Crozet and Kerguelen Islands.
- Preliminary results indicate that depredation in these areas is mainly influenced by mammalian ecology and the conditions of fishing operations.
- A manuscript was recently submitted to Fisheries Research discussing the results obtained from this case study.

Estimating abundances and spatial distributions of great and sooty shearwaters in the Bay of Biscay

Overall conclusions:

- Thanks to multidisciplinary surveys, the distribution and habitat use of species within the Birds and Habitats Directive in the Bay of Biscay can be determined.
- Relative abundance and temporal trends of marine top predators can be estimated with the information provided by these survey, supporting the detection of potential changes in their populations.
- High importance of considering oceanographic, prey and static variables to explain the abundance of marine top predators.
- Spatial predictions obtained matched with abundance patterns.
- Both the survey used (JUVENA) and SAMM (Aerial Monitoring of Marine Megafauna) provide similar orders of magnitude despite methodological differences, areas and sampling times.
- Despite inter-annual variability, some oceanographic phenomena (upwelling, river plumes) occur in similar places year after year, then high density areas could be limited in space.
- Overlap with human pressures, although additional data sources / programs are needed (bycatch, VMS).
- The results obtained are comparable to previous information found in the literature.

GADGET model for cetacean fishery interaction in the Iberian peninsula

A Gadget modelling framework is currently under development in the Iberian Peninsula to analyze the cetacean fishery interaction. The study area comprises the Spanish and Portuguese shelf and oceanic waters (ICES subdivisions VIIIc and IXa); corresponding with the Southern European hake stock distribution and the area of overlap with the common (*Delphinus delphis*) and bottlenose (*Tursiops truncatus*) dolphins populations.

Preliminary results suggest:

- The multispecies model improves the quality of the fit compared with hake single species model (likelihood 1001 vs. 1015).
- The biomass of hake consumed by cetaceans is in similar scale than historical hake catches.

- Hake reference points (F_{msy} , B_{pa} , etc) from the multispecies model have been estimated following the same rationale than in ICES and cetacean by-catch limits (GES-MSFD) were also set.
- Short and medium term projections have been developed.
- The model could provide advice for hake considering impact on hake and cetaceans.

Next steps should focus on uncertainty sources such as:

- Implement dolphin variable abundance considering existing information such as PELACUS survey trends for common dolphin, SCANS-III survey for either dolphins or variability on effort for fleets by-catching dolphins.
- Perform sensitivity analysis for more uncertain parameters (dense-dependent fecundity, M -at-age, etc) should also be analyzed.

5.7 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)

WGSAM has a long-standing experience in discussing fisheries management in a multi species context. This year the Nash equilibrium for the Baltic fish community has been revisited. WGSAM was made aware of a new project on “Ecosystem Based F_{MSY} Values in Fisheries Management”. Next to this, WGSAM discussed EBFM approaches based on ceilings for total system removals or species complexes. First simulation results look promising and WGSAM will keep track on further developments in this area.

Key points of the presentations can be found below. Further details are in the Annex for ToR G.

The effects of density dependent clupeid growth on Nash equilibrium reference points in the Baltic Sea

- Fitting of the clupeid growth functions in the MSI-SOM model to data is improved with intra- and interdependent density dependence.
- The inclusion of density dependent clupeid growth in the MSI-SOM has minor effects on the Nash equilibrium reference points.

Ecosystem F_{MSY} Project

- A project called “Ecosystem Based F_{MSY} Values in Fisheries Management” is running to try and identify multi-species F_{msy} estimates for ICES stocks with analytic assessments across the North Atlantic, and compare these with current single-species based estimates.
- The project does not examine the trade-offs inherent in multispecies management, rather it considers each stock in isolation and attempts to identify the estimates of F_{msy} arising from multispecies models (without changing fishing patterns for other stocks).
- The project will use production models to give a simple route at finding these multispecies F_{msy} estimates, and will also collate existing multi-species or ecosystem F_{msy} estimates for the different stocks. The project will then examine similarities and differences between the multi-species

and single-species estimates, and between the different multispecies models.

Evaluating an ecosystem-based fishery management procedure for Georges Bank using ceilings on system removals

- The ceiling level on total system removals explains most of the variability in performance metrics at the whole ecosystem, aggregate species groups, and single species level
- Implementation of indicator-based harvest control rules also explained a large portion of performance variability when ceilings were set to higher values.

Ecosystem Based Fishery Management in New England, USA

- A proposed EBFM procedure is described, with components illustrated using model simulations. There are six steps outlined:
 - Specify spatial management units
 - Establish specific management objectives and exploitation reference points directed at stock complexes rather than individual species.
 - Establish biomass thresholds (floors) below which the complex as a whole cannot fall (Option 1) or below which no species within the complex can fall (Option 2).
 - Devise an Ecosystem-based Harvest Control Rule based on steps 2 and 3 designed to minimize the risk of overfishing for a range of exploitation rates at the stock complex level.
 - Simulate the performance of a set of scenarios constructed under the EBFM using a suit of metrics including biomass, landings, revenue, probability of breaching a threshold biomass level, maintaining robust size structure of the populations (large fish index), and the stability of the landings.
 - Identify and reconcile tradeoffs.
- We find that low levels of exploitation rate (0.15–0.20) had the best overall performance in terms of biomass, yield, and risk of falling into the depleted status.
- The major tradeoffs involve catch, revenue, and species-complex or species status.

Changes/Edits/Additions to ToR

None

Cooperation with other WG

WGNSSK - Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak

HAWG - Herring Assessment Working Group for the Area South of 62° N

WGMIXFISH - Working Group on Mixed Fisheries Advice

WGHARP - ICES/NAFO/NAMMCO Working Group on Harp and Hooded Seals

Cooperation with Advisory Structures

WGNSSK - Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak

HAWG - Herring Assessment Working Group for the Area South of 62° N

Science Highlights

Please see above under ToRs.

6 Revisions to the work plan and justification

None

7 Next meetings

15–19 October 2018, Paris, France

Annex 1: List of participants

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Annex 2: Recommendations

RECOMMENDATION	ADDRESSED TO
1. An extensive “stock annex” for the North Sea SMS keyrun has been produced during WGSAM 2017. Since the M2 values are used for important North Sea stocks, it is recommended to publish the annex also on the stock annex website. Further keyruns are available from WGSAM (e.g., EwE keyruns for Baltic and North Sea) and a “stock annex” could be provided also for these models for publication on the stock annex website.	ACOM leadership, Secretariat
2. Some ICES assessments make use of the estimated natural mortalities (M1+M2) from SMS and update those in benchmarks. If used, WGSAM does not recommend updating existing data series of natural mortality by simply adding the latest three new years. The time-series as a whole shows patterns which are not retained by this procedure. For example, herring shows an increased natural mortality over the past decade, but adding only the latest three years will give the impression that natural mortality has decreased over the last five years.	WGNSSK, HAWG
3. M2 values produced by WGSAM in key runs are used in several of ICES’ stock assessments to provide catch advice in response to requests from ICES’ clients. WGSAM recommends that in future the group liaise more with ACOM (e.g., as joint SCICOM/ACOM group) to maintain the relevance of its science to the ICES’ advisory process, and so assist the uptake of WGSAM’s work to ICES’ mission.	ACOM leadership

Annex 3: 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.

Ecoregion A: Greenland and Iceland Seas

Ecosystem model for Icelandic waters

- Ecosystem model has been constructed for Icelandic waters using the Atlantis modelling framework.
- Skill assessment has been carried out for the model.
- The model is able to replicate time-series of biomass and landings for the most important commercial groups.
- Sensitivity study was carried out to explore the dynamics of the model

Ecosystem model has been constructed for Icelandic waters using the Atlantis modelling framework. The model is in three dimensional layout and covers an area of 1 600 000 km² that has been divided into 52 boxes (Figure 1.1) and each box can have up to seven layers including a sediment layer. A hydrodynamic model has been built for this area and is used as basis for the oceanography in the Atlantis model, which subsequently controls the advection of plankton and nutrients. The biological model has 52 functional groups: 26 are vertebrates where 8 are at a species level, 16 invertebrates, 5 primary producers, 2 bacteria and 3 detritus groups. Each functional group has its own spatial distribution which is set as a fixed proportion for each box that can be different between seasons (Figure 1.1).

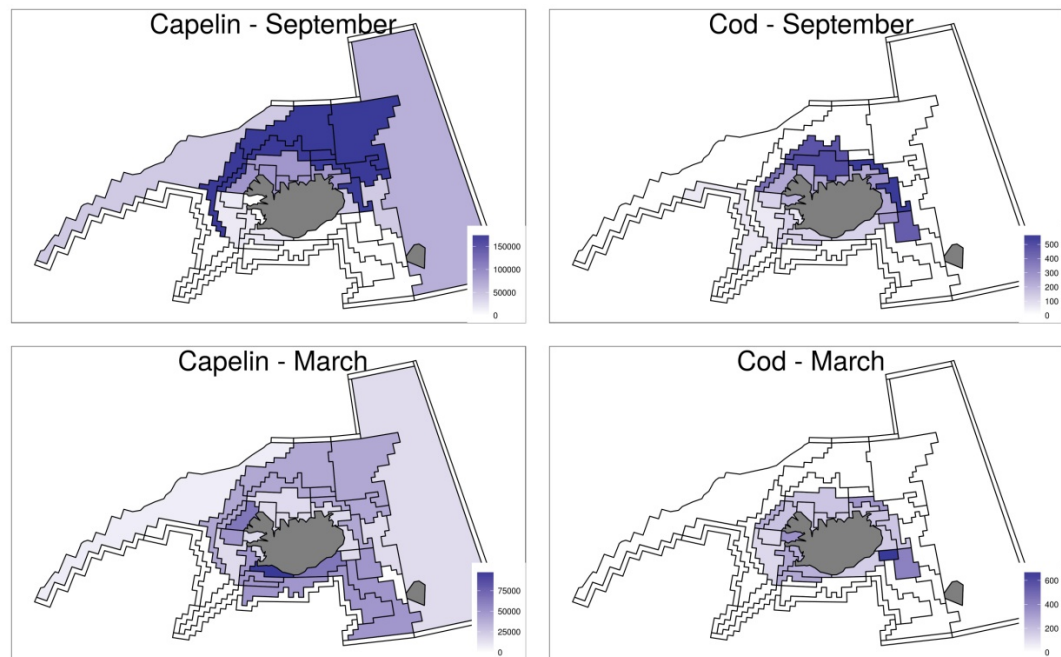


Figure 1.1. The spatial distribution of cod and capelin in March and September.

The Atlantis model also incorporates a fisheries model which is based on selectivity and harvest rates for each targeted group. The harvest rates are allowed to change between years. A skill assessment has been carried out using three metrics: Pearson's correlation, model reliability and model efficiency (Stow *et al.*, 2009). Biomass estimates from assessment models and landings data (Anon, 2016) were used to compare to the model output (Table 1.1). The simulated landings were compared to landings data for 12 commercial groups and all the groups had positive correlation. The reliability index was <1.2 for 7 of the 12 groups and model efficiency was >0 for all groups. Only five groups were compared to biomass estimates. The demersal groups: cod, haddock and saithe had a good fit to the estimates but the pelagic species capelin and herring did not fit as well. The simulated biomass of cod had high correlation with the estimates from the single species assessment model and according to the three metrics the biomass had a better fit than the landings for the cod (Table 1.1 and Figure 1.2). On the other hand landings of the capelin had a better fit than the biomass (Table 1.1 and Figure 1.3).

Table 1.1. Skill assessment: The three metrics model efficiency (MEF), reliability index (RI) and correlation (r) for biomass and landings.

Group	Metrics for landings			Metrics for biomass		
	MEF	RI	r	MEF	RI	r
Cod	0.36	1.05	0.73	0.72	1.05	0.87
Haddock	0.16	1.11	0.71	0.42	1.08	0.82
Saithe	0.63	1.04	0.83	-0.06	1.08	0.63
Herring	0.64	1.53	0.85	-0.93	1.13	0.09
Capelin	0.53	2.51	0.88	-10.79	1.67	0.35
Redfish	0.22	1.10	0.77			
Greenland halibut	0.55	1.30	0.76			
Flatfish	0.15	1.10	0.58			
Other codfish	0.51	1.05	0.79			

Commercial demersal	0.50	1.07	0.77
Blue whiting	0.91	1.90	0.97
Mackerel	0.95	1.36	0.98

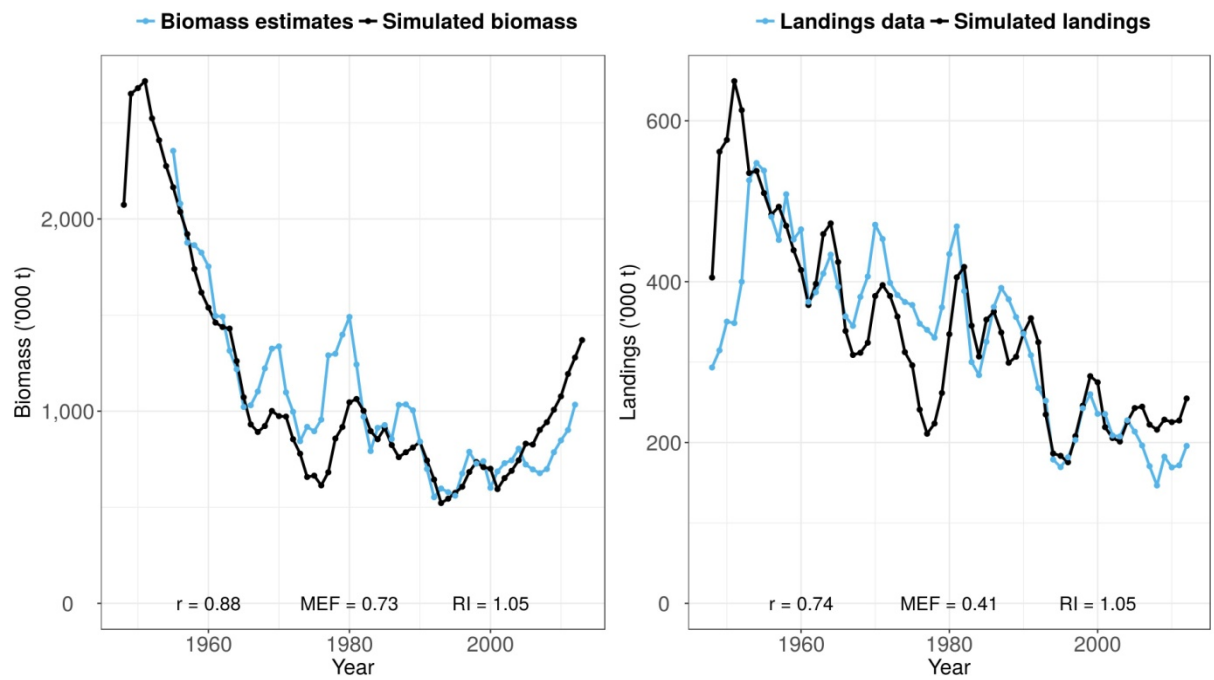


Figure 1.2. Simulated biomass and landings of cod compared to biomass estimates and landings data.

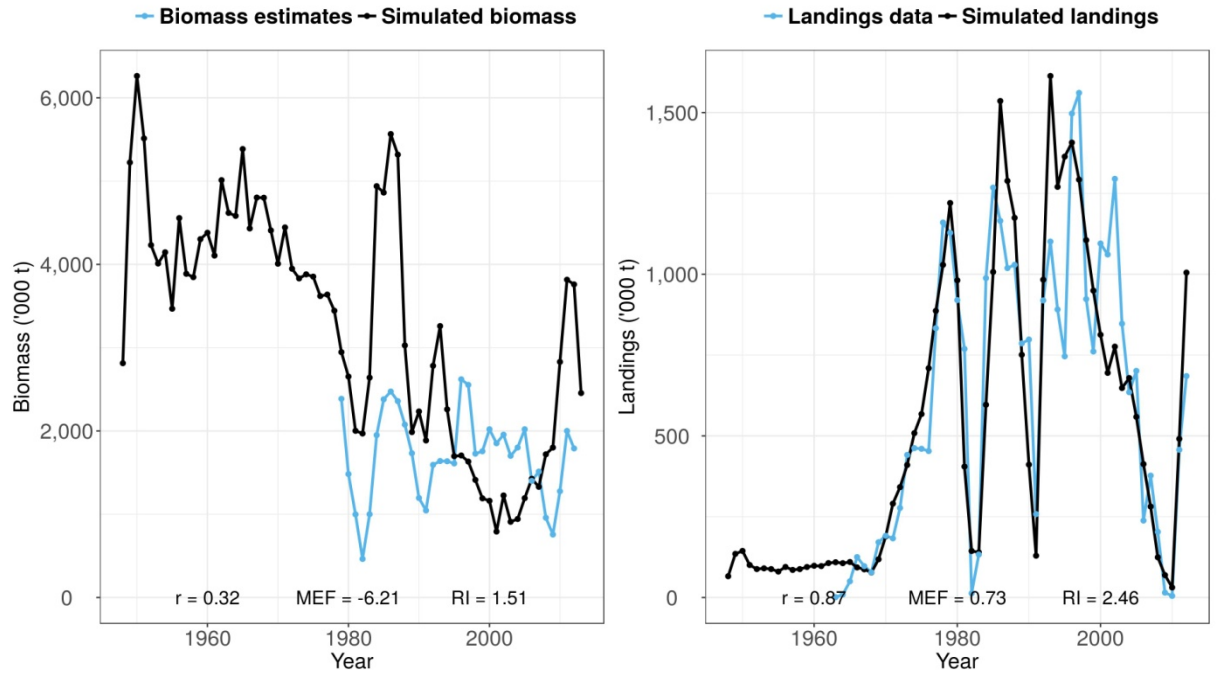


Figure 1.3. Simulated biomass and landings of capelin compared to biomass estimates and landings data.

Sensitivity analysis was carried out where the recruitment parameters for the vertebrate groups were altered by $\pm 20\%$. Growth parameters of large zooplankton, picophytoplankton and diatoms were also perturbed by $\pm 20\%$ and their interactions tested. When only one parameter is perturbed at a time, as was done with the recruitment parameters, a measure of model sensitivity can be calculated as follows:

$$S_{ij} = \frac{V_i(1.2\alpha_j) - V_i(0.8\alpha_j)}{0.4V_i(\alpha_j)}$$

Where S_{ij} is the sensitivity measure for the biomass of group i when maximum recruitment in the Beverton-Holt function is perturbed, $V_i(\cdot)$ is the average biomass of group i for the whole simulated period. The interpretation of S is: when $S = 1$ then 20% change in a parameter results in a 20% change in the output, $S > 1$ means that the change in the output is larger than 20% and $S = 0$ indicates that the output is not affected by the parameter. If the S value is negative then the change in output is in the opposite direction to the change in the parameter.

Altering the recruitment of a group did have an effect on itself but how much that effect was varied between the groups (Figure 1.4). Changing the recruitment of capelin did not have much effect on its own biomass but herring, mackerel and sandeel were very sensitive to a change in the recruitment. Saithe and redfish were the groups that had the most influence on other fish groups in the model.

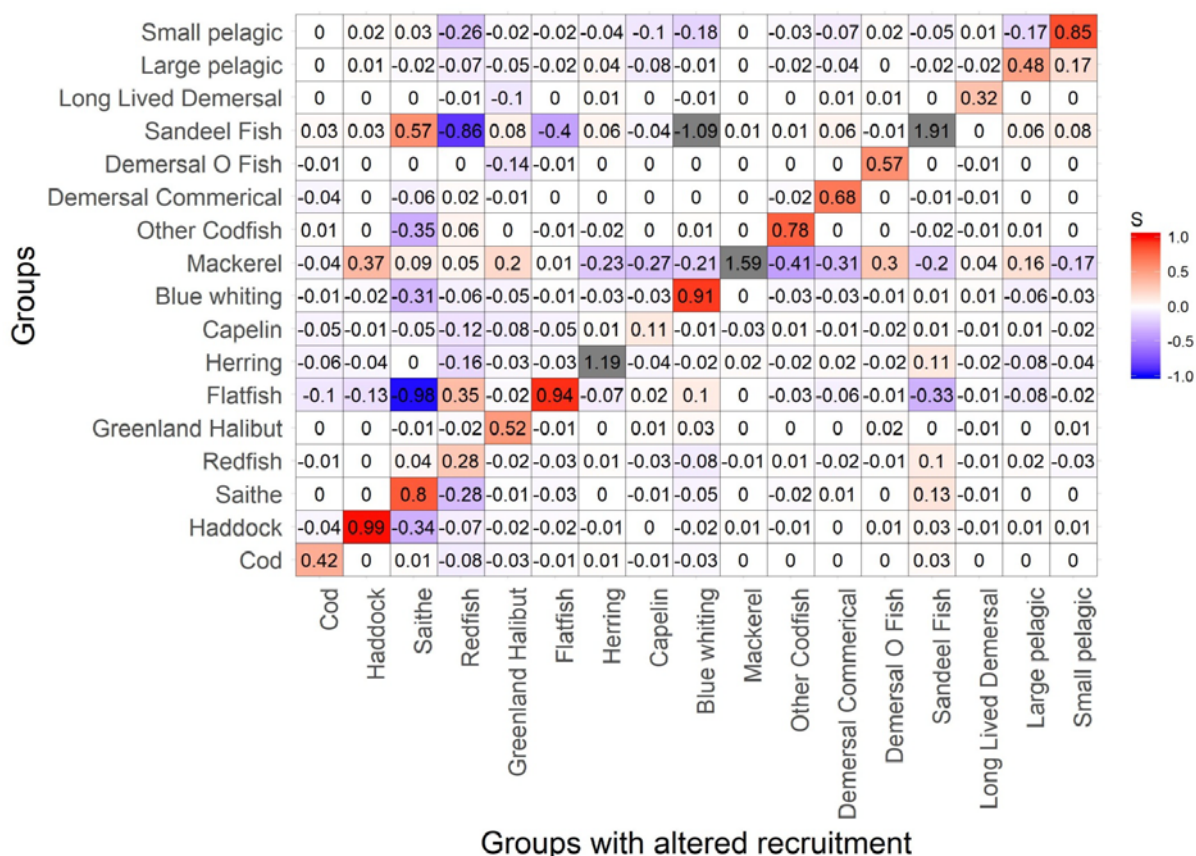


Figure 1.4. Sensitivity study: the metric S for the change in biomass of the fish groups when recruitment is altered for the fish groups.

Anon., 2016. State of Marine Stocks In Icelandic Waters 2015/2016 and Prospects for the Quota Year 2016/2017. Marine Research in Iceland 185. The Marine Research Institution, Reykjavik, Iceland.

Stow, C.A., Jolliff, J., McGillicuddy, D.J., Doney, S.C., Allen, J.I., Friedrichs, M.A., Rose, K.A., Wallhead, P., 2009. Skill assessment for coupled biological/physical models of marine systems. Journal of Marine Systems 76(1), 4–15.

Ecoregion B: Barents Sea

The REDUS project in IMR Norway is focussing on quantifying uncertainty within the assessment process. As part of the, a new MSE tool is being developed to allow greater flexibility in evaluating HCRs and investigating sources of uncertainty. This will work by connecting to external operating models, allowing for the use of multispecies operating models. Such a tool would allow for investigation of multispecies HCRs, but also for investigating if the modelled performance of simpler single species HCRs is altered if considered in a multispecies context.

The HCR for NEA cod (in the Barents Sea) that was adopted by the Norwegian-Russian Fisheries Commission in 2016 calls for higher fishing pressure at high stock sizes, a “so-called double hockey stick”. This implicitly multispecies, as it aims to avoid stock sizes high enough to cause reduced productivity.

The NoBar Atlantis model for the Norwegian and Barents Sea is now operational, and is being used for a number of different projects.

Ecoregion C: Faroes

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

Ecoregion D: Norwegian Sea

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

Ecoregion E: Celtic Seas

Reducing circularity: building a multi-species model from the ground up

- Multispecies size spectrum model of the Celtic Sea with 17 stocks;
- Model fitted without single-species stock assessments – fitting Fishing mortality;
- Included the survey as a fleet in the model and compared the simulated survey and the actual survey.

Spence presented a multispecies size spectrum model of the Celtic Sea (Blanchard *et al.* 2014). This included 17 stocks and was fitted to survey data and landings data. The aim of the project is to fit the model without using single species stock assessment models. This means that the fishing mortality rates, maximum recruitment and background resources are all fitted within the model with measures of uncertainty using Markov Chain Monte Carlo (Spence *et al.* 2016). A survey vessel was included in the model as a fleet and the simulated survey was compared to the actual survey. Early results showed that the model was able to fit trends in fishing mortality well as well as trends in the survey data however it was unable to find absolute values.

For more details email Michael Spence: michael.spence@cefas.co.uk

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

Making the most of survey data: Incorporating age uncertainty when fitting growth parameters

- A new method has been developed that enables the uncertainty in age to be quantified.
- Ignoring uncertainty can lead to inconsistent models of the von Bertalanffy growth curve and management decisions are sensitive to this.
- The method leads to better fitting models and more robust way of fitting the model that allows data from different sources to be included.

A variation of the von Bertalanffy growth function was presented that allows the ages of the individual fish to be continuous as opposed to binned into age groups. Using information from spawning studies the exact age of the fish is inferred, with measures of uncertainty, and then the von Bertalanffy growth curve is fitted to these ages. In the new model the ages are

$$t - s + q,$$

where t is the number of winters survived, s is the spawning time which has a prior with a circular distribution (e.g. von Mises distribution) and q the proportion of the year that the fish was caught in.

It was shown empirically that ignoring this age uncertainty can lead to inconsistent models (Figure 1) and that this can lead to sensitivity in the management models. This new model formulation leads to better fitting models and allows a way of combining data from different studies in different form to fit, with quantifiable measures of uncertainty, von Bertalanffy parameters. Methods of fitting the model to length-stratified sampling are currently being developed with an R package expected in the near future.

See Spence and Turtle (2017) for more details.

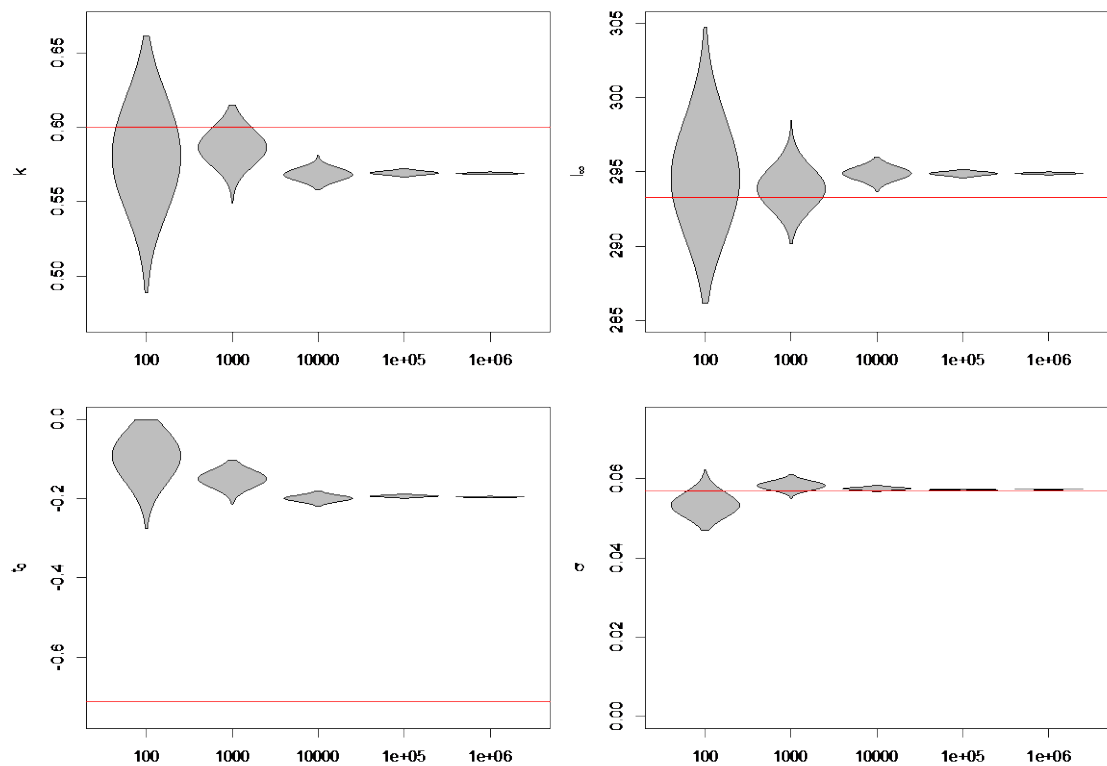


Figure 1. Ignoring uncertainty in spawning times can lead to inconsistent models.

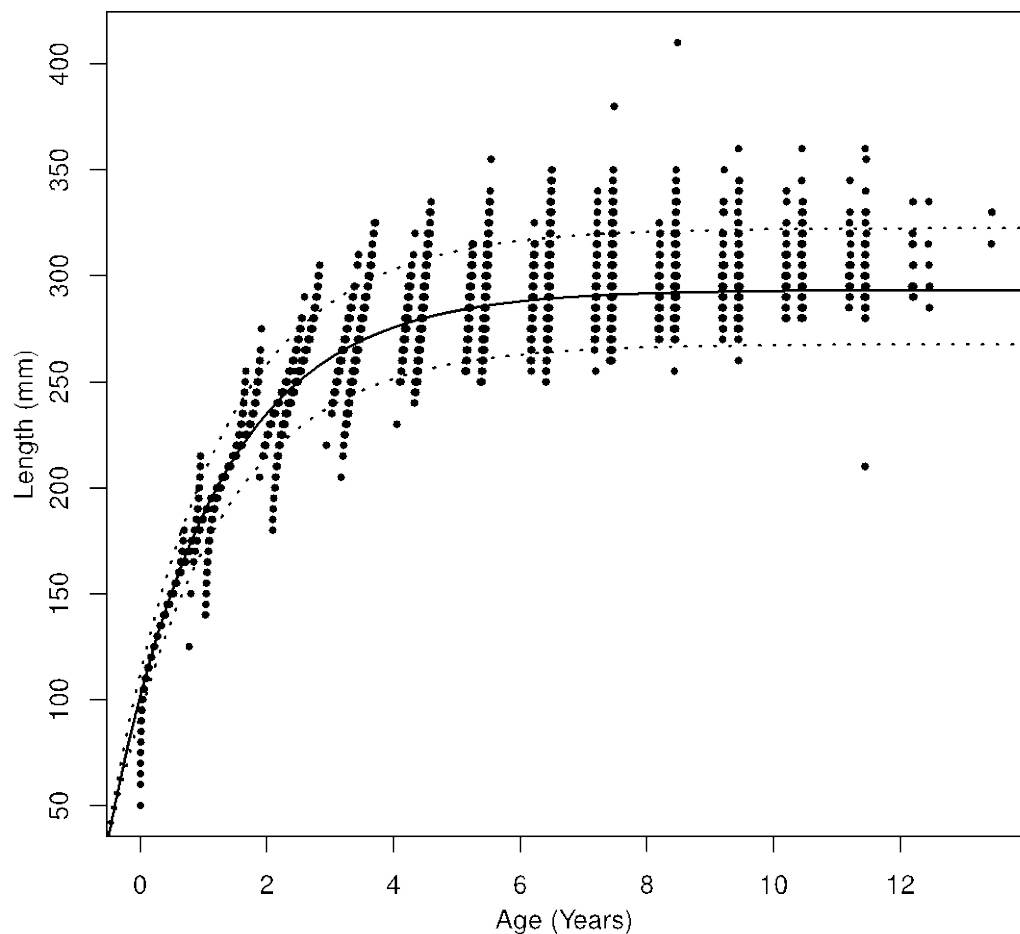


Figure 2. The median fitted model with the points being the mean age of each fish in the survey.

Spence MA and Turtle AJ (2017), Making the most of survey data: Incorporating age uncertainty when fitting growth parameters, *Ecology and Evolution* 7(17):7058–7068.

Ecoregion F: North Sea

Risks and benefits of catching pretty good yield in multispecies mixed fisheries

Robert Thorpe gave a presentation on the “risks and benefits of catching pretty good yield in multispecies mixed fisheries”. This describes a study which uses an ensemble of size-structured multispecies models to investigate the effects of different rates of fishing mortality (F) and fleet configuration on yield, biomass, and risk of stock collapse. The work is relevant because the EU commission has requested ranges of fishing rates that produce > 95% of MSY, rather than point estimates. The request makes sense because it reflects uncertainty in knowledge of the fishery and its management, whilst recognising the need for a negotiation space for the different stakeholders. However, these ranges are constructed stock by stock on a single species basis. They all make sense in isolation, but what about together? Here we answer that question with a multispecies analogue, which takes account of model parameter uncertainty,

management uncertainty, and fleet uncertainty, and evaluates outcomes in terms of risk and reward.

A multispecies maximum sustainable yield (MSY) and associated FMSYs for 21 modelled stocks in the North Sea were defined at the Nash equilibrium, where any independent change in F for any species would not increase that species' MSY. Fishing mortality ranges leading to "pretty good yield" (F-PGY) by species were defined as ranges yielding $> 0.95 \times \text{MSY}$ in an analogue of the ICES process for ranges based on single species assessments. We found that weight and value of yield from the entire fishery increased marginally when all species were fished at the upper end of F-PGY rather than at FMSY, but with small impacts on total fishery yield or value. While fishing anywhere within F-PGY ranges gives managers flexibility to manage trade-offs in multispecies mixed fisheries, our results suggest high long-term yields and disproportionately lower risk of stock collapse are achieved when $F < \text{FMSY}$ for all stocks.

There are 4 main components, a multispecies model framework (Thorpe *et al.* 2015), a definition of relevant fleets (Thorpe *et al.*, 2016), a determination of a multispecies MSY, and a method for evaluating outcomes in terms of risk and reward (Thorpe *et al.*, 2017). The methodology is shown schematically in Figure 1:

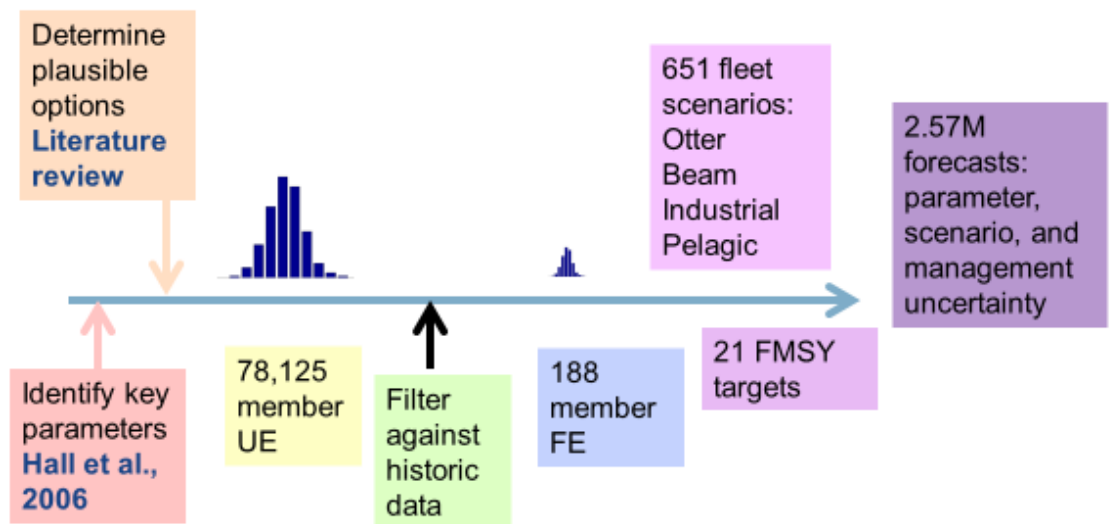


Figure 1. Schematic of study, showing how the multispecies model is tuned against data and then used in assessing outcomes of different fleet combination.

The model is based on the framework of Hall *et al.* (2006), but has been adapted to increase the inter-stock interactions, whilst a hockey-stick spawner recruit relationship has been implemented, along with stochastic recruitment (Thorpe *et al.* 2015, 2016, 2017). We considered results from a 188 member deterministic recruitment ensemble, or a 189 member stochastic ensemble (63 separate models \times 3 realisations of each). The 4 fleets used represent beam trawlers, otter trawlers, industrial trawlers, and pelagic trawlers. 651 patterns of fleet effort were considered, with each pattern being scaled up until the point at which the first choke stock was encountered (hence assuming that there are no discards). 21 possible management targets were considered, covering the F-PGY ranges in 10 intervals from the bottom to FMSY, and from FMSY to the top of the ranges in a further 10 intervals. All stocks were managed to the same point on the ranges.

We calculated gross revenues by multiplying the tonnage of catch by its economic value as determined by reference to UK market prices between 2008 and 2012. These values are shown in Table 1.

Table 1. First market price from the UK market 2008–12 for the model stocks.

STOCK	£ / TONNE	STOCK	£ / TONNE
Sprat	203	Whiting	1017
Norway pout	90	Witch	975
Sandeel	188	Gurnard	399
Poor cod	200	Plaice	1049
Long rough dab	668	Starry ray	736
Dab	668	Haddock	1243
Herring	327	Cuckoo ray	736
Horse mackerel	498	Monkfish	2911
Lemon sole	2829	Cod	2015
Sole	6941	Saithe	924
Mackerel	770		

There is a huge disparity in market values between stocks, with sole worth about 70x more than Norway pout for the same weight.

A 21-stock Nash equilibrium (NE) was calculated and used to define a community MSY. At the Nash equilibrium no stock yield can be improved by changing its mortality alone. The NE was chosen as a representation of MSY because it allowed yield curves to be drawn for all stocks with maximum yields intersecting at the NE, thereby enabling a construction of 95% ranges in a manner analogous to the ICES process for a single species. The NE also has the advantage of a clear definition, whilst one can easily test to see if it has been achieved. If the stocks could be managed as independent units, an NE would emerge if there was no collaboration between the stock managers. Thus the difference between the yield at the NE and the maximum possible yield is a measure of the “prisoner’s dilemma” or the potential benefits of collaboration in the fishery. The way in which this was determined is described in Thorpe *et al.* (2017).

Outcomes of managing to different points in the F-PGY ranges are shown for stochastic recruitment (left) and deterministic recruitment (right) in Figure 3.

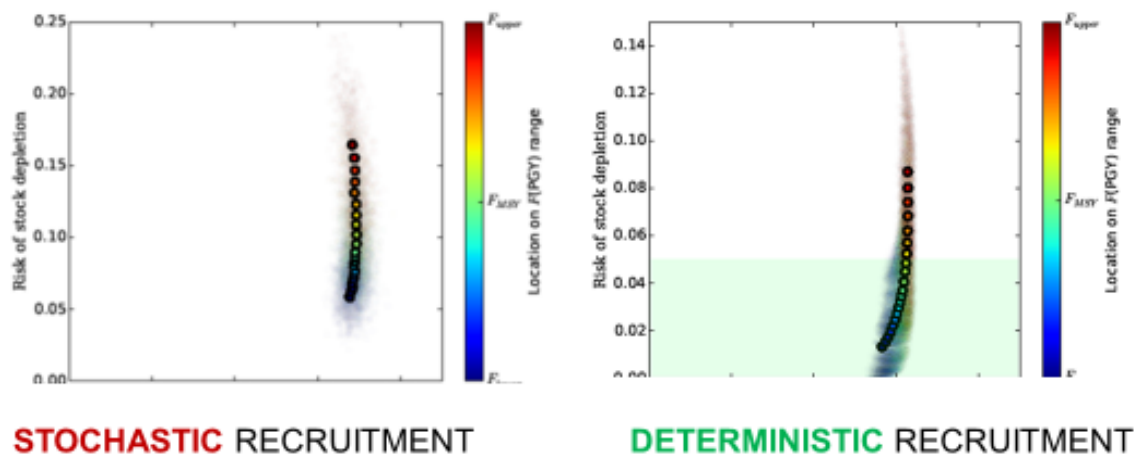


Figure 3: Risk/reward outcomes for different management targets (colours) and fleet options (spread of points of a given colour). Risk is measured in terms of the number of stocks whose biomass is reduced to less than 10% of that when unfished, whilst reward is in terms of gross revenue (catch x price).

The colours reflect the chosen management strategy (e.g. managing to the bottom, middle, or top of the ranges), whilst the spread of points of the same colour reflects the impact of different fleet configurations operating to the same management target. The larger coloured dots show the mean of the fleet outcomes for each management target. Comparing deterministic and stochastic outcomes shows that for low F targets, most of the risk is associated with stochastic recruitment, but for higher F targets, most of the risk is associated with biological process uncertainty rather than recruitment. In both cases, risk increases substantially moving up the F -PGY ranges, but yields only increase modestly, and even go down towards the top of the ranges.

Several conclusions can be drawn from our work. A) model parameter and recruitment uncertainty are both important. Uncertainty is normally treated in an MSE framework, with a big focus on recruitment, but sometimes only one operating model. Ideally MSE design should reflect the sources of uncertainty and consider structural and parameter uncertainty as well as recruitment. B) multispecies F at the NE is much higher than realised F in the mixed fishery. This is because the yields are sensitive to fishing mortality across all stocks, and the yield on one stock can be strongly influenced by fishing on another stock. So as we reduce fishing on large demersals such as cod, the risk to fisheries of their prey, such as herring, may increase. C) yields are sensitive to the fleet dynamics and the operation of choke stocks. D) FMSY ranges have the potential to be a useful management tool by providing short-term flexibility, but long term outcomes are best if we keep $F < F_{MSY}$ for all stocks.

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R.B. Thorpe, S. Jennings, P.J. Dolder (2017) Risks and benefits of catching pretty good yield in multispecies mixed fisheries, *ICES Journal of Marine Science*, DOI 10.1093/icesjms/fsx062

Multispecies and mixed fisheries management strategy evaluation in the North Sea

Robert Thorpe gave a presentation on a management strategy evaluation (MSE) being conducted using a length-structured multispecies and mixed fisheries model of the North Sea fish community. Five candidates for a community MSY (CMSY), the 21-stock Nash equilibrium, one based on single species assessments, and ones based upon the top, middle, and bottom of the ICES “pretty good” yield ranges were evaluated using a variety of Harvest Control Rules (HCR), with outcomes being assessed in terms of average risk of stock depletion and gross revenue (price x catch). The MSE was carried out with an ensemble of 63 models with stochastic recruitment, repeated 100 times for each scenario. In the absence of an HCR, we find that the lower PGY ranges are the safest option and the Nash equilibrium the highest yielding, with the other options being sub-optimal. Application of an HCR cuts risk and reward, the former more than the latter such that the HCR is useful. The impact of the HCR depends on its functional form and the point at which yield is reduced (MSY Btrigger). We find that the optimum choice for CMSY depends on societal views of acceptable risk, with no clearly optimum solution. However the upper part of the PGY ranges is never a good choice.

Previous work with LeMans (Thorpe 2015, 2016, 2017) has focussed on constant harvesting strategies – fishing at the same mortality regardless of stock status to evaluate the long-term impacts of the strategy. But in practice, fishing would not take place in this way, because it would be reduced if the stock status is poor. This is often done via a harvest control rule (HCR), a pre-agreed management procedure which determines how the fishing mortality target varies with stock status. In this study we ask what is the best way of achieving multispecies MSY (assuming this is defined as achieving the maximum possible yield for an acceptable risk using HCRs within a management strategy evaluation framework). Thus we take account of model parameter uncertainty, management target uncertainty, and fleet management uncertainty as well as management implementation uncertainty, evaluating outcomes in terms of risk and reward. A schematic of the experiment design is shown in Figure 1.

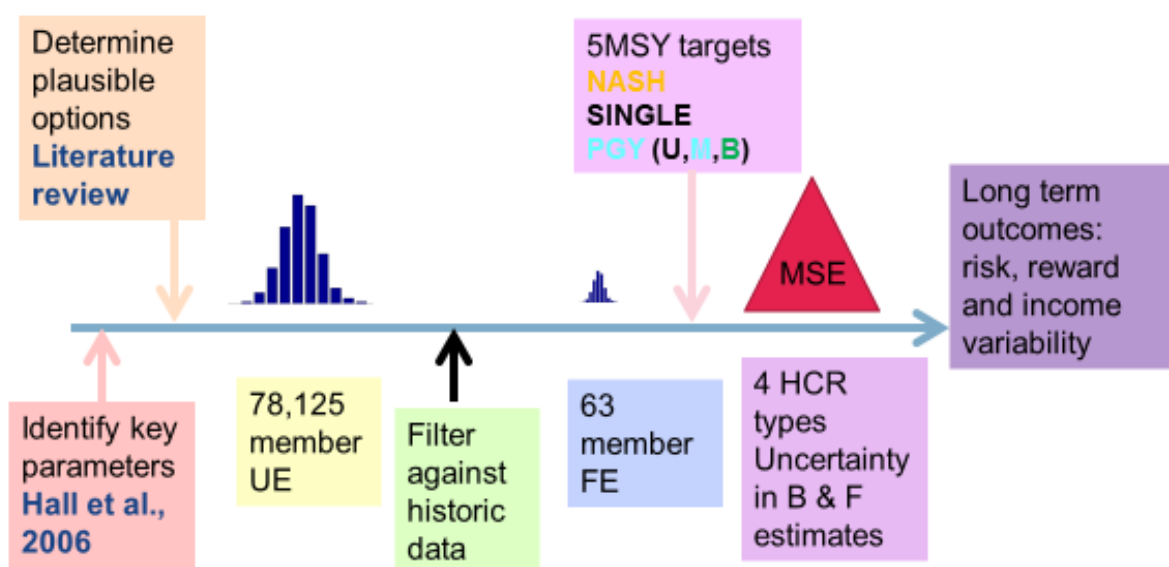


Figure 1. Schematic of the experimental design for the management strategy evaluation.

We use the same 63 member ensemble as in Thorpe *et al.* (2017), with each ensemble being evaluated 100 times to take account of stochastic variation in recruitment. 5 candidates for a community MSY were considered, one based on 2012 single species assessments (Thorpe *et al.* 2015), one based on the Nash equilibrium (Thorpe *et al.* 2017) and three based on the bottom, middle, and top of the “pretty good yield” ranges as defined by ICES. Of the 21 stocks, 7 (cod, haddock, whiting, sole, plaice, herring, and saithe) have published ranges as in Table 1:

Table 1. ICES estimates of “pretty good yield” for 7 North Sea stocks.

STOCK	F-PGY UPPER	F-PGY CENTRAL	F-PGY LOWER
Herring	0.39	0.33	0.24
Sole	0.37	0.20	0.113
Whiting	0.15	0.14	0.14
Plaice	0.30	0.21	0.146
Haddock	0.194	0.194	0.167
Cod	0.46	0.31	0.198
Saithe	0.49	0.36	0.21

The other 14 PGY ranges were generated using the following assumptions. 1) The average F across all stocks was maximised, 2) no fleet can have more than three times the effort of another, when they are both expressed as effort relative to the average effort between 1990 and 2010, and 3) there are no discards, so the fishery is limited as soon as the first choke limit is reached. Figure 2 shows the Fs that result from these assumptions.

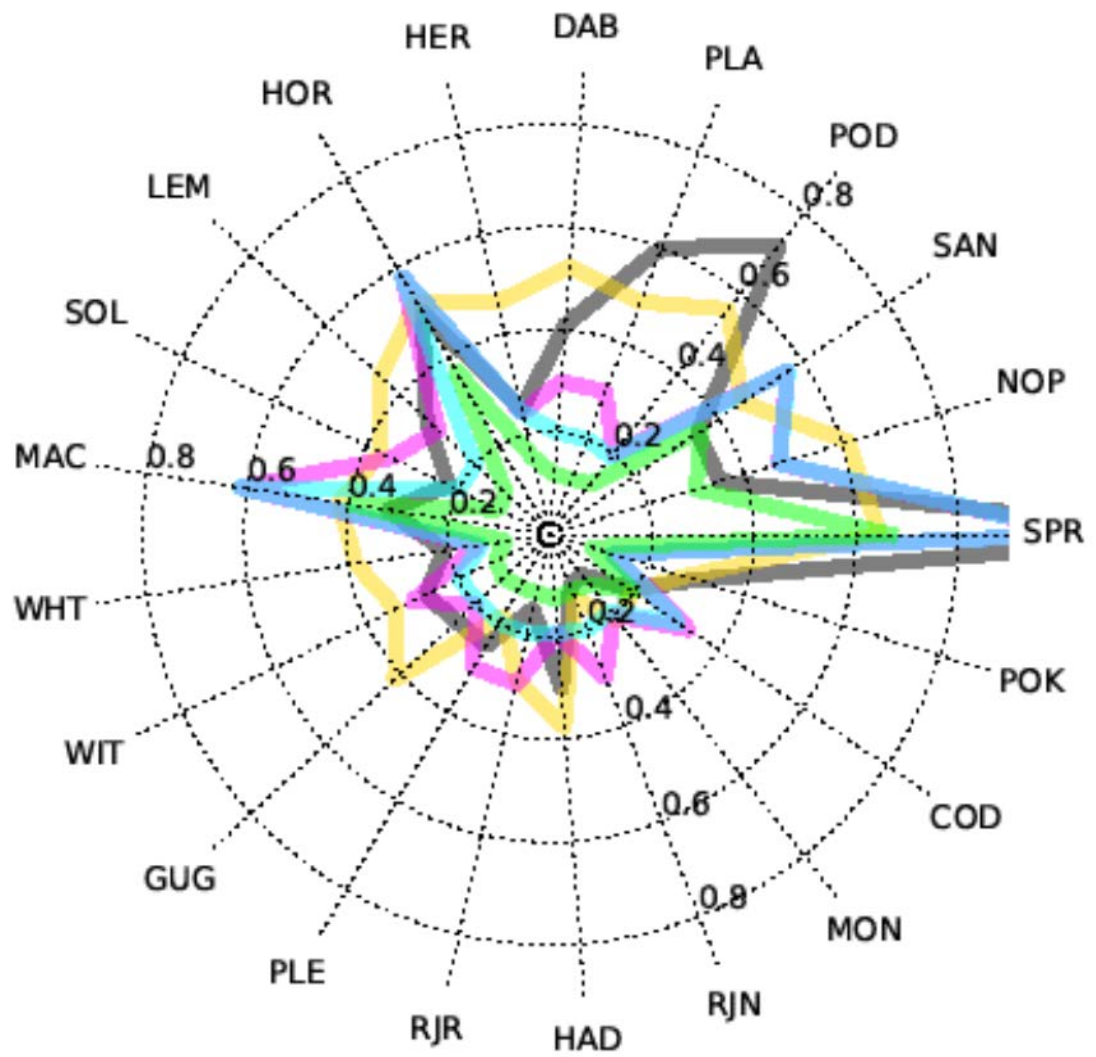


Figure 2. Radar plot of community MSY for a) estimates based on single species assessments (black), b) 21-stock stochastic Nash equilibrium (gold), c) upper PGY ranges (magenta), d) mid PGY ranges (cyan), and e) lower PGY ranges (green).

Our management strategy evaluation considered 4 types of harvest control rules, which are shown in Figure 3.

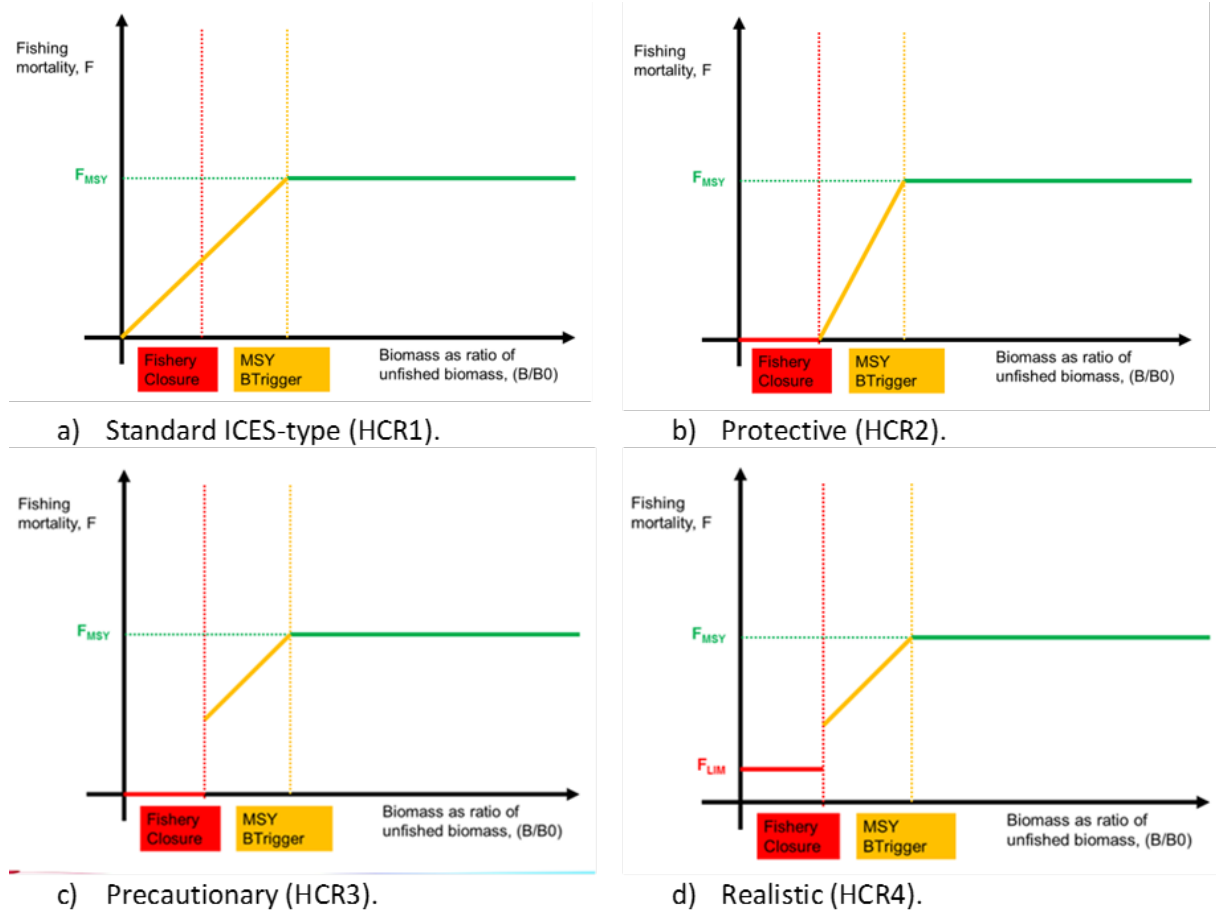


Figure 3. Schematic of the 4 types of harvest control rules used in the management strategy evaluation.

Within the MSE, the ensemble model acts as the operating model. Stock status is assessed by taking the ensemble mean biomasses adjusted by a log-normal error term of given size from 0 to 50%. The harvest control rule is then used to generate a target F for the stock, which is implemented with lognormal uncertainty of given size from 0 to 30%. Stock status is assessed annually, after which the newly ascertained F is applied for the next year.

For the reference case with constant F and no HCR, results are shown in Figure 4.

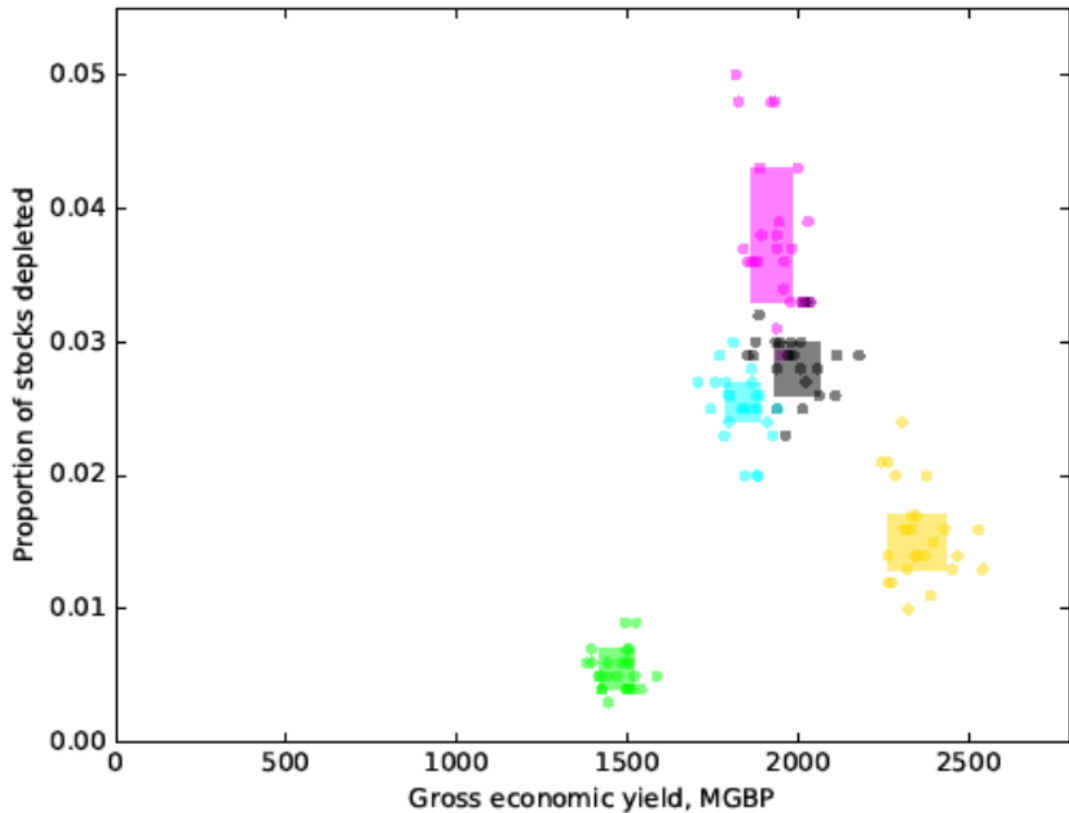


Figure 4. Risk – reward outcomes for constant F strategies. Black = single species, Gold = Nash, GreEn = L-PGY, Cyan = M-PGY, Magenta = U-PGY.

We find that the Nash equilibrium gives the highest yield, and L-PGY is the safest. A choice between these would depend on societal risk appetite. The other solutions are sub-optimal. Application of an HCR reduces both risk and yield, the former more than the latter. The nature of the reduction depends upon the form of the HCR, the choice of MSY Btrigger, and the level at which a stock is considered depleted.

Overall we find the following:-

- a) The best outcome depends upon societal views of risk and reward – there is no choice of CMSY that is clearly optimal, although some can be dismissed as sub-optimal.
- b) The upper PGY ranges are never a good choice (consistent with Thorpe *et al.* 2017).
- c) The annual operation of an HCR reduces both risk and yield. Yield reductions are more modest than those of risk, making the HCR a valuable management tool, independent of its form (amongst those considered here), definition of risk, or definition of MSY Btrigger.

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SMS (Stochastic Multispecies Model) in the North Sea

A new SMS keyrun is presented under ToR B.

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 Institute of Sea Fisheries (TI-SF) 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 1). 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.

TI-SF started 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). Different species models are currently tested to provide habitat preferences as input to Ecospace (Figure 2). Until next year it is planned to have a calibrated operational model.

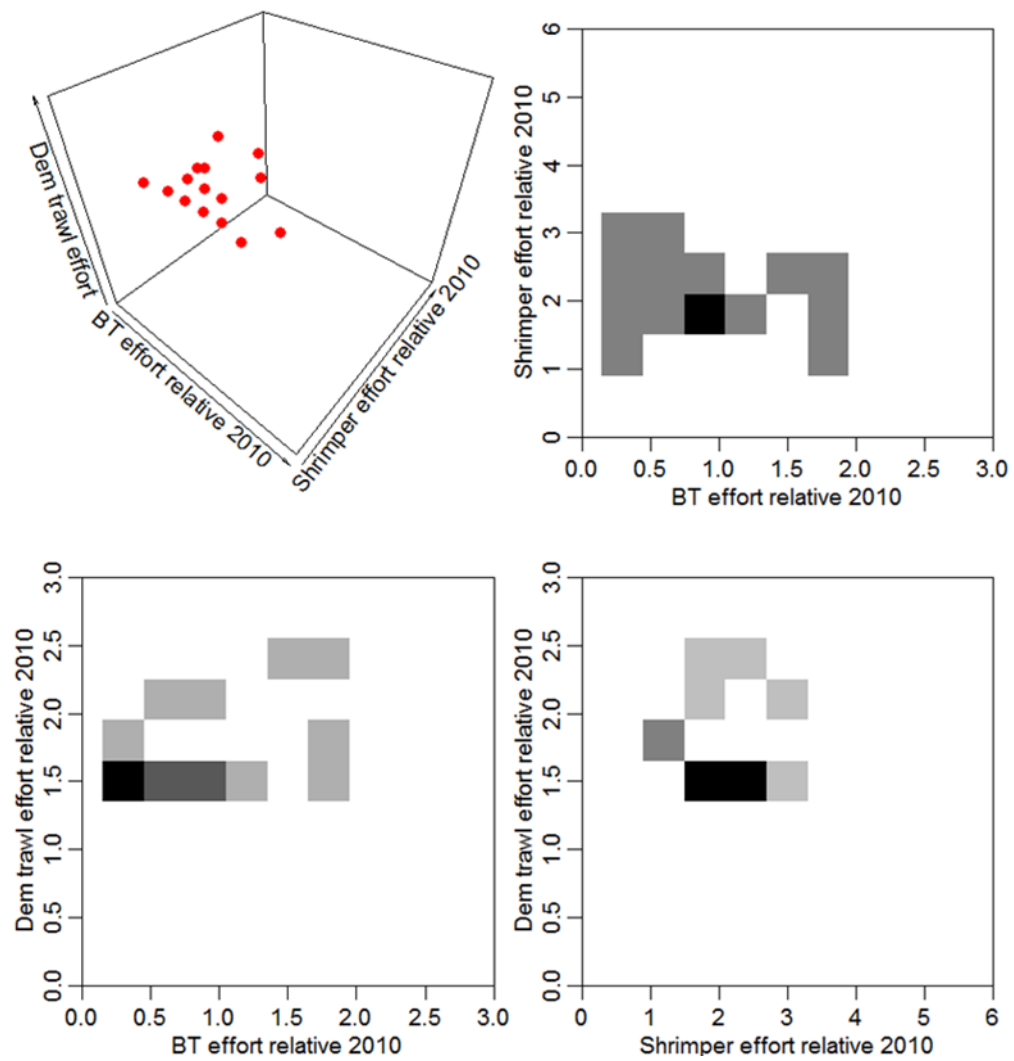


Figure 1. Spheres indicate effort regimes that lead to all four scope species — cod, plaice, sole and brown shrimp — to be simultaneously caught at 30% of their respective maximal possible catches.

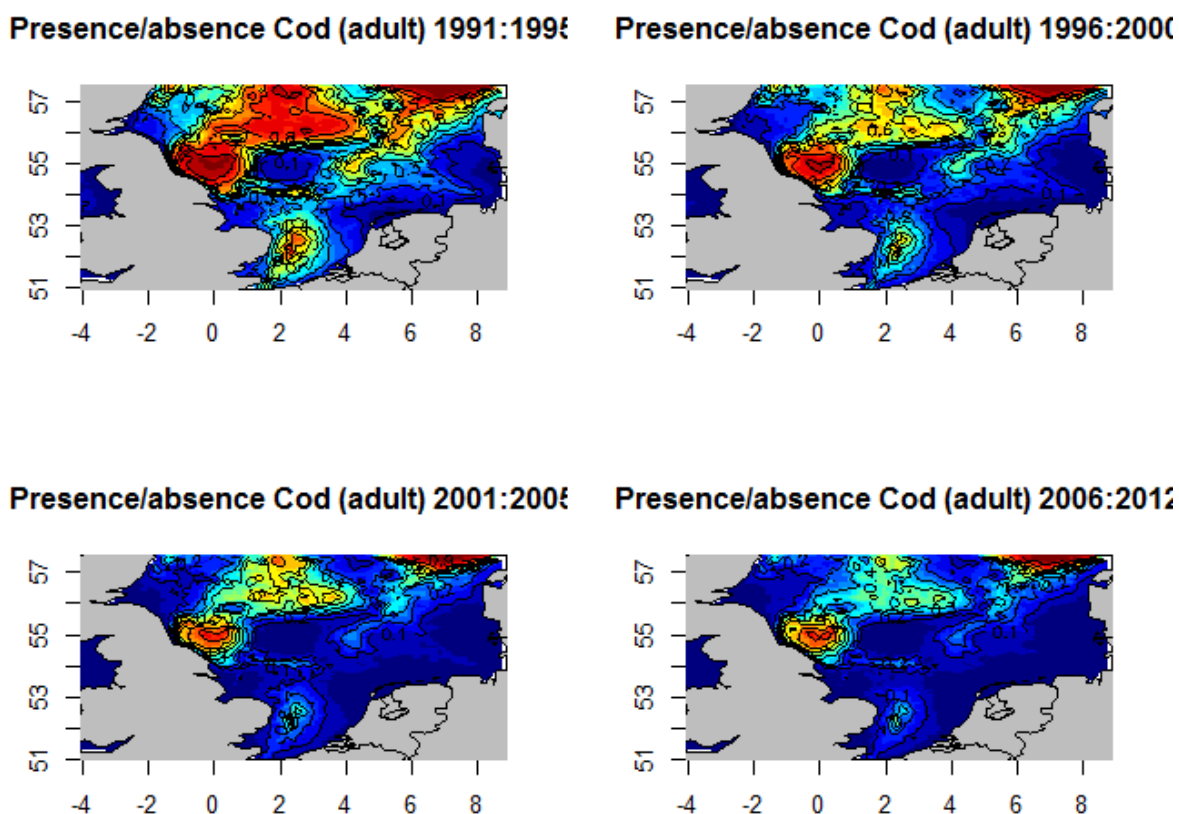


Figure 2. Distribution of cod in the North Sea modelled with a presence/absence model. Different model types (e.g., Presence absence vs. Hurdle model vs. Vast) will be tested to derive input for habitat preferences in Ecopsace.

Ecopath with Ecosim in the English Channel

A new 2D spatial model of the extended area of the Bay of Seine has recently been completed and is being calibrated for Ecosim and Ecospace modelling. This work was done using the EwE software to evaluate the impact of the recently installed windmill farms to the biological resources and exploitations in the Bay of Seine and adjacent areas. The model developed is an improved version of the Seine estuary EwE models developed by Tecchio *et al.* (2015) and the Courseulle-sur-Mer EwE model developed by Raoux *et al.* (2017). Six EwE models were developed by Tecchio *et al.* (2015) considering only 15 groups per model to show how the trophic structure change considering 6 habitat types (Figure 1). Within the downstream part of the estuary proper, we identified three estuarine habitats, i.e. the northern channel (Fosse Nord, FN), the southern channel (Fosse Sud, FS) and the central navigation channel (Chenal, CH). Extending into the eastern Seine Bay, we identified 3 other spatial marine compartments based on their sediment composition and benthic habitat type (E4, E14, and E2), following guidelines by the European Union Nature Information System (EUNIS classification of benthic habitats). Due to the general current regimes in the Bay, and as the discharge plume of the Seine river flows primarily adjacent to the southern coast, the marine habitats under its influence were mainly E4 and E14, while the E2

was considered the habitat under the least influence of the plume (Salomon and Breton, 1991).

The trophic structures of the 6 different habitat models are shown in Figure 2. Keystonness analyses were also performed to compare the 6 different EwE habitat models developed (Figure 3). Ecological network analyses (ENA) were also performed directly in each EwE habitat model. The extended Bay of Seine EwE model includes 43 groups where both the Ecosim and Ecospace modules are being calibrated.

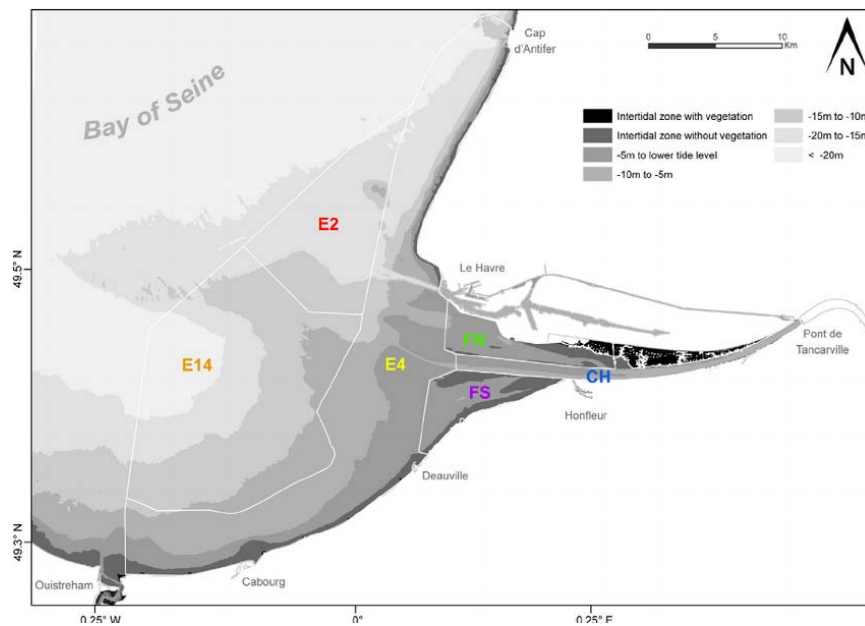


Figure 1. Map of the study area in the Seine estuary, northern France, with bathymetry and showing the subdivision of the six modelled habitats. Source: Le Havre and Rouen port authorities, SHOM, and GIP Seine-Aval.

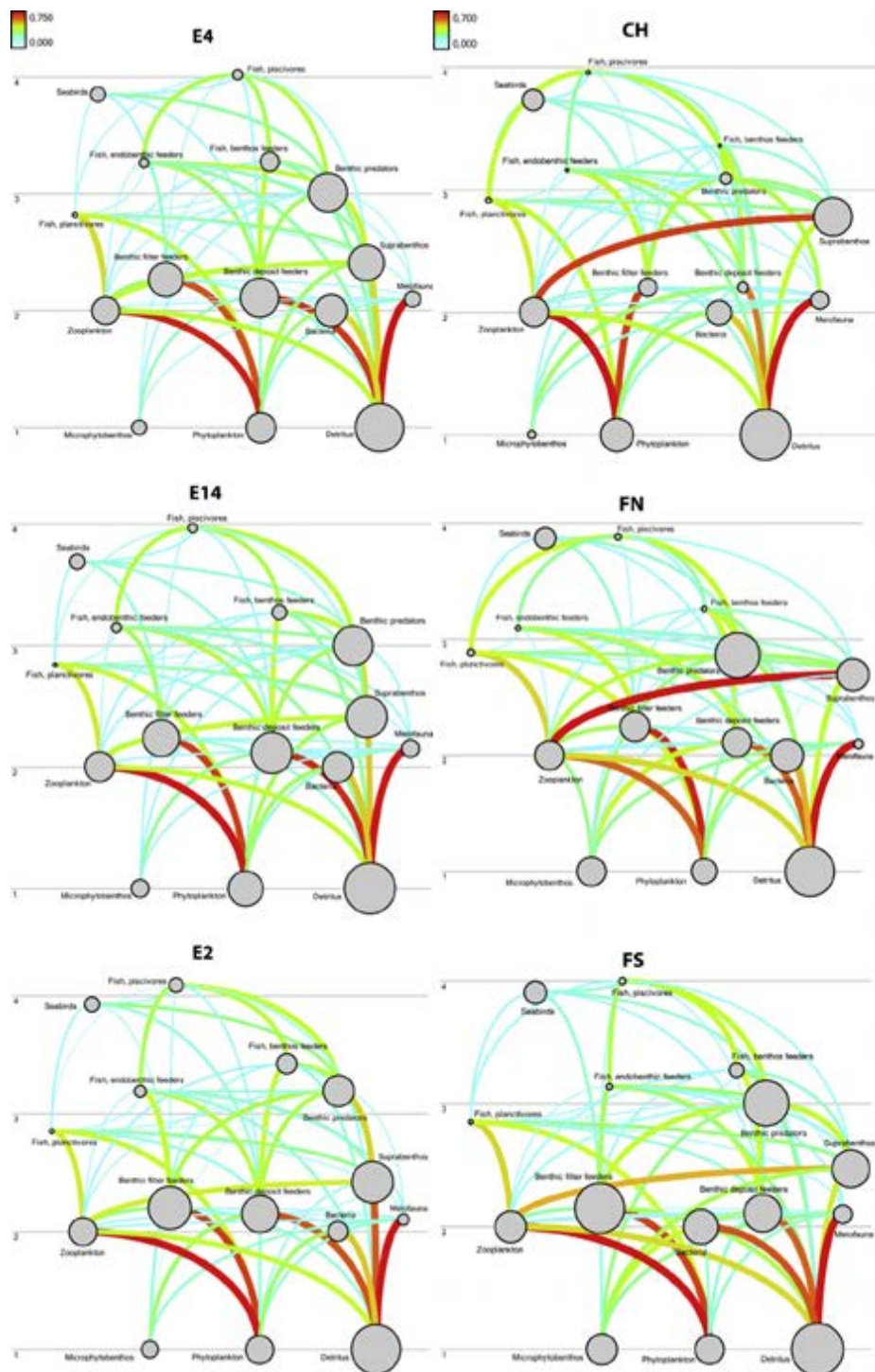


Figure 2. Flows and biomasses of the three estuarine habitats (E4, E14, E2, CH, FN, FS). Flows values by diet are indicated by the colour scale, while group biomasses are logarithmically proportional to the area of their respective circles. Groups are vertically positioned by their calculated trophic level.

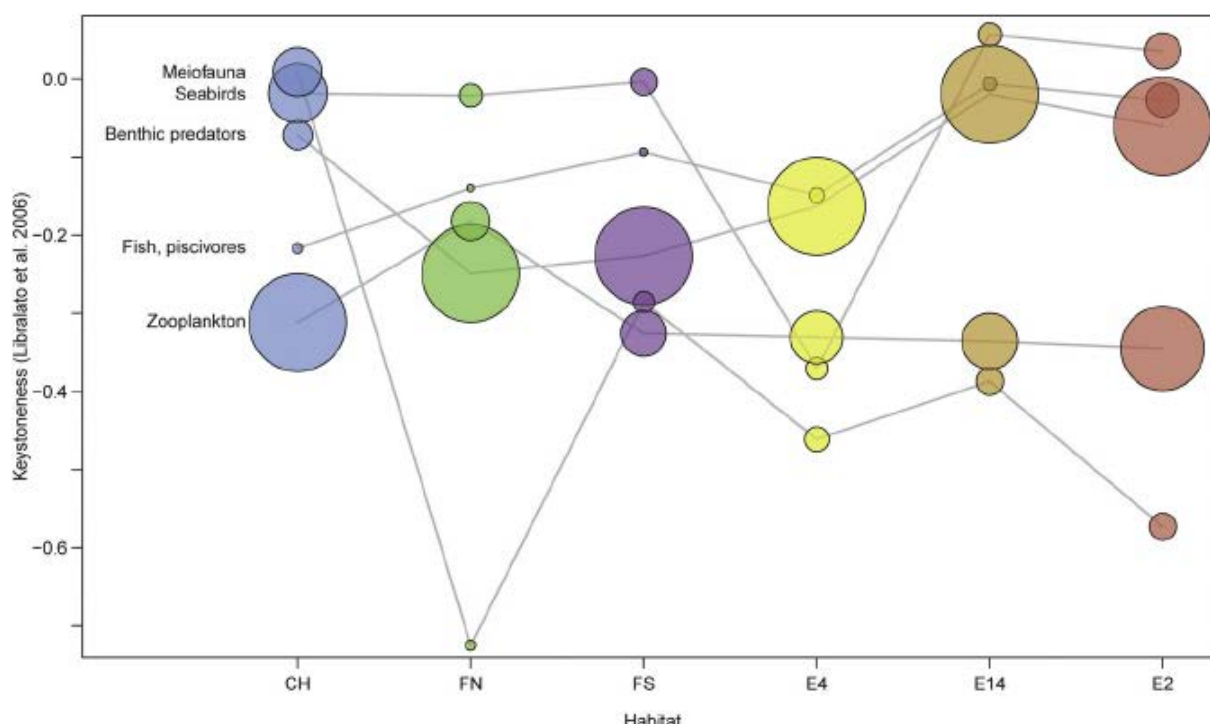


Figure 3. Keystone index by Libralato *et al.* (2006) by functional group for each of the six modelled habitats (indicated also by colour codes). The area of circles is proportional to each group biomass, relative to its habitat. Only groups with high keystone index (>0.55) are represented.

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Ecoregion G: South European Atlantic Shelf

The Gadget multispecies model in the Bay of Biscay

A multispecies model is being built using Gadget in the Bay of Biscay aiming at assessing the most relevant stock for the Basque fisheries operating in that area from a more holistic perspective and evaluating the effects of existing intra and inter-species relationships on the dynamics of these stocks.

At this point, single species models are operational for 6 stocks that inhabit and coexist in the study area: Northern and Southern hake, anchovy, Northern sardine, Western horse mackerel, Mackerel. And a new single-species model for Southern sardine is under construction.

Based on these models, a multispecies model has been set up aiming at linking all these species through the trophic relationships existing between them, using data and information from Velasco (2007), which is mainly focused on hake.

Some genetic studies are being developed in parallel to obtain new information about the diet of these individuals and look for trophic relationships between them, other

than the ones identified by the previously mentioned study. This is still work in progress and will hopefully be shown next year.

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TheEwE model in the Bay of Biscay

And Ecopath with Ecosim (EwE) model has been used to analyze the food web dynamics of the marine ecosystem of the Bay of Biscay (French Continental shelf) and see how the implementation of different management measures can affect it both in the short and in the long term. As an example of that, the potential effects of the Landing Obligation have been analyzed, using the outputs from a bioeconomic model developed by Prellezo *et al.* (2016) using FLBEIA (Garcia *et al.*, 2017) to condition or force the forecast simulations in Ecosim (Andonegi& Prellezo, in prep).

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GADGET model for cetacean fishery interaction in the Iberian peninsula (Camilo Saavedra and Santiago Cerviño)

A Gadget modelling framework is currently under development in the Iberian Peninsula to analyze the cetacean fishery interaction. The study area comprises the Spanish and Portuguese shelf and oceanic waters (ICES subdivisions VIIIc and IXa); corresponding with the Southern European hake stock distribution and the area of overlap with the common (*Delphinus delphis*) and bottlenose (*Tursiops truncatus*) dolphins populations. The main challenge to build this model was the scarcity of cetacean information. Cetacean model parameters (e.g. natural mortality, growth, fecundity, consumption) were derived from the analysis of stranded and by-caught individuals, as well as from existing information in other areas. By modelling cetacean abundance, predation and the mortality caused by their interaction with the fishery we can explore the effects of fisheries management measures and the tradeoffs between two different targets, i.e. maximize the fisheries yield and keep dolphin populations in a healthy status. This work is headed by IEO and carried on within the project Mare-Frame (EU FP7 613571).

Preliminary results suggest:

- The multispecies model improves the quality of the fit compared with hake single species model (likelihood 1001 vs. 1015).

- The biomass of hake consumed by cetaceans is in similar scale than historical hake catches.
- Hake reference points (Fmsy, Bpa, etc) from the multispecies model have been estimated following the same rationale than in ICES and cetacean by-catch limits (GES-MSFD) were also set.
- Short and medium term projections have been developed.
- The model could provide advice for hake considering impact on hake and cetaceans

Next steps should focus on uncertainty sources such as:

- Implement dolphin variable abundance considering existing information such as PELACUS survey trends for common dolphin, SCANS-III survey for either dolphins or variability on effort for fleets by-catching dolphins.
- Perform sensitivity analysis for more uncertain parameters (dense-dependent fecundity, M-at-age, etc.) should also be analyzed.

Conceptual model

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

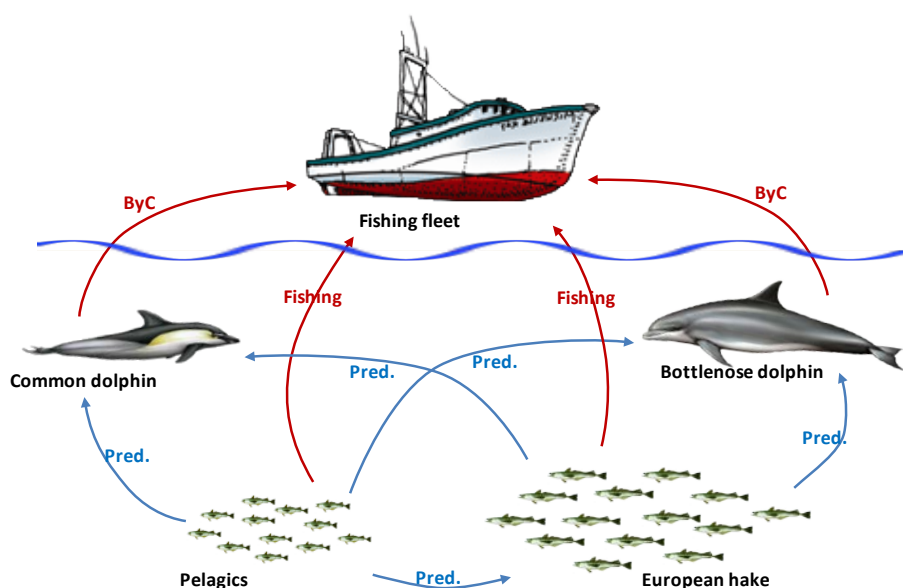


Figure shows conceptual model interactions among fishery, cetaceans, hake and pelagic.

Settings and assumptions

The main assumption of the model selected is that populations are closed populations. The model assumes that there is no migratory flow between the southern and northern stocks of hake, or between dolphin stocks with the rest of their populations. Moreover, the abundance of dolphins stocks has been stabilized over time and only varies when the model is projected under different scenarios. The abundance used was calculated only in shelf waters, assuming that dolphins inhabiting this area have complete access to the prey species distributed in this area, not deeper than 500 m. The natural mortality of hake excluding predation (M1) has been considered constant for every age class as in the original single-species model. Natural mortality of dolphins is also constant for the study period but different for every age class. However, bycatch mortality was estimated as equal proportion for every age class and sex.

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

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

Parameterization and model fitting

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

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

Conclusions

A GADGET multi-species model was developed to explore the interactions between the Southern stock of the European hake (*Merluccius merluccius*) and the fleet exploiting hake and two important hake predators: common and bottlenose dolphins. It extends the current GADGET Southern hake assessment model used by ICES to give advice to EU. This extension considers two species of small cetaceans, Common and Bottlenose dolphins, which are considered the main hake predators in the area. The model provides a new platform to give ecosystem advice considering trade-offs between fisheries and cetacean. The modelling exercise shows that it is possible to build this multispecies model with the existing cetacean information. However there are still some issues to address such as exploring options to allow variable cetacean abundance (Saavedra *et al.*, 2017a and b) or explore model sensitivity to more uncertain parameters.

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Ecoregion H: Western Mediterranean Sea

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

Ecoregion I: Adriatic–Ionian Seas

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

Ecoregion J: Aegean–Levantine

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

Ecoregion K: Oceanic northeast Atlantic

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

Ecoregion L: Baltic Sea

Please see specific reports from the Baltic Sea under ToRs C and G.

Ecoregion M: Black Sea

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

Ecoregion: Canadian Northwest Atlantic

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

Ecoregion: US Northwest Atlantic

There are updates to several models ongoing in the Northeast US; other work is reported under ToRs C, D, and G.

Atlantis update

The Northeast US Atlantis model (Link *et al.*, 2010) is being updated to the current Atlantis codebase. To complete this, new hydrography inputs from a regional oceanographic model are required. This work is in progress. Calibration and skill assessment against regional datasets are expected within the next year. Contact: Ryan Morse (Ryan.Morse@noaa.gov).

Rpath multistep and linked geographic regions

Recent modifications to the Rpath code (R implementation of Ecopath with Ecosim) allow users to run simulations in multiple steps. Self-contained within R, this allows the model to evaluate its current state and modify parameters if necessary. This can allow users to set-up various harvest control rules to conduct management strategy evaluations or allow multiple models to be linked via migration terms. This latter usage will allow for various connectivity studies either between adjacent ecological production regions as demonstrated by Lucey *et al.* or between estuaries, near shore regions, and open ocean. The code is available at https://github.com/slucy/Rpath_Dev. Contact: Sean Lucey (Sean.Lucey@noaa.gov)

Comparative performance of multispecies models

A project is in progress to develop and evaluate two multispecies models for tactical use in providing fisheries management advice. A general multispecies statistical catch-at-age model (Curti *et al.*, 2013) and a length-based multispecies model (Gaichas *et al.*, 2017) are fit to a dataset of the Georges Bank fish community to evaluate model performance. The primary objective of this work is to assess the impact of structural uncertainty in length- and age-based multispecies population dynamic models on the estimation of underlying population parameters, with a focus on biomass, recruitment, mortality, and predation as metrics of performance. Contact: Jason Boucher (jason.boucher@noaa.gov)

Ecosystem indicator reporting for fisheries management

State of the Ecosystem (SOE) reports were produced in a new format for the New England and Mid-Atlantic Fishery Management Councils in 2017. The format is brief (total <20 pages) and provides ecosystem information in a management-relevant context by aligning ecosystem indicators with management objectives, including seafood production, profits, employment, provision of recreational opportunities, stability, and cultural practices and attachments, as well as ecological objectives such as maintaining habitat, trophic structure, biomass and productivity. In the Mid-Atlantic region, indicators from these reports are being incorporated within a risk assessment framework to address Ecosystem Approach to Fishery Management policy objectives. Reports for the Mid-Atlantic and New England can be found online respectively: http://www.mafmc.org/s/Tab02_2017-04_State-of-the-Ecosystem-and-EAFM.pdf and

http://s3.amazonaws.com/nefmc.org/2_2016-State-of-the-Ecosystem-Report.pdf . Contacts: Sarah Gaichas and Sean Lucey (Sarah.Gaichas@noaa.govSean.Lucey@noaa.gov)

Current ecosystem considerations were summarized in a risk analysis framework to provide ecological context to the stock status of the 19 stocks assessed in the Operational Assessment, including condition factor, productivity analyses and habitat modelling as an alternative index of biomass. Potential impacts of the recent record high fall bottom and sea surface temperatures with subsequent thermal habitat reductions and range shifts were assessed for each stock, taking into account each stock's thermal preferences and vulnerability to climate change. Contact: Laurel Smith (Laurel.Smith@noaa.gov)

Herring Management Strategy Evaluation (MSE)

Models of herring, predator populations, and fishery economics were linked to evaluate harvest control rules for Atlantic herring. In 2016, New England Fishery Management Council (Council) initiated a stakeholder process to elicit objectives and performance measures for a management strategy evaluation (MSE) to develop harvest control rules that considered the role of herring as forage in the ecosystem. Models were developed and applied between stakeholder workshops held in May and December 2016.

The general objective for the Council was to answer "how do changes in herring population abundance affect predator populations?" This is a different and more complex question than that addressed in the 2012 benchmark herring assessment "how much herring is consumed by predators?" Council specifications and time constraints and did not permit development of integrated multispecies models (existing models account for predation mortality on herring, but not "bottom up" herring impacts on predators), nor spatial or seasonal models accounting for migrations of wide-ranging predators into or out of the Northeast US shelf ecosystem. At the initial stakeholder workshop, it was agreed that separate "general predator" models linked to herring would be a reasonable approach, with the goal of developing one model for each of the four predator categories: highly migratory fish, groundfish, seabirds, and marine mammals.

Predators were therefore modelled with fairly simple delay-difference population dynamics that allowed different predator population processes to be dependent on some aspect of herring population status, following (Plagányi and Butterworth, 2012). Each predator model takes output from the herring OM as input, and outputs performance metrics identified at the stakeholder workshop as in Figure 1. While this allows "bottom up" effects of herring on predators to be examined, this configuration does not consider "top down" effects of predators on herring, or simultaneous interactions of multiple predators with herring.

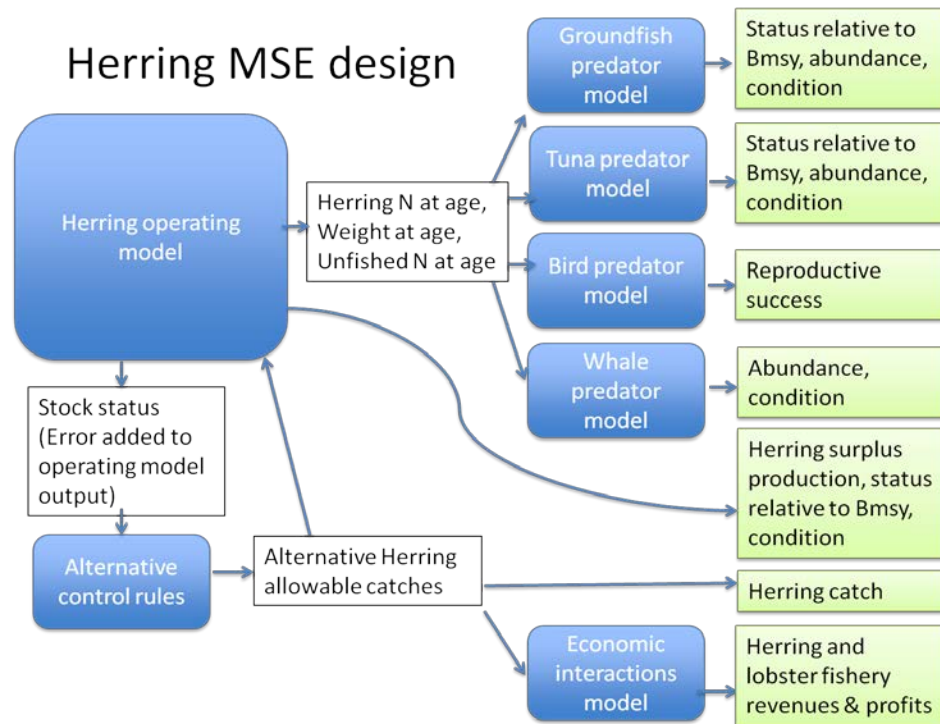


Figure 3. Planned MSE design, models in blue, outputs linking models in white boxes, performance measures linked to stakeholder objectives in green.

There were two components of predator modelling for the herring MSE: a predator population model, and a herring-predator relationship model to link herring with predator populations. Predator population models were based on either the most recent stock assessment for the predator or from observational data from the Northeast US shelf. Herring-predator relationships were based on either peer-reviewed literature or observational data specific to the Northeast US shelf. There was sufficient information for population modelling of Bluefin tuna (representing tuna), common terns (representing birds), and spiny dogfish (representing groundfish). Modelled herring relationships included herring condition (population average weight) affecting tuna condition (population average weight), herring total biomass affecting tern reproductive success, and herring total abundance affecting dogfish survival. However, data were insufficient to build a population model for any marine mammal species in the region. Potential effects of changes in herring production and/or biomass on marine mammals were instead evaluated using an updated version of an existing food web model for the Gulf of Maine (Link *et al.*, 2006, 2008, 2009) and incorporating food web model parameter uncertainty. Full documentation is available at:

http://s3.amazonaws.com/nefmc.org/4.TechnicalDetails_AtlHerringMSE_2017Feb24.pdf

The herring MSE included 8 herring operating models bracketing uncertainty in herring productivity (steepness and M), herring growth (observed weight at age from the 1970s vs. the recent period of poorer growth), and assessment bias; for each operating model 5,460 control rules were tested. For each control rule, 100 replicate simulations reflecting stochastic herring recruitment variability were run for 150 years each. Each of these simulated time-series was passed to each predator model and multiple output metrics were recorded based on the final 50 years of each replicate simulation.

We found that operating model configuration drives tuna average weight. Separating operating models with historical herring weight at age) from those with recent weight at age demonstrates the primary contrast in tuna results. After this difference in operating models is accounted for, there is far less contrast in the median performance of different control rules for tuna.

Examining tern productivity results by operating model shows little contrast across operating model uncertainties, but differences in performance between control rule types. The biomass-based control rule implemented for 3 years with a constraint of 15% change between specifications showed a wide range of variability in performance across control rule variants, as did the constant catch control rules.

Cases of poor status observed for dogfish were limited to two control rule types within the herring operating models specified with high natural mortality and low stock-recruit steepness representing a poor herring productivity state. The control rule types performing poorly for dogfish under poor herring productivity were the same performing poorly for tern productivity. Also similar to terns, herring average weight did not affect dogfish results.

Overall, food web modelling showed that a simulated increase in herring production in the Gulf of Maine may produce modest but uncertain benefits to marine mammal predators, primarily because increased herring was associated with decreases in other forage groups also preyed on by marine mammals. Three food web perturbations were compared: a forced 10% increase in herring (Small pelagics-commercial) production relative to the base model, a "low herring" run that forced herring biomass from current to half of current over 10 years and then held it at half of current for the remainder of the 100 year run, and a "high herring" run that forced herring biomass from current to 150% of current over 10 years and then held it at 150% of current biomass for the remainder of the 100 year run. All runs were compared to the unperturbed baseline run for each ensemble member, and proportional differences in each group's production from base (averaged over the final 10 years of the run) are reported.

In comparing all three scenarios (using only the 50% interquartile ranges) it becomes clear that the two biomass scenarios (red and blue) represent rather extreme changes in the ecosystem relative to the 10% change in production (Fig 2). Further, increasing herring biomass in the ecosystem (red) had a wider range of results, and therefore uncertainty, relative to decreasing herring biomass (blue).

In all but the extreme increase in herring biomass scenario, response of predator productivity (including that of marine mammals) was damped relative to the change in herring production. The 1.5x herring biomass increase scenario resulted in generally similar patterns of response across species as the more modest 10% production increase scenario, although the uncertainty in response increased disproportionately as indicated by the extent of the 90% interquartile range of productivity. This suggests that the impacts of greatly increased herring biomass in the Gulf of Maine ecosystem may be more uncertain than greatly decreased herring biomass.

The advantage of using a full food web model to address the impacts of changing herring biomass is that it integrates both bottom up and top down food web responses and tradeoffs between species that could not be considered in the more detailed modelling of herring relationships with individual predators. In particular, the tradeoff between increased herring and decreased productivity of other forage groups demonstrated in these scenarios has the potential to diminish any expected benefits to predators from "leaving more herring in the water" when herring is con-

sidered in a single species context, even as an important forage fish. Predators in the Gulf of Maine and throughout the Northeast US shelf ecosystem tend to be opportunistic and rely on many prey, so tradeoffs between prey types caused by management for one prey species should be weighed carefully.

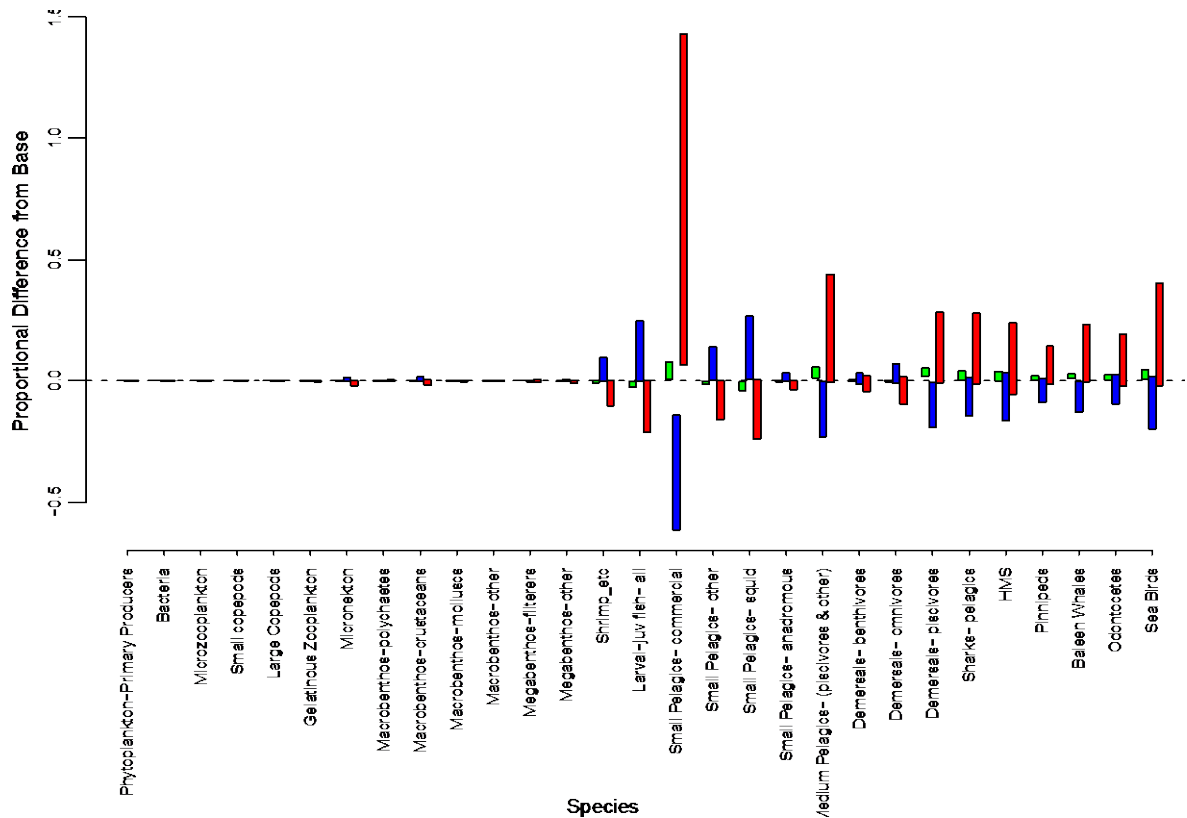


Figure 4. Relative change in group production from a forced 1.5 x base biomass increase for Small Pelagics-commercial. Boxes represent 50 percent of model results of model results within the 5677 member ensemble.

Our models were designed for evaluating alternative herring control rules, not predator stock assessment and population prediction. We caution against generalizing results for these particular predators to other predators, as population parameters and herring relationships differ.

Overall lessons from this process can inform future work. First, isolating a clear herring-predator relationship from observations is difficult or impossible when other factors dominate predator dynamics. Second, even with good observations, the modelled herring-predator relationship may require strong assumptions and not be statistically significant due to the many other factors affecting predators (e.g. terns). Third, apparent positive herring-predator relationships may not arise from the modelled mechanism (e.g. dogfish). Finally, a clear herring predator relationship is not satisfactory when it does not answer the question of interest to stakeholders (e.g. tuna).

Although we selected predators with high herring diet proportions, observed predator population responses to herring alone do not dominate dynamics, and our herring-predator relationship models reflect that. Predator responses to aggregate prey dynamics are likely to be much clearer than responses to individual prey in the Northeast US ecosystem given its food web structure. While modelling this is a more complex and time-consuming undertaking, the results may give clearer advice for

managers making decisions regarding multiple simultaneously exploited prey and predators within the ecosystem. Contact: Sarah Gaichas (Sarah.Gaichas@noaa.gov)

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Annex 4: 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 eco-system models for different ICES regions

Stock Annex for the ICES North Sea SMS configuration

Working Group Working Group on Multispecies Assessment Methods (WGSAM)

Date: November 2017 (after the WGSAM 2017 meeting in October)

Predatory species Assessed species: Cod, haddock, saithe, whiting, mackerel

Species with given input population size: North Sea horse mackerel, western horse mackerel, grey gurnard, starry ray, hake, fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin, razorbill, grey seal, harbour porpoise

Prey species Assessed species: Cod, haddock, herring, Norway pout, southern North Sea sandeel, northern North Sea sandeel, sprat, whiting,

Stock Assessor Morten Vinther

Summary

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

In the present SMS analysis the following predator and prey stocks were available: predators and prey (cod, whiting, haddock), prey only (herring, sprat, northern and southern sandeel, Norway pout), predator only (saithe, mackerel), no predator prey interactions (sole and plaice) and 'external predators' (8 species of seabirds, starry ray, grey gurnard, North Sea horse-mackerel, western horse-mackerel, hake, grey seals and harbour porpoise). The population dynamics of all species except 'external predators' were estimated within the model.

2017 key run

A key run for the North Sea SMS model, including data for the period 1974–2016 was produced at the 2017 WGSAM. This key run replaces the key 2014 key run. The new key run includes revision and updates to the input data and a few modifications of the structure of the model.

All stock assessment models were updated with the most recent data and stock numbers were corrected where the stock area did not correspond to the key run area (the North Sea proper, division IV). New estimates of quarterly mean weight at age in the stock produced for stocks where this information was not available from the stock assessments. These values were lower than previous estimates and this increased the range of age groups of cod consumed by marine mammals to also include significant impacts on cod of age 3. To improve the inclusion of mackerel in the model, this species was included as a fully modelled predator in the model and the proportion of the mackerel stock which occurs in the North Sea in each quarter was reviewed and new estimates produced. Consumption (ration) of the main fish predators, including mackerel and horse mackerel, was revised to reflect the most recent knowledge of evacuation rates leading to changes for mackerel and horse mackerel (lower consumption rates). Finally, the quarterly overlap of the species with sandeel was evaluated and adapted to better mirror the stomach contents observed. Diet data for the predatory fish were bias corrected to take into account that evacuation rate is a function of prey energy density, prey armament and ambient temperature. This correction gave in general lower diet proportion of the SMS prey fish and higher proportion of “other food” compared to the observed stomach contents which previously have been used directly as diet. Diet data for harbour porpoise were corrected for differences in residence time of otoliths from different species and size of prey and the resulting consumption showed a larger contribution from sandeel and herring while whiting was less important than previously estimated.

1. Model description

The SMS model (Lewy and Vinther, 2004) is a stock assessment model including biological interaction estimated from a parameterised size dependent food selection function. The model is formulated and fitted to observations of total catches, survey CPUE and stomach contents for the main stocks in the North Sea. Parameters are estimated by maximum likelihood and the variance/covariance matrix is obtained from the Hessian matrix.

The following predator and prey stocks are available:

- predators and prey (cod, whiting, haddock),
- prey only (herring, sprat, northern and southern sandeel, Norway pout),
- predator only (saithe and mackerel),
- no predator prey interactions (sole and plaice) and
- ‘external predators’ (8 seabird species, starry ray, grey gurnard, North Sea horse-mackerel, western horse-mackerel, hake, grey seals and harbour porpoise).

The population dynamics of all stocks except ‘external predators’ are estimated within the model.

A detailed description of the model can be found in Appendix 1.

2. Input data

The description of input data is divided into four main sections:

Analytical assessment stocks: Stocks for which analytical age-based assessments are done by ICES or can be done from data available from ICES. Data input are similar to those applied by ICES “single species” assessments used for TAC advice, with some additional data.

External predator stocks: Stocks for which stock numbers are assumed known and given as input to SMS.

Diet and ration data: Diet data and food ration data for all predators (analytical stocks and external predators) derived from observed stomach contents data

Additional data: Miscellaneous data

2.1 Analytical assessment stocks

This group of stocks includes:

Cod
Haddock
Whiting
Saithe
Mackerel,
Herring
Northern sandeel
Southern sandeel
Sprat
Norway pout
Plaice
Sole

“Single species” input data, by default given by quarterly time steps, include

Catch at age in numbers (file canum.in)
Proportion of the catch at age landed (file proportion_landed.in)
Mean weight at age in the catch (file weca.in)
Mean weight at age in the stock (file west.in)
Proportion mature at age (file propmat.in)
Proportion of M and F before spawning (file proportion_M_and_F_before_spawning.in)
M, single species natural mortality at age (file natmor.in)
Survey catch at age and effort (file fleet_catch.in)

SMS uses quarterly time steps so input catch data should preferably also be given by quarter. Most of the ICES North Sea stock assessments are however done using annual time steps (see table below).

Table 2.1.1. Overview of “dynamic” stocks used in SMS and their basis from ICES single species advice

Species	SMS		ICES Assessment				
	Species	Max age	Stock area	First	Age range	time	catch

	code			year	(data)	step	categories
Cod	COD	10+	North Sea, eastern English Channel, Skagerrak	1963	1–15	year	D+L
Whiting	WHG	8+	North Sea and eastern English Channel	1978	1–15	year	D+I+L
Haddock	HAD	10+	North Sea, West of Scotland, Skagerrak	1972	1–15	year	D+I+L
Saithe	POK	10+	North Sea, Rockall and West of Scotland, Skagerrak and Kattegat	1967	3–10+	year	D+L
Herring	HER	9+	North Sea, Skagerrak and Kattegat, eastern English Channel	1947	0–8+	year	C
Northern sandeel	NSA	4+	Mix of sandeel stocks	1986	0–4+	semester	C
Southern sandeel	SSA	4+	Mix of sandeel stocks	1983	0–4+	semester	C
Sprat	SPR	3+	North Sea	1974	0–3+	quarter	C
Norway pout	NOP	3	North Sea, Skagerrak, and Kattegat	1984	0–3+	quarter	C
Plaice	PLE	10	North Sea, Skagerrak	1957	1–10+	year	D+L
Sole	SOL	10	North Sea	1957	1–15+	year	D+L

Quarterly catch data

Quarterly catch at age number for cod, whiting, haddock, saithe and herring were provided by ICES assessment groups up to 2003. However, such data have not routinely been reported since. Most stocks data before 2013 did not include discards, as those were not considered in the ICES assessment. In addition, stock areas for the ICES assessments have changed for many stocks since 2003. For example, haddock area 6a (West of Scotland) was joined with the previously used stock area North Sea and Skagerrak in 2014. These changes in both stock areas and the addition of discards make it almost impossible to use the older time-series of catches.

Some quarterly catch data, including discards, can be found in the ICES InterCatch database (kindly provided by Henrik Kjems at ICES). InterCatch data include national catch information used to derive the total international catch data for ICES stock advice. For each year, stock and nation (and fleet) a total annual catch weight is provided often divided into landings and discards. In addition national catch at age in numbers and mean weight by the year or quarter can optionally be provided using the same aggregation level as for the total catch weight. InterCatch data including quarterly catch data, but the data series includes only the most recent years

Table 2.1.2. Year range for available InterCatch data (August 2017)

	InterCatch years
Cod	2002–2016
Whiting	2011–2016
Haddock	2010–2016
Saithe	2002–2016

Mackerel	2015–2015
Plaice	2011–2016
Sole	2011–2015

Table 2.1.3. Year range for quarterly data from assessment reports or produced by the stock coordinator (*)

Herring	2005–2016*
Northern sandeel	1982–2016*
Southern sandeel	1982–2016*
Sprat	1974–2016
Norway pout	1982–2016

Unfortunately, the quarterly catches provided did not appear to be updated back in time in response to e.g. benchmark decisions on changes in stock area. Further, discards were not consistently reported in the time period. Hence, the quarterly catch data could not be used for whiting, haddock, saithe, mackerel, plaice and sole. **Annual catch data as provided for the ICES single species assessment are therefore used for cod, whiting, haddock, saithe, mackerel, plaice and sole. Data by quarter were available from assessments or stock coordinators for herring, sandeel stocks, sprat and Norway pout (Table 2.1.3).**

For stocks with annual catch data it is assumed that annual F is distributed equally over the year, that is $F_{t,q} = F_{t,A2,q}$ in the F model is set to the same value for all quarters (see Appendix 1, equation 3 for details).

For some stocks annual catch data are divided in landings and discards, and in some cases industrial by-catch (Table 2.1.1). The proportion of the catch at age landed as use in SMS is derived by year and age from landings (landings and industrial by-catch) and discards number at age. This proportion is assumed the same for all quarters.

Cod

Catch data

Annual catch data (catch at age in number and mean weight at age, for landings and discards and combined) are available from the ICES assessment working group for the North Sea stocks (see ICES WGNSSK, 2017). For cod, annual scaling factors of

observed catches, 1993–2005, are estimated by the ICES SAM assessments. The input catch numbers are raised by this factor before used in SMS.

Survey data

Survey data are copied from the single species assessment (see table below where α and β is the timing of the survey, given as proportion of the year)

	Name	Years	Ages	α and β	Source
1	IBTS Q1, Gam	1983–2017	1–5	0–0.25	WGNSSK 2017
2	IBTS Q3, Gam	1992–2016	1–4	0.5–0.75	WGNSSK 2017

Biological data

Proportion mature and single species natural mortality (M) data are copied from the assessment.

The single species assessment assumes that mean weight at age in the stock is equal to mean weight at age in the catch. This gives bias (overestimate) of the mean weight of the youngest age classes, as the larger individuals within an age class are more likely to be retained in the fishing gear.

In SMS it is assumed that the mean weight at age for age 2 and younger is constant over the years. Data from the old North Sea MSVPA (ICES CM 1997/Assess:16) are used for these younger ages. MSVPA data give weight by age and quarter, but the weights do not change between years. For age 3 and older, the ratio between weight per quarter (and age) as specified in MSVPA data is maintained but raised to the annual mean weight used in single species assessment. Raising is done from the simple mean of quarterly mean weights and the annual single species mean weight in the particular year. The mean weight for quarter 1 will thereby be lower than the single species stock weights, which lead to a smaller SSB (quarter 1) in SMS, compared to the single species SSB. This was changed from previous practice in 2017 to ensure that a consistent method was used in all years. Figure 2.1.1 compares the two sets of mean weights.

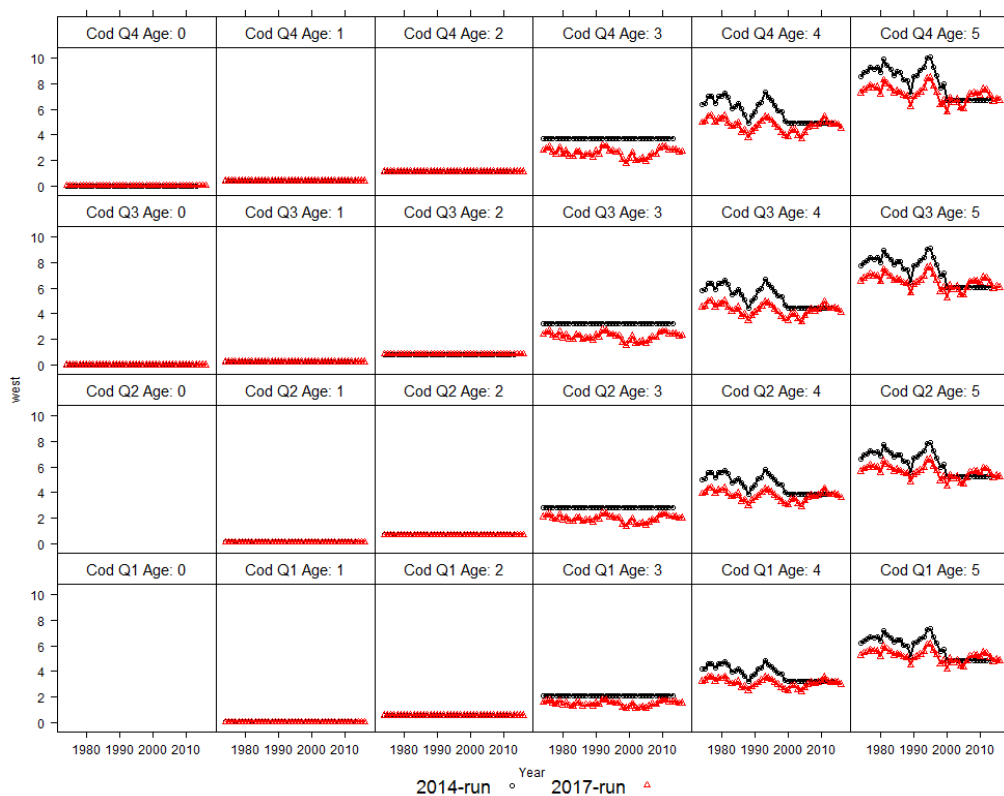


Figure 2.1.1. Mean weight at age in the sea of cod by quarter as used in the 204- and 2017 key run.

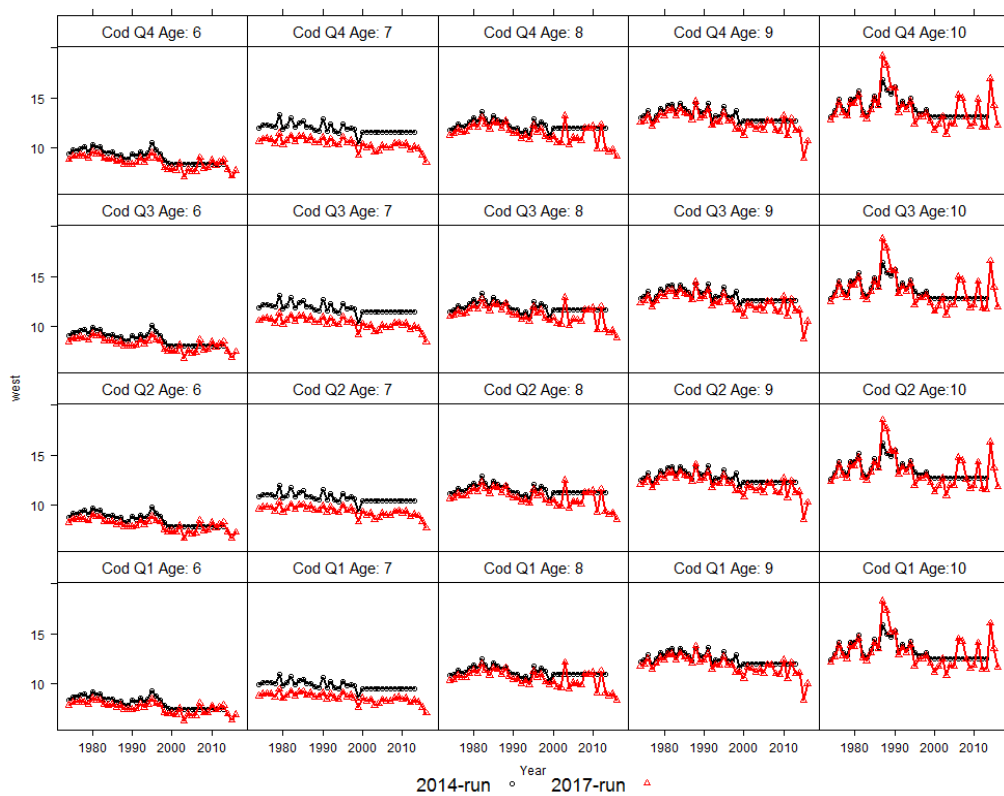


Figure 2.1.1 (continued) Mean weight at age in the sea of cod by quarter as used in the 2014- and 2017 key run.

Stock distribution

The ICES “North Sea cod” includes the stock areas, North Sea, Skagerrak and the eastern Channel (see Table 2.1.1). SMS calculates predation mortalities for the fish within the North Sea, so data on the proportion of the fish stock within the North Sea is needed, ideally by year, quarter and age.

The NS-IBTS covers the North Sea, Kattegat, Skagerrak and the English Channel (just Quarter 1 since 2007), **and provides data to assess distribution of cod, whiting and Norway pout** but less relevant data for haddock and saithe where IBTS only partly covers the stock area. Herring is not included because IBTS data do not separate between the North Sea and the Western Baltic stocks, which both are found in high proportions in the Kattegat and Skagerrak. The plaice population is not divided between areas as plaice is not a predator or prey in the SMS model, such that a population split does not affect the other species.

The distribution of the cod and whiting stocks were determined from the IBTS quarter 1 and quarter 3 survey data. Average CPUE by species, year, quarter, age and ICES rectangle and were downloaded from ICES DATRAS database (data type “CPUE per age per subarea”, survey NS-IBTS, quarter 1 and 3).

The proportion of the stock within the North Sea area was calculated from

1. Mean CPUE within each ICES roundish area, year and quarter is calculated as a simple mean of the “CPUE per age per subarea” (subarea=ICS rectangle)
2. An index for stock abundance per area (North Sea, Skagerrak, Kattegat and English Channel) is calculate as the sum of average roundfish area cpue, weighted by the area (km²) of the roundfish areas.
3. The proportion of the stock within the North Sea is finally calculated by year and quarter from the index per area.

The smoothed value and potential significant trend the proportions [0;1] within the North Sea was subsequently analysed by a gam model (beta distributed data on (0,1) with logit link function) with the proportion as a function of (spline smooth) of year.

Results for cod

The observed proportion of the stock within and outside the North Sea is shown for Quarter 1 (Figure 2.1.3) and quarter 3 (**Figure 2.1.4**) and **Figure 2.1.5** show the observed proportion within the North Sea (excluding the English Channel data, as those exist only for the last 10 years) and the fitted proportion assuming a smooth temporal change. There is a highly significant trend for age 1 and age 2 in quarter 1. In quarter 3, the trend for age 3 is statistical significant, but the temporal change in proportion is limited. Even though it is not statistical significant, the trend for age 1 and age 2 in quarter 3 follows the general trend for the same age groups in quarter 1 (**Figure 2.1.6**)

The proportion of cod stock within the Eastern Channel based on survey data cannot be determined for a longer time-series. Available data suggest a proportion below 5%. The commercial catch of cod is mainly determined by the individual TACs for three areas North Sea, Skagerrak and the English Channel (east and western combined), however catch data reported to ICES (WGNSSK, 2017) show that 4% of the cod stock catch has been taken from the Eastern Channel for the years 2007–2016. This proportion, if it is representing the stock distribution, is small and therefore ignored for SMS purposes.

For Quarter 1, the fitted survey proportions for age 1 to 5+ are used to exclude cod in the Skagerrak/Kattegat from the SMS consumption model. For quarter 3, only data back to 1991 are available. The difference between the fitted proportions by quarter for age 1 and older is quite small (Figure 2.1.6), and therefore the Quarter 1 proportions are assumed to apply also to quarter 3. For age 0 in quarter 3, the observations are highly variable and it is therefore assumed that the proportion of age 0 in quarter 3 follows the proportion of age 1 in quarter 1. These methods result in the proportion of the stock within the North Sea presented in Table 2.1.4. The proportions are assumed to be the same for all quarters.

Table 2.1.4. Proportion of the cod stock within the North Sea (ICES subarea 4) by year and age as used in SMS.

	Age				
	0&1	2	3	4	5+
1974	0.94	0.91	0.89	0.93	0.96
1975	0.93	0.91	0.89	0.93	0.96
1976	0.92	0.90	0.88	0.93	0.96
1977	0.91	0.90	0.88	0.93	0.95
1978	0.91	0.90	0.88	0.93	0.95
1979	0.90	0.89	0.88	0.93	0.95
1980	0.89	0.89	0.88	0.93	0.95
1981	0.88	0.89	0.88	0.93	0.95
1982	0.86	0.88	0.88	0.93	0.95
1983	0.85	0.88	0.88	0.93	0.95
1984	0.84	0.88	0.88	0.92	0.95
1985	0.82	0.87	0.88	0.92	0.95
1986	0.81	0.87	0.87	0.92	0.95
1987	0.79	0.86	0.87	0.92	0.95
1988	0.78	0.86	0.87	0.92	0.95
1989	0.76	0.85	0.87	0.92	0.95
1990	0.74	0.85	0.87	0.92	0.95
1991	0.73	0.85	0.87	0.92	0.95
1992	0.71	0.84	0.87	0.91	0.95
1993	0.69	0.84	0.87	0.91	0.95
1994	0.68	0.83	0.87	0.91	0.94
1995	0.66	0.83	0.86	0.91	0.94
1996	0.65	0.82	0.86	0.91	0.94
1997	0.63	0.82	0.86	0.91	0.94
1998	0.62	0.81	0.86	0.91	0.94
1999	0.61	0.80	0.86	0.90	0.94
2000	0.60	0.80	0.86	0.90	0.94
2001	0.59	0.79	0.86	0.90	0.94
2002	0.58	0.79	0.86	0.90	0.94
2003	0.57	0.78	0.85	0.90	0.94
2004	0.57	0.77	0.85	0.90	0.94
2005	0.56	0.77	0.85	0.90	0.94
2006	0.56	0.76	0.85	0.89	0.94
2007	0.55	0.76	0.85	0.89	0.94
2008	0.55	0.75	0.85	0.89	0.94
2009	0.55	0.74	0.85	0.89	0.93
2010	0.55	0.74	0.85	0.89	0.93

2011	0.55	0.73	0.85	0.89	0.93
2012	0.55	0.72	0.84	0.89	0.93
2013	0.55	0.71	0.84	0.88	0.93
2014	0.55	0.71	0.84	0.88	0.93
2015	0.55	0.70	0.84	0.88	0.93
2016	0.55	0.69	0.84	0.88	0.93

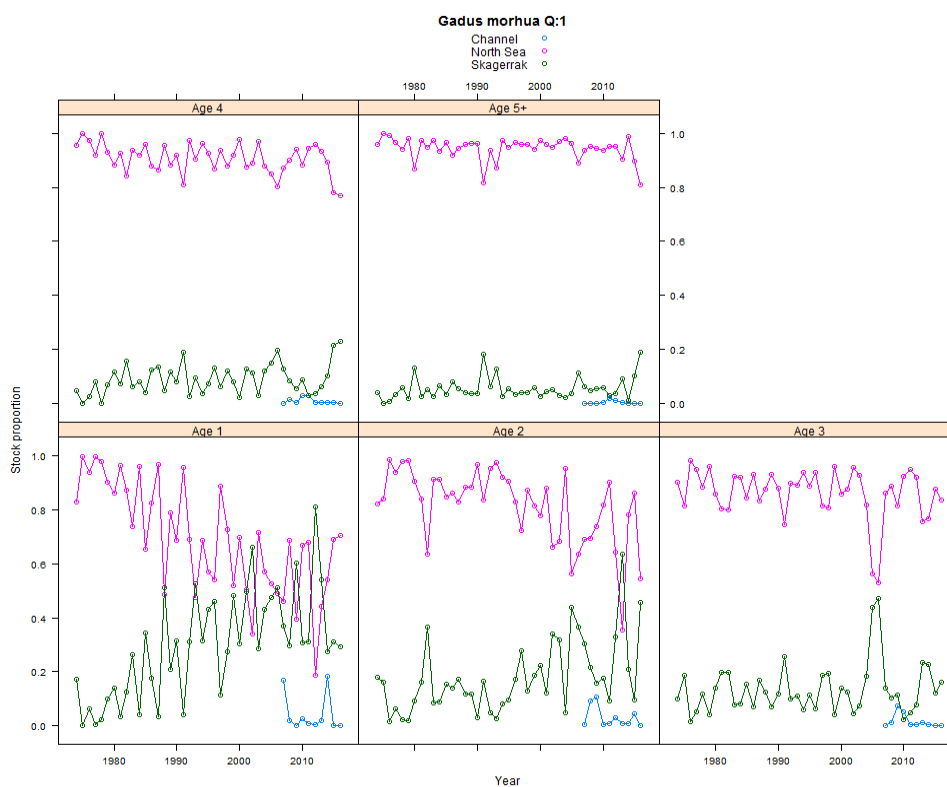


Figure 2.1.2. Stock distribution, Cod quarter 1. Please note that data for the English Channel were available since 2007.

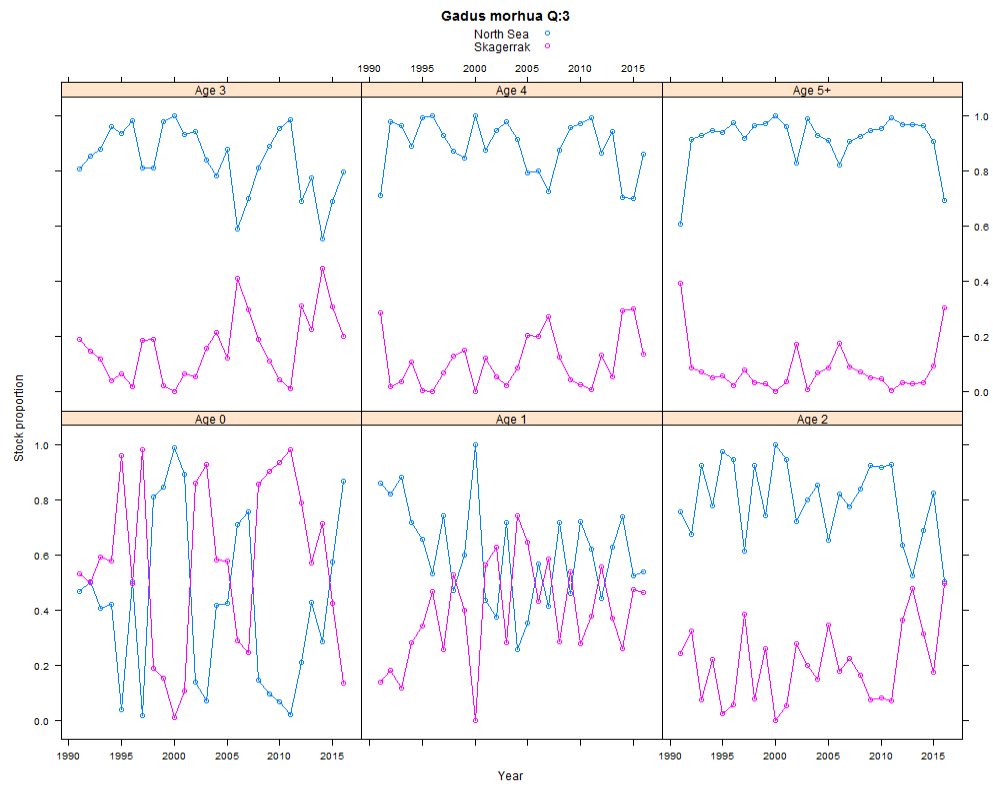


Figure 2.1.3. Stock distribution, Cod quarter 3.

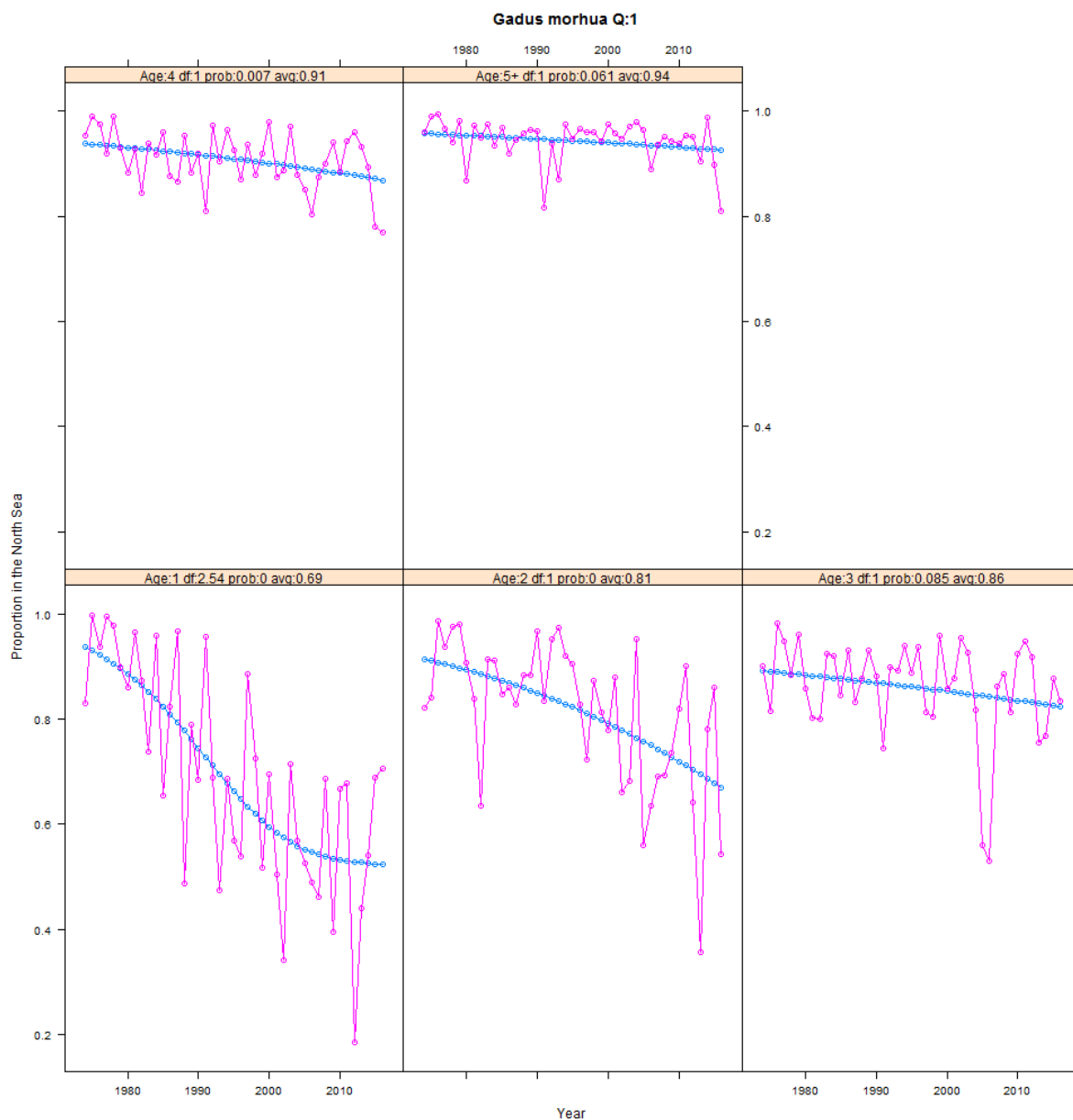


Figure 2.1.4. Observed and fitted proportion of the cod stock (North Sea & Skagerrak data) within the North Sea. For each age the degree of freedom for the fit, the significance of the fit and the average proportion is shown.

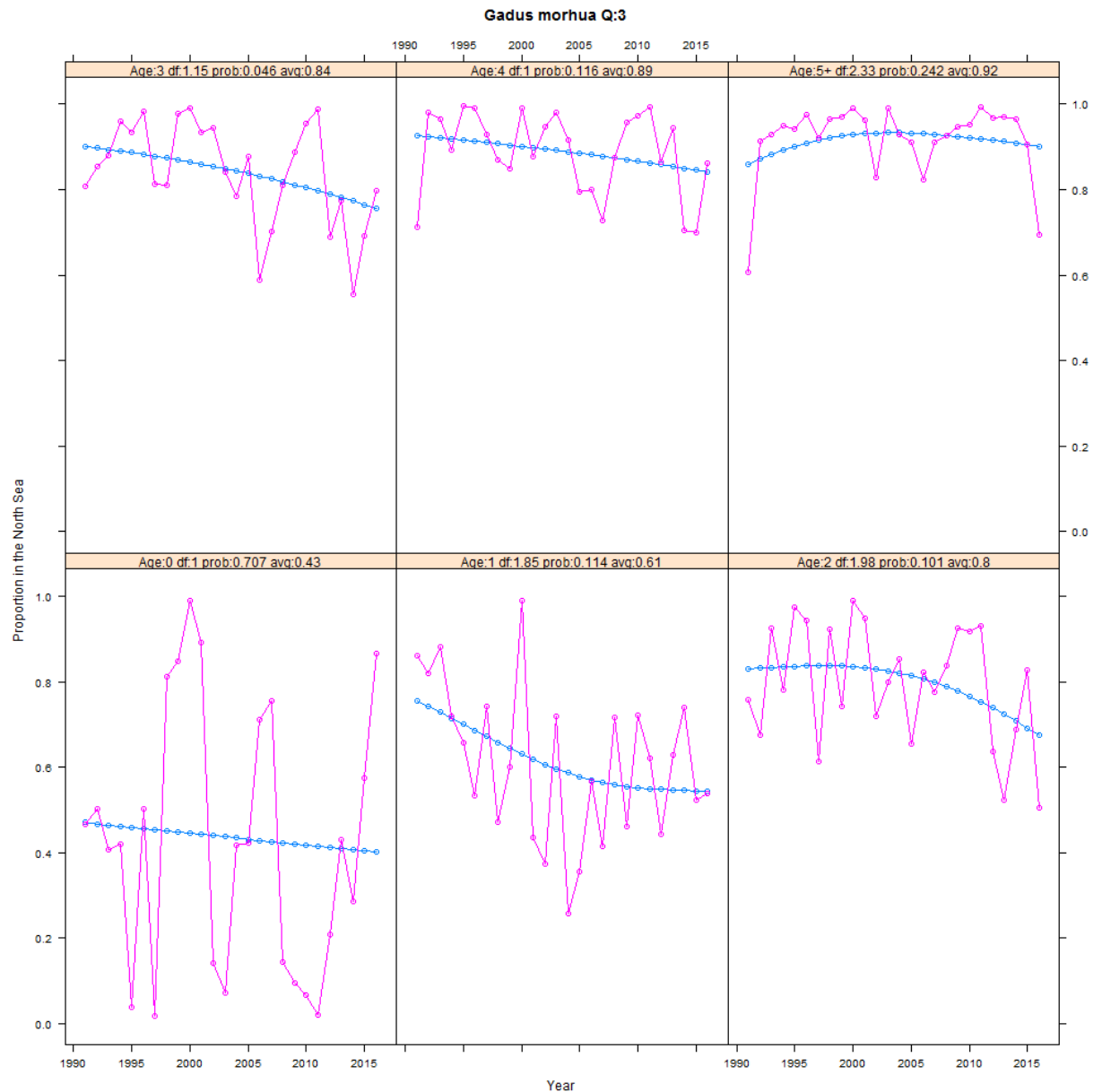


Figure 2.1.5. Observed and fitted proportion of the cod stock (North Sea & Skagerrak data) within the North Sea. For each age the degree of freedom for the fit, the significance of the fit and the average proportion is shown.

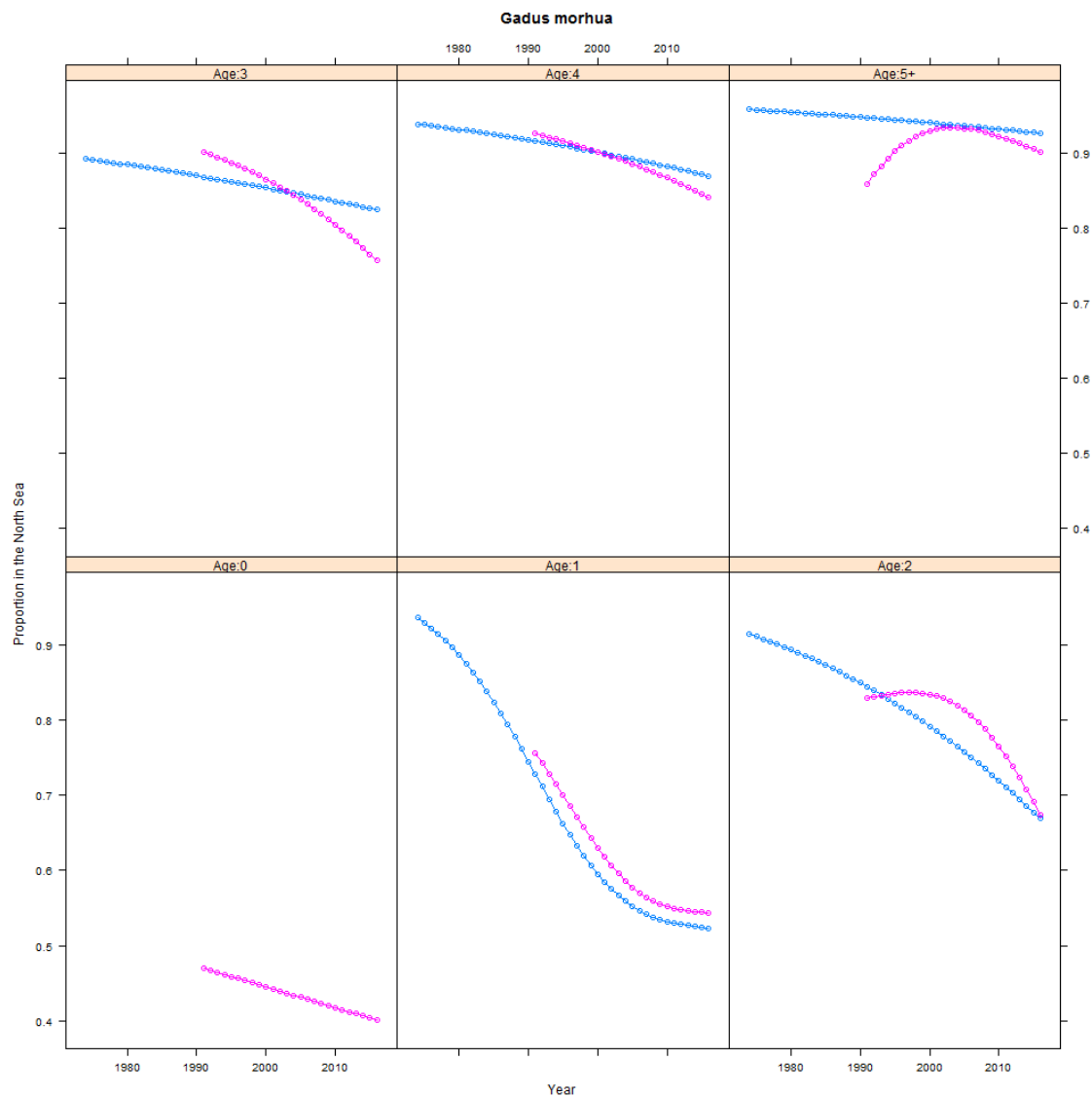


Figure 2.1.6. Fitted proportion of the cod stock (North Sea & Skagerrak data) within the North Sea for quarter 1 (1974–2016) and quarter 3 (1991–2016).

Whiting

Catch data

Annual catch at age data are available from the assessment (WGNSSK, 2017) since 1978. Catch data 1974–1977 from MSVPA (ICES CM 1997/Assess:16) were not updated. It is assumed that the proportion landed for the period 1974–1977 is equal to the average proportion landed 1987–1992.

Survey data

Survey data are copied from the single species assessment.

	Name	Years	Ages	alfa	and	Source
--	------	-------	------	------	-----	--------

				beta	
1	IBTS Q1	1978–2017	1–5	0–0.25	WGNSSK 2017
2	IBTS Q3	1991–2016	0–5	0.5–0.75	WGNSSK 2017

Biological data

Proportion mature and M data are copied from the single species input.

The single species assessment assumes that mean weight at age in the stock is equal to mean weight at age in the catch. Mean weight at age in the stock used in SMS was derived as for cod for ages 0–2. Mean weights at age for ages 3 and older were assumed equal to mean weight in the catch. Applied mean weight at age in the sea can be found in Appendix 2.

Stock distribution

Survey data for the English Channel are only available for Quarter 1 since 2007 (Figure 2.1.7) but show that the proportion within the Channel is variable but low, and decreasing by age. Estimates of commercial catches within each area (WGNSSK 2017) show that the proportion of catches from the North Sea decreases from around 90% in 1995 to around 75% in 2015, but the trend is not statistically significant. Based on the short survey time-series and commercial catch statistics, it is assumed that 90% of the ICES (North Sea & eastern English Channel) whiting stock is situated within the North Sea. This is assumed for all years, quarter and ages in SMS.

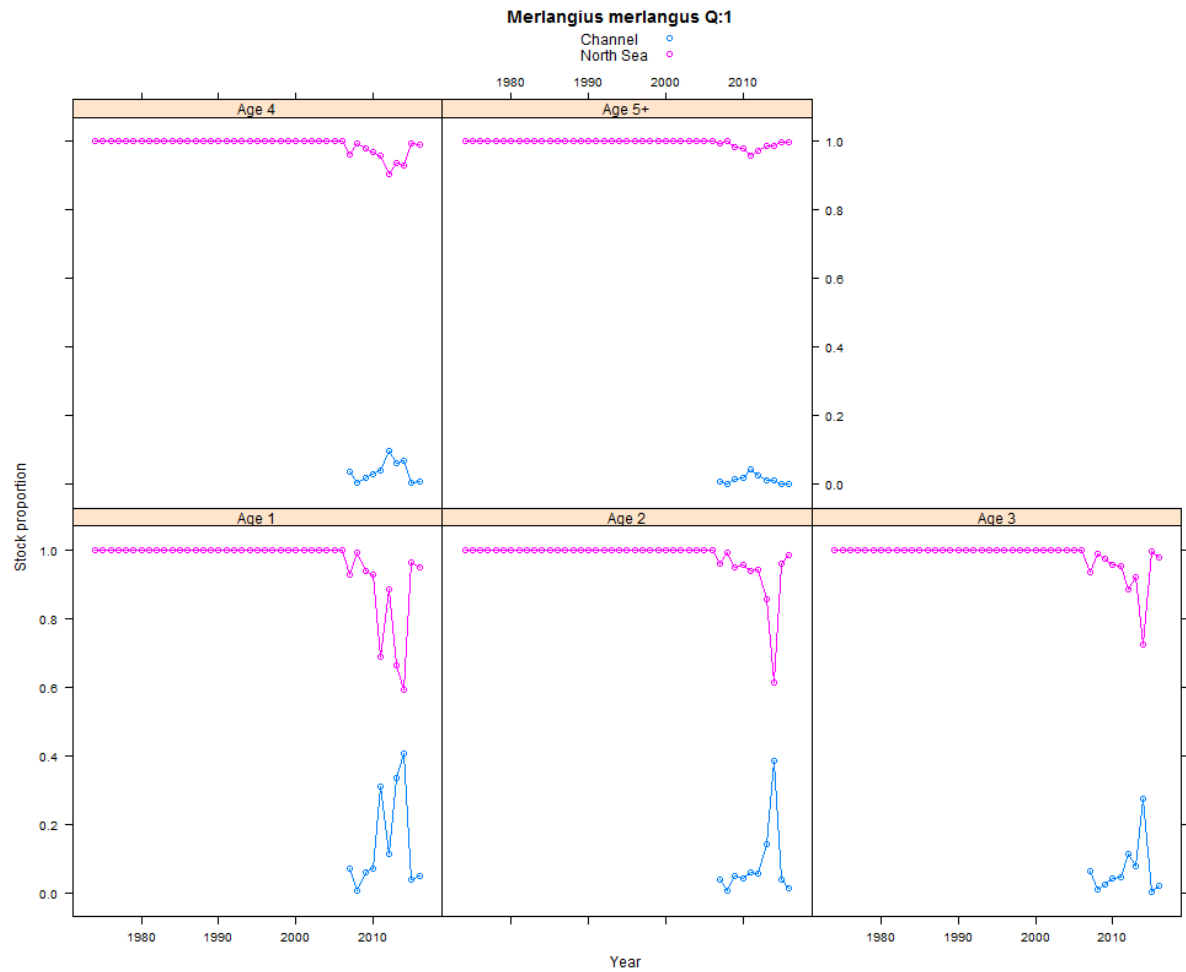


Figure 2.1.7. Stock distribution, Whiting quarter 1. Please note that data for the English Channel were available since 2007.

Haddock

Catch data

Annual catch at age data are available from the assessment (WGNSSK, 2017) since 1965, and were used in SMS.

Survey data

Survey data are copied from the single species assessment (survey 1 and 2).

	Name	Years	Ages	alfa and beta	Source
1	IBTS Q1	1974–2017	1–5	0–0.25	WGNSSK 2017
2	IBTS Q3	1991–2016	0–5	0.5–0.75	WGNSSK 2017

Biological data

Proportion mature data are copied from the single species input (WGNSSK, 2017).

The single species assessment assumes that mean weight at age in the stock is equal to mean weight at age in the catch. Mean weight at age in the stock used in SMS for ages 0–2 was derived as for cod. Mean weights at age for ages 3 and older were assumed equal to mean weight in the catch. Applied mean weight at age in the sea can be found in Appendix 2.

Stock distribution

Survey data for Area 6 are not analysed here. Catch data (WGNSSK 2017) show that 12% of the catches are taken “West of Scotland”. For SMS, it is assumed that 88% of the stock is within the North Sea for all years, quarters and ages. For age 1 and older, a variable but small proportion is found in Skagerrak/Kattegat. This proportion is however ignored in SMS.

Saithe

Catch data

Annual catch at age data are available from the assessment (WGNSSK, 2017) since 1967, and were used in SMS.

Survey data

Survey data (fleet 1) are copied from the single species assessment. With this tuning fleet only, the SMS assessment gives a rather different assessment result compared with the ICES single species assessment. The ICES assessment make use of a combined (commercial CPUE) biomass index, which cannot be used in SMS. To get a more consistent SMS assessment the stock numbers estimated by ICES the single species assessment were used a survey data (fleet 2). Saithe in SMS acts as predator only and the stock dynamic of other SMS species does not affect saithe, which makes it possible to use this approach to get a more consistent (compared to the ICES assessment) result. A CV of 0.3 ($\text{rlnorm}(x, \text{meanlog}=0, \text{sdlog}=0.3)$) was assumed for this artificial index for all ages and years.

	Name	Years	Ages	alfa and beta	Source
1	IBTS Q3	1992–2016	3–8	0.62–0.62	WGNSSK 2017
2	Stock assessment N	1997–2016	3–9	0–0	WGNSSK 2017

Biological data

Proportion mature and M are copied from the single species input (WGNSSK, 2017).

The single species assessment assumes that mean weight at age in the stock is equal to mean weight at age in the catch. Mean weight at age in the stock used in SMS for ages 0–2 was derived as for cod. Mean weights at age for ages 3 and older were assumed equal to mean weight in the catch. Applied mean weight at age in the sea can be found in Appendix 2.

Stock distribution

90.6% of saithe are assumed to be present in the North Sea following the historical distribution of TAC between areas 6 and 4+3.

Mackerel

The ICES assessment of this Northeast Atlantic mackerel is conducted with data from 1980 for age 0–12+ (WGWIDE 2017). Given the wide stock area of the mackerel, mackerel found in the North Sea constitutes a low and variable proportion of the full stock. The inclusion of mackerel as one assessed stock rather than two external predators (western and North Sea mackerel) is new in 2017 key run and follows the decisions made at the mackerel benchmarks, that mackerel in Northeast Atlantic is one stock (with three spawning components: western, southern, and North Sea).

Catch data

Annual catch numbers and mean weight at age in the catch are copied from the ICES assessment (WGWIDE 2017).

For the period before 1980 (1974–1979) estimates of total catch weight are provided by WGWIDE (Table 8.3.1.1, WGWIDE 2016)

Year	Total catch weight (tonnes)
1974	607586
1975	784014
1976	828235
1977	620247
1978	736726
1979	843155

Catch at age and quarter for the period 1974–1979 are derived from single species stock numbers in 1980 (WGWIDE 2017) assuming a similar exploitation pattern as in 1980–1984 estimated by the single species assessment and the total catch weight 1974–1979. Mean weight at age in the catch 1974–1979 was similarly derived from the mean of observed mean weight 1980–1984.

Survey data

The mackerel assessment uses an SSB index (from egg-sampling) and tagging data in addition to two cpue indices. Due to uncertain catch at age data in the first half of the time-series and other issues, the assessment is highly sensitive to the survey data used in the assessment. To get an assessment result which is close to the single species output, estimated stock numbers from the single species assessment are used as CPUE indices in the SMS model. A CV of 0.3 ($\text{rlnorm}(x, \text{meanlog}=0, \text{sdlog}=0.3)$) was assumed for this artificial index for all ages and years. **(after looking at the SMS estimates of uncertainties on mean F and SSB, which is very low, the CV for the artificial should have been set higher!)**

	Name	Years	Ages	alfa and beta	Source
1	Swept area	2010–2017	3–10	0.58–0.75	WGWIDE 2017

2	Stock assessment N	1980–2016	0–9	0–0	WGWISE 2017
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Biological data

Constant quarterly mean weight at age data in the sea are copied from the MSVPA input data (ICES CM 1997/Assess:16) and as basis for all years. The plus group (10+) mean weight is calculated as a simple mean of ages 10–12 in the MSVPA data. Where annual catch mean weight is available (1980–2016) from the assessment (WGWISE, 2017), these were used to scale the year independent MSVPA data in a similar way as for cod (Figure 2.1.8).

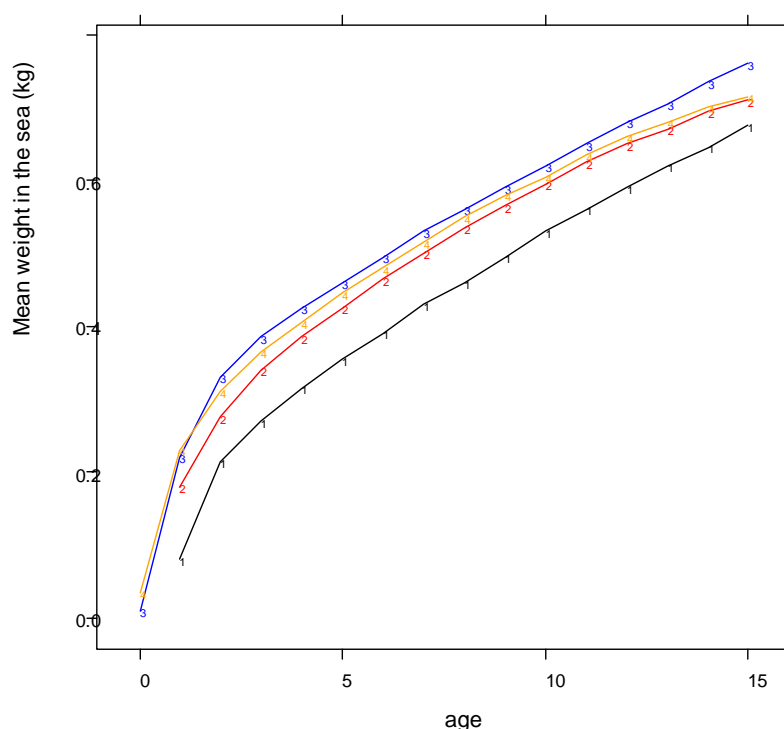


Figure 2.1.8. Mean weight at age in the sea by quarter as used in MSVPA (ICES CM 1997/Assess:16) and used as basis for SMS input.

Proportion mature and natural mortality (M) data are copied from the ICES assessment (1980-) and the 1980 values are copied to 1974–1979.

Stock distribution

Historically, information on the proportion of the mackerel stocks (at that time the western and North Sea stocks) which was inside the North Sea was provided by the relevant assessment working groups (see Table 2.1.5 and Table 2.1.6 below). However, data have not been updated by the assessment working groups since 1997. The proportion of the stock by spawning component (North Sea, Western and Southern) can be estimated from the egg-survey data and an additional assumption on the relative size of the North Sea component, which not has been surveyed at the same time (Table 2.1.7).

WGSAM (2017) reviewed the historical information from catch distribution together with the reported proportions. In later years, the proportion of the catches of the Northeast Atlantic mackerel taken in the North Sea has decreased and the majority of the catches seem to have been taken in areas north of the North Sea (Figure 2.1.8).

Table 2.1.5. Percentage of the west mackerel stock to be present in the North Sea. Data from: Tabel 7.4 ICES CM 1990/Assess:19 for juveniles age group 1 and 2; Table 2 from ICES CM 1989/H:20 for 3+ for the period 1974–85; and Table 12.3 from ICES CM 1997/Assess:3

	Q1			Q2			Q3			Q4		
	Age			Age			Age			Age		
	1	2	>2	1	2	>2	1	2	>2	1	2	>2
year	0	0	0	0	0	0	0	0	30	0	0	5
1974												
1975	0	0	0	0	0	0	0	0	70	0	0	10
1976	0	0	0	0	0	0	0	0	15	0	0	5
1977	0	0	0	0	0	0	0	0	5	0	0	5
1978	0	0	0	0	0	0	0	0	10	0	0	5
1979	0	0	0	0	0	0	0	0	25	0	0	10
1980	0	0	10	0	0	5	0	0	40	0	0	25
1981	0	0	10	0	0	5	0	0	45	0	0	35
1982	0	5	10	5	5	5	10	10	45	10	10	35
1983	0	5	10	10	5	5	10	20	45	10	20	35
1984	0	5	10	15	5	5	25	30	45	25	30	35
1985	0	5	10	20	5	5	30	80	45	30	100	35
1986–1989	0	20	20	40	20	10	60	100	50	60	70	70
1990–1997	0	10	10	20	10	5	30	50	50	30	70	70

Table 2.1.6. Percentage of the North Sea mackerel component to be present in the North Sea. Data from: Fig app 1–2 ICES CM 1985/Assess:7 for period 1974–1984; Fig 9.1 and 9.2 ICES CM 1986/Assess:12 for period 1985; and Table 8.3 ICES CM 1987/Assess:11 for 1986–1997

	Q1			Q2			Q3			Q4		
	Age			Age			Age			Age		
	1	2	>2	1	2	>2	1	2	>2	1	2	>2
year	70	70	30	70	70	90	80	80	80	85	85	55
1974												
1975-1984	70	70	30	70	70	90	80	80	80	85	85	55
1985	95	95	45	95	95	80	80	80	80	90	90	65
1986-1997	100	80	80	100	100	100	100	100	50	100	80	70

Table 2.1.7. WGSAM 2017 estimates of relative contribution from the North Sea, Western and southern components estimated from the egg-survey data (1989, 1992, 1995, 1998, 2001, 2004, 2007, 2010, 2013 and 2016) and assumptions about the relative contributions from the North Sea component. Data for the period before 1989 are copied from Table 2.4.4.2 ICES CM 2005/ACFM:08.

Year	North Sea	Western	Southern
1974	0.221	0.651	0.128
1975	0.205	0.668	0.128
1976	0.201	0.671	0.128
1977	0.177	0.695	0.128
1978	0.136	0.736	0.128
1979	0.125	0.747	0.128
1980	0.116	0.756	0.128
1981	0.081	0.786	0.133
1982	0.080	0.792	0.128
1983	0.074	0.798	0.128
1984	0.037	0.835	0.128
1985	0.037	0.835	0.128
1986	0.037	0.835	0.128
1987	0.037	0.835	0.128
1988	0.037	0.835	0.128
1989	0.037	0.835	0.128
1990	0.037	0.835	0.128
1991	0.037	0.835	0.128
1992	0.037	0.835	0.128
1993	0.037	0.835	0.128
1994	0.037	0.835	0.128
1995	0.029	0.842	0.129

1996	0.029	0.842	0.129
1997	0.029	0.842	0.129
1998	0.029	0.764	0.207
1999	0.029	0.764	0.207
2000	0.029	0.764	0.207
2001	0.029	0.847	0.124
2002	0.029	0.847	0.124
2003	0.029	0.847	0.124
2004	0.029	0.872	0.099
2005	0.029	0.872	0.099
2006	0.029	0.872	0.099
2007	0.029	0.858	0.113
2008	0.029	0.858	0.113
2009	0.029	0.858	0.113
2010	0.029	0.777	0.194
2011	0.029	0.777	0.194
2012	0.029	0.777	0.194
2013	0.029	0.748	0.223
2014	0.029	0.748	0.223
2015	0.029	0.748	0.223
2016	0.038	0.856	0.105

Using the available proportion of the stock by component (Table 2.1.7) and the proportion of each component within the North Sea (Table 2.1.5 and Table 2.1.6), it is possible to calculate the proportion of Northeast Atlantic mackerel within the North Sea (Figure 2.1.9)

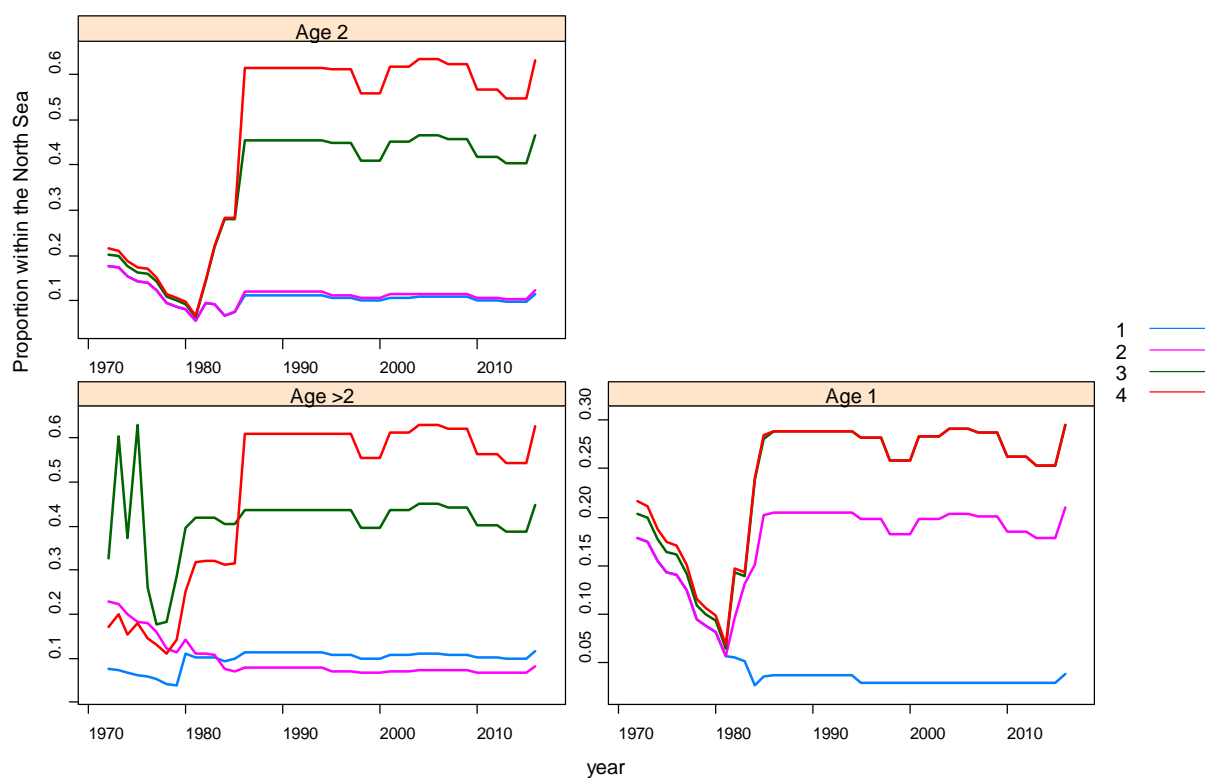


Figure 2.1.9. Preliminary estimate of proportion of the Northeast Atlantic Mackerel stock by age group and quarter (1–4) within the North Sea calculated from stock distributions presented in Table 2.1.4 - Table 2.1.6.

This proportion presented in the figure assumes however that the proportions of the various components have been constant since 1997, which is not the case. The spatial catch distribution show a northerly and easterly expansion of the catch areas (WGWISE, 2017) which also is reflected in the catch proportion from the North Sea (Figure 2.1.10). The contribution of North Sea catches has roughly been halved in the period 2000–2016. Using this trend as an indicator of the proportion of the total stock within the North Sea since 2000, the proportion estimated (Figure 2.1.9) becomes smaller for the period since 2000 (Figure 2.1.11)

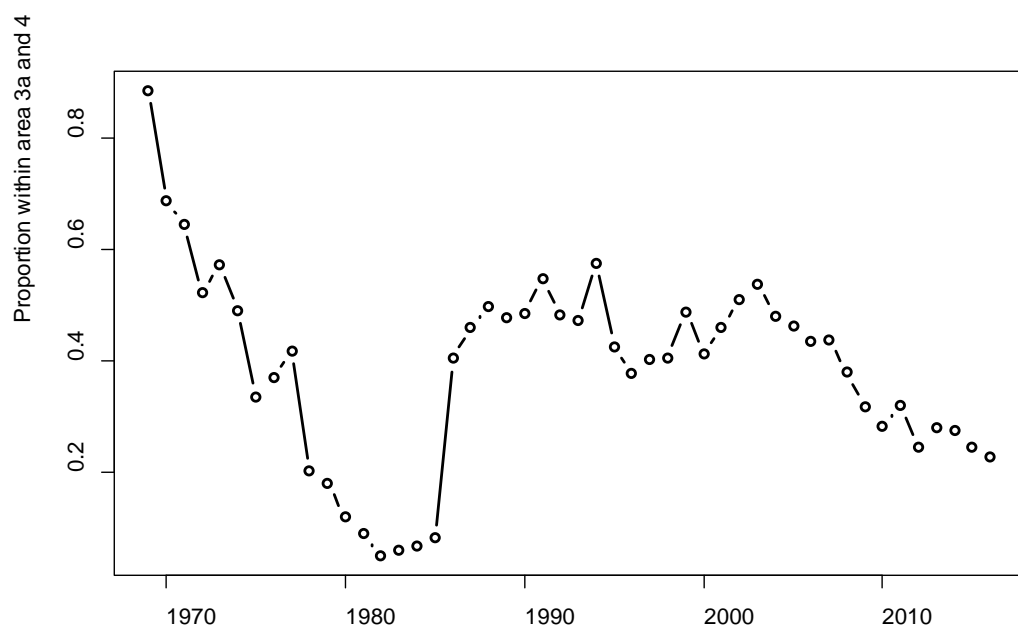


Figure 2.1.10. Proportion of mackerel catches in the North Sea. Data from WGWIDE 2017.

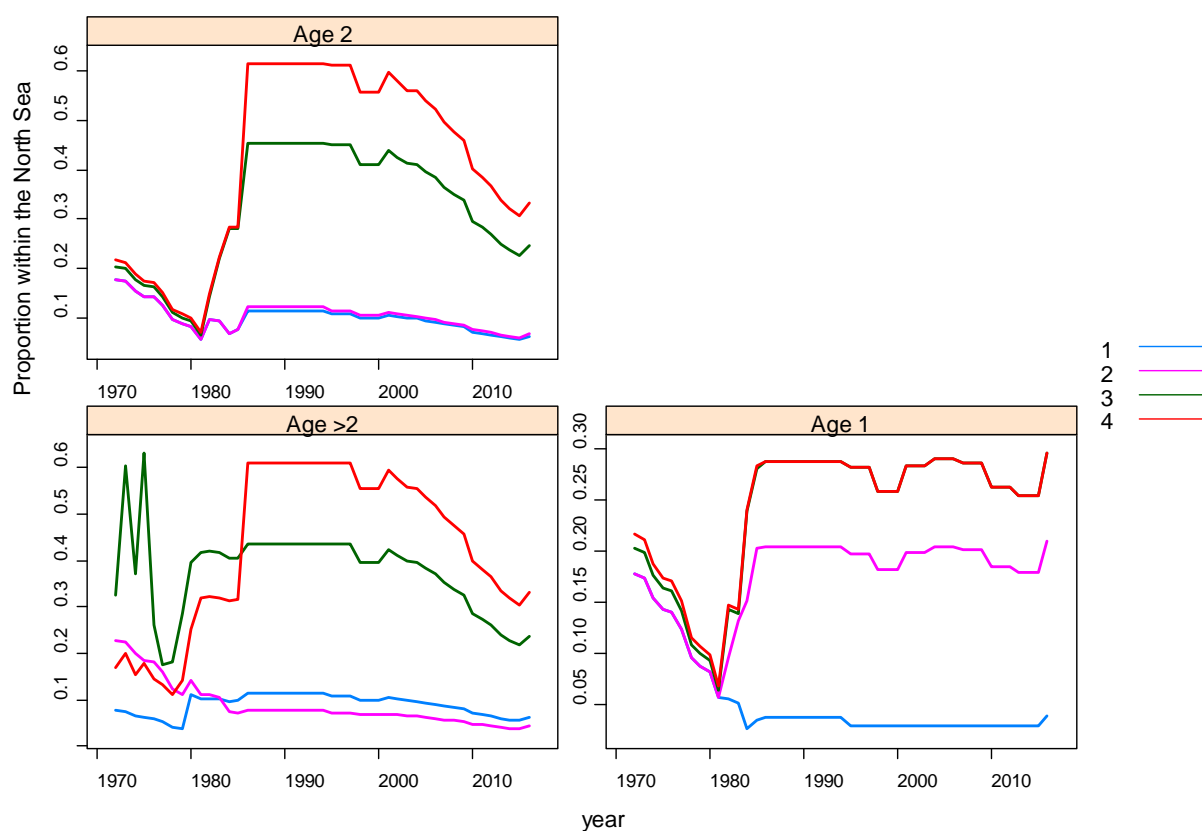


Figure 2.1.11. Estimate of proportion of the Northeast Atlantic Mackerel stock by age group and quarter (1–4) within the North Sea calculated from stock distributions presented in Table 4 - Table 6 and the trend in proportions caught within the North Sea since 2000 (Figure 2.1.10)

WGSAM, 2107 concluded to use the proportion of the stock within the North Sea as presented by Figure 2.1.11. It was recognised that this estimate is based on a series of assumptions, however the estimate seems the best available.

Herring

In 2017, the age range was changed from 0–7+ to 0–9+ to follow the single species configuration.

Catch data

Annual catch exist for the period since 1947 (HAWG, 2017). Quarterly data, 2005–2016 are available from the stock coordinator (Norbert Rohlf) and from the 2007 key run (1974–2004). The existing quarterly data were adjusted such that the sum of quarterly catch numbers summed up to the annual numbers used by HAWG.

Survey data

Survey data are copied from the single species assessment (survey 1–3).

	Name	Years	Ages	alfa and beta	Source
1	HERAS	1989–2016	1–7 (9)	0.54–0.56	HAWG 2017
2	IBTS Q1	1984–2017	1–5	0.08–0.17	HAWG 2017
3	MIK	1992–2017	0–0	0–0	HAWG 2017

Sandeel

The ICES sandeel assessments (2017) for the North Sea area include 6 individually assessed stocks. Ideally SMS should follow the same division to provide relevant natural mortalities for sandeel in the different stocks. However, using all stocks separately would give problems with limited catch at age and diet data availability for some of the stocks. Instead, sandeel in SMS are divided using the previously used Northern and Southern sandeel areas (Figure 2.1.12).

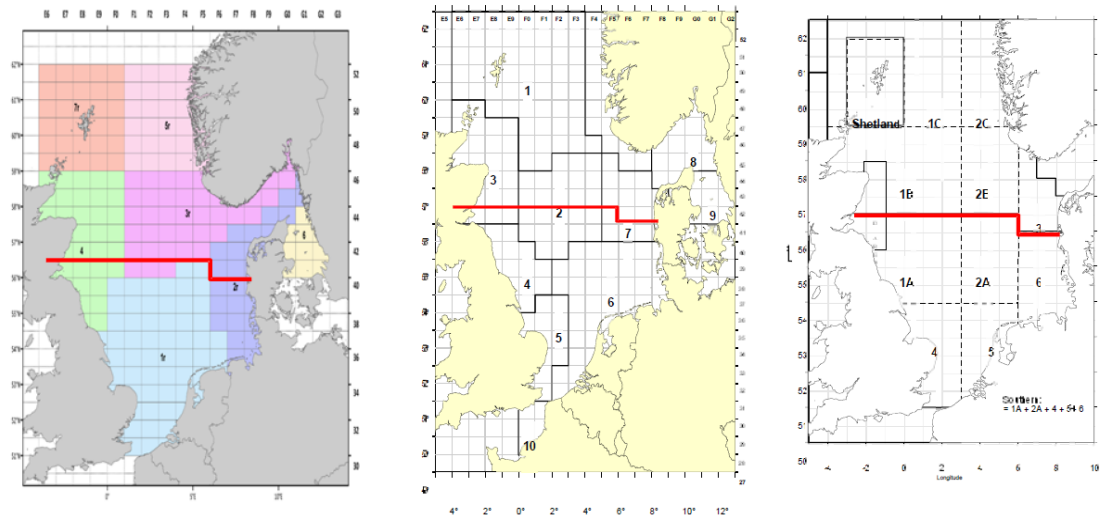


Figure 2.1.12. Sandeel stock and data compilation areas: The left plot shows the stock areas as applied by ICES in 2017. The red line shows the division between the previously used “Northern” and “Southern” sandeel areas. The plot in the middle shows the ICES roundfish areas, which are used as strata in the compilation of stomach content data. The right plot shows the northern and southern areas with sampling areas.

Catch data since 1983 are available by ICES rectangle (HAWG 2017, Anna Rindorf pers. comm.) and were aggregated into the two stocks. Data 1974–1982 are available from the 1999 ICES assessment, where assessment data are aggregated into a Northern and Southern stock. In the estimation of sandeel as prey it is assumed that sandeel found in stomachs from fish sampled in roundfish area 1, 2, 3 and 7 are northern sandeel and southern sandeel are from roundfish area 4, 5 and 6. This split aligns fairly well with the two stock areas (Figure 2.1.12).

Estimating mean weight in the stock is a special concern for sandeel, as weight of 1-year olds and older fish in the catch in the months from July onwards is likely to be biased towards lower mean weights due to differences in the onset of burying of large and small sandeel (Pedersen *et al.* 1999, Rindorf *et al.* 2016). Moreover, weight in the catch of 0-group is highly variable as the 0-group fishery only occurs in part of the time-series and the exact timing of it varies. The stock mean weight of sandeel age 1+ in quarter 2 and 3 were estimated from the long-term (1982–2016) mean catch weight in the first and second half year, respectively. Quarter 1 mean weight was estimated as 79% of that in quarter 2 to reflect the recorded difference in condition between the two quarters (Rindorf *et al.* 2016). Quarter 4 mean weight was estimated as 89% of that in quarter 3, accounting for half the condition loss between quarter 3 and quarter 1 (Rindorf *et al.* 2016). The mean weight of 0-groups in quarter 4 was estimated as the long term average weight of 0-group in the catch the second half year. The 0-group in quarter 3 is assumed to be the half of the mean weight in quarter 4. This procedure was used as the mean weight of 0-groups in catches in quarter 3 was substantially higher than that observed in the stomachs, indicating that the fisheries selection may exclude smaller individuals.

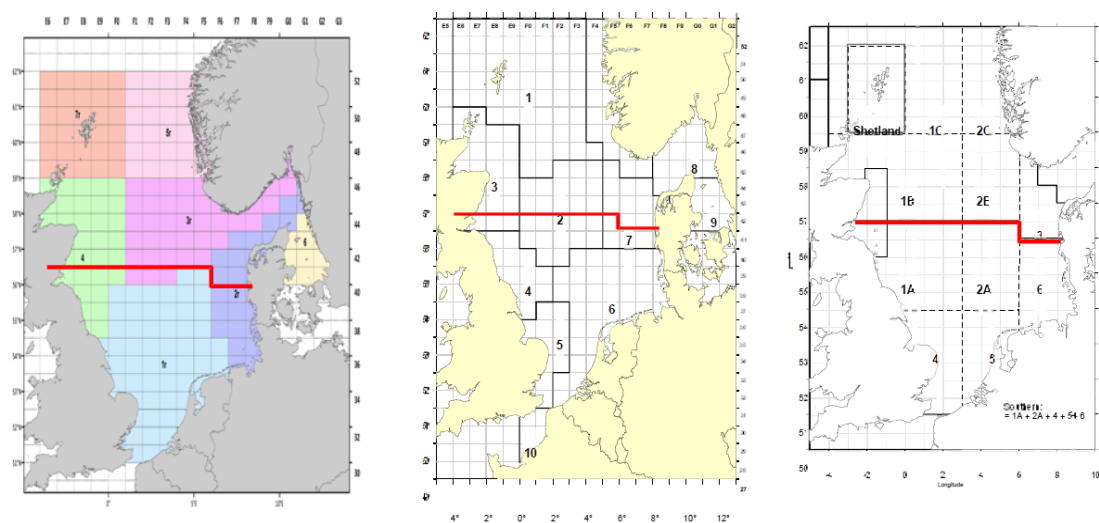


Figure 2.1.13. Northern and southern sandeel areas in relation to (2017) assessment areas (left), roundfish areas (middle) and historical industrial sampling areas (right).

Survey data

Survey data are derived from data used in the single species assessments in areas 1–3 using the same model but deriving sandeel surveys indices for the northern and southern North Sea. In addition to this, three commercial time-series were used to parallel the use of effort tuning of F in the sandeel assessment. These commercial CPUE time-series replace the effort time-series used by the ICES single species effort. A separate time-series was used for the north and south sandeel, but fleet 4 was only used for northern sandeel, as there has not historically been a substantial fishery in the southern North Sea in the second half year.

	Name	Years	Ages	alfa and beta	Source
1	Dredge survey	2004–2016	0–1	0.75–1	HAWG 2017
2	Commercial 1 half year	1982–1989	1–3	0.25–0.5	HAWG 2017
3	Commercial 1 half year	1999–2016	1–3	0.25–0.5	HAWG 2017
4	Commercial 2 half year	1976–2004	1–3	0.25–0.5	Sandeel assessment 2005

Sprat

The single species sprat assessment (HAWG, 2017) uses a single species version of SMS with quarterly time steps, which gives data similar to the data used in the multi-species SMS. The single species assessment uses however a life cycle year from July to June, which is different than the calendar year used in SMS multispecies. To correct for that, year, quarter and age in single species data are transformed to multispecies data by the following rule:

If singles species quarter is Q1 or Q2 then multispecies Quarter=single species Q + 2
 If singles species quarter is Q3 or Q4 then {
 multispecies Quarter=single species Q – 2
 multispecies Year=single species Year + 1
 multispecies Age=single species Age + 1
 }

Catch data

Quarterly catch data are copied from the single species assessment (HAWG, 2017), using the above mentioned data transformation of year, quarter and ages.

Survey data

Survey data are copied from the single species assessment (survey 1–3).

	Name	Years	Ages	alfa and beta	Source
1	IBTS Q1	1975–2017	1–3+	0.0–0.0	HAWG 2017
2	HERAS	2001–2016	1–3+	0.25- 0.50	HAWG 2017
3	IBTS Q3	1991–2016	1–3+	0.5–0.75	HAWG 2017

Biological data

Proportion mature, stock mean weight and M data are copied from single species data. Applied mean weight at age in the sea can be found in Appendix 2.

Norway pout

The single species sprat assessment (WGNSSK, 2017) uses quarterly data for the period since 1974. To accommodate mortality due to spawning stress, the oldest age group (age 3) in the SMS model run is not a plus group (i.e. all Norway pout die when turning 4 years).

Catch data

Quarterly catch data are copied from the single species assessment.

Survey data

Survey data are copied from the single species assessment (survey 1–3).

	Name	Years	Ages	alfa and beta	Source
1	EGFS	1982–1991	0–3	0.5- 0.75	WGNSSK 2017
2	EGFS	1992–2017	0–2	0.5- 0.75	WGNSSK 2017
3	IBTS Q1	1974–2017	1–3	0.0–0.25	WGNSSK 2017

Biological data

Proportion mature, stock mean weight and M data are copied from single species data. Applied mean weight at age in the sea can be found in Appendix 2.

Plaice**Catch data**

Annual catch at age data are available from the assessment (WGNSSK, 2017) since 1957, and were used in SMS.

Survey data

Survey data are copied from the single species assessment (survey 1–3) with:

	Name	Years	Ages	alfa and beta	Source
1	BTS-Isis-early	1985–1995	1–8	0.66–0.75	WGNSSK 2017
2	BTS-Combined	1996–2016	1–9	0.66–0.75	WGNSSK 2017
3	SNS1	1974–1999	1–6	0.66–0.75	WGNSSK 2017
4	SNS2	2000–2016	1–6	0.66–0.75	WGNSSK 2017
5	IBTS Q3	1997–2016	1–9	0.63–0.63	WGNSSK 2017
6	IBTS Q1	2007–2016	1–7	0.10–0.10	WGNSSK 2017

Biological data

Proportion mature data are copied from the single species input (WGNSSK, 2017).

The single species assessment assumes that mean weight at age in the stock is equal to mean weight at age in the catch. Mean weight at age in the stock used in SMS for ages 0–2 was derived as for cod. Mean weights at age for ages 3 and older were assumed equal to mean weight in the catch.

Sole**Catch data**

Annual catch at age data are available from the assessment (WGNSSK, 2017) since 1957, and were used in SMS.

Survey data

Survey data are copied from the single species assessment (survey 1–2).

	Name	Years	Ages	alfa and beta	Source
--	------	-------	------	---------------	--------

1	BTS-Isis	1985–2016	1–9	0.66–0.75	WGNSSK 2017
2	SNS	1974–2016	1–6	0.66–0.75	WGNSSK 2017

Biological data

Proportion mature data are copied from the single species input (WGNSSK, 2017).

The single species assessment assumes that mean weight at age in the stock is equal to mean weight at age in the catch. Mean weight at age in the stock used in SMS for ages 0–2 was derived as for cod. Mean weights at age for ages 3 and older were assumed equal to mean weight in the catch.

2.2 External predators

The “external predator” group includes predators for which the stock numbers are given by input. The list of species includes:

- Birds
 - Fulmar
 - Guillemot
 - Herring Gull
 - Kittiwake
 - GBB. Gull
 - Gannet
 - Puffin
 - Razorbill
- Fish
 - Starry ray
 - Grey gurnards
 - Western horse mackerel
 - North Sea horse mackerel
 - Hake
- Mammals
 - Grey seal
 - Harbour porpoise

Time-series of their abundance are given in Figure 2.2.1.

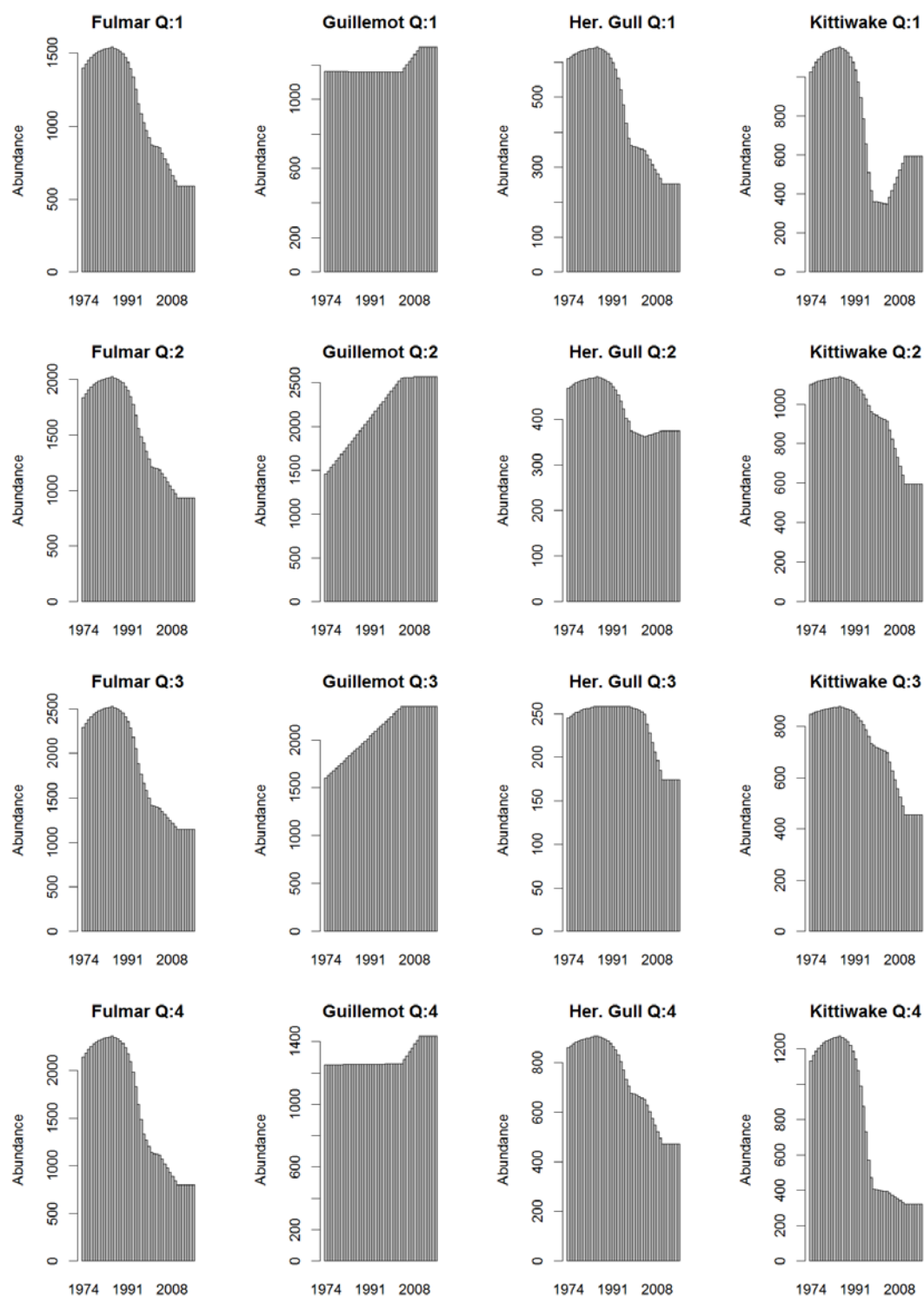


Figure 2.2.14. Estimates as used by SMS of the abundance of “external predators” present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

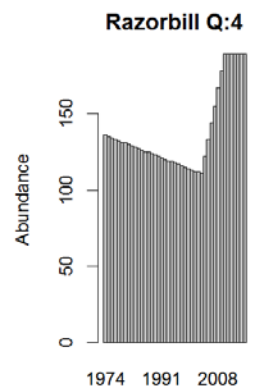
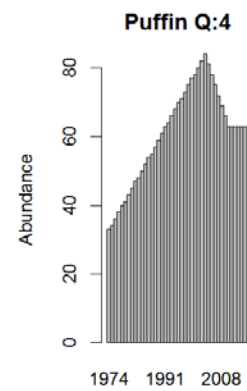
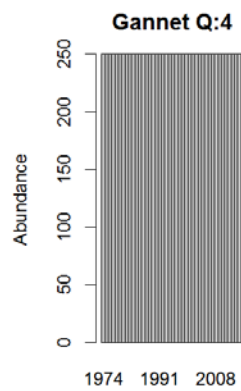
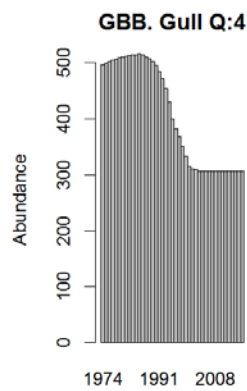
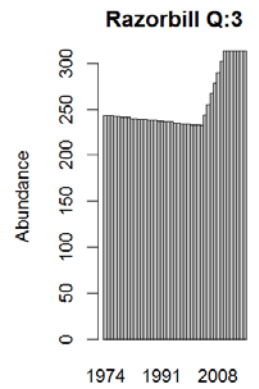
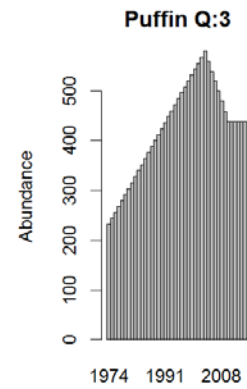
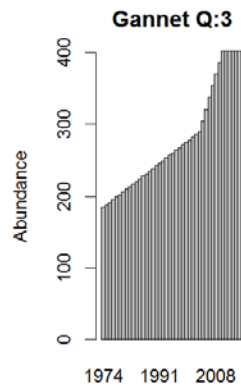
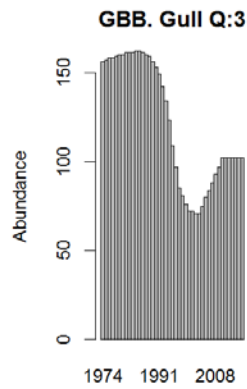
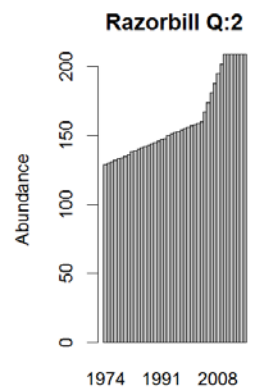
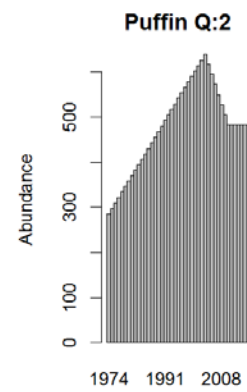
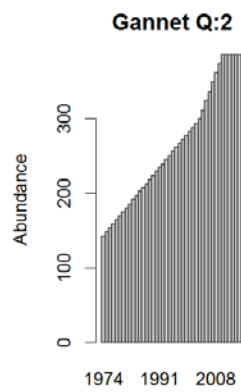
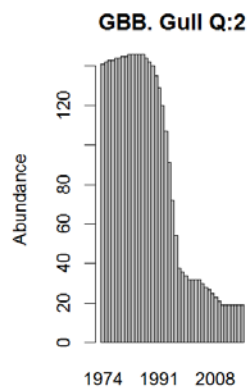
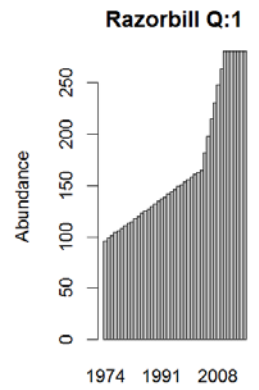
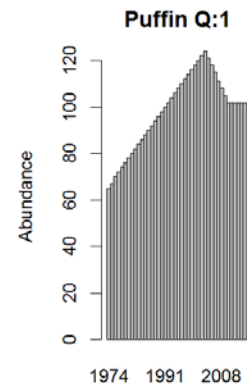
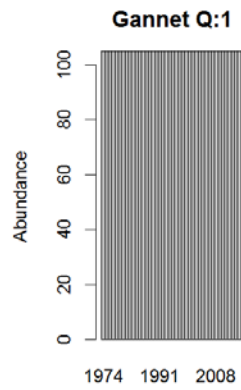
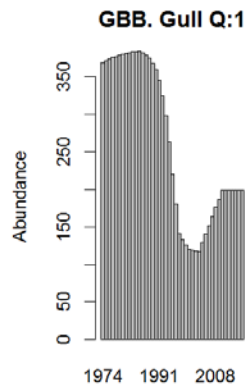


Figure 2.2.1 (Continued) Estimates as used by SMS of the abundance of “external predators” present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

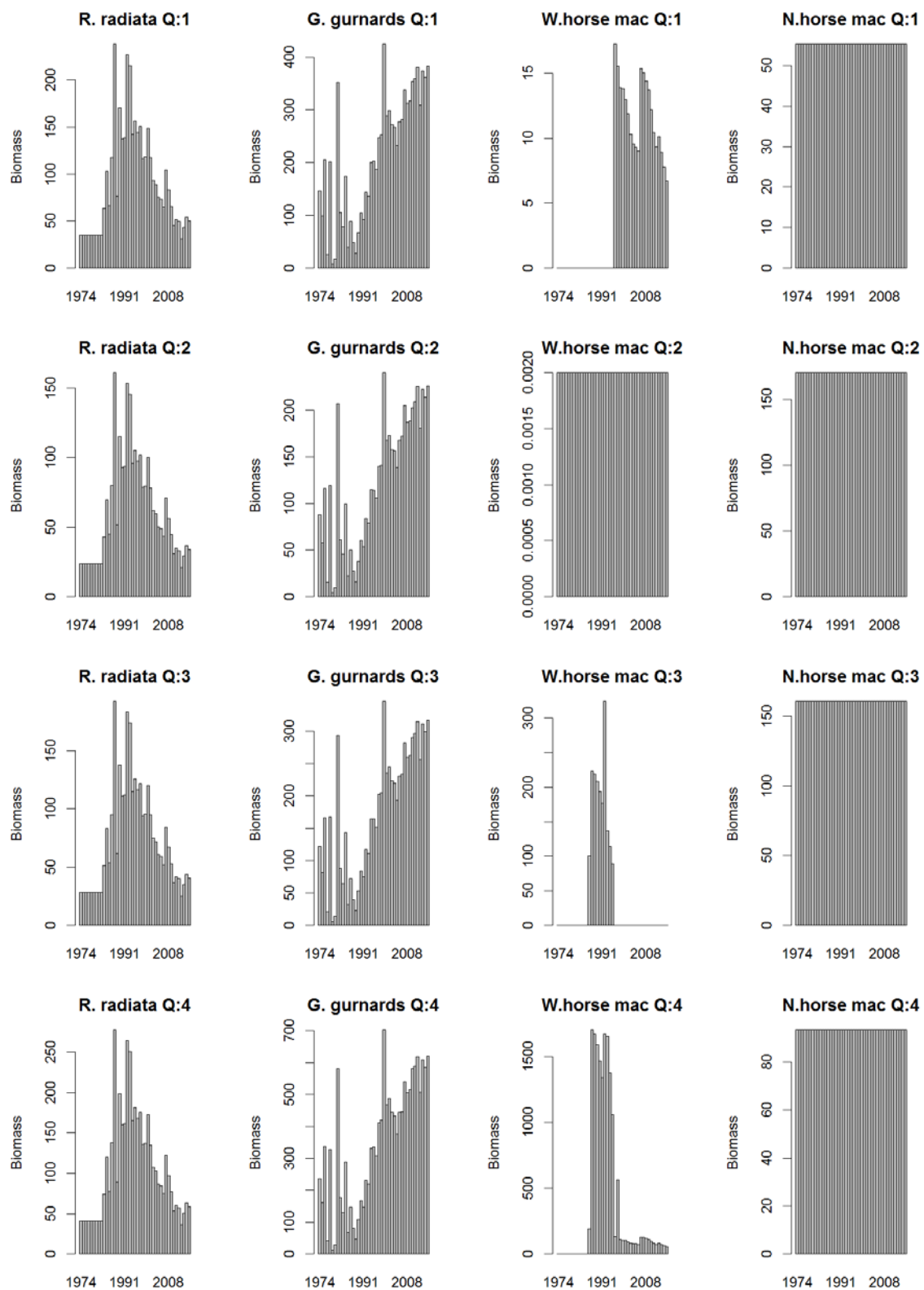


Figure 2.2.1 (Continued) Estimates as used by SMS of the abundance of “external predators” present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

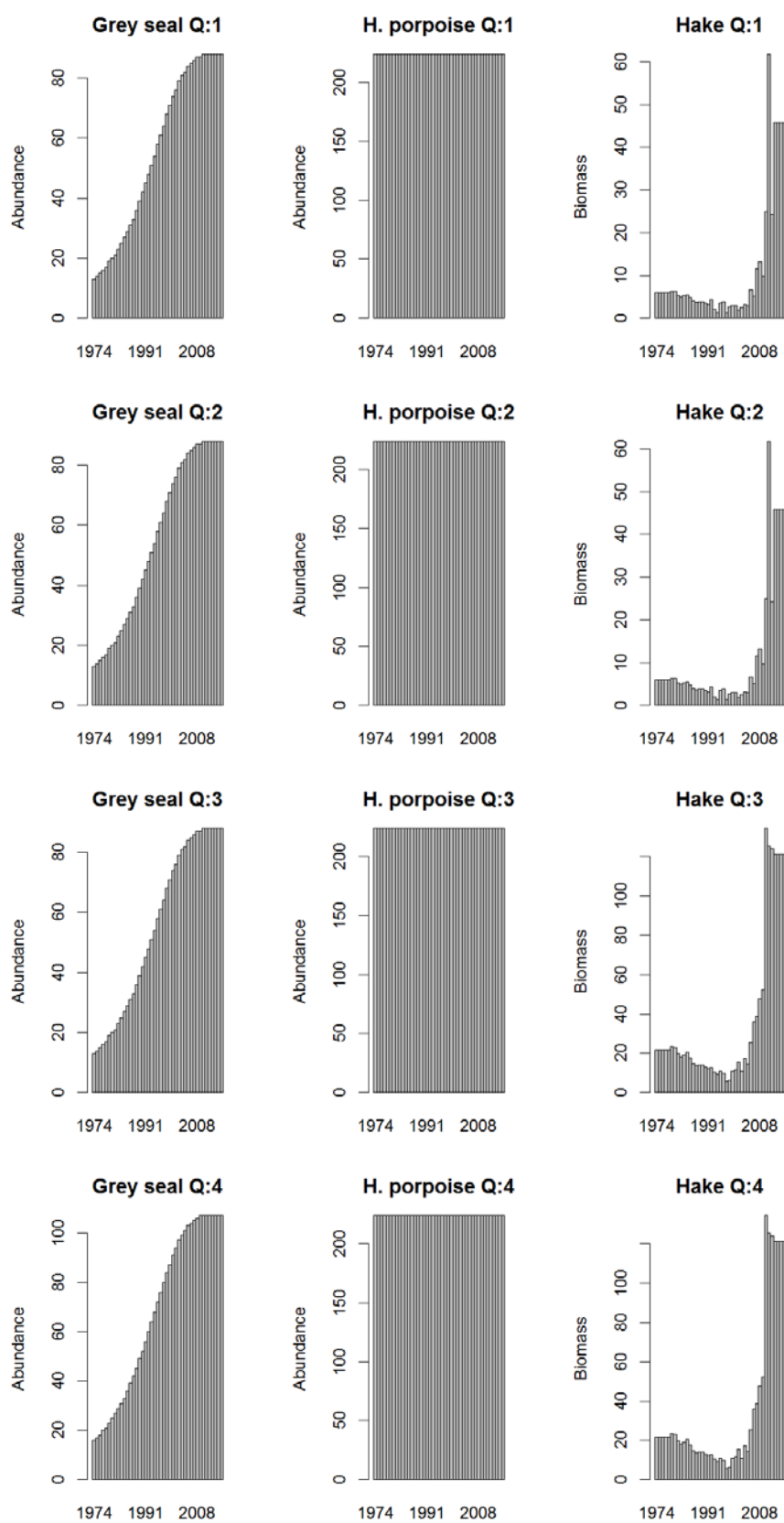


Figure 2.2.1 (Continued) Estimates as used by SMS of the abundance of “external predators” present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

Birds

Numbers of seabirds in the North Sea were calculated using two sources: counts of seabirds at sea and counts of seabirds staying in the colony while breeding or attending nest sites. Seabirds at sea have systematically been recorded in the North Sea since 1979, with a joint database, the European Seabirds at Sea Database (ESAS), existing since 1991. The ESAS database version 4.1 (as of September 2004) contained data from seabirds at sea counts over the period 1979 to 2004. Coverage of the North Sea over years and seasons was unequal. Yearly distance travelled ranged between 4,407 and 301,293 km. As seabirds are partly on land while breeding and also at other times of the year, conversion factors based on breeding population numbers were used to derive population numbers from number recorded at sea. Data from breeding population numbers were taken from published accounts, from national databases and from ICES Working Group on Seabird Ecology reports. Energy requirements for chicks were also estimated and expressed as numbers of adults as these are not covered by the energy budgets for adults. All these numbers derived from land/colonies were then added to the numbers calculated for the sea areas from the ESAS database.

Because of the rather limited temporal coverage of the data, at-sea numbers for each quarter of a year were estimated for two time periods only, 1979–1991 and 1992–2004. Data were calculated separately for six sub regions. The data obtained by this procedure were treated differently afterwards depending on bird species. From known trends in breeding population numbers over the last decades and from trends in small subsets of the North Sea, different models were applied to calculate numbers at sea for all years and quarters from 1963 to 2004. For four species (northern gannet, common guillemot, Atlantic puffin, razorbill), a linear trend was assigned to the population trend as this has more or less been the case for the overall breeding bird numbers (counts of breeding birds are not available on an annual or biannual basis for the whole North Sea). This is certainly a simplification of the real situation but should reflect the overall trends. For the other four species (northern fulmar, herring gull, great black-backed gull and black-legged kittiwake), a logistic model was applied as all four species showed substantial increases from the 1960s to the 1980s/90s and declines afterwards. The derivation of seabird data was updated with more recent years and trends in ICES WGSAM 2011, and has not been updated since. Therefore, populations from 2011 onwards were assumed constant.

Starry rays and grey gurnards

The time-series of grey gurnard and starry ray (*Amblyraja radiata*) are estimated from IBTS CPUE by length, scaling the time-series CPUE index to a “known” average biomass. For **starry ray** an average biomass of 100 kt over the years 1977–1988 is suggested by Sparholt and Vinther (1991). Sparholt (1990) estimated the average biomass of **grey gurnards**, 1983–1985, in the range 48 kt (IYFS Q1 data) to 146 kt (EFGS Q3). Another estimate (Daan *et al.*, 1990) estimated the average biomass of grey gurnards to 205 kt based on EFGS Q3 data 1977–1986, using the method of Sparholt.

The stock number per length class, year and quarter is derived from a generalized linear model (SAS procedure Genmod) of CPUE (number per hour) assuming a Poisson distribution and using a log-link function. CPUE was modelled by individual size classes from the explanatory variables: year, quarter, roundfish area and gear. Data were extracted from ICES DATRAS (data type: CPUE per length per haul) for the period since 1974. Quarter 1 data were used for the whole period; quarter 3 since 1991 and quarter 2 and quarter 4 for the period 1991–1997. Data from the early part of the time-series seem not to have recorded starry ray or gurnards even though it was

noted that all species were recorded. All records from individual cruises (year, quarter and vessel) with no recorded catch of starry ray or gurnards in any haul were excluded from the analysis.

The total average biomass is divided into size classes from the average observed cpue and mean weight in the years 1991–1997 where data exist for all 4 quarters. By using this method it is assumed that catchability is independent of size, which is probably not the case for smaller individuals. The average stock estimate in thousands tonnes by size classes are shown in the table below.

	Species	
	Grey gurnard	Starry ray
Size cm group		
00–10	0.04	-
10–20	22.52	0.39
20–30	124.04	4.11
30–99	58.40	95.50
All	205.00	100.00

The model “year-effects” for starry ray are more uncertain for the period prior to 1981 and these data were finally allocated to one year, “pre-1981”. The year effect for “pre-1981” was used for stock estimate for 1974–1981.

For both species, the published biomass estimates are very uncertain and they are not used directly in SMS. For starry ray it is assumed that the stock has an average biomass of 100 kt over the years 1982–2013. The final year, 2013, was used in the 2014 key-run and this year has been maintained as there are recent trends in the biomass. For grey gurnards and average biomass of 205 kt is assumed for the years 1977–2013, where the year range is chosen mainly for stability reasons.

Horse mackerel

ICES considers horse mackerel (*Trachurus trachurus*) in the northeast Atlantic to be separated into three stocks. The southern stock is found in the Atlantic waters of the Iberian Peninsula, the North Sea stock in the eastern English Channel and North Sea area, and the western stock on the northeast continental shelf of Europe, stretching from the Bay of Biscay in the south to Norway in the north. ICES makes an analytical (absolute) assessment of the western stock, while the North Sea stock is assessed from survey indices and an absolute stock biomass is not estimated.

Previously, ICES has stated that about 7% of the combined western and North Sea mackerel stock resides in the North Sea. WGSAM 2017 decided to assume that the North Sea stock development followed that of the western stock and total North Sea horse mackerel biomass was therefore 7.5% of the biomass of the western mackerel. Lately, an increasing proportion of the North Sea horse mackerel was caught in fisheries in the English Channel in the 4th quarter. However, this change in quarter 4 dis-

tribution does not necessarily reflect changes in quarter 2 and 3 distribution, and as these are the quarters where the main feeding takes place. Therefore, WGSAM considered that North Sea horse mackerel were all present in the North Sea.

The western horse mackerel stock assessment reports have previously reported the proportion of western horse mackerel entering the North Sea in each quarter (Table 2.2.1).

Table 2.2.8. Percentage of the western horse mackerel stock entering the North Sea by quarter.
Sources: Tab 12.3 in ICES CM 2000/ACFM:5 for 1998; tab 12.2 in ICES CM 1999/ACFM:6 for 1997; tab 12.x in ICES CM 1998/Assess:6 for 1996; tab 12.5 in ICES CM 1997/Assess:3 for 1995; tab 12.5 in ICES CM 1996/Assess:7 for 1994; tab 18.5 in ICES CM 1995/Assess:2 for 1993; tab 16.5 in ICES CM 1993/Assess:19 for 1992; tab 13.5 in ICES CM 1992/assess:17 for 1991)

	Age 1–4				age >4			
	Q1	Q2	Q3	Q4	Q1	Q2	Q3	Q4
1974–1985	0	0	0	0	0	0	0	0
1986	0	0	0	0	0	0	5	10
1987	0	0	0	0	0	0	5	40
1988	0	0	0	0	0	0	5	40
1989	0	0	0	0	0	0	5	40
1990	0	0	0	0	0	0	5	40
1991	0	0	0	0	0	0	5	40
1992	0	0	0	0	0	0	10	55
1993	0	0	0	0	0	0	5	65
1994	0	0	0	0	0	0	5	65
1995	0	0	0	0	0	0	5	65
1996	0	0	0	0	1	0	0	10
1997	0	0	0	0	1	0	0	50
1998–2016	0	0	0	0	1	0	0	10

This information has not been available since 1998, but the proportion of western stock horse mackerel caught in the North Sea (all horse mackerel caught in subarea 4a) is still reported (Figure 2.2.2). Based on these data, it was decided to assume that 10% of the western horse mackerel stock was present in the North Sea in quarter 4. In quarters 2 and 3, no western horse mackerel were present in the North Sea. In quarter 1, horse mackerel are not feeding and hence it is not relevant to know their abundance in the North Sea.

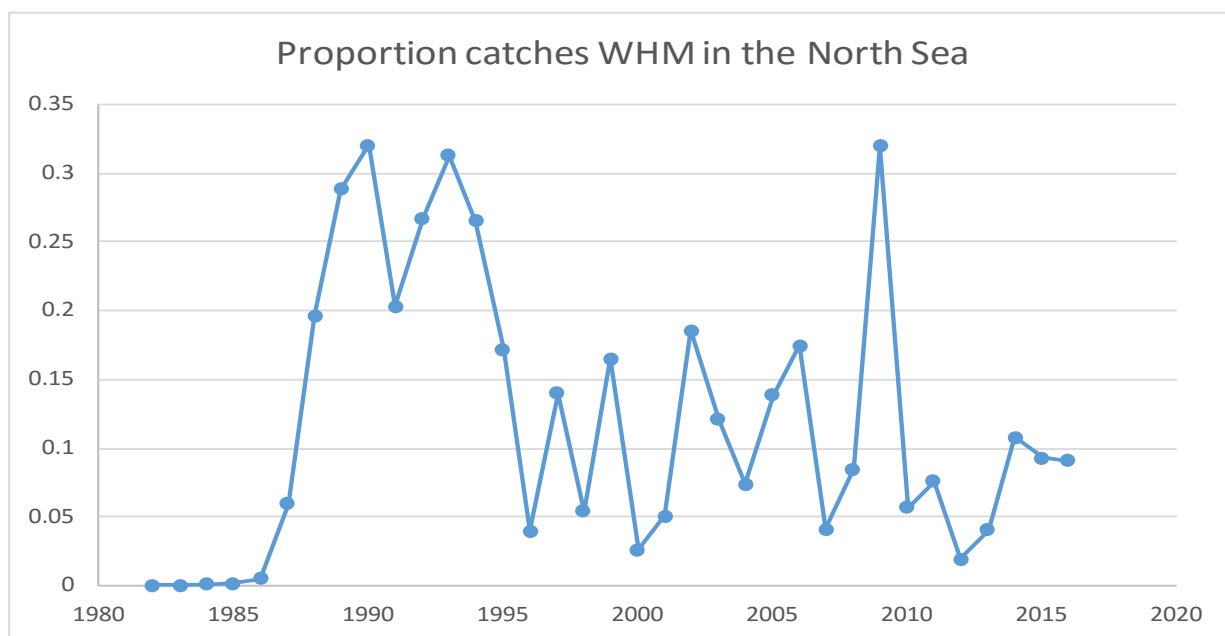


Figure 2.2.15. Proportion of western horse mackerel catches in the North Sea (data from WGWISE 2017)

Hake

Hake was included in the 2014 key run as an “external predator”. Since 2000, considerably more hake has been caught in the IBTS survey and information from the fishing industry points to increasing hake abundance in the North Sea. In 2014, there was only an assessment for northern hake. This assessment included all sea areas from the northern Bay of Biscay up to the Norwegian Sea. Three different surveys (IBTS, SWC-IBTS, EVOHE; all from the second half of the year) were available for the years 1997-2013 from DATRAS to calculate the proportion of the total Northern hake abundance and biomass resident in the North Sea in the second half of the year. When using CPUE per rectangle * number of rectangles in the survey area as index, it was estimated that 10- 15% of the hake stock in numbers were in the North Sea (WGSAM 2014) while the biomass percentage is much larger as the hake found in the North Sea in the second half of the year are larger than average. For the years before 1997, it is assumed that the proportion of the northern hake stock in the North Sea stays constant at the average from the years 1997–2001, i.e. before the CPUEs started to increase in the IBTS.

The overall biomass and abundance present in the North Sea was divided into size categories by using the size distribution observed in Q3 IBTS hake catches. Based on the diet composition it was decided to have three size classes of hake in the model (<250mm; 250 - <600mm; ≥600mm). From the biomass and abundances obtained for the second half of the year, the abundances in the first half of the year were calculated by multiplying the abundances in the second half of the year with the ratio of CPUEs per size class observed between the 1st quarter and 3rd quarter IBTS. While small and medium sized hake are present in both quarters, large hake are caught to a much lesser extent in the first quarter.

For the 2017 key run, the development in IBTS Q1 and Q3 was inspected and as there was no trend in the catch rates since 2013, the stock numbers in 2014–2016 were assumed equal to stock numbers in 2013.

Grey seal

The abundance of grey seals was estimated using a demographic model fitted to pup production estimates, and estimates of adult numbers based on haulout counts in the North Sea and Orkney for the period 1984 to 2009 (Buckland *et al.* 2004; Thomas 2011). Populations prior to 1984 are estimated assuming exponential growth in the period up to 1990 (using 1984–1990 to estimate parameters). For 2010 onwards, the value in 2009 is used as populations are assumed to be levelling off.

Harbour porpoise

The abundance of cetaceans in the North Sea is monitored during aerial and boat-based sightings surveys, with corrections to take account of the detectability of the animals (Hammond *et al.* 2002). Harbour porpoise population size was assumed to be constant over the period and set to the average of the number of porpoises in the North Sea proper in the two SCANs years (224 100).

2.3 Diet and ration data

Seabirds

Average bird diet data of ten species for the most recent 25 years were estimated as part of the BECAUSE project. For each bird species estimated data include biomass eaten for each prey species and the minimum, mean and maximum length of the prey. There were no further data on size or age distribution available.

Mammals

Data on grey seals

Seal diet data derived from scats were sampled in 1985 and 2002 at haulout sites around the UK coast. Recently, data from 2010/2011 were also presented by Hammond and Willis (2016), but these data were not available to WGSAM. However, they confirm the previous estimates of high gadoid consumption, with very large cod and ling recorded in the scats.

An aggregated estimate of grey seal diet composition based on the 1985 and 2002 collections was calculated for each of these years weighted according to the number of seals using each haulout site. The sizes of fish consumed by the seals were inferred from otolith measurements which are corrected for the effects of digestion. The resulting size distribution for sandeels in grey seal diet suggests that a considerable proportion of the diet in 1985 consisted of sandeels greater than 20cm in length. Because sandeels caught by the fishery are generally smaller than this, there is some uncertainty whether these sandeels are *Ammodytes marinus*, and it has been suggested that they may instead be a different sandeel species such as *Hyperoplus lanceolatus*. To avoid this problem, sandeel larger than 20 cm were assumed to be 'other food'. Net consumption was assumed to be 5.5 kg per seal per day.

Data on harbour porpoise

Decadal diet composition (proportion per species and 1 cm length group) was derived from Danish and UK samples assuming that DK and UK samples each represented 50% of the population except in the 1980's where only Danish samples were available (table 2.3.1). Unfortunately, the number of stomachs was too low to allow quarterly diet composition to be estimated and all diets were assumed to be derived from their 3rd quarter, at this is the quarter where fish recruits in the SMS model and as such have the full size range of fish sizes. Stomach data from each decade were assigned to years, 1985, 1995 and 2005 respectively. Daily consumption was set to 2.4 kg (Sophie Smout, University of St. Andrews, pers. Comm.).

Table 2.3.9. Number of harbour porpoise stomachs analysed per country and decade.

Decade	UK	Denmark
1980–1989	0	40
1990–1999	46	62
2000–2009	56	10

In 2011 and 2014/2015, no correction for differences in evacuation times between prey were applied. In 2017, the data were corrected to account for the fact that residence time of otoliths in the stomach of harbour porpoise depends on the otolith size. A simple model describing this relationship as a power function of otolith length was suggested by Ross *et al.* (2016). Using this model, the bias originating from differential residence time of fish prey otoliths was remedied by applying the correction factor $l_o^{-1.5}$ to the observed numbers of the six prey fish cod, whiting, Norway pout, sandeel, herring and sprat by length class. l_o is the otolith length, which was calculated from the otolith length – total fish length relationships compiled by Leopold *et al.* (2001). The two datasets from UK and DK were merged for each of the three decades 1985–1994, 1995–2004, and 2005–2014, giving equal weight to the data from the two countries.

The corrected size distributions of the six fish species were scaled to the fraction of the food (mass) requirement of the harbour porpoise population in the North sea constituted by these species (*i.e.*, 87.0%, 82.2% and 69.8% of total food requirement for the decades 1985–1994, 1995–2004, and 2005–2014, respectively). Weight-length relationships from the 3rd quarter were used, which is also a change from previously. The correction compared to previously resulted in a 50% increase in herring, 267% increase in sandeel, a 54% decrease for whiting and smaller changes for other species (Figure 2.3.1).

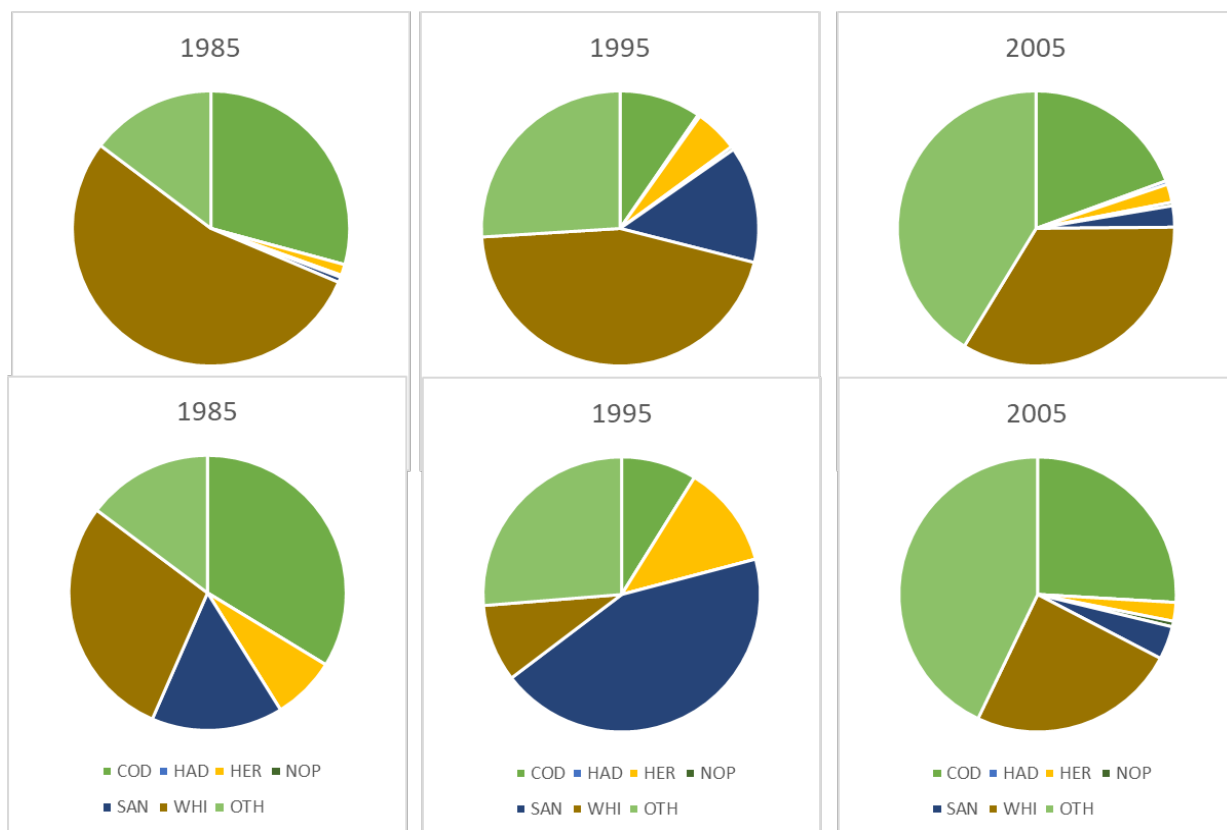


Figure 2.3.16. Harbour porpoise stomach content recorded (top) and consumption rates after correcting for differences in residence times (bottom).

Fish stomach data

An international stomach sampling program was initiated in 1981 to collect stomach contents data from economically important piscivorous fish species in the North Sea. The sampling program was under the auspices of ICES with the purpose to collect data on “who eats whom” of the exploited fish in the North Sea for use in fish stock assessment. Stomachs were sampled from saithe, cod, haddock, whiting and mackerel. Stomach sampling continued in the period 1981 to 1991 with inclusion of more fish species. The highest sampling intensity was in 1981 and 1991. Further information on the background for the ICES stomach sampling project are given in Daan (1989), ICES 1989 and ICES 1997.

Stomach contents data on exchange format are available from ICES (<http://ices.dk/marine-data/data-portals/Pages/Fish-stomach.aspx>)

Compilation of stomach contents data

Stomach contents data are given by year, quarter, predator, predator length/age, prey and prey length/age. The compilation of the individual stomach samples from a trawl haul into average diet of the North Sea follows the technique given by ICES 1996 and is briefly described below. Most stomachs have been pooled within a haul for each of the predator length groups considered.

For each haul the stomach samples for a given species and length class include the information on the number of a) empty stomachs; b) stomach with skeleton remains

only; c) stomach with food and d) stomach with food, but regurgitated. In most cases stomachs within a haul are pooled at the time of sampling for each predator size class. Only stomach contents from the feeding, non-regurgitated stomachs were recorded and later bulked to save time. In the calculation of the average stomach content, it was assumed that the regurgitated stomachs had similar stomach content as the (valid) feeding fish.

First the average stomach content per ICES roundfish area is calculated using stomach data from the ICES rectangles available. If more than one sample is taken from a rectangle, the average stomach content for a predator length class is calculated as a weighted mean, using the number of stomachs sampled as weights. The average stomach content of a given predator and length class in a roundfish area are calculated as a weighted mean of the average stomach content per ICES square weighted by the square root of the arithmetic mean of the observed CPUEs within a rectangle.

Partly digested prey items are in some cases not fully identified to species level or size class. In such cases a species or size redistribution of unidentified items was made accordingly to the observed diet (see ICES, 1997 for details).

The length based observations were optionally transformed into age-groups using an age-length-key (ALK) given by quarter and roundfish area. The ALKs were derived from quarterly surveys or alternatively from commercial catches. Stomach contents data by ages are however not used by SMS.

For a given predator the average North Sea stomach contents by quarter were finally calculated as a weighted mean of the average stomach contents by roundfish area. The quarterly proportions of the stock in the roundfish areas of the total North Sea stock of a given predator were used as weighting factors. The spatial distribution of the predators and age length keys by roundfish area were derived from quarterly surveys or commercial catches.

Estimation of food ration from stomach contents data

Food rations (evacuation rate of stomach contents) are estimated from the observed stomach contents and using the methods suggested by Andersen & Beyer (2005a,b). This model takes into account the differences in evacuation rates between prey types due to their energy density and their resistance to digestion (armament).

Ration (R) (per hour) by prey group (i) for an individual stomach or a pool of stomachs are calculated from:

$$R = \sum_i \rho M_i b_i e^{\delta T} L^\lambda E^{-\xi} K \left(\frac{N_A}{N_F} \right)^{\alpha-1} S^\alpha$$

M= armament of individual prey (group) i

b=proportion of prey (group) i

T= temperature (°C)

L= length (cm) of the predator

E= average energy density (kJ/g wet weight) of the stomach (or of the pooled stomach sample)

N = Number of stomachs in the sample, total (A) and with food (F)

S = average stomach contents in grams

ρ , δ , λ , m and K = parameters to the model

Table 2.3.10. Parameter values of the generic cylinder model of gastric evacuation.

Species	ρ	λ	δ	m	α	K
Cod	0.00224	1.30	0.083	-0.85	0.5	0.85
Haddock	0.00191	1.30	0.083	-0.85	0.5	0.85
Saithe	0.00171	1.35	0.081	-0.85	0.5	0.85
Whiting	0.00171	1.35	0.081	-0.85	0.5	0.85
Mackerel	0.00174	1.30	0.080	-0.85	0.5	0.85

The estimated rations by individual strata (year, quarter, predator and predator size class used in sampling) are combined into one equation for ration from mean weight ($\text{ration} = a \cdot W^b$) where a and b dependent on quarter (Table 2.3.3).

Table 2.3.11. Parameters for estimating quarterly ration per individual from its mean weight (ration= $a \cdot W^b$)

Species	Quarter	a	b				
					4	0.420	0.593
				10 G. gurnards	1	0.423	0.867
					2	0.702	0.790
01 Fulmar	1	34.420	0.000		3	0.786	0.702
	2	28.720	0.000		4	0.592	0.771
	3	27.091	0.000				
	4	34.420	0.000				
02 Guillemot	1	32.456	0.000				
	2	32.258	0.000	Species	Quarter	a	b
	3	32.828	0.000				
	4	32.148	0.000	11 W. horse mac	1	0.000	0.000
03 Her. Gull	1	28.550	0.000		2	0.000	0.000
	2	33.688	0.000		3	4.507	1.765
	3	36.829	0.000		4	1.573	1.035
	4	62.300	0.000	12 N. horse mac	1	0.000	0.000
04 Kittiwake	1	21.865	0.000		2	3.155	1.765
	2	20.971	0.000		3	4.507	1.765
	3	20.971	0.000		4	1.573	1.035
	4	21.865	0.000	13 Grey seal	1	477.855	0.000
05 GBB. Gull	1	42.956	0.000		2	438.480	0.000
	2	43.412	0.000		3	382.284	0.000
	3	44.178	0.000		4	708.882	0.000
	4	48.950	0.000	14 H. porpoise	1	219.000	0.000
06 Gannet	1	84.200	0.000		2	219.000	0.000
	2	89.900	0.000		3	219.000	0.000
	3	89.900	0.000		4	219.000	0.000
	4	84.200	0.000	15 Hake	1	0.772	0.761
07 Puffin	1	14.950	0.000		2	2.180	0.802
	2	15.084	0.000		3	1.302	0.825
	3	15.084	0.000		4	1.527	0.766
	4	14.950	0.000	16 Cod	1	0.900	0.786
08 Razorbill	1	20.116	0.000		2	1.212	0.786
	2	20.916	0.000		3	1.247	0.786
	3	21.159	0.000		4	1.390	0.786
	4	20.116	0.000	17 Whiting	1	0.426	0.683
09 A. radiata	1	0.198	0.548		2	0.455	0.683
	2	0.186	0.509		3	0.679	0.683
	3	0.236	0.463		4	0.574	0.683

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18 Haddock	1	0.323	0.714
	2	0.446	0.714
	3	0.594	0.714
	4	0.588	0.714
19 Sai the	1	0.394	1.045
	2	1.139	1.045
	3	0.604	1.045
	4	0.706	1.045
20 Mackerel	1	0.101	1.443
	2	1.283	1.443
	3	1.444	1.443
	4	0.220	1.443

Calculated consumption rates expressed as daily ration per kg body weight (Figure 2.3.2) generally decreased with size of the predator with the exception of mackerel, saithe and horse mackerel, where consumption increased with predator size. All three species feed mostly on zooplankton at small ages, and the estimates may be a result of underestimation of zooplankton consumption. This should have a limited effect on fish consumption (the amount eaten will be smaller but the relative contribution of fish will be higher).

The consumption in percent body weight for hake was assumed to be the same as for saithe at a similar weight and North Sea horse mackerel consumption was assumed identical to that of western horse mackerel. Following the estimation of all daily consumption rates, daily consumption in weight for each predator age group was estimated using the actual weight at age in the stock of that age group. Previously, a constant ration in weight was used for each age group, but given the recent decrease in mean weight of predators (particularly saithe but also cod), this practice was changed. Similarly, all mean weights at age in the stock of prey fish were updated with annually observed values to account for recent persistent changes in mean weight at age of forage fish.

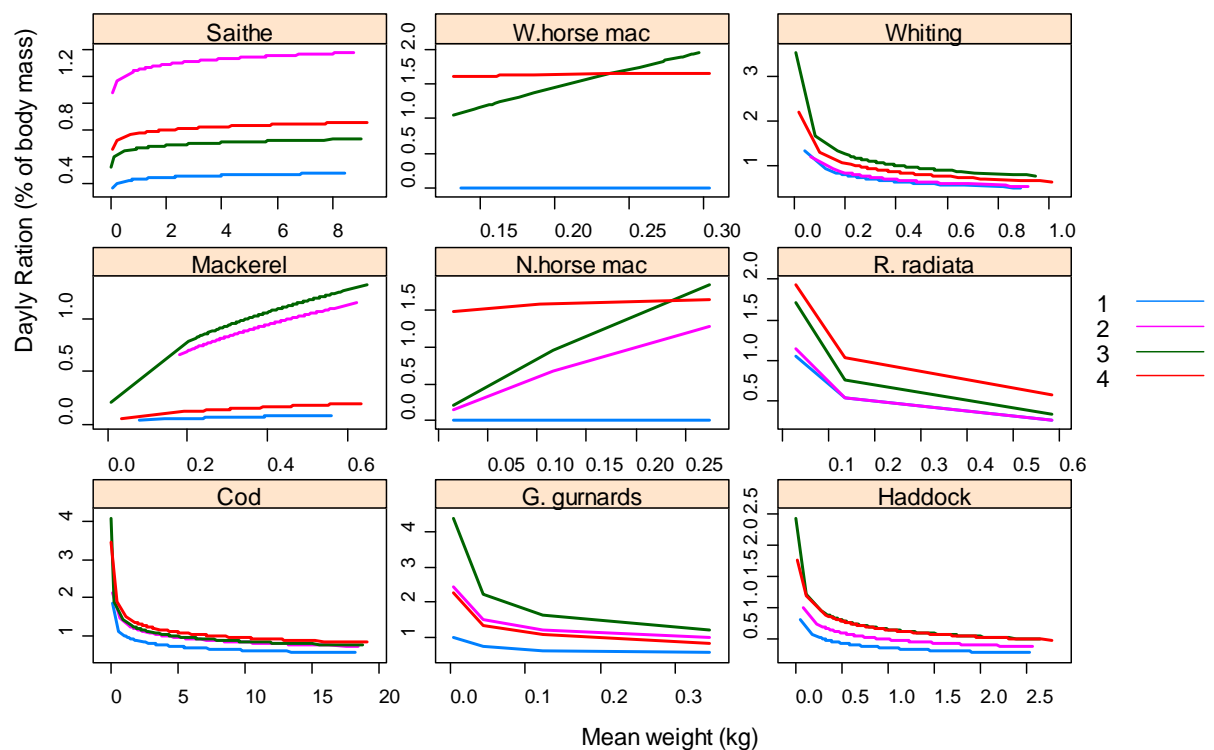


Figure 2.3.17. Daily consumption rates as used in SMS calculated from the method of Andersen. Colours show quarter of the year.

Estimation of diet from stomach contents

The diet of fish species was estimated from the observed stomach contents, taking the prey and temperature dependence into account as done for the calculation of food ration.

Stomachs were firstly pooled into one sample including stomachs from a predator, predator size class, year, quarter and roundfish area, from which the diet was derived. Average temperature for this stratum was derived from temperature by ICES rectangle weighted by the number of stomachs sampled in the rectangles. The outline of the method to derive diet at population levels is described in 2.3.3.1.

Compared to the observed stomach content the estimate of diet shows a relative larger proportion of “other food” and thereby a lower proportion of fish prey (mainly because the energy contents in most fish is higher compared to invertebrates). An example is shown in Table 2.3.4, where the ratio between the new and old estimate is shown for the predators cod and whiting.

Table 2.3.12. Ratio between observed stomach content and the estimated diet data used in SMS for cod in 1991, quarter 2 and 3.

		Predator size class (lower length in mm)												
		100	120	150	200	250	300	350	400	500	600	700	800	1000
Quarter	prey	0.52	.	0.6	0.77	.	0.73	0.6	.
2	COD
	HAD	.	.	.	0.55	.	.	0.6	0.59	0.8	0.82	0.72	0.68	0.72
	HER	0.47	0.64	0.54	0.73	0.75	0.75	0.63	0.68
	NOP	0.52	0.61	0.59	0.79	0.78	0.82	0.69	0.72
	NSA	.	.	0.48	0.52	0.53	0.55	0.65	0.63	0.81	0.82	0.89	0.69	0.72
	OTH	.	1	1.04	1.05	1.12	1.29	1.39	1.32	1.25	1.22	1.26	1.34	1.48
	SPR	0.41	0.47	.	0.47	0.76	0.64	0.61	0.62	0.8
	SSA	.	.	0.47	0.46	0.44	0.5	0.68	0.61	0.7	0.66	0.65	0.59	0.87
	WHG	.	.	.	0.46	.	.	0.59	0.61	0.77	0.8	0.79	0.61	0.71
3	COD	.	0.82	.	0.52	0.67	0.67	0.71	0.65	0.71	0.79	0.86	0.76	0.86
	HAD	.	.	.	0.49	0.63	0.64	0.7	0.7	0.75	0.75	.	0.75	0.86
	HER	0.37	.	0.75	0.71	0.71	0.77	0.69	0.8
	NOP	0.96	0.82	.	0.49	0.65	0.66	0.68	0.69	0.68	0.74	0.78	0.75	0.86
	NSA	.	.	.	0.5	0.63	0.6	0.69	0.7	0.68	0.78	0.83	0.74	.
	OTH	1	1.01	1	1.26	1.55	1.36	1.19	1.51	1.35	1.57	1.6	1.33	1.04
	PLE	0.61
	SOL	0.78
	SPR	0.42	.	0.64	.	0.38	0.42	.	.
	SSA	0.62	0.4	0.34	0.37	.	.	0.27	.	.
	WHG	0.64	0.43	0.53	0.69	0.69	0.75	0.53	0.67	.

Table 2.3.13. Ratio between observed stomach content and the estimated diet data used in SMS for whiting in 1991, quarter 2 and 3.

		Predator size class (lower length in mm)							
		100	120	150	200	250	300	350	400
Quarter	prey	.	0.91	0.84	0.95	0.98	.	.	.
2	COD
	HAD	.	.	0.87	0.86	0.92	.	.	.
	HER	.	.	0.9	.	0.87	0.92	0.86	0.87
	NOP	.	.	0.97	0.89	0.93	0.9	0.93	0.91
	NSA	.	0.99	0.97	0.88	0.92	0.86	0.93	0.91
	OTH	1	1.01	1.04	1.17	1.14	1.23	1.22	1.23
	SPR	.	.	0.85	0.88	0.92	0.95	0.92	0.92
	SSA	0.98	0.86	0.9	0.92	0.99	1.03	1.02	0.99
	WHG	.	0.88	0.82	0.97	0.99	0.98	0.95	0.92
3	COD	.	.	.	0.7	0.95	0.88	.	.
	HAD	1.06	1	0.63	0.77	0.94	1.04	1.08	1.15
	HER	.	.	0.46	0.74	0.87	0.93	0.96	0.85
	NOP	1.05	1.02	0.56	0.79	0.93	1.04	1.08	1.02
	NSA	1.03	1.01	0.62	0.79	0.92	1.02	1.05	1.03
	OTH	0.98	0.97	1.07	1.35	1.5	1.27	1.29	2.33
	SPR	.	.	0.59	0.57	0.75	0.78	0.65	.
	SSA	.	.	0.57	0.79	0.9	0.84	0.72	.
	WHG	1.05	0.88	0.4	0.73	0.92	0.95	1.05	0.93

Appendix 3 provides an overview of diet data as used by SMS by the individual predators and size class. Number of stomachs sampled is also presented in Appendix 3.

Size distribution of predator and prey size classes used for stomach observations

Most of the sampled stomachs have been pooled into size classes, e.g. saithe 300–400 mm in the 1981 sampling, such that information on the individual fish does not exist. Similarly, size of prey item was pooled within size classes, e.g. herring 150–200 mm, in the compilation of stomach contents data. The size distribution and mean length of the individual size classes (and they differs between sampling years) was derived from the size distribution of fish in the sea (or actually in the trawl) estimated from IBTS 1991–1997 data. Sandeel are not caught during IBTS and data from the Danish commercial fishery 1987–2003 were used instead for this prey species. For both data sources, data from several years were combined into one average quarterly size distribution.

This size distribution was then used to split total biomass eaten on age groups using a length-weight relation, and length to age keys from the quarterly IBTS data 1991–1997.

Both the sandeel fishery and IBTS use trawls with a small mesh size, but nevertheless, fish smaller than 5–7 cm are hardly caught. As data are not available to correct for this underrepresentation of the smallest fish, it is ignored in the SMS run, such that the size distribution used by SMS has probably fewer very small fish compared to the size distribution in the sea.

New stomach data

New data were collected in 2013 on mackerel diet composition. Unfortunately, the length of the prey items was not recorded, and therefore, the data cannot be used without assigning the prey types to specific length groups. This estimation could not be performed at WGSAM in 2017, but should be the focus of work in preparation for the next key run.

References

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2.4 Other input data

In addition to the data mentioned above SMS uses data on predator-prey overlap, length-weight relations, residual natural mortality (M1) and age to length keys (ALK)

Predator prey overlap

Predator prey species overlap is a quarter dependent parameter used in the calculation of food suitability (see equation 8 in Appendix 1). By default the spatial overlap is set to one, but it can also be estimated within SMS for a few combinations. "Spatial overlap" does also include vertical overlap, e.g. sandeel as prey when they are available in the water column (mainly quarter 2 and 3) and buried in the sediment (mainly quarter 4 and 1). For some seabirds (fulmar, kittiwake, gannet and razorbill) the spatial overlap is set to 20 for quarter 2 and 3 to reflect the high proportions of sandeel in their (or their chicks') diet. The value 20 was chosen based on a few trial runs, where 20 gave a sufficient fit to data.

Length-weight relations

Conversion from length into weight is used for some SMS configuration. The parameters values are shown below.

Table 2.4.14. Length (mm) weight (kg) relation parameters: $\text{Weight} = a * \text{length}^b$

Species	a	b	source
G. gurnards	6.20000e-09	3.10000	Coull <i>et al.</i> 1989
horse mac	1.05000e-08	2.96220	Silva <i>et al.</i> 2013
Hake	6.59000e-09	3.01700	Fishbase
Cod	2.04750e-08	2.85710	Coull <i>et al.</i> 1989
Whiting	1.05090e-08	2.94560	Coull <i>et al.</i> 1989
Haddock	1.82120e-08	2.82680	Coull <i>et al.</i> 1989
Saithe	2.83220e-08	2.73740	Coull <i>et al.</i> 1989
Mackerel	3.81000e-09	3.21000	Coull <i>et al.</i> 1989
Herring	6.03000e-09	3.09040	Coull <i>et al.</i> 1989
Sandeel	2.66875e-09	3.06000	Stock coordinator
Nor. pout	7.50000e-09	3.02440	Silva <i>et al.</i> 2013
Sprat	8.72900e-10	3.47460	Stock coordinator
Plaice	1.51000e-08	2.88760	Silva <i>et al.</i> 2013
Sole	8.00000e-09	3.04999	Silva <i>et al.</i> 2013

References

Coull K. A., Jermyn A. S., Newton A. W., Henderson G.I. and Hall W.B. 1989. Length weight Relationships for 88 Species of Fish Encountered in the North East Atlantic. Scottish Fisheries Research Report Number 43: 81pp.

Silva J. F., Ellis J. R. and Ayers R. A. 2013. Length-weight relationships of marine fish collected from around the British Isles. Sci. Ser. Tech. Rep., Cefas Lowestoft, 150: 109pp.

Age to length conversion keys

SAM is an age-length based model, where stock dynamic (N, F, M2 etc.) is by age classes while predation is calculated on the basis of the sizes of predators and preys. This means that e.g. stock numbers at age has to be converted into stock number at size class for the calculation of M2.

For each species, age and quarter the proportion of stock numbers by size classes used at the 1991 stomach sampling are derived from the derived from the size distribution of fish in the sea (or actually in the trawl) estimated from IBTS 1991–1997 data. Sandeel are not caught during IBTS and data from the Danish commercial fishery 1987–2003 were used instead for this species. For both data sources, data from several years were combined into one average quarterly size distribution. Both the sandeel fishery and IBTS use trawls with a small mesh size, but nevertheless, fish smaller than 5–7 cm are hardly caught. As data are not available to correct for this bias, it is ignored in the SMS run, such that the size distribution used, has probably fewer very small fish compared to the size distribution in the sea.

An example of the age to length conversion keys is shown in the table below.

Table 2.4.15. Example of age to length conversion key: Whiting. The table shows the percentage of a given size class for a given age and quarter.

		Size class (lower limit in mm)													All
		50	60	70	80	100	120	150	200	250	300	350	400	500	
Age	Quarter														
0	3	2.0	8.1	16.8	35.9	21.1	11.0	5.2	100.0
	4	.	1.0	2.0	5.0	15.3	31.0	42.7	3.0	100.0
1	1	.	.	1.0	2.0	3.8	31.4	50.8	11.1	100.0
	2	2.0	14.8	67.5	15.7	100.0
	3	1.0	2.0	28.6	59.4	9.0	100.0
	4	2.0	11.4	70.3	16.3	100.0
2	1	4.1	62.4	32.1	1.4	.	.	.	100.0
	2	0.1	6.6	63.6	28.6	1.2	.	.	.	100.0

	3	0.0	0.7	31.8	59.9	7.6	.	.	.	100.0
	4	0.1	34.2	56.1	9.5	.	.	.	100.0
3	1	0.2	16.2	66.2	17.4	.	.	.	100.0
	2	17.2	67.5	15.3	.	.	.	100.0
	3	0.2	7.8	60.8	27.6	3.5	.	.	100.0
	4	0.0	3.6	60.8	31.3	4.3	.	.	100.0
4	1	0.2	4.0	49.6	39.3	6.9	.	.	100.0
	2	4.6	58.4	31.2	5.8	.	.	100.0
	3	2.2	38.7	45.4	11.9	1.9	.	100.0
	4	1.9	47.4	37.1	11.3	2.3	.	100.0
5	1	0.8	39.9	42.6	14.2	2.4	.	100.0
	2	3.1	46.8	36.1	11.4	2.7	.	100.0
	3	0.6	32.0	48.8	14.2	4.4	.	100.0
	4	44.3	42.1	10.5	3.1	.	100.0
6	1	0.2	38.6	45.0	11.1	5.1	.	100.0
	2	4.1	43.7	37.5	11.2	3.6	.	100.0
	3	34.3	42.2	18.3	5.1	.	100.0
	4	0.7	43.9	46.0	7.0	2.4	.	100.0
7	1	25.5	58.0	9.7	6.7	.	100.0
	2	28.0	48.1	17.6	6.4	.	100.0
	3	1.7	76.1	14.6	7.6	.	100.0
	4	25.8	60.2	10.6	3.4	.	100.0
8	1	32.3	44.2	14.8	5.8	2.9	100.0
	2	19.0	49.0	26.9	5.0	.	100.0
	3	22.0	47.8	22.2	8.0	.	100.0
	4	70.5	26.4	1.1	2.1	100.0

Residual natural mortality (M1)

M1 (residual natural mortality) by quarter is set to

0.05 for the species cod, whiting, haddock, saithe, the two sandeel stocks, Norway pout, sprat and

0.0375 for mackerel, and

0.025 for herring, plaice and sole

M1 for non-prey species is the annual natural mortality (M) used in the single species assessment divided on 4 quarters.

3. Model configuration

The configuration of the SMS model aims firstly to mimic the results from ICES single species assessment models when SMS is run in single species mode (no estimation of predation mortality) using the same annual M values as the single species assessment, and secondly to configure options for predation mortality as concluded at the last key run (if not changed).

Appendix 4 presents the SMS configuration (option files) used for the 2017 key-run.

3.1 Fishing mortality

SMS uses a separable F model while some of the ICES single species models use a more flexible model for F (e.g. SAM using random walk F, or XSA where F are estimated directly from catch observation). Further, some models use types of abundances indices (e.g. SSB or tagging data) and estimate process noise, which have not been implemented in SMS. The SMS single species assessment will therefore not be able to replicate the ICES single species output, but the results should be quite close.

In Appendix 5, the stock summaries from ICES single species assessment are compared with the summaries from the SMS runs using fixed M. The differences are commented below.

Cod

The 2017 SMS model run for cod in single species mode mirrors the ICES assessment in the development of F (Appendix 5 figure A5.1). SSB is somewhat lower due to the use of quarter 1 mean weight in the stock in SMS whereas the ICES assessments use annual average weight at age when estimating SSB. SMS uses the ICES mean weights as an annual mean weight, but uses a fixed quarterly growth increment factor, which means that mean weight in quarter 1, as used in the calculation of SSB, becomes smaller in SMS than in the ICES assessment. Recruitment in SMS is always at age zero in quarter 3, while the ICES assessment uses age 1 at the beginning of the year. This difference in recruitment timing makes it difficult to compare the two recruitment estimates.

Whiting

The whiting assessment has undergone an inter-benchmark between the 2015 and 2017 multispecies key run. While the 2017 SMS run mirrors the development in F from the ICES assessment directly (Appendix 5 figure A5.2), the SSB level is lower in the SMS assessment than in the ICES assessment, part of which is explained by the difference in mean weight at age (quarterly vs annual).

Haddock

The 2017 SMS assessment of haddock followed the trend of F and SSB from the ICES assessment quite well, but F is larger and SSB is lower in the SMS run (Appendix 5 figure A5.3).

Saithe

F and SSB are quite similar between the two runs (Appendix 5 figure A5.4), but recruitment seems different due to recruitment at age 0 in SMS and at age 3 in ICES assessment. The 2017 SMS model run for saithe estimated a higher recruitment in later years. The high similarity between the two assessment was only possible because of the stock numbers of ages 3–9 from the ICES assessment were introduced as survey tuning series into the 2017 SMS model run for 1997–2016, assuming a CV of 0.3 for this “survey” time-series. As this species is only a predator in the model, this means that the natural mortalities induced by saithe are consistent with the stock size as estimated in the latest ICES assessment, however uncertainties of SMS output in general might be biased.

Mackerel

In general, the SMS assessment is similar to the ICES assessment for recruitment and from 1990 onwards for F and SSB (Appendix 5 figure A5.5). It does however not fully mirror the development in SSB in the ICES assessment for the period from 1980 to 1990. For this period, the ICES model is down-weighting the observed catches but this feature was not implemented in the SMS version. If the ICES assessment is more correct, this may lead to an underestimation of natural mortality of sprat and sandeel in the period from 1980 to 1990 as the abundance of large mackerel may be underestimated.

Herring

The 2017 SMS assessment of herring follows the ICES assessment reasonably well, even though the development of F is smoother in the ICES assessment which models F as a random walk process (Appendix 5 figure A5.6). Difference in spawning time in the two models will give differences in estimated SSB, but does not fully explain the difference in SSB from the two models.

Norway pout

There has been a benchmark in 2016, and the 2015 SMS run is based on different data and therefore not strictly comparable. The ICES assessment estimates SSB on November 1st, whereas the SMS uses SSB by January 1st, and since natural mortality is larger than growth in the period between the two, the ICES values are substantially lower than the

SMS ones. The 2017 SMS run shows similar developments in F and recruitment as the ICES assessment (Appendix 5 figure A5.7).

Sandeel

The sandeel are assessed in substocks that are not identical to those in the multispecies SMS implementation. Therefore, the results were compared to the 2015 key run with the changes made in the 2016 sandeel benchmark. This included the division of effort (here CPUE) series into 5 periods (-1988, 1989–1998, 1999–2004, 2005–2009, 2010–), that was also implemented in the 2017 SMS run. Both sandeel stocks showed similar dynamics in the two runs (figure not shown).

Sprat

The 2017 SMS output is not directly comparable with the ICES assessments for SSB as SSB in SMS was estimated on January 1st whereas the ICES assessment uses the estimate by July 1st. Further, the age-range for F -bar is different (age 1.5–age 2.5 in the ICES assessment). Recruitment is directly comparable as date is the same and here the two assessments show the same temporal pattern (Appendix 5 figure A5.8).

Plaice and sole

Plaice and sole are not a predators or preys in SMS, so the final SMS assessment is equal to the single species SMS presented (Appendix 5 figure A5.8 and A5.9). The stock dynamics are estimated quite similarly from the two models, but SMS has a more modest increase in plaice SSB in recent years compared to the ICES assessment.

3.2 Configuring predation mortality options

The SMS model has two options for size preferences of predators: either prey are taken according to their abundance in the environment (no size selection) within the observed predator-prey size range; or it can be assumed that a predator has a preferred prey size ratio and that a prey twice as big as the preferred size is as attractive as another half the prey size (log-normal distribution). In 2011, sensible size preferences could only be estimated for around half the fish species and the parameters for the remaining predators were close to the bounds. This corresponds to a situation where the data do not contain sufficient information to estimate the size preference parameters. This was also the case for grey seals. For harbour porpoise, modelling size selection as non-uniform resulted in a greater preference and hence natural mortality of 1-year old cod and a lower consumption of 0- and 2-year old cod. Predicted recruitments, F s and SSBs were virtually identical. The likelihood of the model was improved by 10 with two 2 parameters added, which indicted as statistical significant improvement of the fit (X^2 test). Inspection of the fit revealed, however, that the size distribution in the diet predicted with size selection was substantially narrower than the observed.

WGSAM 2011 considered that size selection should either be for all predators or none, or at least consistent within groups such as fish and mammals. Given that the model likelihood was only slightly improved by introducing size selection, that fitting parameters close to their bounds may give unwanted results inside the model (for technical reasons) and that the fits of the diets themselves were not improved for all species, it was decided

to use uniform selection for all predator species, as done since the 2007 key run. This practice was continued in the 2017 key run, such that model options for predation mortality have been kept constant since the 2014 key run, except for harbour porpoise.

With the change in mean weight at age for cod, cod at age 3 obtained a smaller mean weight which gave a steep increase in M2 for age 3, as the diet data show that harbour porpoise can eat the (now smaller) age 3 cod. WGSAM 2017 discussed this issue a lot and concluded that the available diet data for harbour porpoise was not sufficient to justify such an increase in M2. Technically, the configuration of size selection was changed from “uniform size selection” to “Constraint uniform size selection” (see equation 13 in Appendix 1) such that the harbour porpoise could not eat cod older than 2 years (implemented by a predator:prey size range). For the other preys eaten by porpoise the constraints in size selection were set to the observed value such that the size selection model in practise was not change for these preys.

4. Other issues

The SMS model, and input and output can be found at Github: https://github.com/ices-eg/wg_WGSAM

The Github include several directories:

- NortSeaKeyRun_2014: The SMS North Sea key run made at the 2014 WGSAM, including data for the period 1974–2013. The version here has been corrected in 2015 for an input error.
- NortSeaKeyRun_2017: The SMS North Sea key run made at the 2017 WGSAM, including data for the period 1974–2016.
- input_output: Detailed presentation of input and output file for the 2017 key run
- SMS_ADMB: AD Model Builder source code for the SMS North Sea program
- SMS_R_prog: R scripts for preparing, running and presenting results from a SMS run

5. Results of the 2017 North Sea SMS key run

Substantial changes of input data to the new key run and ICES benchmarks for some of the stocks since the 2014 key run have produced stock summaries (recruitment, mean F and SSB) from the 2017 key run that is somewhat different from the summaries from the 2014-key run. However, the new estimated predation mortalities (M2) are consistent with the M2 values from the previous key run. The robustness of the estimate of predation mortality corresponds well to the conclusion made by the long row of ICES working groups using the SMS model or previous model versions like 4M and MSVPA, that the estimate of M2 is robust to e.g. changes in consumption rates and the amount of “other food” in the diet, level of M1 or amount of other predators.

Key run summary sheet

Area	North Sea
Model name	SMS
Type of model	Age-length structured statistical estimation model
Run year	2017
Predatory species	Assessed species: Cod, haddock, saithe, whiting, mackerel Species with given input population size: North Sea horse mackerel, western horse mackerel, grey gurnard, starry ray, hake, fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin, razorbill, grey seal, harbour porpoise
Prey species	Cod, haddock, herring, Norway pout, southern North Sea sandeel, northern North Sea sandeel, sprat, whiting,
Time range	1974–2016.
Time step	Quarterly
Area structure	North Sea
Stomach data	Fish species: 1981, 1985, 1986, 1987, 1991, 2005, 2013 Grey seals: 1985, 2002 Harbour porpoise: Decadal 1985, 1995, 2005
Purpose of key run	Making historic data on natural mortality available and multispecies dynamic
Model changes since last key run	All time-series updated. Mackerel included as a modelled stock. Proportion of the stock within the North Sea given as input and used for estimating M2. Daily food ration of changed for the main fish species. Bias correction of diet composition of harbour porpoise and the main predatory

	fish.
Output available at	https://github.com/ices-eg/wg_WGSAM
Further details in	Report of the Working Group on Multispecies Assessment Methods 2017

5.1 Results of the 2017 key run

The input and output from the model are comprehensive and cannot all be presented in this report. This report presents only the key-output.

Detailed input- and output data on ASCII and HTML files, and presented on graphs can be downloaded from WGSAM SharePoint/data/North_Sea_key_run or from https://github.com/ices-eg/wg_WGSAM

The structure of data in the “input_output” directory to be downloaded is:

Input

Configuration

Option files for SMS configuration

c.obs

plots of observed catch numbers at age from the 2014- and 2017 key runs

OtherPredators

plots of stock size of external predators from the 2014- and 2017 key runs

West

plots of mean weight at age in the sea from the 2014- and 2017 key runs

PropMat

plots of proportion mature at age in the sea from the 2014- and 2017 key runs

Ration

plots of consumption (food ration) at age from the 2014- and 2017 key runs

StomachContents

plots of relative stomach contents

Output

Diagnostics

Diagnostic overview file

Residual plots (catch and survey)

Stomachs

Plots of observed and predicted stomach contents

StockSummary

Stock summaries as plots and tables

ASCII files with all input and output variables by year, quarter, species and age

Uncertainties

Coefficient of variations of estimated recruitment, mean F, SSB and M2

NaturalMortalities

Tables with M2 and $M=M1+M2$ values

PartialM2

Plots of M2 by year (and quarter) for each age group of prey species, showing the partial M2 from each predator

WhoEatsWhom

Plots of biomass eaten by various combinations of predator and preys.

CSV files with the same information (on three aggregation levels).

Comparisons

Plots of stock summaries from the 2014- and 2017 key runs

Plots of M2 at age from the 2014- and 2017 key runs

ICEScomparison

Comparison of ICES single species assessment and SMS in single species mode

Retrospective

Plots of stock summaries, retrospective analysis 2013 to 2016

Plots of M2 at age, retrospective analysis 2013 to 2016

Text in **bold** shows directory names.

The key-run including executable and source file for SMS can be found in the directory **SMS-key-run-2017**

Model diagnostics

The population dynamics of all species except ‘external predators’ were estimated within the model. The key-run converged and the uncertainties of parameters and key output variables were obtained from the inverse Hessian matrix. Key diagnostics (Table 5.1.1) show a reasonable fit for catch and survey indices data for most species. For Norway pout and sprat the fit to catch data is poor; however better for survey indices. The two sandeel stocks show a reasonable fit to catch data in the main fishing season (quarter 2) but the fit is poor for quarter 3. Stock recruitment relationships are estimated quite well (reasonable sigma value) for the stocks except for haddock.

Table 5.1.1. Key SMS model diagnostics.

November 14, 2017 13:16:46 run time:362 seconds

objective function (negative log likelihood): -5126.37

Number of parameters: 1817

Number of observations used in likelihood: 15348

Maximum gradient: 0.00958871

Akaike information criterion (AIC): -6618.74

Number of observations used in the likelihood:

	Catch	CPUE	S/R	Stomach	Sum
Species: 1, Fulmar	0	0	0	144	144
Species: 2, Guillemot	0	0	0	144	144
Species: 3, Her. Gull	0	0	0	168	168
Species: 4, Kittiwake	0	0	0	132	132
Species: 5, GBB. Gull	0	0	0	204	204
Species: 6, Gannet	0	0	0	96	96
Species: 7, Puffin	0	0	0	96	96
Species: 8, Razorbill	0	0	0	132	132
Species: 9, A. radiata	0	0	0	64	64
Species:10, G. gurnards	0	0	0	149	149
Species:11, W.horse mac	0	0	0	14	14
Species:12, N.horse mac	0	0	0	34	34
Species:13, Grey seal	0	0	0	54	54
Species:14, H. porpoise	0	0	0	19	19
Species:15, Hake	0	0	0	33	33
Species:16, Cod	430	275	43	881	1629
Species:17, Whiting	344	356	43	586	1329
Species:18, Haddock	430	376	43	130	979
Species:19, Saithe	344	290	43	188	865
Species:20, Mackerel	430	433	43	105	1011
Species:21, Herring	1634	440	43	0	2117
Species:22, N. sandeel	774	204	43	0	1021
Species:23, S. sandeel	774	144	43	0	961
Species:24, Nor. pout	602	300	43	0	945
Species:25, Sprat	516	241	43	0	800
Species:26, Plaice	430	779	41	0	1250
Species:27, Sole	387	530	41	0	958
Sum	7095	4368	512	3373	15348

unweighted objective function contributions (total):

	Catch	CPUE	S/R	Stom.	Stom N.	Penalty	Sum
Fulmar	0.0	0.0	0.0	-323.9	0.0	0.00	-324
Guillemot	0.0	0.0	0.0	-204.3	0.0	0.00	-204

Her. Gull	0.0	0.0	0.0	-388.3	0.0	0.00	-388
Kittiwake	0.0	0.0	0.0	-237.4	0.0	0.00	-237
GBB. Gull	0.0	0.0	0.0	-502.8	0.0	0.00	-503
Gannet	0.0	0.0	0.0	-135.6	0.0	0.00	-136
Puffin	0.0	0.0	0.0	-104.2	0.0	0.00	-104
Razorbill	0.0	0.0	0.0	-149.9	0.0	0.00	-150
A. radiata	0.0	0.0	0.0	-35.8	0.0	0.00	-36
G. gurnards	0.0	0.0	0.0	-80.0	0.0	0.00	-80
W.horse mac	0.0	0.0	0.0	2.1	0.0	0.00	2
N.horse mac	0.0	0.0	0.0	-9.8	0.0	0.00	-10
Grey seal	0.0	0.0	0.0	-124.7	0.0	0.00	-125
H. porpoise	0.0	0.0	0.0	-26.7	0.0	0.00	-27
Hake	0.0	0.0	0.0	-17.9	0.0	0.00	-18
Cod	-416.4	-137.5	-5.5	-1463.6	0.0	0.00	-2023
Whiting	-241.5	-193.6	-20.9	-668.0	0.0	0.00	-1124
Haddock	-208.0	-96.3	15.1	-82.1	0.0	0.00	-371
Saithe	-282.2	-107.4	-17.5	-108.0	0.0	0.00	-515
Mackerel	-410.2	-145.7	-11.1	-83.2	0.0	0.00	-650
Herring	272.9	-106.8	-6.3	0.0	0.0	0.00	160
N. sandeel	126.0	27.6	8.2	0.0	0.0	0.00	162
S. sandeel	78.9	-51.3	1.9	0.0	0.0	0.00	30
Nor. pout	178.0	0.4	-8.0	0.0	0.0	0.00	170
Sprat	364.2	7.1	-6.3	0.0	0.0	0.00	365
Plaice	-398.1	-150.5	-26.3	0.0	0.0	0.00	-575
Sole	-445.6	123.5	-3.0	0.0	0.0	0.00	-325
Sum	-1382.1	-830.6	-79.6	-4743.9	0.0	0.00	-7036

sqrt(catch variance) ~ CV:

Cod

1	0.627
2	0.146
3	0.146
4	0.146
5	0.146
6	0.146
7	0.230
8	0.230
9	0.437

10	0.437
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Whiting

0	1.147
1	0.482
2	0.198
3	0.198
4	0.198
5	0.312
6	0.312
7	0.312
8	0.312

Haddock

0	0.795
1	0.468
2	0.228
3	0.228
4	0.228
5	0.228
6	0.365
7	0.365
8	0.622
9	0.622
10	0.622

Saithe

3	0.451
4	0.451
5	0.197
6	0.197
7	0.197
8	0.255
9	0.255
10	0.255

Mackerel

1	0.396
2	0.414
3	0.204
4	0.204
5	0.204

6	0.204
7	0.204
8	0.204
9	0.204
10	0.204

Herring

	season			
age	1	2	3	4
0			0.717	0.906
1	0.898	0.650	0.643	0.554
2	0.898	0.650	0.643	0.554
3	0.898	0.650	0.643	0.554
4	0.898	0.650	0.643	0.554
5	0.898	0.650	0.643	0.554
6	0.898	0.650	0.643	0.554
7	0.898	0.650	0.643	0.554
8	1.350	0.788	0.604	1.070
9	1.350	0.788	0.604	1.070

N. sandeel

	season			
age	1	2	3	4
0			1.351	
1		0.559	1.321	
2		0.559	1.321	
3		0.559	1.321	
4		1.188	1.131	

S. sandeel

	season			
age	1	2	3	4
0			1.414	
1		0.445	1.085	
2		0.445	1.085	
3		0.445	1.085	

4 0.445 1.085

Nor. pout

season

age	1	2	3	4
0			1.414	1.226
1	0.708	0.687	0.633	0.729
2	0.708	0.687	0.633	0.729
3	0.956	1.042	1.132	0.969

Sprat

season

age	1	2	3	4
1	1.398	1.414	1.023	0.850
2	1.414	1.092	1.016	0.848
3	1.414	1.414	1.181	1.414

Plaice

1	0.391
2	0.156
3	0.239
4	0.239
5	0.239
6	0.239
7	0.239
8	0.239
9	0.239
10	0.239

Sole

2	0.415
3	0.174
4	0.174
5	0.174
6	0.174
7	0.174
8	0.174
9	0.174

10 0.174

sqrt(Survey variance) ~ CV:

Cod	age 0	age 1	age 2	age 3	age 4	age 5
age 6						
Cod IBTS Q1		0.55	0.31	0.31	0.31	0.31
Cod IBTS Q3		0.47	0.36	0.36	0.36	
Whiting						
Whg IBTS Q1 1974-1988		0.41	0.43	0.43	0.43	0.43
Whg IBTS Q1 1989-		0.40	0.32	0.32	0.32	0.32
Whg IBTS Q3	0.67	0.38	0.28	0.28	0.28	0.28
Haddock						
Had IBTS Q1		0.51	0.51	0.51	0.51	0.51
Had IBTS Q3	0.52	0.28	0.28	0.51	0.51	0.51
Saithe						
Pok IBTS Q3				0.96	0.96	0.68
0.68						
Pok SAM N				0.22	0.22	0.22
0.22						
Mackerel						
Mac Swept area		0.89	0.60	0.60	0.60	0.60
0.60						
Mac SAM assessment	0.39	0.39	0.39	0.39	0.39	0.39
0.39						
Her HERAS		0.44	0.29	0.37	0.37	0.37
0.37						
Her IBTS Q1		0.31	0.85	0.85	0.85	0.85
Her MIK	0.46					
N. sandeel						
Nsa Dregde 2004-	0.66	0.58	0.58			
Nsa Commercial 1982-1998		0.62	0.62	0.62		
Nsa Commercial 1999-		0.79	0.54	0.54		
Nsa Commercial 2 1976-2004	1.21					
Nsa acoustic		0.78	0.78	0.67	0.67	
S. sandeel						
Ssa Dregde 2004-	0.48	0.85	0.85			
Ssa Commercial 1982-1998		0.58	0.43	0.43		
Ssa Commercial 1999-2004		0.20	0.20	0.20		
Ssa Commercial 2005-2009		0.20	0.20	0.20		
Ssa Commercial 2010-2016		0.49	0.49	0.49		
Nor. pout						
Nop ENGFS 1982-1991	1.12	0.47	1.17	1.17		
Nop ENGFS 1992-	0.96	0.37	0.64			
Nop IBTS Q1 1974		0.54	0.59	0.59		

Nop SGOGFS 1998	0.62	0.46	0.46		
Sprat					
Spr IBTS Q1 1974-		0.62	0.62	0.62	
Spr HERAS-Acoustic 2003-		0.40	0.47	0.47	
Spr IBTS Q3 1991-		0.77	0.64	0.86	
Plaice					
Ple BTS 1985-1995		0.49	0.49	0.65	0.65
0.63					0.63
Ple BTS-1996-		0.32	0.36	0.46	0.46
0.46					0.46
Ple SNS 1974-1999		0.42	0.42	0.91	0.91
1.18					1.18
Ple SNS 2000-		0.34	0.34	0.73	0.73
1.29					1.29
Ple IBTS Q3 1997-		0.45	0.33	0.36	0.36
0.36					0.36
Ple IBTS Q1 2007-		0.33	0.33	0.25	0.25
0.41					0.41
Sole					
Sol BTS 1985-		0.53	0.41	0.60	0.60
1.20					0.60
Sol SNS 1974-		0.55	0.59	0.59	0.59
1.27					1.27
Recruit-SSB		alfa	beta	recruit s2	re-
cruit s					
Cod	Hockey stick -break.:		135.736	1.180e+005	0.253
0.503					
Whiting	Hockey stick -break.:		83.438	1.840e+005	0.111
0.334					
Haddock	Hockey stick -break.:		40.148	1.000e+005	1.042
1.021					
Saithe	Ricker:		2.842	4.491e-006	0.163
0.404					
Mackerel	Geometric mean:		15.242		0.202
0.449					
Herring	Ricker:		58.172	8.901e-007	0.275
0.524					
N. sandeel	Ricker:		2327.920	3.987e-006	0.540
0.735					
S. sandeel	Ricker:		1286.318	2.588e-006	0.402
0.634					
Nor. pout	Ricker:		1442.563	4.459e-006	0.246
0.496					
Sprat	Hockey stick -break.:		949.209	9.000e+004	0.259
0.509					
Plaice	Ricker:		6.068	1.611e-006	0.102
0.319					
Sole	Ricker:		7.224	2.554e-005	0.318
0.564					

Retrospective analysis for M2

The retrospective analysis of M2 shows a consistent estimate of predation mortalities (Figure 5.1.1 to Figure 5.1.8). As for all other retrospective assessment analysis, this analysis also shows that values (M2) in the terminal year of the time-series have larger uncertainties; however this uncertainty is not huge. The largest retrospective variability is seen for southern sandeel (Figure 5.1.6), which is probably due to the variability in the stock number estimate from catch and survey observation, rather than due to variability from one year to next in parameter estimates related to predation. Southern sandeel assessment make use of a short survey indices time-series, 2010–2016, which provide uncertain and variable stock estimates when reduced further in the retrospective analysis.

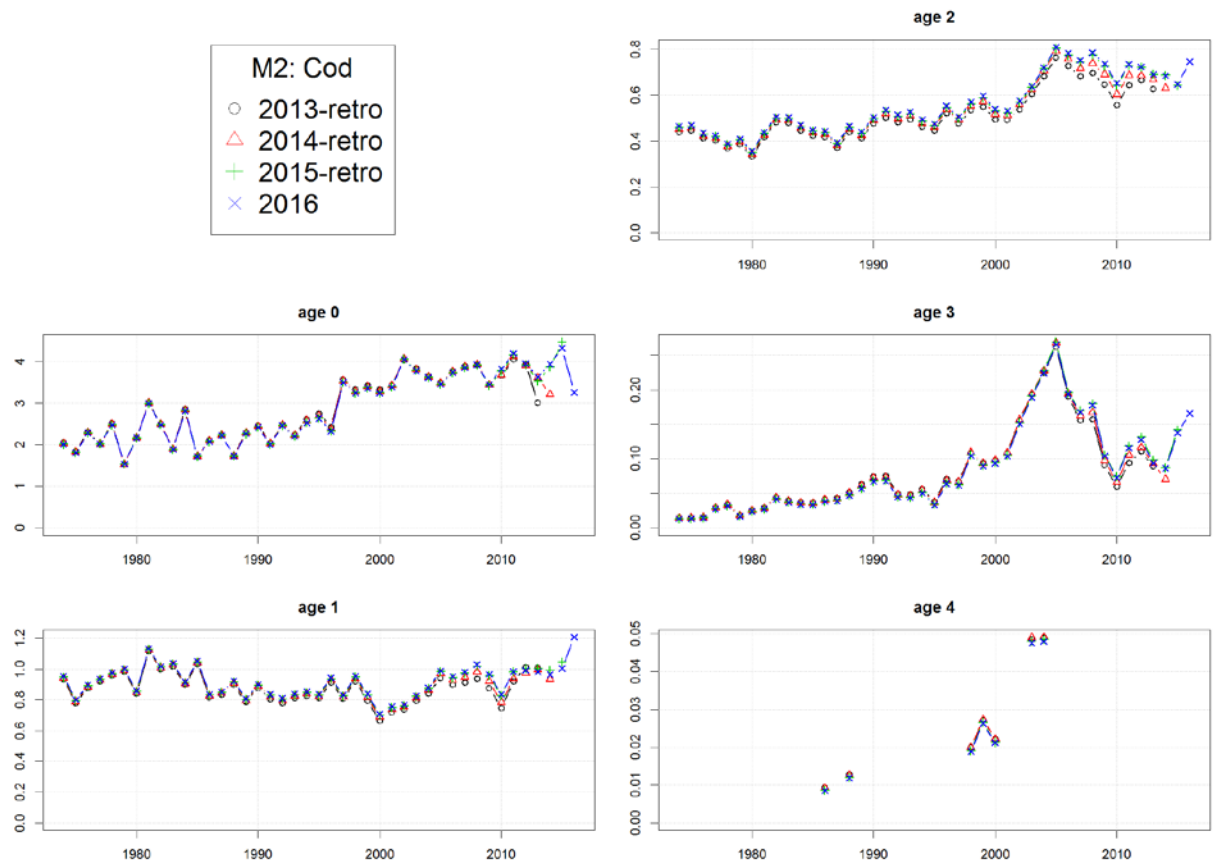


Figure 5.1.1 Retrospective analysis of M2 for cod.

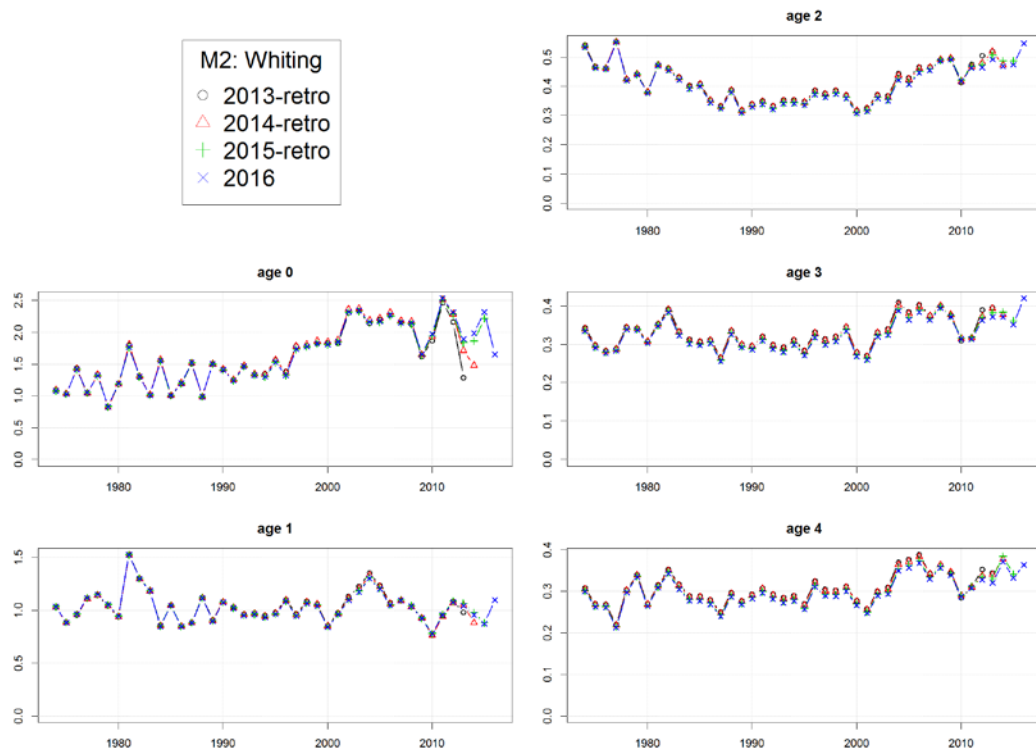


Figure 5.1.2. Retrospective analysis of M2 for whiting.

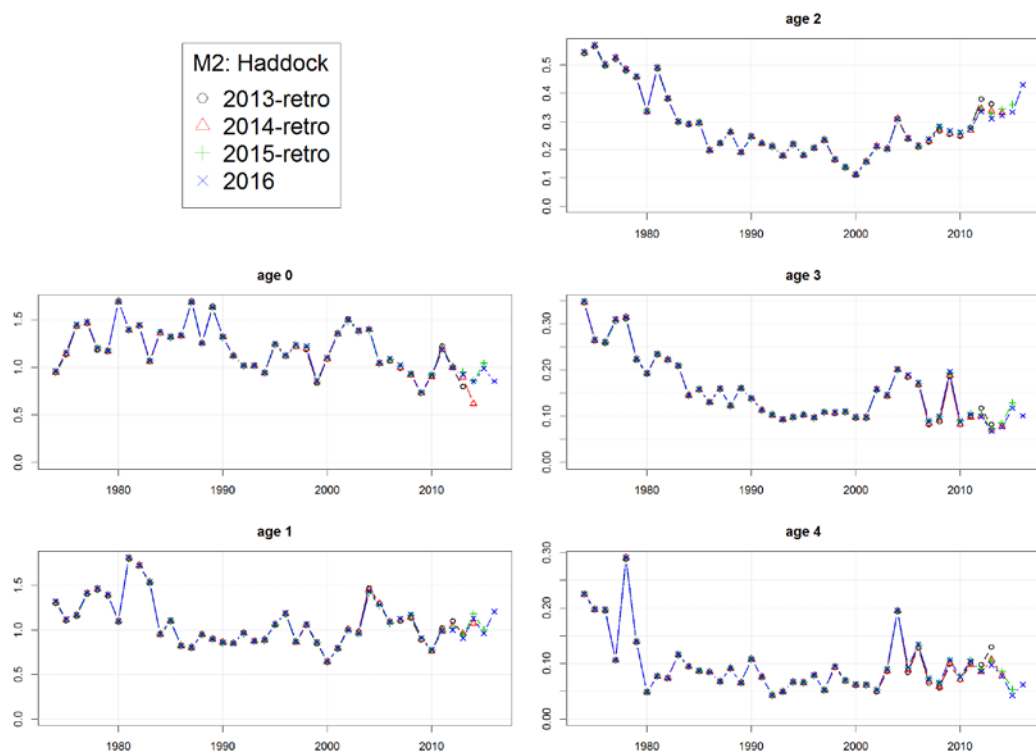


Figure 5.1.3. Retrospective analysis of M2 for haddock.

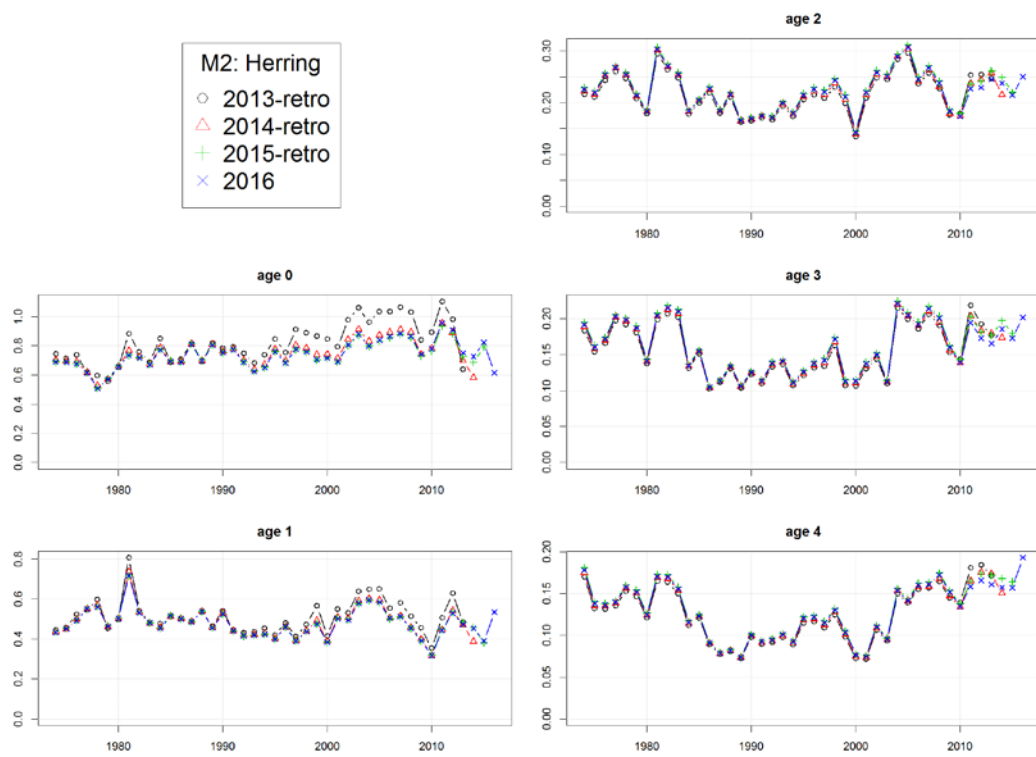


Figure 5.1.4. Retrospective analysis of M2 for herring.

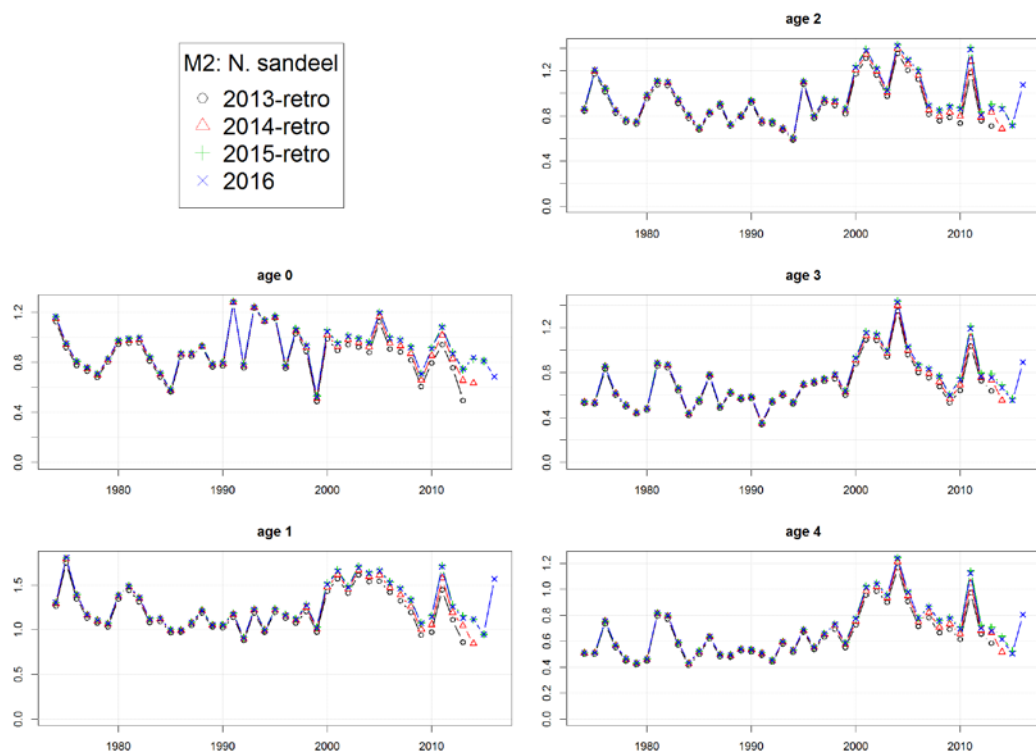


Figure 5.1.5 Retrospective analysis of M2 for northern sandeel.

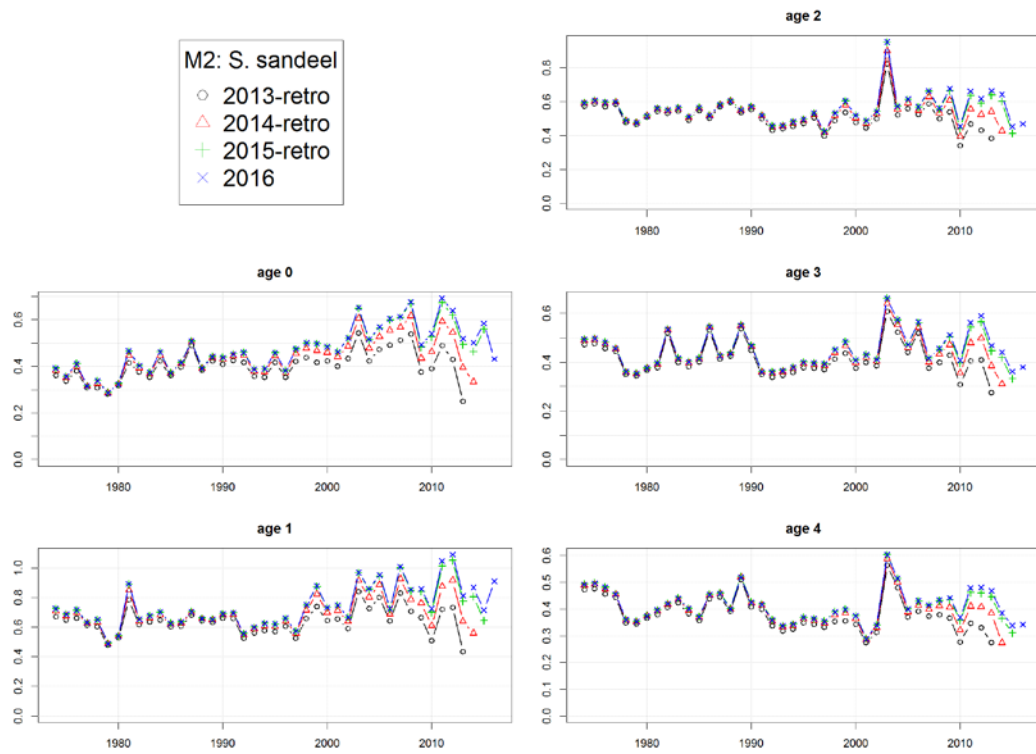


Figure 5.1.6 Retrospective analysis of M2 for southern sandeel.



Figure 5.1.7 Retrospective analysis of M2 for Norway pout.

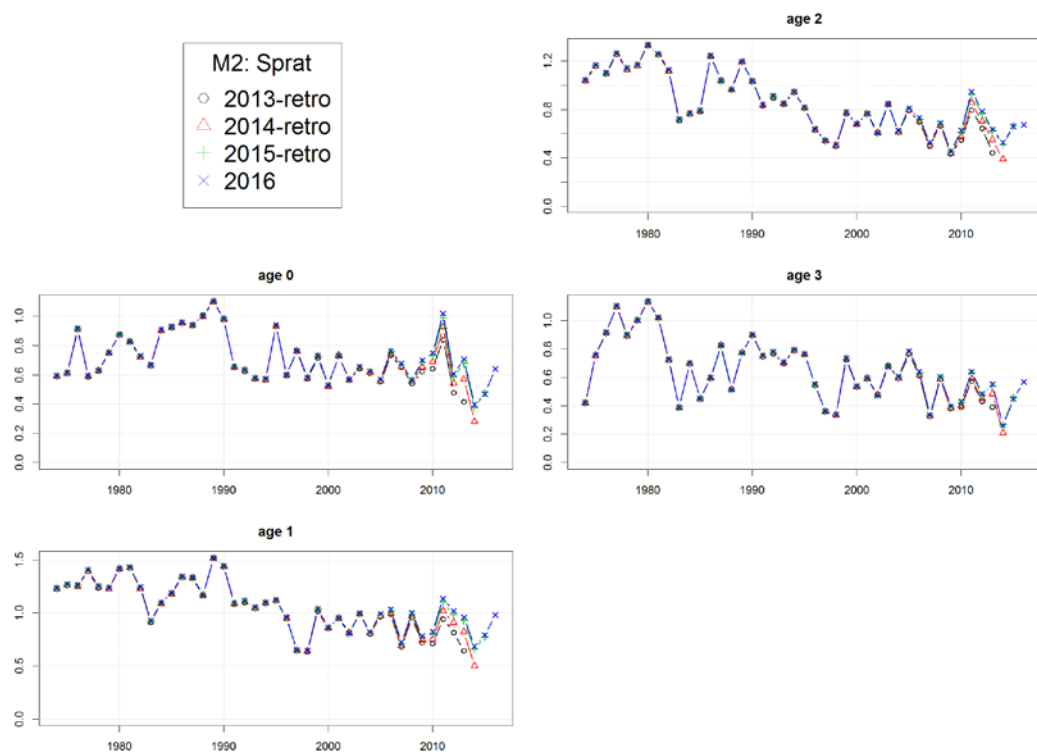


Figure 5.1.8 Retrospective analysis of M_2 for sprat.

Stock summary results

The stock summaries are presented in Figure 5.1.9 to Figure 2.1.13.

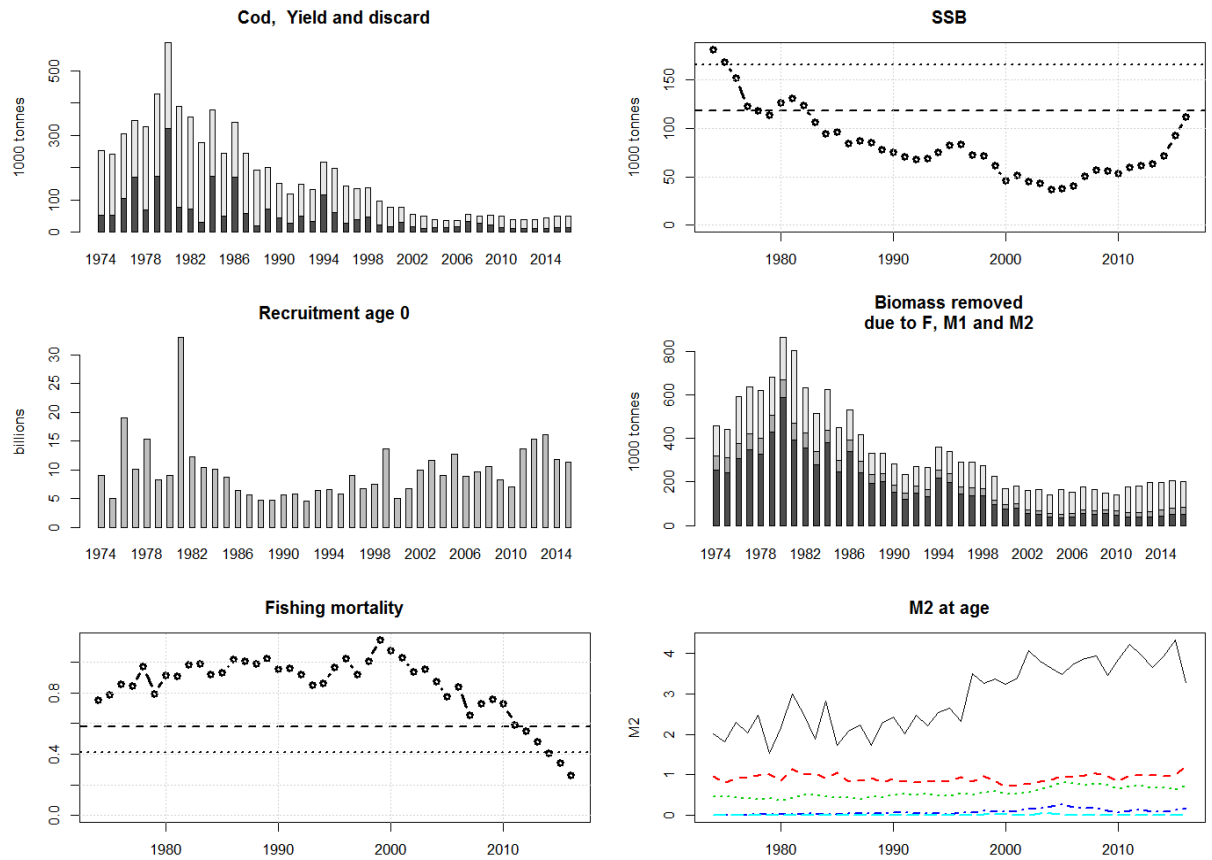


Figure 5.1.9. SMS output for cod. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

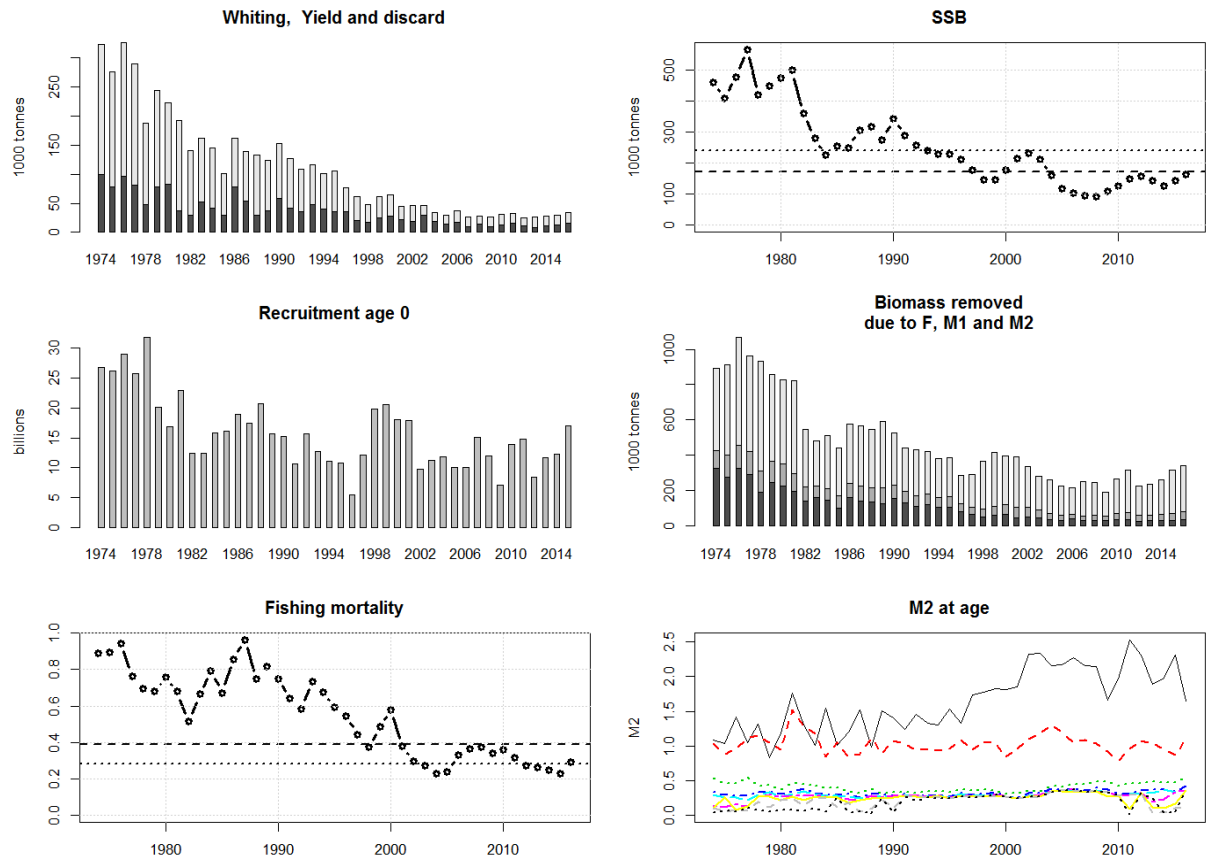


Figure 5.1.10. SMS output for whiting. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

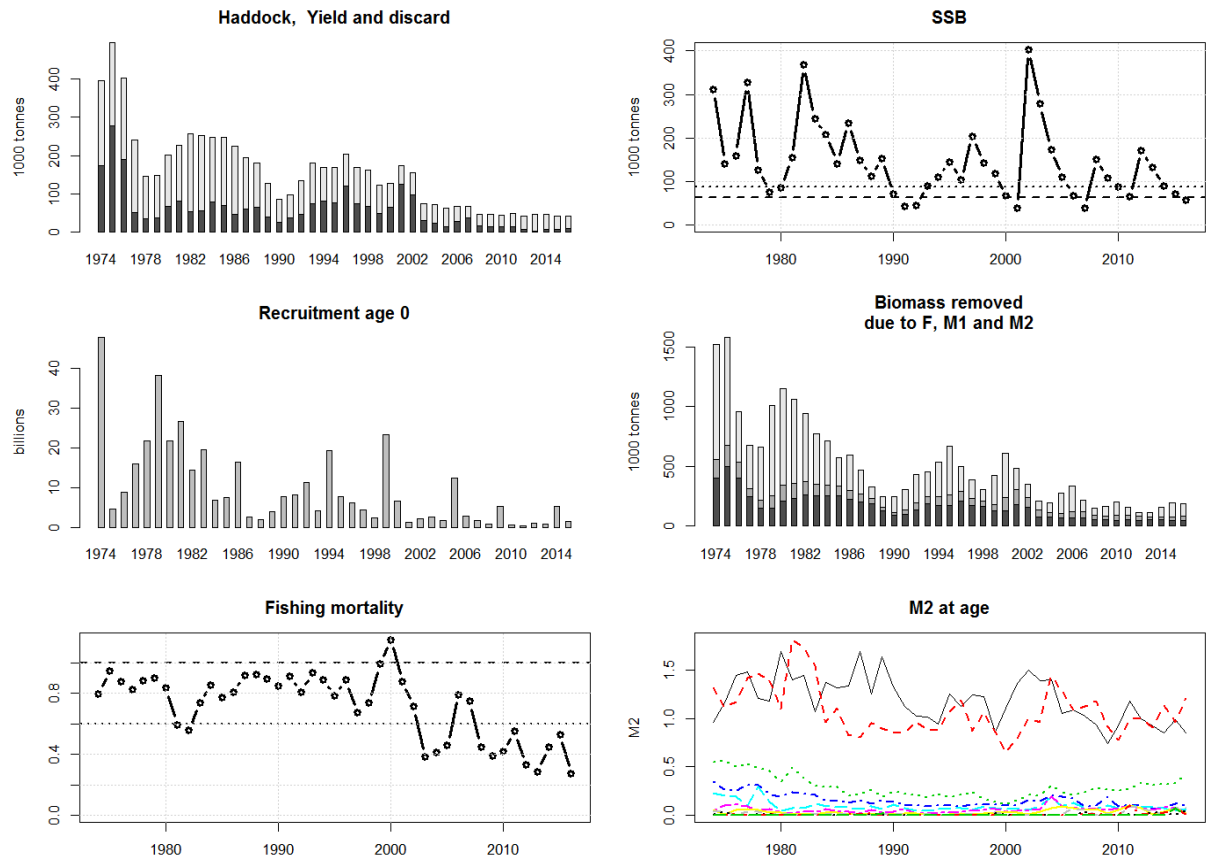


Figure 5.1.11. SMS output for haddock. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

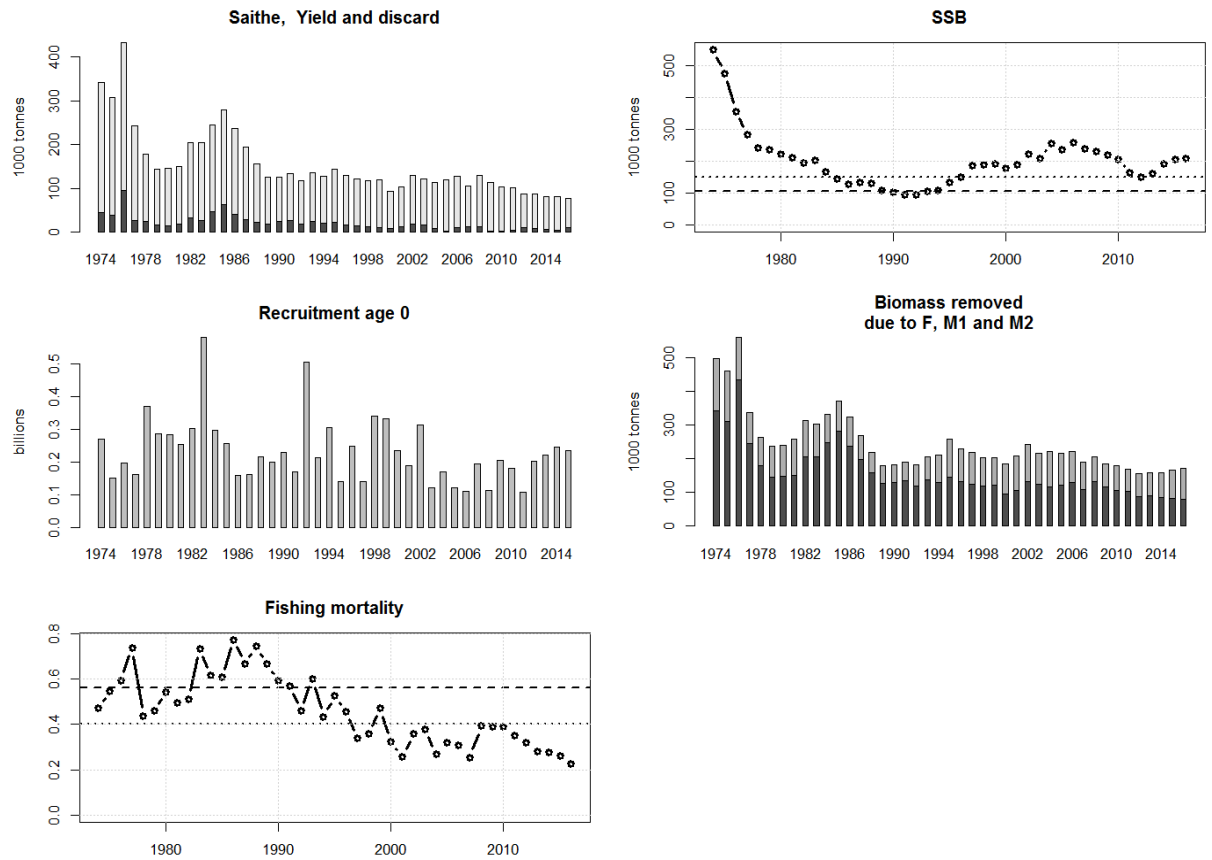


Figure 5.1.12. SMS output for saithe. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB and Biomass removed due to fishery (F).

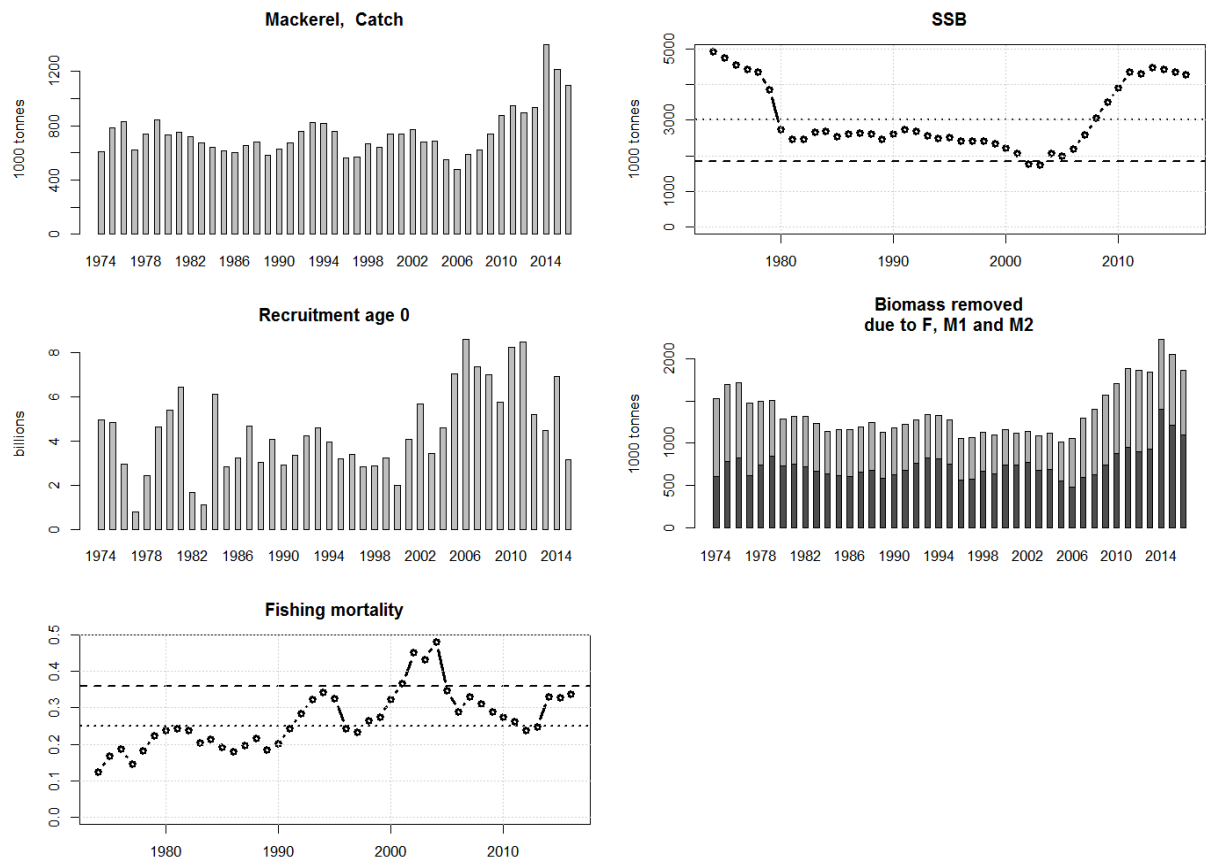


Figure 5.1.13. SMS output for Mackerel. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB and Biomass removed due to fishery (F).

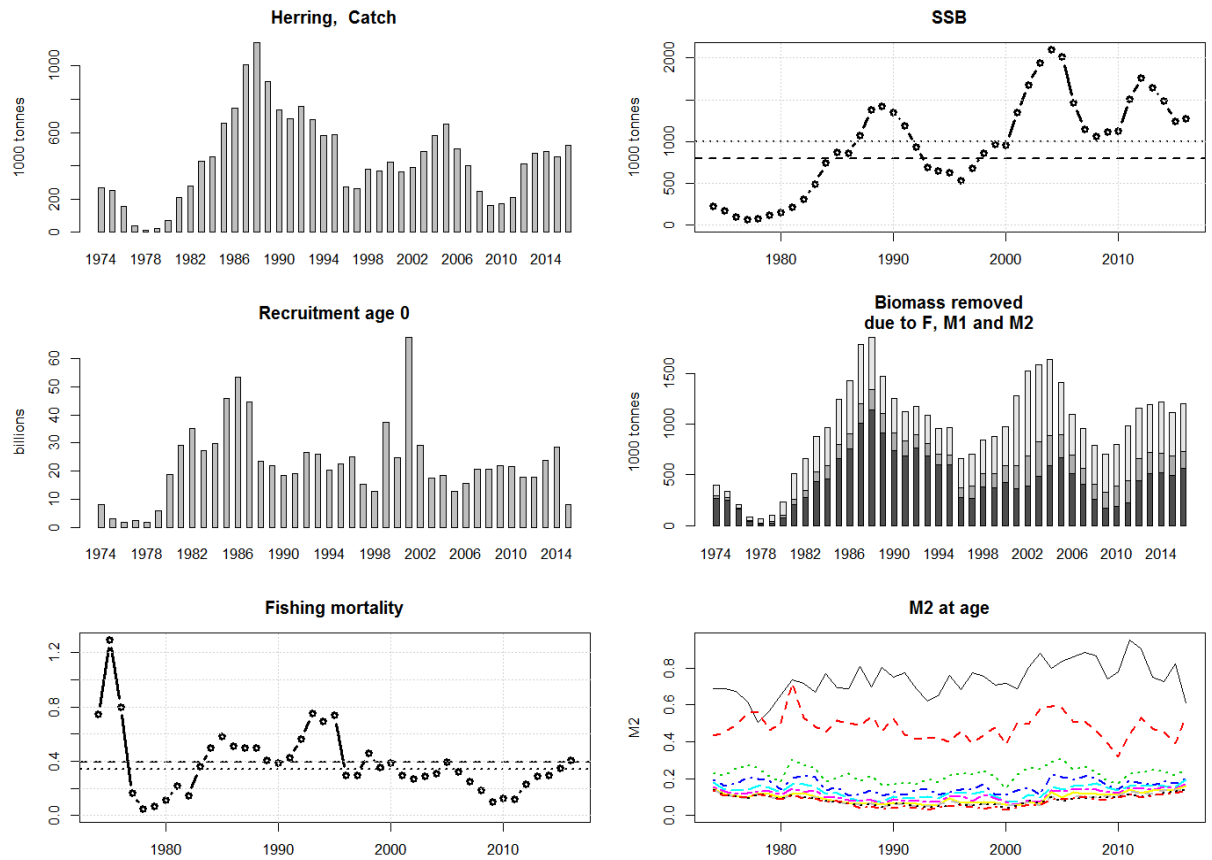


Figure 5.1.14. SMS output for Herring. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

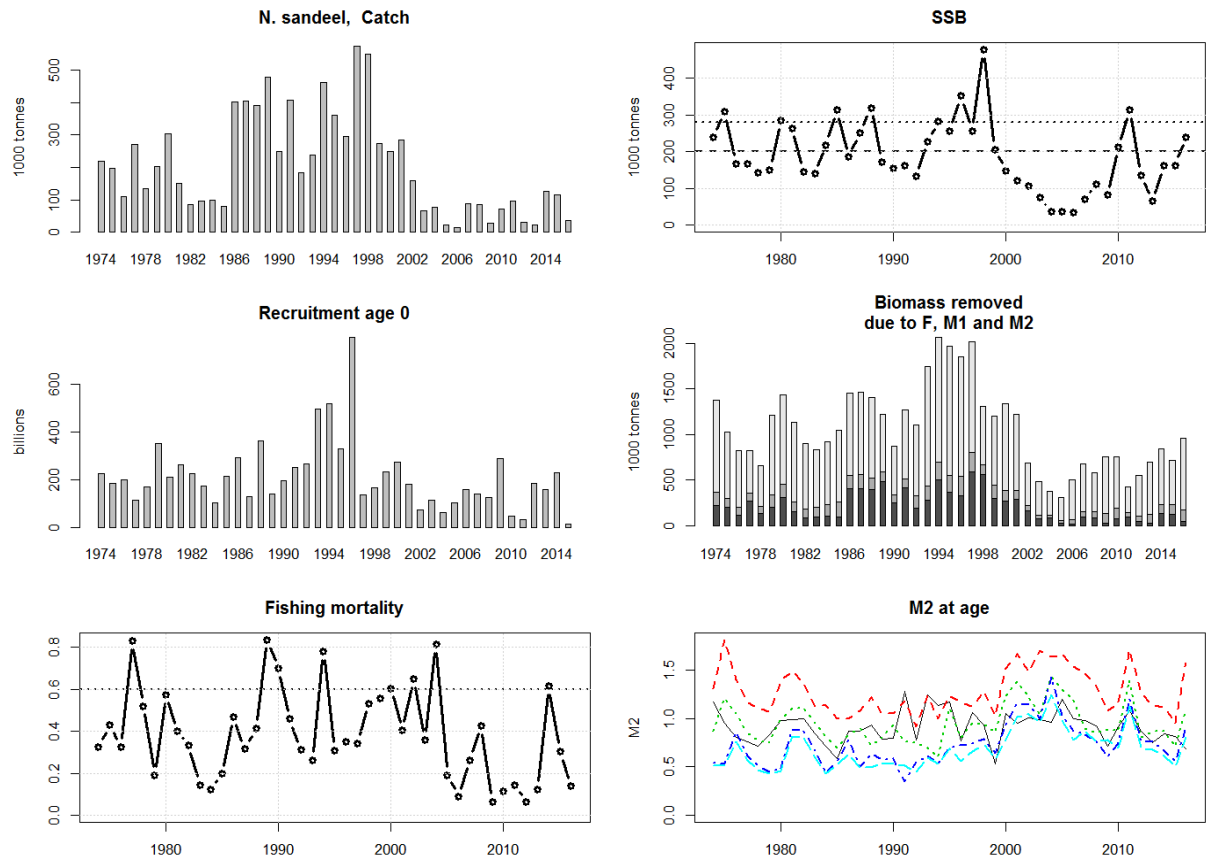


Figure 5.1.15. SMS output for Northern Sandeel. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

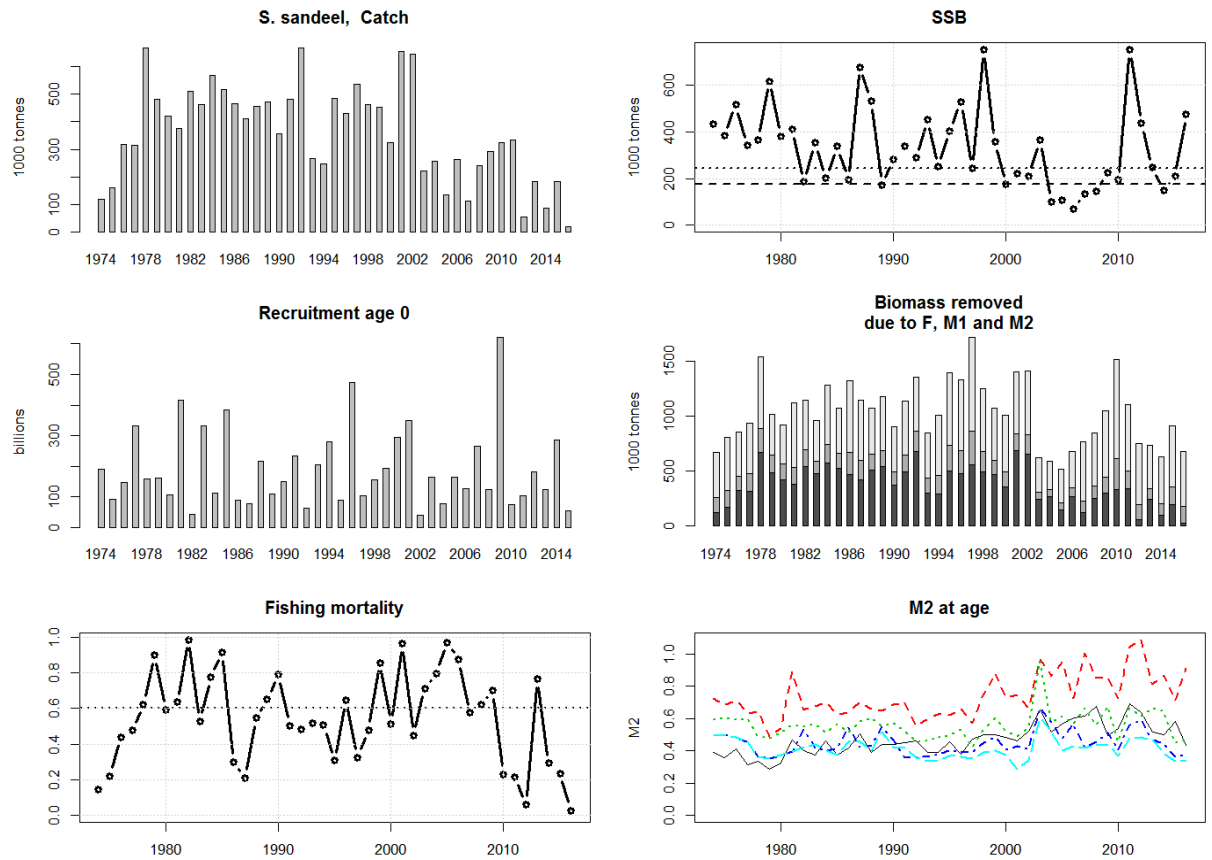


Figure 5.1.16. SMS output for Southern Sandeel. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

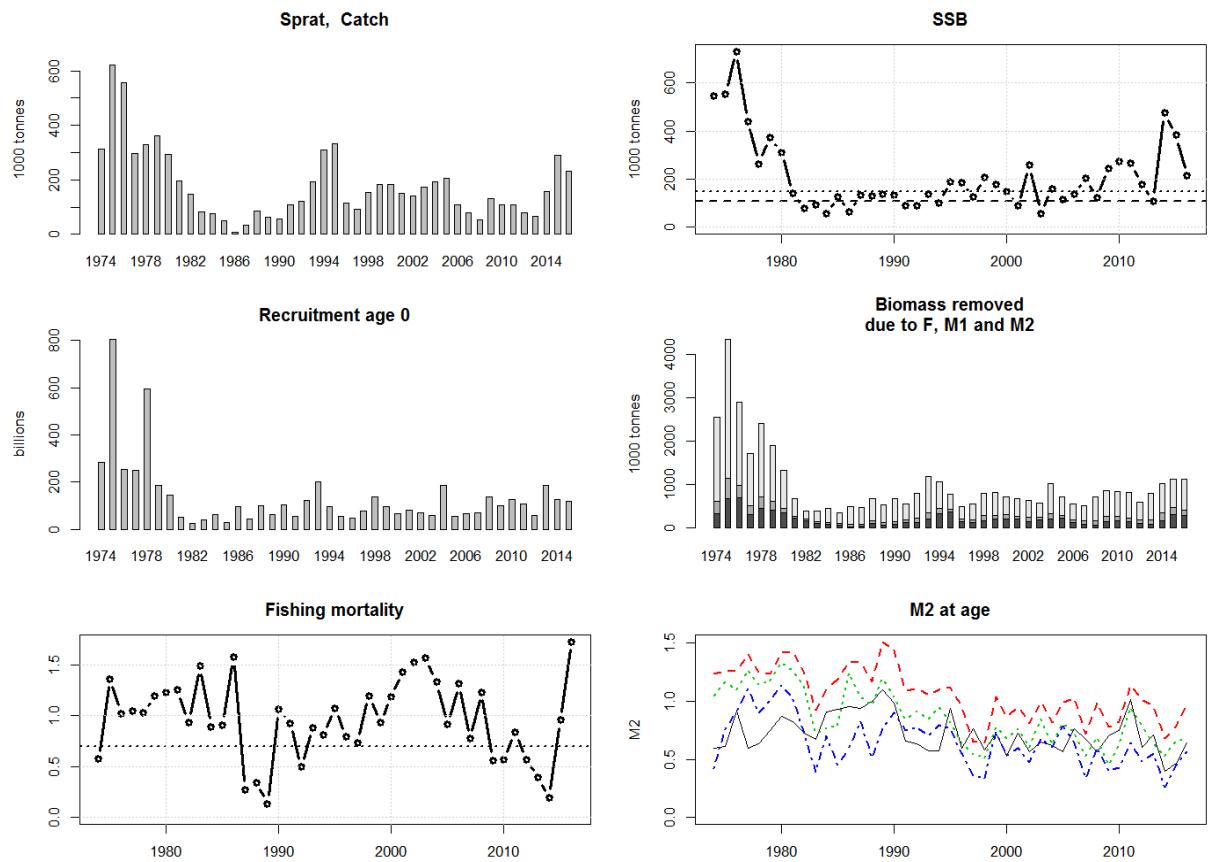


Figure 5.1.17. SMS output for Sprat. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

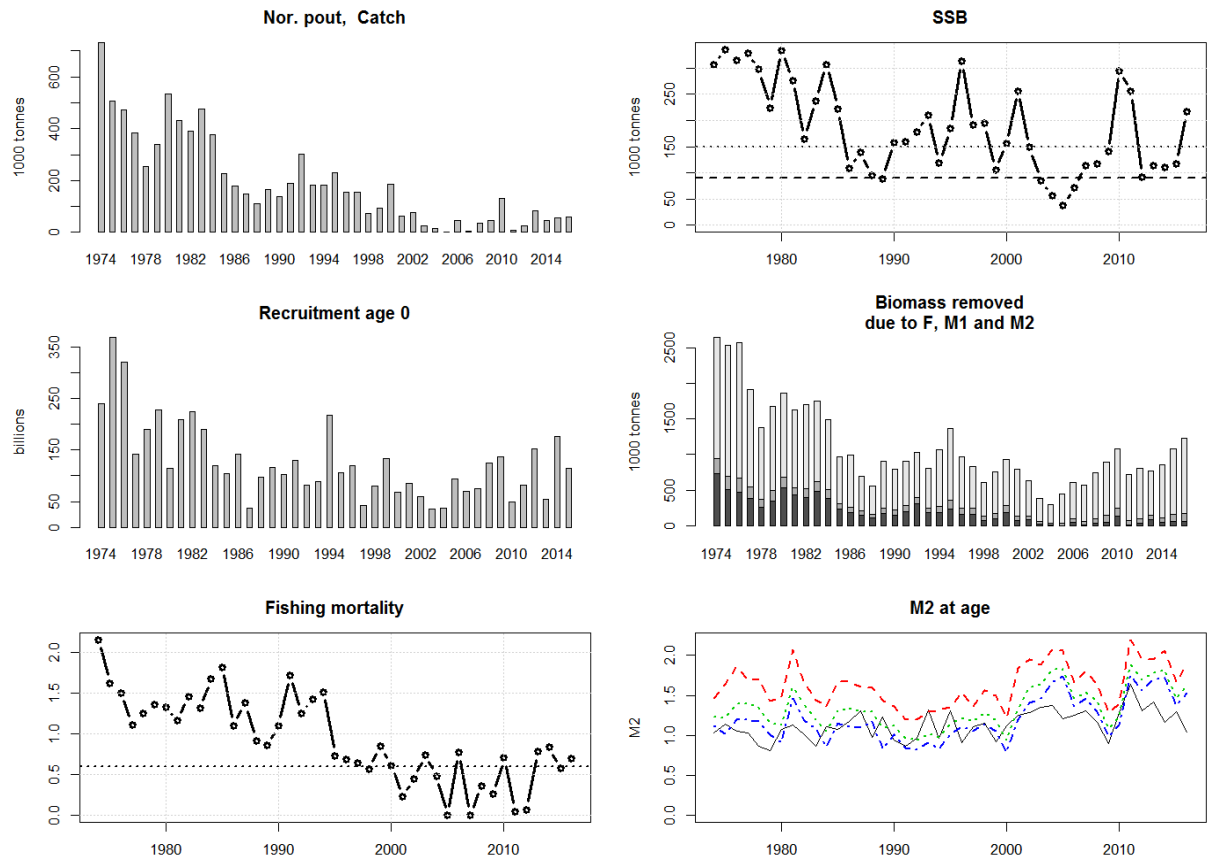


Figure 5.1.18. SMS output for Norway pout. Catch weight divided into yield (landings) and discards, Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

Who eats whom

Eaten biomass by predator

Biomass of eaten SMS prey species biomass decreased from more than 6 billion tons in the mid-seventies to around 3 billion tonnes in recent years (Figure 5.1.19).

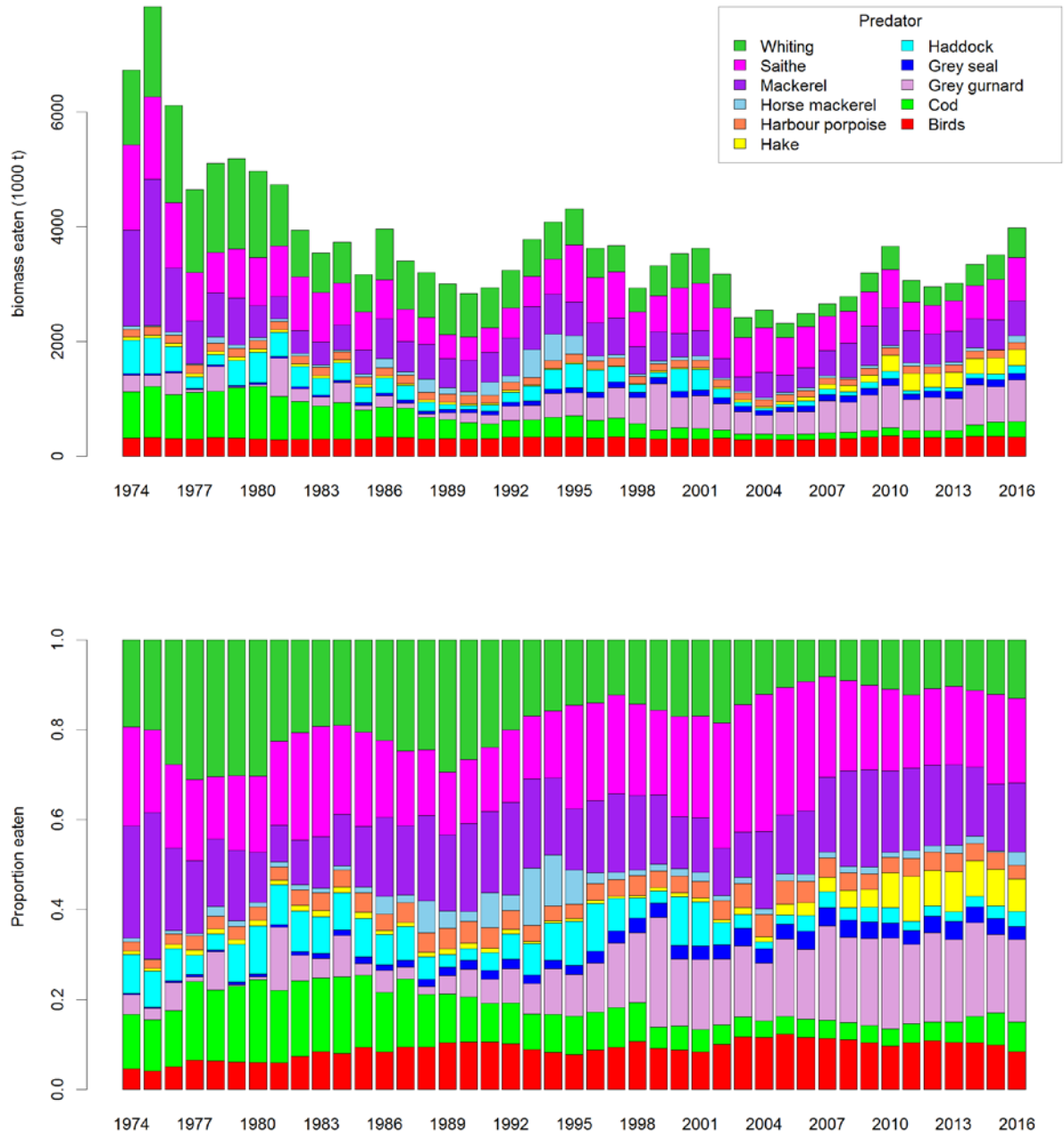


Figure 5.1.19. Eaten total biomass of prey species by individual predator (groups). Upper figure shows the absolute weight eaten and the lower figure shows relative weight eaten.

Eaten biomass by prey

The eaten biomass of the individual SMS prey species (Figure 5.1.20) follows in general the prey stock sizes.

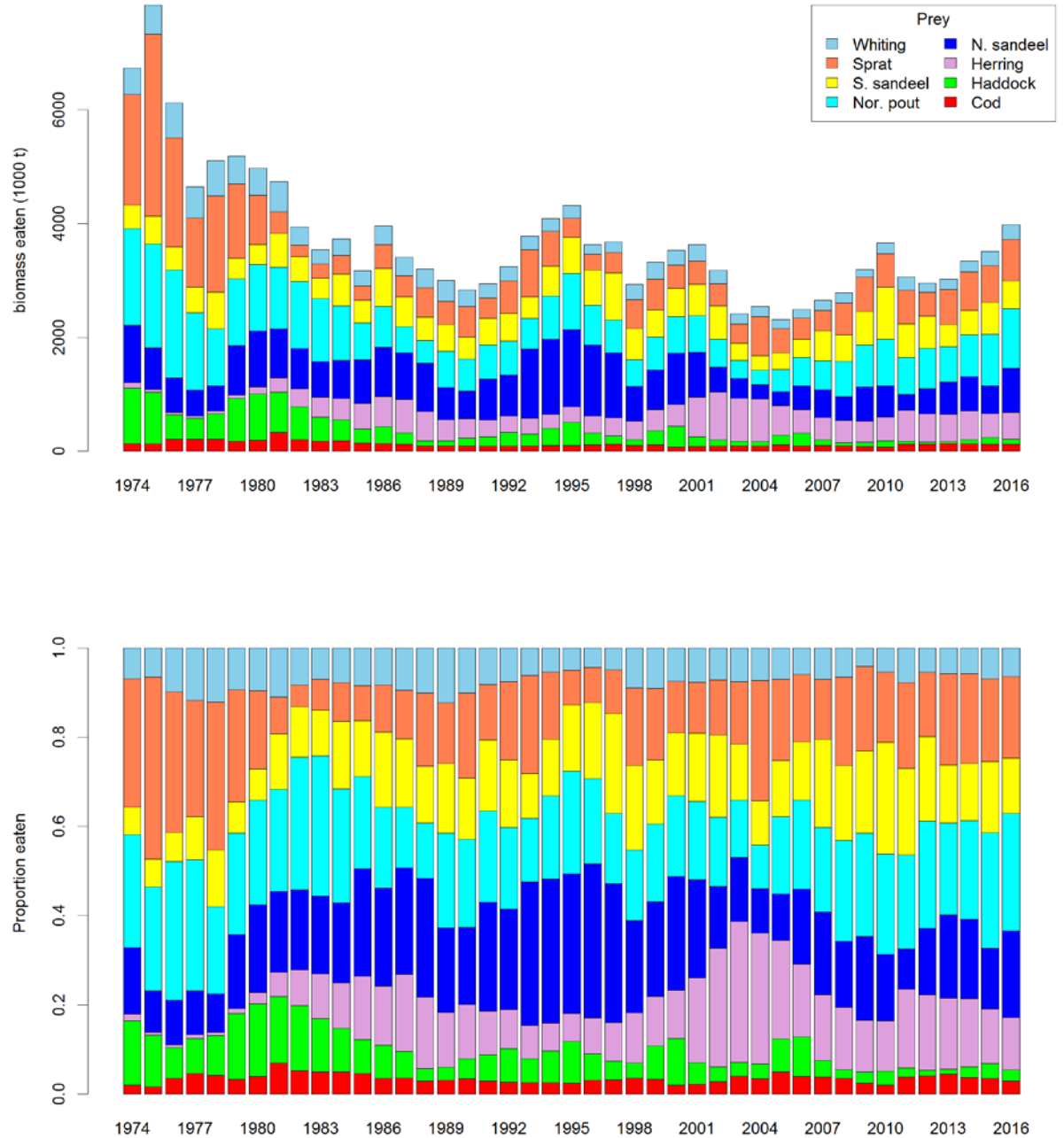


Figure 5.1.20. Eaten biomass of the individual prey species. Upper figure shows the absolute weight eaten and the lower figure shows relative weight eaten.

Eaten biomass by individual prey species

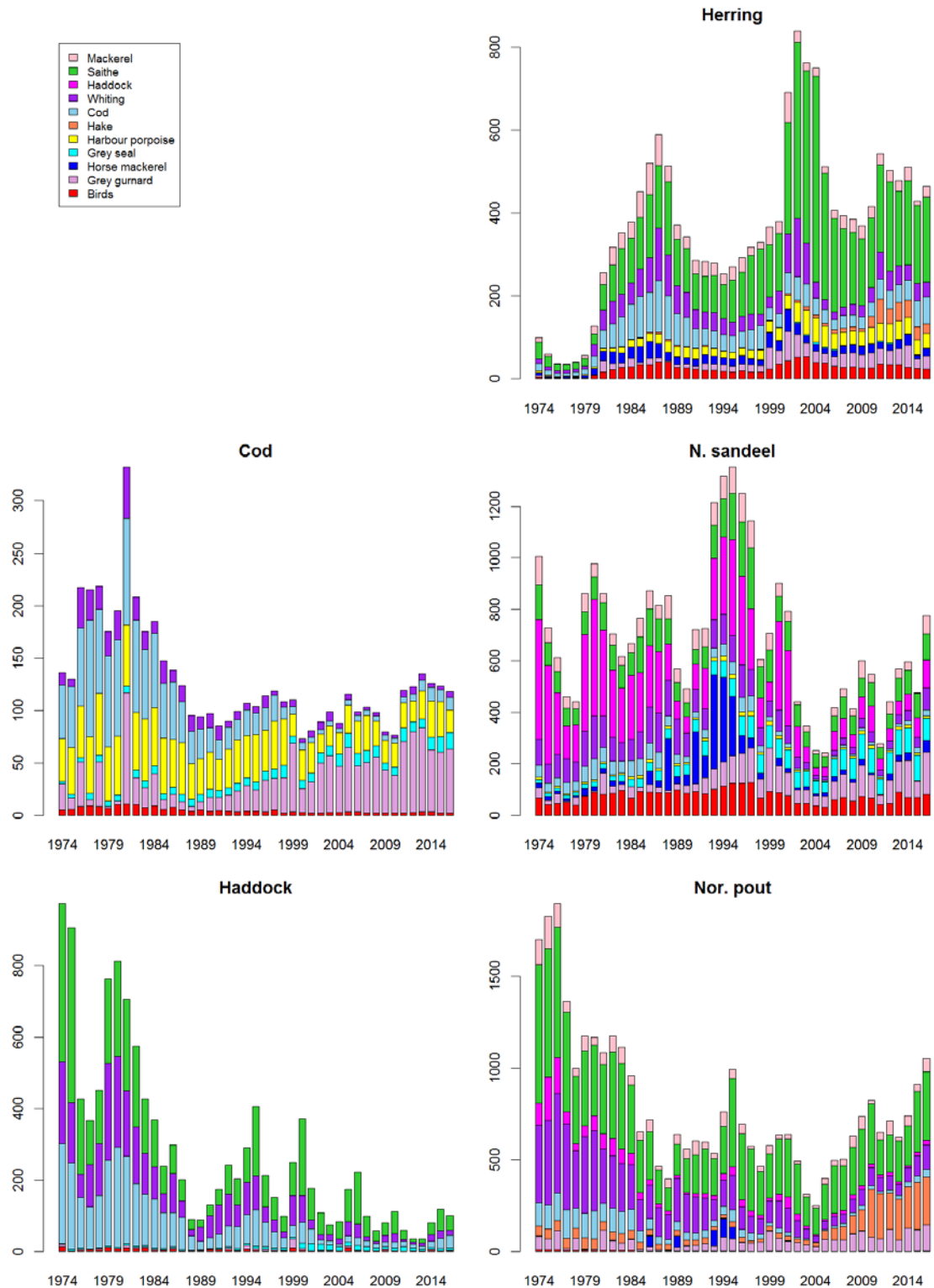


Figure 5.1.21. Eaten biomass of the individual prey species by predator (groups).

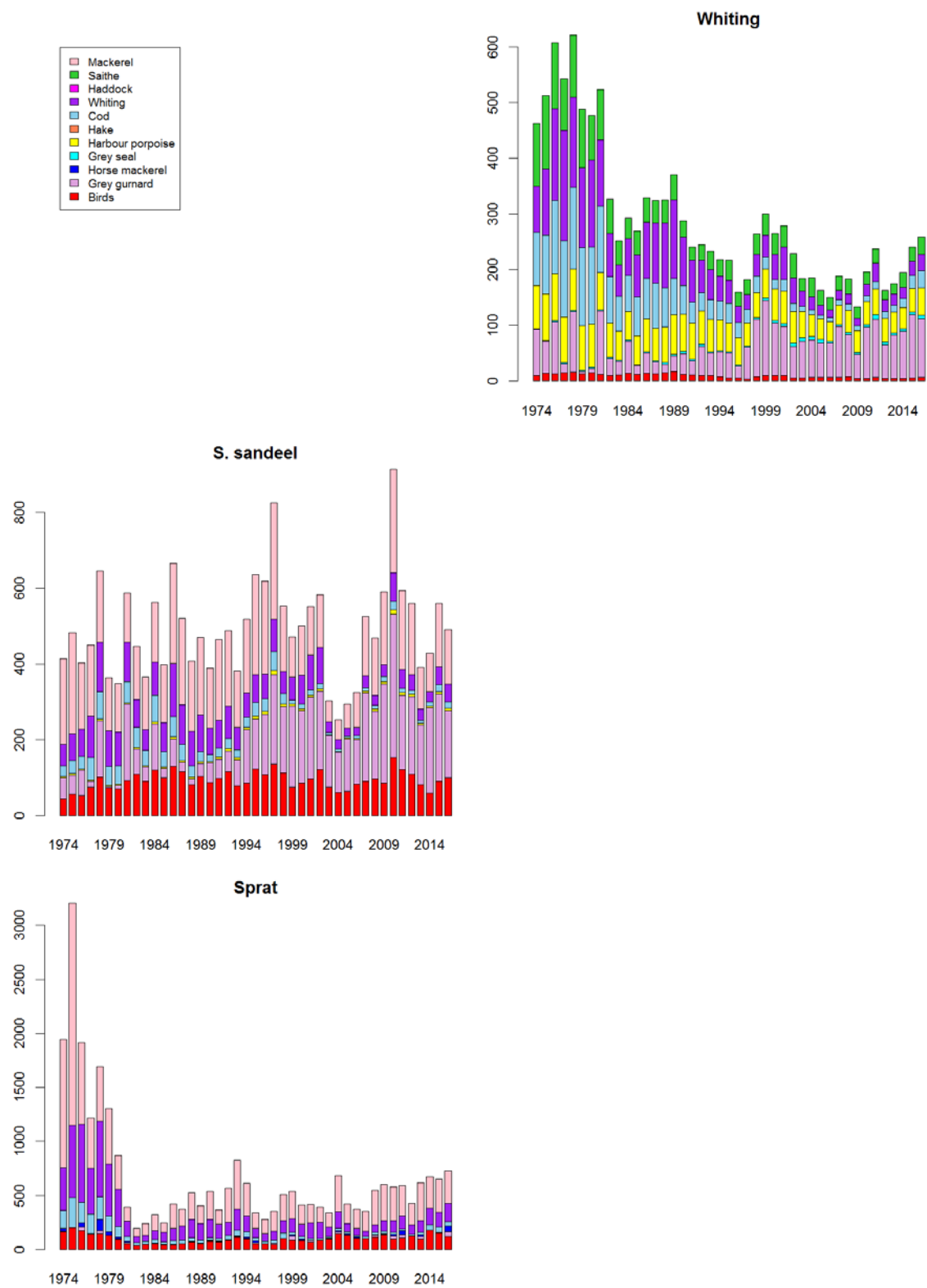


Figure 5.1.21. (continued). Eaten biomass of the individual prey species by predator (grouped for birds and horse mackerel).

Predation mortalities (M2)

The overall picture of M2 at age (Figure 5.1.22) is highly variable between species. For cod and whiting the steep increase in abundance of the predator grey gurnard has led to increase in M2 of 0-group fish in recent years. Further, mortality of 3-year old cod has increased substantially as a result of the recent increase in grey seal abundance. Haddock natural mortality particularly of age 2 fish has decreased over time with the decreased in the biomass of large cod followed by an increase in most recent years. The same trend is seen for 2+herring, but here the effect is counteracted in later years as the biomass of large hake has increased. Similarly, the decrease in herring natural mortality induced by cod is counteracted by an increase in grey gurnard predation.

The two sandeel stocks show markedly different patterns in the main predators, with cod, mackerel, whiting, saithe, seabirds and in later years grey seals all exerting a significant impact on northern sandeel whereas grey gurnards, mackerel, whiting and seabirds are the main predators on southern sandeel. Natural mortality of southern sandeel seems to have increased over the period whereas that of northern sandeel and has fluctuated without a clear trend. Natural mortality of Norway pout increased in the late 1990s whereas the mortality of sprat has decreased more or less monotonically since the mid-1980s.

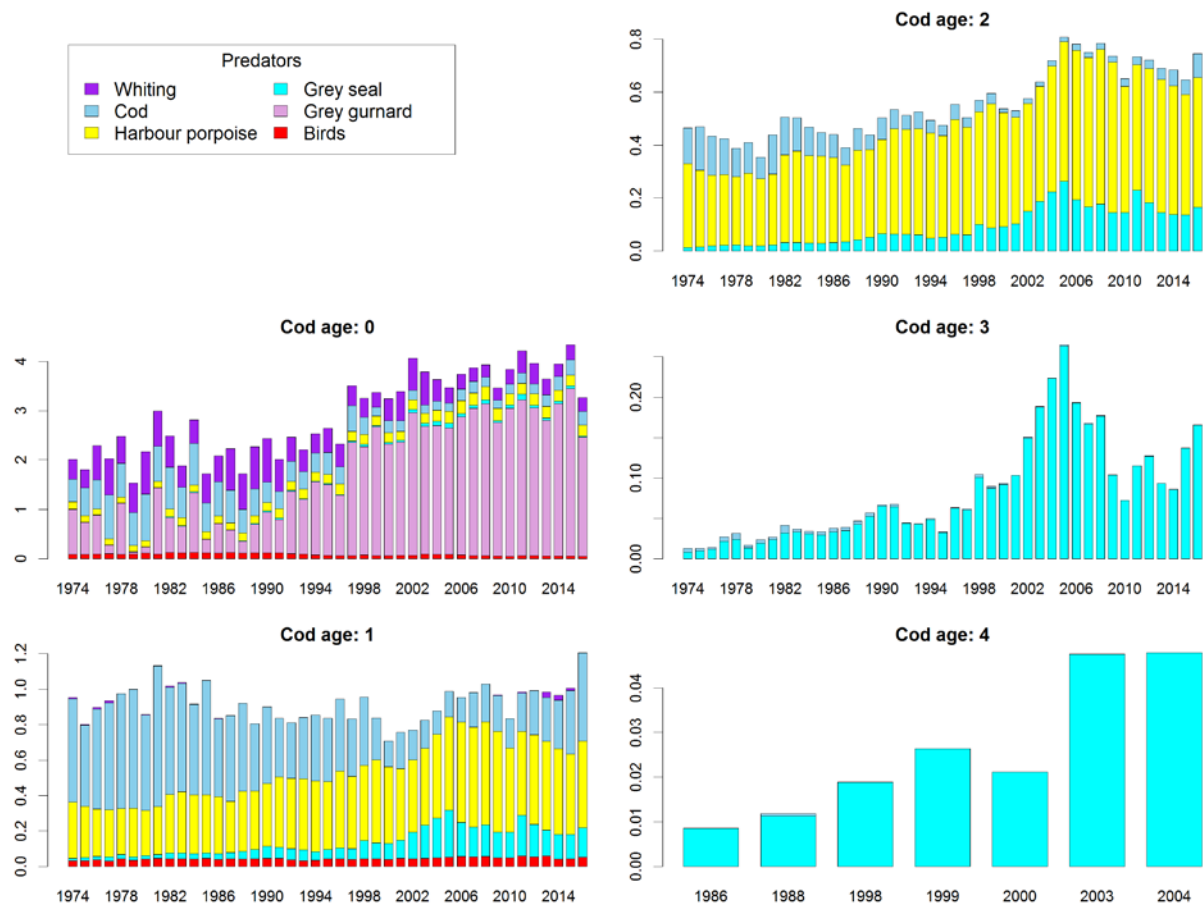


Figure 5.1.22. Annual predation mortality (M2) by prey species and age inflicted by predator species.



Figure 5.1.22. (continued). Annual predation mortality (M2) by prey species and age inflicted by predator species.

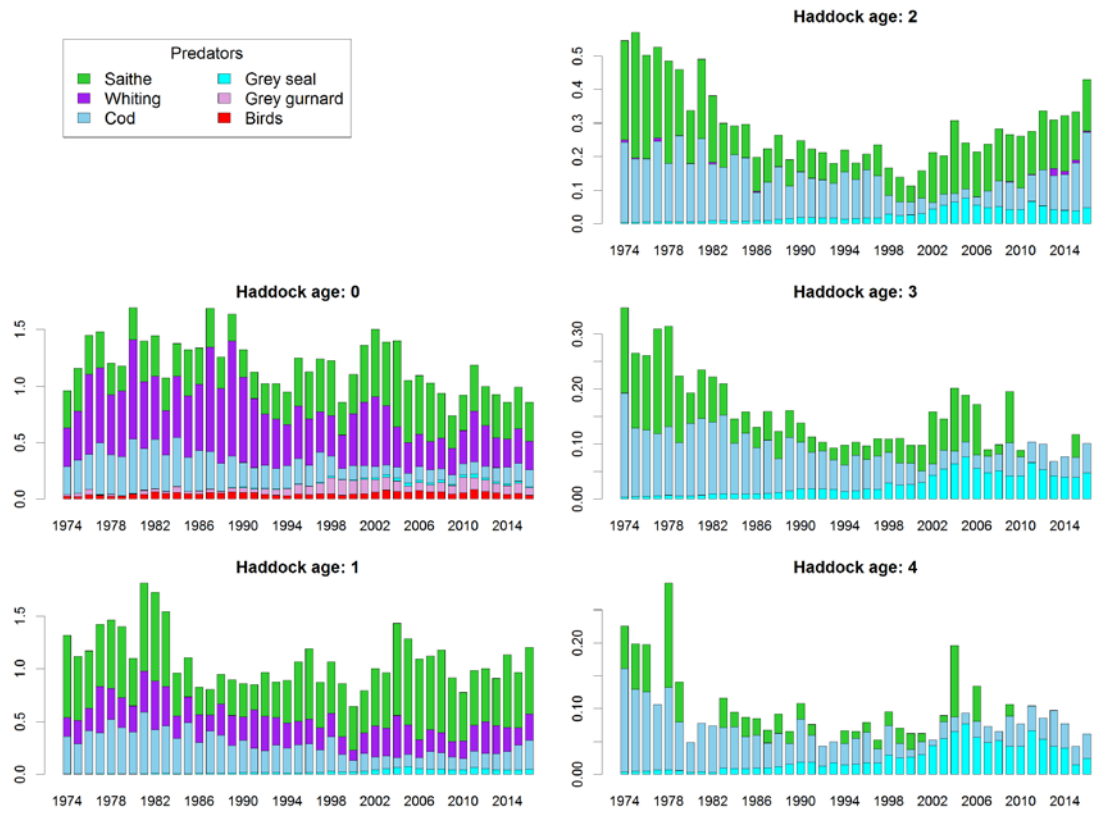


Figure 5.1.22. (continued). Annual predation mortality (M2) by prey species and age inflicted by predator species.

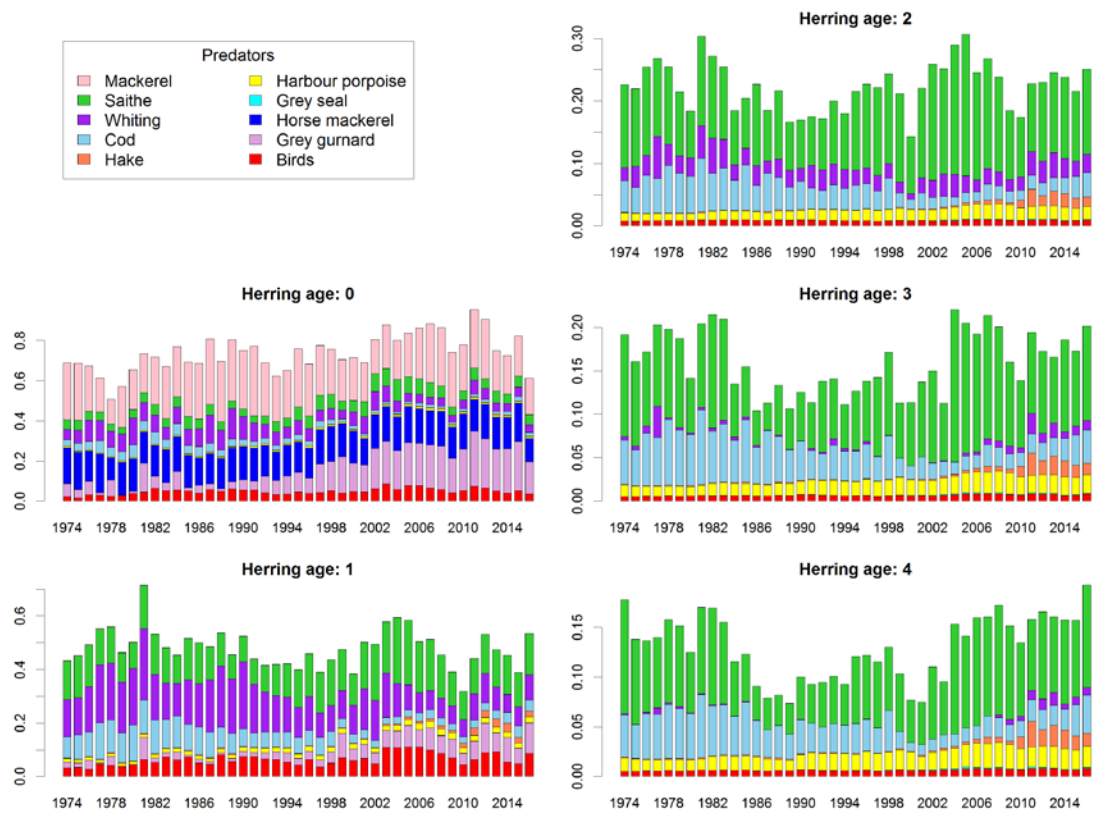


Figure 5.1.22 (continued). Annual predation mortality (M_2) by prey species and age inflicted by predator species.

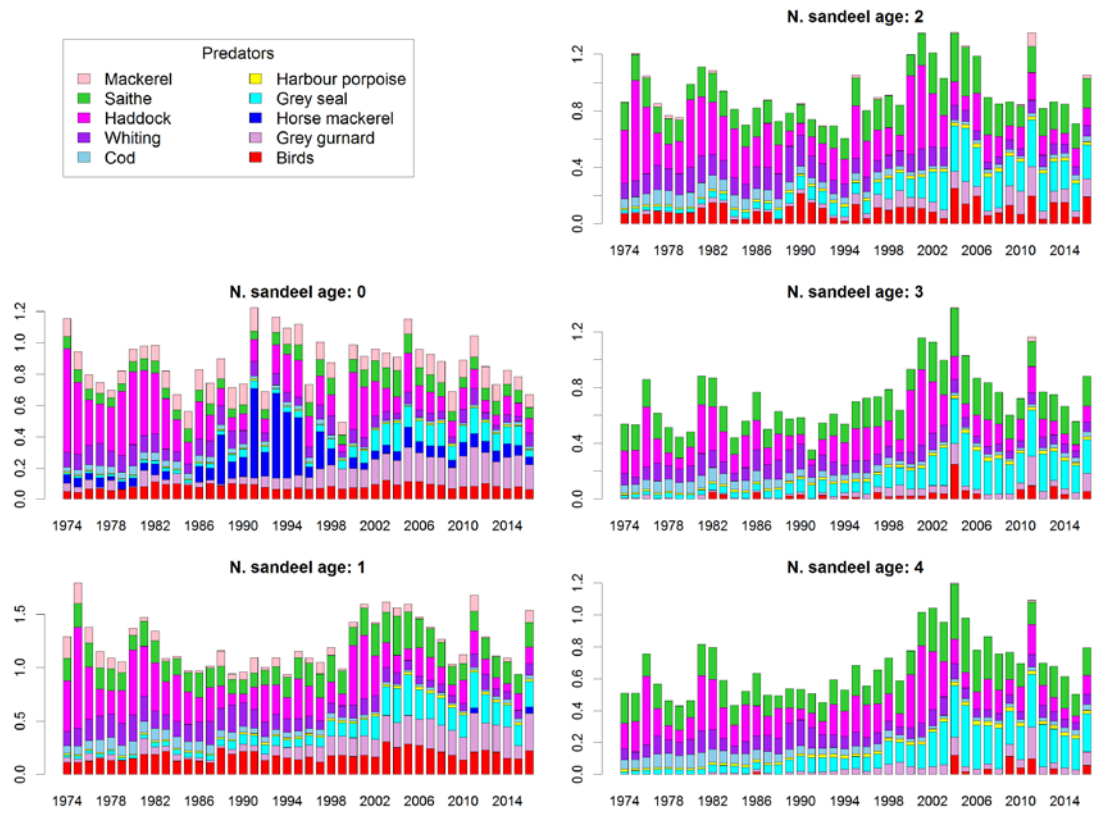


Figure 5.1.22 (continued). Annual predation mortality (M2) by prey species and age inflicted by predator species.

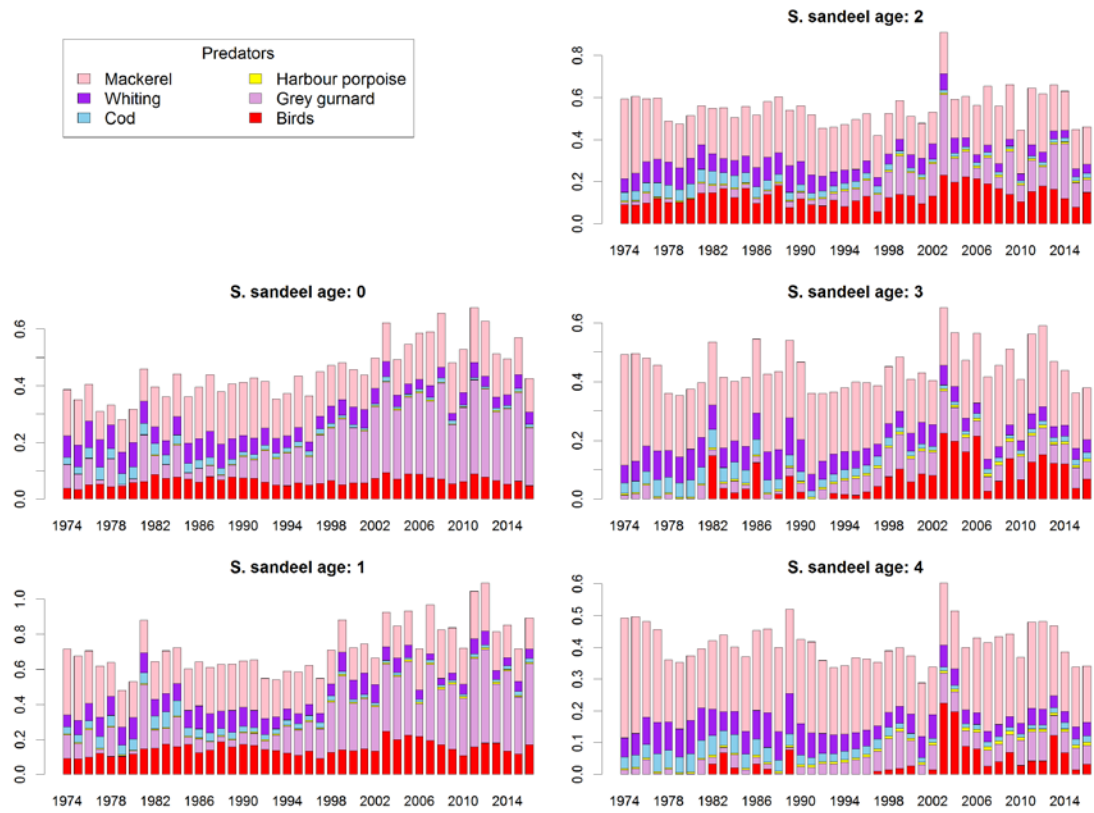


Figure 5.1.22 (continued). Annual predation mortality (M_2) by prey species and age inflicted by predator species.

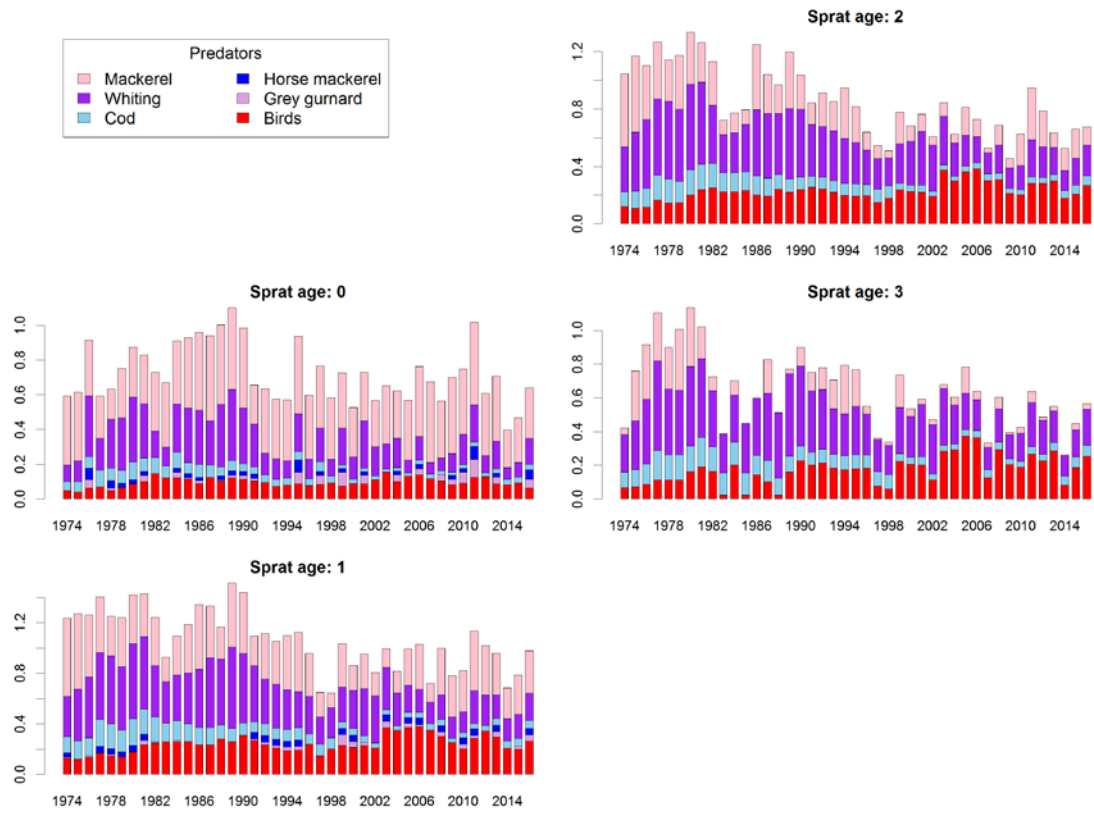


Figure 5.1.22 (continued). Annual predation mortality (M2) by prey species and age inflicted by predator species.

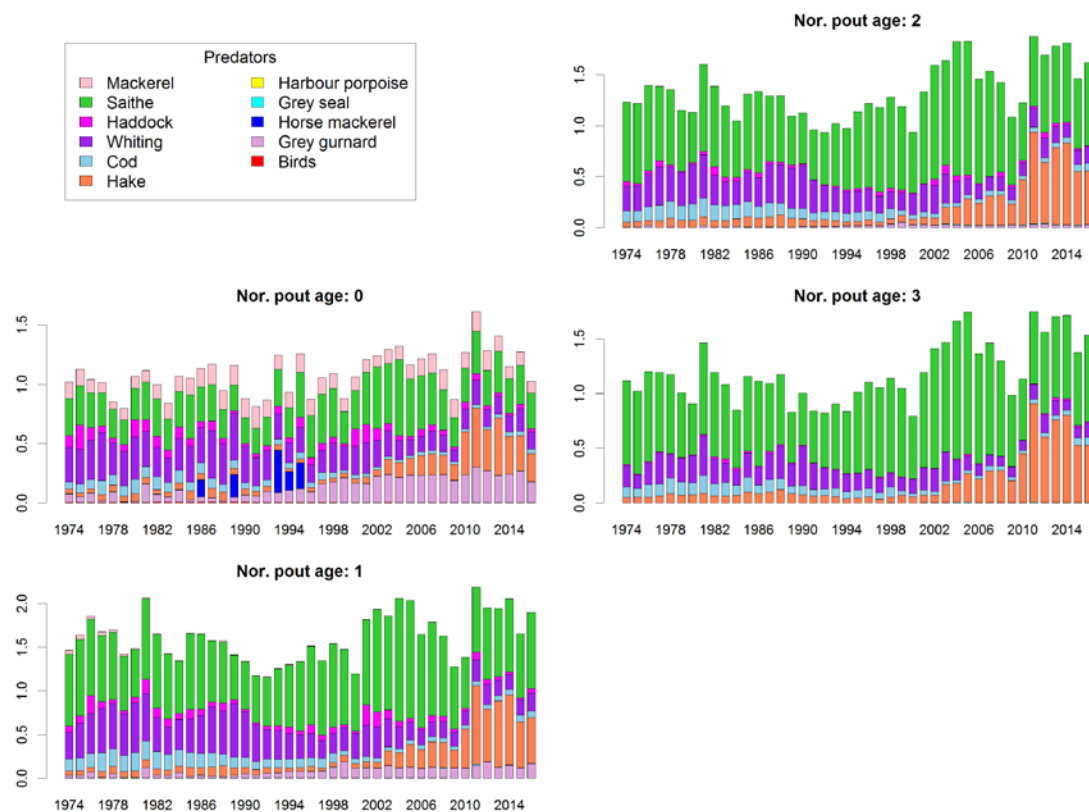


Figure 5.1.22 (continued). Annual predation mortality (M2) by prey species and age inflicted by predator species.

Uncertainties of key output

SMS estimate the uncertainties of selected output variables using the Hessian delta-method approximation. Most variables like stock number and F for dynamic species are estimated within the model, while other variables like the stock numbers of “external predators” are assumed known without errors. This combination of estimated and assumed “known” variables will probably lead to an underestimate of the uncertainties of e.g. predation mortality. This section presents the uncertainties of SSB, mean F , recruitment and $M2$.

Uncertainties of SSB

The uncertainties presented as a Coefficient of Variation (1 standard deviation of the value divided by the value itself) of SSB (Figure 5.1.23) show the highest uncertainties for the prey species Southern. sandeel, Northern sandeel, sprat and Norway pout. The uncertainties for mackerel and for saithe seem too low, probably because of the use of stock numbers from the ICES assessment as artificial survey indices in SMS (see sec 2.1.6.2). A higher CV on the artificial indices should probably have been used to better reflect the uncertainties in the SMS assessment!

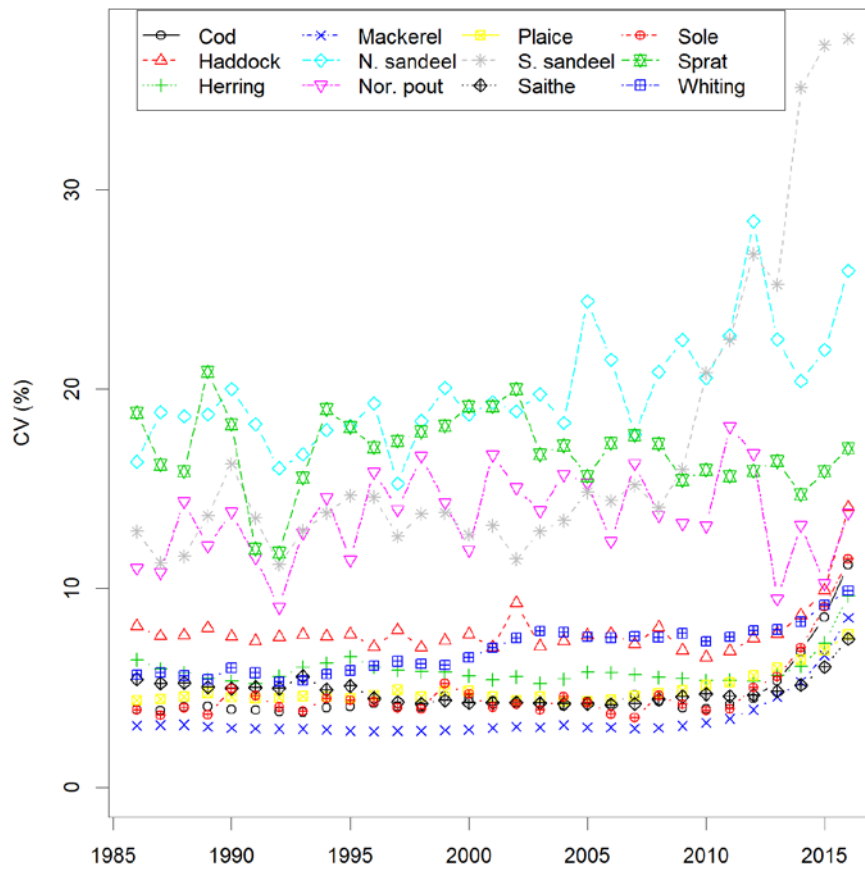


Figure 5.1.23. Uncertainties (1 sd / value) of estimated SSB as estimated by SMS.

Uncertainties of mean F

The uncertainties of mean F show a similar pattern as for SSB with the highest CVs are estimated for the prey species. F has been close to zero for some years for herring which gives a very high CV in some years. For Norway pout, catches are set to zero for a few years, which result in a low (0) CV.

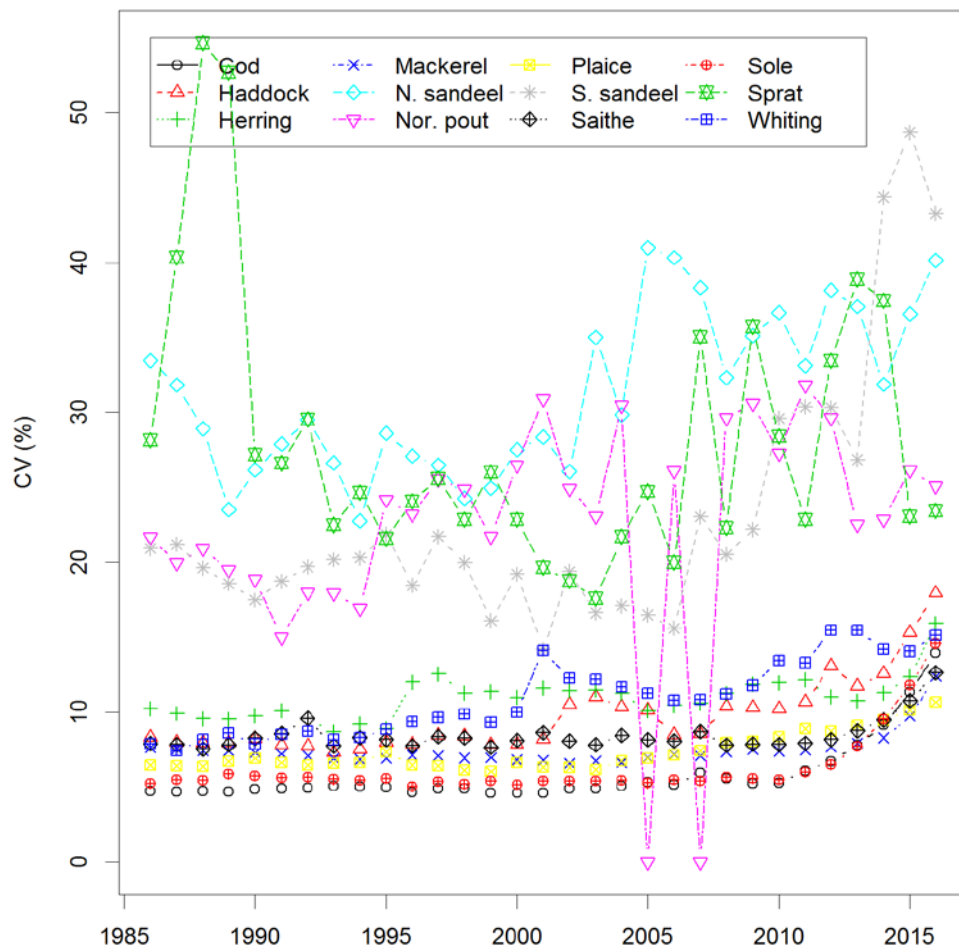


Figure 5.1.24. Uncertainties (1 sd / value) of estimated mean F as estimated by SMS.

Uncertainties of recruitment

The uncertainties of recruitment are very high (>50%) for the most recent years (Figure 5.1.25 left panel). Further back in time, the CV is highest for cod, the two sandeel stocks, sprat and whiting. For mackerel and saithe the CV is too low as for SSB.

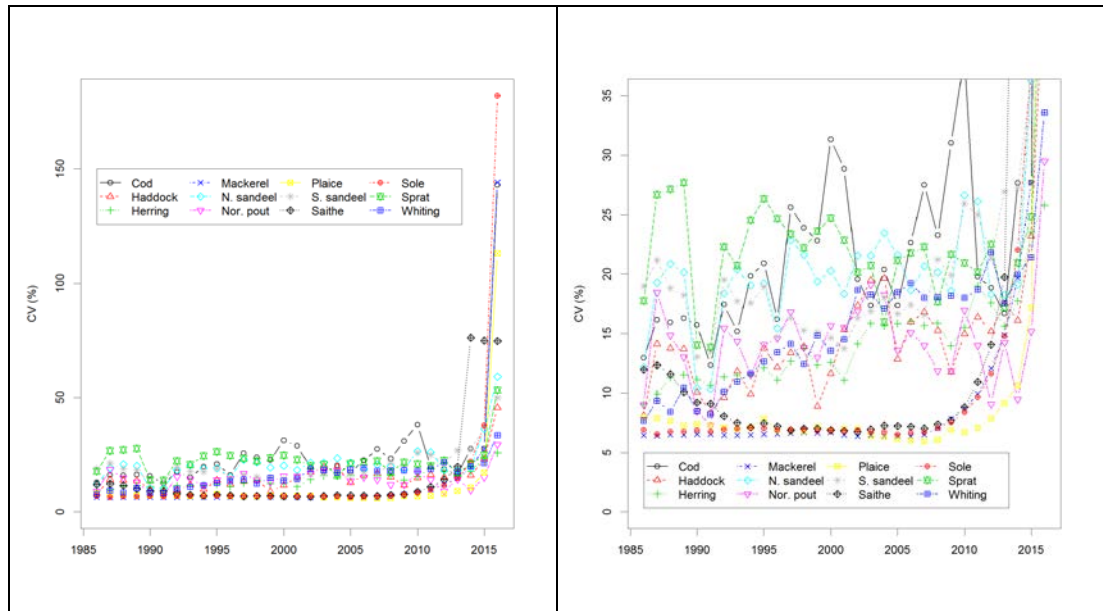


Figure 5.1.25. Uncertainties (1 sd / value) of estimated recruitment as estimated by SMS. Left panel show the full range of uncertainties and the right panel show uncertainties up to 35 %.

Uncertainties of Predation mortality(M2)



Figure 5.1.26. Uncertainties (1 sd / value) of estimated predation mortality (sum of quarterly M2) as estimated by SMS.

The CVs of M2 are typically in the range 5–10% (Figure 5.1.26), which is in the same range as CV of mean F for the predator species (Figure 5.1.24) and CV of M2 is below the CV of mean F for prey species. For age 0 the CV of M2 increases significantly, due to the uncertainty on recruitment in the most recent year. CV is lowest for all ages for the species Norway pout and northern sandeel, which might be due to the (too) low uncertainty on abundance of their main predators saithe and mackerel. Saithe is also a main predator on herring, but the CV on herring M2 is relatively high for all ages. CV of M2 is relatively high for cod ages 1 and 2. It is mainly cod itself, with a low uncertainty on stock abundance of older cod (SSB, Figure 5.1.23) and marine mammals, with stock abundance given as input, that predate on cod ages 1–2. The CV on M2 seems therefore mainly to arise from high uncertainties on the model parameters for predation from marine mammals and older cod.

Uncertainties presented as CV may give a biased impression for low values (of the “mean”). Figure 5.1.27 to Figure 5.1.29 show the estimated M2 values for ages 0–2, with added lines for plus-minus 2 times the standard deviation. The overall picture is that the annual M2 values are statistically different for both examples of M2 without no temporal trend (e.g. ages 0–1 for Norway pout) and examples with a trend (e.g. cod age 0 and age 2.)

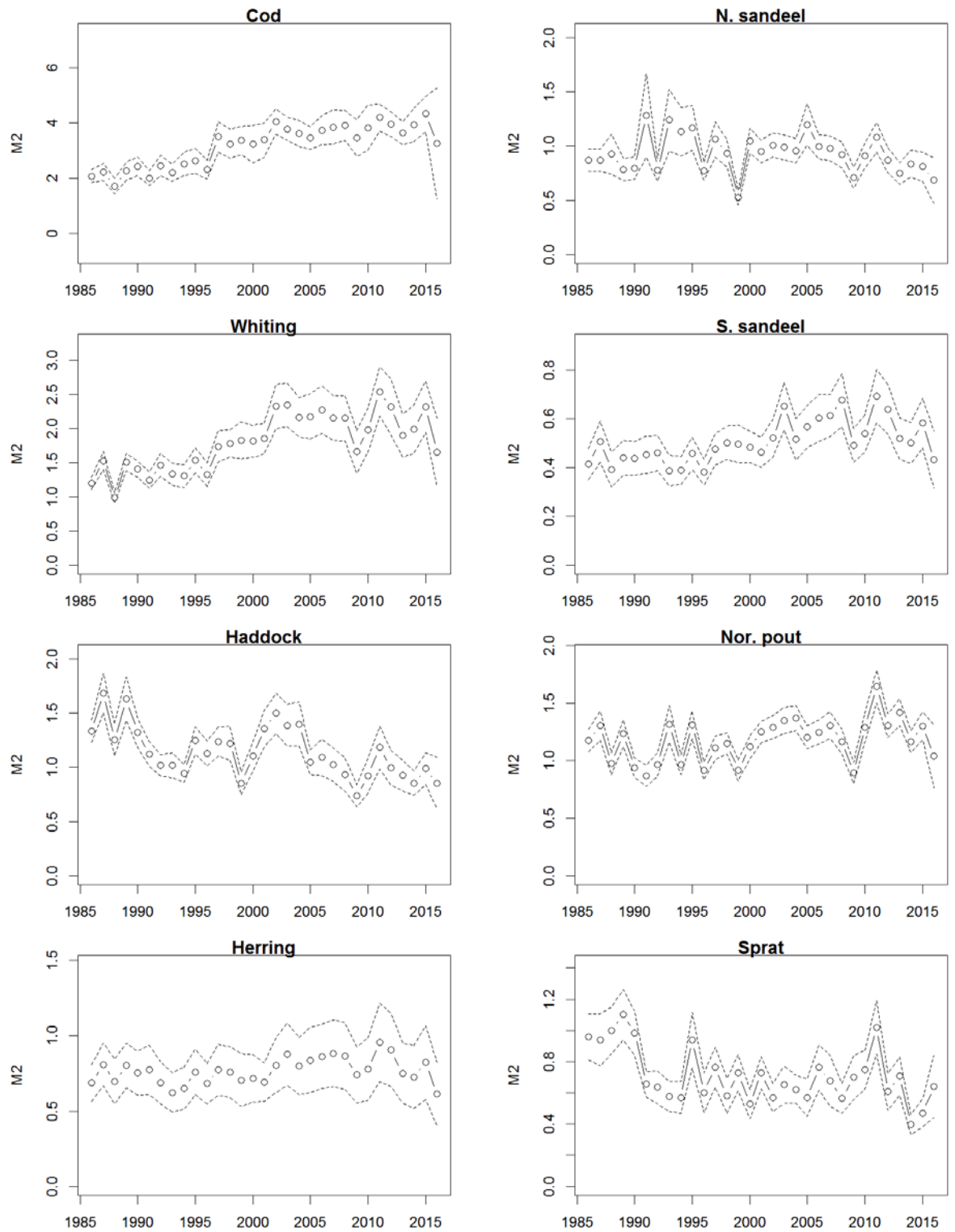


Figure 5.1.27. M2 value with plus-minus 2 times the standard deviation as estimated by SMS.

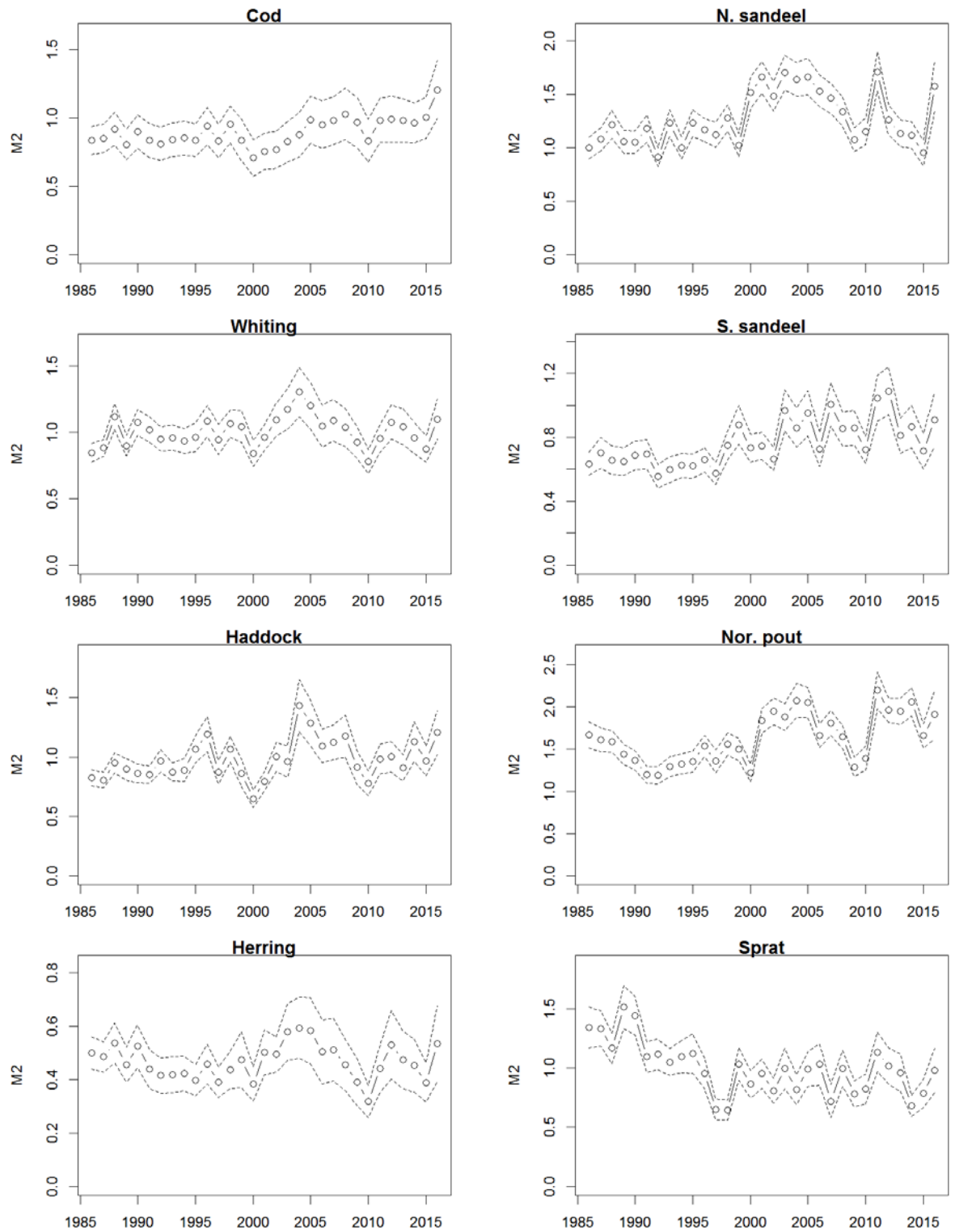


Figure 5.1.28. M2 value with plus-minus 2 times the standard deviation as estimated by SMS.

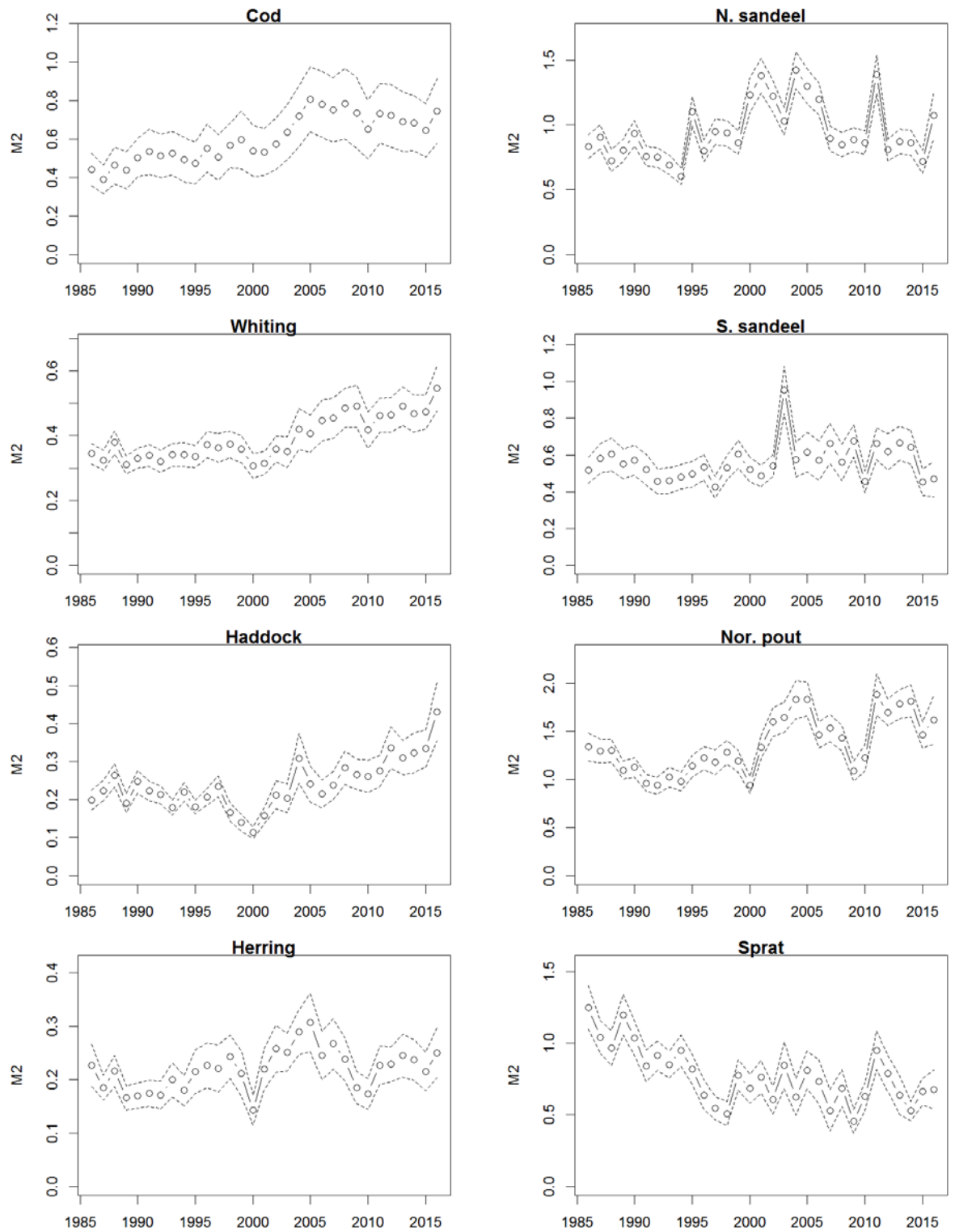


Figure 5.1.29. M2 value with plus-minus 2 times the standard deviation as estimated by SMS.

Whiting : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3	4	5	6	7	8
1974	1.180	1.231	0.733	0.534	0.499	0.347	0.307	0.307	0.244
1975	1.130	1.083	0.662	0.491	0.462	0.317	0.462	0.259	0.266
1976	1.518	1.161	0.659	0.477	0.462	0.349	0.278	0.278	0.249
1977	1.154	1.316	0.750	0.483	0.413	0.335	0.335	0.264	0.299
1978	1.418	1.348	0.620	0.539	0.497	0.468	0.468	0.399	0.271
1979	0.931	1.250	0.639	0.537	0.534	0.491	0.465	0.316	0.255
1980	1.295	1.143	0.577	0.504	0.464	0.464	0.422	0.422	0.275
1981	1.865	1.724	0.670	0.548	0.509	0.486	0.468	0.448	0.279
1982	1.404	1.497	0.654	0.583	0.542	0.481	0.418	0.336	0.264
1983	1.113	1.382	0.622	0.523	0.505	0.482	0.474	0.474	0.299
1984	1.650	1.047	0.592	0.501	0.477	0.475	0.457	0.446	0.252
1985	1.107	1.244	0.599	0.497	0.477	0.460	0.452	0.321	0.452
1986	1.298	1.044	0.544	0.502	0.468	0.435	0.383	0.383	0.241
1987	1.627	1.080	0.524	0.456	0.440	0.425	0.421	0.255	0.255
1988	1.086	1.314	0.579	0.527	0.486	0.478	0.439	0.292	0.228
1989	1.604	1.097	0.510	0.492	0.468	0.461	0.446	0.432	0.446
1990	1.510	1.272	0.529	0.486	0.482	0.482	0.452	0.303	0.242
1991	1.342	1.219	0.539	0.509	0.496	0.482	0.482	0.472	0.467
1992	1.562	1.149	0.521	0.487	0.481	0.480	0.473	0.481	0.404
1993	1.429	1.159	0.541	0.480	0.472	0.471	0.462	0.462	0.462
1994	1.402	1.132	0.541	0.499	0.477	0.477	0.477	0.457	0.448
1995	1.631	1.161	0.535	0.472	0.456	0.456	0.449	0.449	0.443
1996	1.426	1.283	0.572	0.518	0.511	0.478	0.478	0.469	0.469
1997	1.837	1.145	0.562	0.499	0.489	0.477	0.464	0.464	0.463
1998	1.878	1.265	0.574	0.507	0.488	0.474	0.469	0.469	0.469
1999	1.924	1.241	0.559	0.534	0.500	0.493	0.483	0.483	0.493
2000	1.910	1.040	0.507	0.469	0.466	0.466	0.466	0.466	0.466
2001	1.948	1.159	0.515	0.460	0.447	0.447	0.442	0.447	0.447
2002	2.422	1.294	0.559	0.520	0.489	0.470	0.465	0.465	0.465

2003	2.438	1.374	0.550	0.524	0.493	0.490	0.465	0.462	0.465
2004	2.263	1.501	0.620	0.587	0.549	0.549	0.549	0.549	0.530
2005	2.273	1.399	0.607	0.564	0.556	0.554	0.552	0.554	0.556
2006	2.372	1.245	0.646	0.584	0.568	0.566	0.528	0.562	0.566
2007	2.253	1.290	0.654	0.563	0.530	0.530	0.535	0.535	0.530
2008	2.249	1.235	0.686	0.595	0.556	0.541	0.541	0.556	0.547
2009	1.757	1.122	0.691	0.571	0.539	0.539	0.465	0.539	0.539
2010	2.074	0.978	0.617	0.513	0.487	0.483	0.487	0.483	0.487
2011	2.635	1.154	0.663	0.514	0.507	0.502	0.297	0.502	0.221
2012	2.414	1.275	0.664	0.562	0.527	0.527	0.527	0.519	0.519
2013	1.993	1.241	0.691	0.571	0.520	0.389	0.319	0.238	0.444
2014	2.086	1.156	0.668	0.571	0.571	0.435	0.306	0.234	0.234
2015	2.417	1.071	0.673	0.551	0.532	0.532	0.368	0.307	0.241
2016	1.751	1.297	0.746	0.620	0.563	0.559	0.559	0.339	0.559

Haddock : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3	4	5	6	7	8	9	10
1974	1.059	1.519	0.746	0.548	0.426	0.244	0.254	0.244	0.214	0.200	0.200
1975	1.258	1.320	0.771	0.466	0.398	0.301	0.214	0.238	0.238	0.206	0.200
1976	1.547	1.371	0.703	0.461	0.397	0.313	0.260	0.202	0.204	0.220	0.200
1977	1.578	1.621	0.727	0.509	0.306	0.294	0.270	0.237	0.201	0.200	0.203
1978	1.304	1.665	0.685	0.514	0.491	0.257	0.256	0.235	0.212	0.200	0.200
1979	1.276	1.599	0.660	0.424	0.340	0.263	0.231	0.213	0.213	0.209	0.200
1980	1.794	1.301	0.537	0.393	0.248	0.227	0.227	0.208	0.205	0.205	0.202
1981	1.498	2.015	0.690	0.435	0.277	0.223	0.212	0.215	0.203	0.202	0.203
1982	1.545	1.925	0.582	0.422	0.274	0.236	0.207	0.206	0.206	0.200	0.200
1983	1.172	1.742	0.500	0.410	0.316	0.238	0.215	0.215	0.202	0.204	0.204
1984	1.475	1.158	0.492	0.346	0.295	0.269	0.226	0.212	0.201	0.200	0.202
1985	1.421	1.308	0.496	0.358	0.287	0.242	0.232	0.210	0.203	0.200	0.200
1986	1.437	1.025	0.399	0.331	0.285	0.236	0.213	0.215	0.205	0.208	0.200
1987	1.787	1.006	0.424	0.359	0.267	0.221	0.208	0.207	0.208	0.206	0.201
1988	1.354	1.150	0.464	0.323	0.291	0.258	0.210	0.204	0.208	0.218	0.202
1989	1.733	1.097	0.391	0.361	0.265	0.229	0.223	0.206	0.201	0.201	0.218
1990	1.422	1.062	0.447	0.339	0.308	0.240	0.215	0.210	0.202	0.201	0.200
1991	1.222	1.051	0.423	0.314	0.276	0.263	0.223	0.206	0.204	0.201	0.200
1992	1.121	1.165	0.413	0.303	0.243	0.218	0.221	0.203	0.201	0.200	0.200
1993	1.120	1.075	0.380	0.293	0.249	0.221	0.212	0.211	0.201	0.201	0.200
1994	1.046	1.090	0.420	0.298	0.267	0.227	0.204	0.203	0.203	0.201	0.200
1995	1.350	1.267	0.381	0.304	0.266	0.226	0.208	0.202	0.205	0.204	0.200
1996	1.228	1.390	0.407	0.297	0.279	0.241	0.216	0.225	0.200	0.201	0.202
1997	1.340	1.072	0.435	0.309	0.252	0.249	0.210	0.210	0.202	0.200	0.200
1998	1.323	1.265	0.367	0.309	0.295	0.255	0.223	0.204	0.204	0.202	0.200
1999	0.956	1.063	0.339	0.310	0.270	0.265	0.233	0.227	0.220	0.201	0.200
2000	1.203	0.847	0.313	0.298	0.262	0.238	0.235	0.203	0.206	0.200	0.200
2001	1.460	0.995	0.358	0.298	0.262	0.245	0.228	0.204	0.200	0.202	0.200
2002	1.600	1.201	0.413	0.359	0.252	0.251	0.215	0.215	0.201	0.200	0.200

2003	1.489	1.163	0.404	0.346	0.290	0.264	0.234	0.203	0.201	0.200	0.200
2004	1.501	1.632	0.508	0.401	0.396	0.391	0.274	0.204	0.201	0.200	0.200
2005	1.150	1.485	0.441	0.389	0.293	0.296	0.293	0.208	0.202	0.201	0.200
2006	1.192	1.291	0.415	0.372	0.334	0.275	0.275	0.273	0.203	0.201	0.200
2007	1.125	1.324	0.438	0.290	0.273	0.268	0.260	0.260	0.240	0.208	0.200
2008	1.035	1.377	0.483	0.299	0.266	0.266	0.266	0.205	0.205	0.207	0.200
2009	0.839	1.112	0.466	0.396	0.306	0.255	0.221	0.208	0.201	0.201	0.202
2010	1.021	0.977	0.461	0.289	0.277	0.277	0.261	0.204	0.201	0.201	0.201
2011	1.284	1.184	0.476	0.304	0.304	0.304	0.298	0.290	0.212	0.298	0.200
2012	1.097	1.202	0.536	0.299	0.285	0.273	0.273	0.273	0.201	0.211	0.201
2013	1.027	1.110	0.510	0.268	0.297	0.253	0.214	0.238	0.238	0.210	0.200
2014	0.955	1.330	0.523	0.277	0.277	0.288	0.237	0.222	0.213	0.228	0.200
2015	1.089	1.165	0.534	0.317	0.243	0.276	0.276	0.276	0.207	0.268	0.271
2016	0.955	1.405	0.630	0.301	0.262	0.224	0.216	0.277	0.245	0.208	0.200

Herring : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3	4	5	6	7	8	9
1974	0.739	0.534	0.326	0.292	0.278	0.253	0.247	0.239	0.239	0.236
1975	0.737	0.552	0.319	0.261	0.238	0.229	0.215	0.210	0.208	0.208
1976	0.725	0.594	0.354	0.272	0.236	0.218	0.208	0.203	0.203	0.203
1977	0.665	0.652	0.368	0.303	0.240	0.219	0.199	0.194	0.192	0.199
1978	0.558	0.661	0.355	0.298	0.258	0.231	0.223	0.217	0.217	0.217
1979	0.621	0.564	0.315	0.287	0.252	0.232	0.217	0.204	0.204	0.202
1980	0.706	0.603	0.284	0.242	0.225	0.213	0.195	0.186	0.186	0.186
1981	0.785	0.817	0.403	0.304	0.270	0.247	0.218	0.209	0.209	0.209
1982	0.768	0.632	0.371	0.315	0.270	0.227	0.220	0.198	0.198	0.192
1983	0.721	0.581	0.355	0.310	0.255	0.235	0.203	0.197	0.192	0.192
1984	0.822	0.555	0.284	0.235	0.216	0.195	0.186	0.177	0.177	0.173
1985	0.742	0.616	0.305	0.255	0.223	0.197	0.180	0.174	0.172	0.179
1986	0.737	0.600	0.327	0.205	0.191	0.182	0.170	0.163	0.163	0.163
1987	0.859	0.586	0.285	0.213	0.179	0.176	0.165	0.158	0.152	0.138
1988	0.747	0.637	0.316	0.233	0.182	0.182	0.168	0.162	0.158	0.151
1989	0.854	0.557	0.266	0.206	0.174	0.163	0.160	0.146	0.146	0.134
1990	0.803	0.625	0.270	0.225	0.200	0.187	0.170	0.163	0.163	0.141
1991	0.823	0.540	0.275	0.214	0.193	0.176	0.164	0.159	0.164	0.149
1992	0.738	0.516	0.271	0.238	0.195	0.178	0.161	0.150	0.150	0.141
1993	0.673	0.520	0.300	0.241	0.201	0.172	0.154	0.148	0.146	0.130
1994	0.701	0.523	0.280	0.211	0.192	0.171	0.157	0.157	0.148	0.145
1995	0.810	0.499	0.315	0.226	0.221	0.199	0.190	0.179	0.151	0.148
1996	0.733	0.559	0.326	0.238	0.222	0.205	0.166	0.166	0.157	0.154
1997	0.825	0.490	0.321	0.242	0.215	0.177	0.166	0.153	0.151	0.142
1998	0.808	0.537	0.343	0.271	0.230	0.211	0.174	0.160	0.158	0.133
1999	0.756	0.576	0.312	0.213	0.203	0.191	0.168	0.160	0.160	0.147
2000	0.766	0.484	0.243	0.214	0.177	0.174	0.157	0.154	0.141	0.128
2001	0.741	0.602	0.320	0.238	0.174	0.159	0.155	0.155	0.153	0.141
2002	0.855	0.595	0.358	0.250	0.210	0.180	0.171	0.171	0.179	0.156

2003	0.929	0.679	0.351	0.213	0.196	0.175	0.164	0.156	0.161	0.154
2004	0.850	0.695	0.389	0.321	0.253	0.236	0.224	0.203	0.203	0.188
2005	0.888	0.684	0.407	0.305	0.241	0.229	0.197	0.181	0.173	0.189
2006	0.913	0.604	0.345	0.292	0.260	0.241	0.225	0.204	0.194	0.190
2007	0.934	0.613	0.368	0.314	0.261	0.238	0.217	0.202	0.193	0.200
2008	0.915	0.555	0.338	0.301	0.272	0.243	0.218	0.198	0.196	0.184
2009	0.791	0.491	0.284	0.260	0.252	0.228	0.210	0.201	0.197	0.188
2010	0.830	0.418	0.274	0.239	0.235	0.222	0.213	0.208	0.201	0.199
2011	1.005	0.542	0.327	0.294	0.258	0.249	0.243	0.231	0.216	0.216
2012	0.957	0.630	0.329	0.273	0.266	0.246	0.223	0.216	0.202	0.195
2013	0.801	0.574	0.345	0.266	0.261	0.239	0.236	0.222	0.209	0.208
2014	0.777	0.554	0.338	0.286	0.257	0.253	0.234	0.234	0.222	0.215
2015	0.873	0.489	0.315	0.273	0.257	0.243	0.238	0.236	0.229	0.223
2016	0.664	0.635	0.350	0.302	0.293	0.270	0.259	0.247	0.240	0.233

Northern sandeel : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3	4
1974	1.266	1.506	1.064	0.739	0.711
1975	1.053	2.012	1.407	0.734	0.711
1976	0.907	1.596	1.247	1.056	0.955
1977	0.861	1.372	1.050	0.817	0.767
1978	0.811	1.310	0.967	0.712	0.663
1979	0.930	1.273	0.950	0.645	0.630
1980	1.076	1.590	1.187	0.682	0.662
1981	1.090	1.691	1.309	1.084	1.017
1982	1.098	1.562	1.301	1.070	0.998
1983	0.942	1.322	1.147	0.863	0.791
1984	0.812	1.331	1.011	0.640	0.632
1985	0.681	1.198	0.899	0.757	0.722
1986	0.971	1.197	1.033	0.980	0.834
1987	0.970	1.279	1.107	0.700	0.698
1988	1.028	1.417	0.925	0.827	0.693
1989	0.886	1.254	1.004	0.774	0.738
1990	0.896	1.252	1.135	0.785	0.735
1991	1.382	1.381	0.959	0.553	0.708
1992	0.881	1.110	0.950	0.746	0.653
1993	1.341	1.430	0.892	0.811	0.795
1994	1.234	1.198	0.805	0.738	0.730
1995	1.268	1.429	1.304	0.899	0.886
1996	0.872	1.365	1.001	0.922	0.755
1997	1.163	1.320	1.147	0.944	0.856
1998	1.035	1.477	1.137	0.986	0.931
1999	0.627	1.225	1.063	0.837	0.788
2000	1.146	1.715	1.432	1.129	0.975
2001	1.053	1.861	1.581	1.355	1.217
2002	1.109	1.681	1.420	1.338	1.244

2003	1.089	1.902	1.230	1.194	1.155
2004	1.058	1.837	1.624	1.628	1.441
2005	1.299	1.864	1.500	1.228	1.180
2006	1.094	1.731	1.400	1.066	0.979
2007	1.079	1.663	1.094	1.034	1.064
2008	1.021	1.535	1.049	0.966	0.957
2009	0.809	1.275	1.085	0.803	0.978
2010	1.010	1.353	1.064	0.939	0.896
2011	1.180	1.911	1.591	1.392	1.324
2012	0.970	1.462	1.008	0.967	0.899
2013	0.848	1.335	1.072	0.956	0.879
2014	0.938	1.317	1.064	0.865	0.814
2015	0.912	1.151	0.916	0.755	0.704
2016	0.785	1.774	1.275	1.093	1.005

Southern sandeel : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3	4
1974	0.491	0.924	0.793	0.693	0.692
1975	0.457	0.886	0.806	0.696	0.696
1976	0.510	0.915	0.795	0.681	0.681
1977	0.416	0.829	0.798	0.656	0.656
1978	0.438	0.849	0.689	0.560	0.560
1979	0.385	0.688	0.675	0.552	0.552
1980	0.424	0.739	0.715	0.574	0.574
1981	0.565	1.091	0.761	0.596	0.596
1982	0.503	0.853	0.750	0.735	0.620
1983	0.472	0.876	0.763	0.615	0.639
1984	0.561	0.901	0.711	0.601	0.601
1985	0.472	0.824	0.764	0.615	0.571
1986	0.515	0.834	0.718	0.745	0.654
1987	0.607	0.901	0.781	0.626	0.658
1988	0.491	0.856	0.804	0.635	0.600
1989	0.540	0.850	0.751	0.751	0.719
1990	0.538	0.888	0.771	0.667	0.625
1991	0.552	0.895	0.720	0.560	0.617
1992	0.559	0.755	0.656	0.559	0.559
1993	0.488	0.797	0.659	0.565	0.536
1994	0.489	0.824	0.681	0.578	0.543
1995	0.558	0.819	0.696	0.599	0.568
1996	0.482	0.858	0.732	0.596	0.563
1997	0.575	0.773	0.625	0.592	0.553
1998	0.601	0.951	0.732	0.651	0.589
1999	0.597	1.076	0.804	0.683	0.599
2000	0.585	0.933	0.720	0.608	0.574
2001	0.563	0.947	0.687	0.630	0.488
2002	0.621	0.866	0.742	0.611	0.539

2003	0.752	1.168	1.153	0.862	0.803
2004	0.616	1.059	0.774	0.774	0.715
2005	0.668	1.151	0.815	0.672	0.600
2006	0.703	0.924	0.770	0.764	0.630
2007	0.713	1.207	0.863	0.616	0.615
2008	0.776	1.053	0.761	0.655	0.633
2009	0.592	1.057	0.877	0.711	0.642
2010	0.640	0.924	0.655	0.607	0.569
2011	0.792	1.245	0.862	0.762	0.679
2012	0.739	1.289	0.818	0.790	0.681
2013	0.619	1.011	0.865	0.668	0.668
2014	0.602	1.066	0.843	0.641	0.585
2015	0.684	0.914	0.652	0.561	0.538
2016	0.532	1.110	0.669	0.579	0.542

Norway pout : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3
1974	1.130	1.666	1.434	1.320
1975	1.236	1.842	1.421	1.221
1976	1.148	2.060	1.596	1.401
1977	1.126	1.880	1.591	1.392
1978	0.962	1.904	1.554	1.372
1979	0.906	1.626	1.354	1.207
1980	1.179	1.689	1.333	1.108
1981	1.225	2.268	1.806	1.661
1982	1.113	1.856	1.590	1.391
1983	0.962	1.636	1.398	1.285
1984	1.212	1.560	1.250	1.050
1985	1.174	1.872	1.512	1.356
1986	1.275	1.868	1.538	1.313
1987	1.403	1.808	1.496	1.292
1988	1.073	1.791	1.499	1.372
1989	1.335	1.638	1.297	1.030
1990	1.039	1.564	1.325	1.204
1991	0.966	1.394	1.161	1.039
1992	1.065	1.393	1.138	1.023
1993	1.418	1.495	1.224	1.104
1994	1.066	1.523	1.176	1.036
1995	1.408	1.553	1.339	1.211
1996	1.014	1.736	1.421	1.305
1997	1.210	1.559	1.380	1.252
1998	1.249	1.759	1.483	1.342
1999	1.017	1.697	1.390	1.248
2000	1.219	1.417	1.142	0.991
2001	1.349	2.035	1.535	1.393
2002	1.390	2.146	1.796	1.610

2003	1.447	2.081	1.843	1.666
2004	1.469	2.276	2.028	1.863
2005	1.305	2.255	2.033	1.944
2006	1.348	1.861	1.662	1.566
2007	1.407	2.007	1.737	1.660
2008	1.264	1.844	1.628	1.500
2009	0.994	1.488	1.287	1.184
2010	1.390	1.592	1.426	1.332
2011	1.743	2.398	2.081	1.949
2012	1.405	2.161	1.898	1.758
2013	1.518	2.146	1.984	1.903
2014	1.266	2.261	2.013	1.914
2015	1.401	1.860	1.665	1.576
2016	1.138	2.109	1.818	1.734

Sprat : Natural mortality (sum of quarterly M1+M2)

Year/Age	0	1	2	3
1974	0.693	1.434	1.246	0.622
1975	0.714	1.468	1.369	0.959
1976	1.016	1.460	1.303	1.115
1977	0.693	1.604	1.468	1.303
1978	0.732	1.450	1.343	1.098
1979	0.851	1.438	1.372	1.205
1980	0.974	1.619	1.535	1.335
1981	0.927	1.630	1.461	1.221
1982	0.829	1.441	1.330	0.927
1983	0.769	1.125	0.923	0.588
1984	1.008	1.295	0.970	0.901
1985	1.028	1.388	0.994	0.649
1986	1.058	1.543	1.447	0.797
1987	1.040	1.532	1.241	1.026
1988	1.101	1.366	1.167	0.712
1989	1.202	1.715	1.398	0.972
1990	1.083	1.641	1.237	1.099
1991	0.755	1.294	1.043	0.952
1992	0.735	1.314	1.113	0.979
1993	0.677	1.250	1.051	0.905
1994	0.670	1.297	1.148	0.993
1995	1.038	1.322	1.017	0.964
1996	0.699	1.156	0.838	0.750
1997	0.865	0.849	0.745	0.561
1998	0.682	0.844	0.708	0.537
1999	0.827	1.234	0.977	0.934
2000	0.627	1.063	0.882	0.736
2001	0.829	1.151	0.964	0.794
2002	0.667	1.006	0.805	0.672

2003	0.753	1.193	1.045	0.877
2004	0.722	1.018	0.825	0.805
2005	0.668	1.190	1.011	0.983
2006	0.863	1.230	0.930	0.839
2007	0.775	0.920	0.729	0.533
2008	0.665	1.196	0.886	0.804
2009	0.800	0.982	0.654	0.597
2010	0.849	1.022	0.827	0.628
2011	1.118	1.334	1.148	0.839
2012	0.707	1.218	0.988	0.685
2013	0.807	1.158	0.836	0.750
2014	0.496	0.883	0.726	0.461
2015	0.570	0.988	0.861	0.649
2016	0.740	1.178	0.875	0.767

5.2 Comparison with the 2015 key run

Since the last key run in 2014, (which was updated in 2015) there have been several changes in input data to the SMS:

- Update of “single species data” (catch at age numbers, mean weights, proportion mature, survey indices etc.) with use of the most recent ICES assessment input data. Re-estimation of quarterly mean weight at age in the sea from ICES annual data and quarterly differences from existing SMS data. Some stocks have been benchmarked since the 2014 key run, giving substantial changes in both the ICES and the SMS assessments
- Inclusion of mackerel as a dynamic species, which replaces the “external predators” North Sea mackerel and Western stock mackerel. With both approaches the proportion of the north Atlantic mackerel within the North Sea needs to be known. In lack of a documented time-series for that, WGSAM made their own estimate of stock distribution, where used in SMS.
- Re-calculation of “single species data” for the two sandeel stocks, as the present ICES stock areas for sandeel fit poorly into the northern and southern sandeel areas used in SMS.
- Update of consumption estimates (daily ration) of fish predators, particularly mackerel and horse mackerel using updated parameter for the evacuation model.

- Bias correction of diet estimate from observed stomach contents taking variable evacuation rate of prey species, stomach fullness and temperature into account for the fish stocks (cod, whiting, haddock saithe and mackerel) and taking variable evacuation rates of otolith (sizes) into account for harbour porpoise.
- Inclusion of distribution of fish stocks making calculations of M2 based only on the predator and prey stock numbers within the North Sea area.

The following sections describes the changes in the main output variable between the (in 2015 updated) 2014 key run and the new 2017 key run.

Cod

The main differences for cod between the two key runs are a somewhat higher recruitment in the last two decades in the 2017 key run (Figure 5.2.1, upper panel). The higher recruitment fits very well to the higher M2 in the new run (Figure 5.2.1, upper panel).

There has been very little change in the predation mortality of cod of age 1 and 2 between the 2014 and 2017 key runs.

Predation mortality of cod age 3 has increased substantially since the last key run. This is a result of the updated time-series for weight at age of cod in the stock which results in lower mean weights which are consistently within the range which can be consumed by both harbour porpoise and grey seals. Harbour porpoise has been observed to eat 3.029 kg cod, grey seal up to 4.066 kg. SMS uses the mean weight within prey size range group such that these values are translated into 1.773 kg and 2.754 kg respectively. This means that 4-year olds are not consistently included in the diet since they are only occasionally below 4 kg. The high predation mortality of age 3 cod around year 2000 corresponds to the increase in the grey seal population (see section 5.3). When age 4 is predated on, M2 is always low.

Whiting

Recruitment is pretty much the same in the two runs which is in line similar M2 for ages 0–1 (Figure 5.2.2). Predation mortality of older age groups has decreased in the new run. This is partly a result of the correction of harbour porpoise consumption to account for longer residence times of whiting otoliths in porpoise stomachs than those of e.g. herring and sandeel. The bias correction of diet data for fish predator may also influence.

Haddock

Predation of haddock is largely the same between the two key runs (Figure 5.2.3). The two series may be different due to the updated time-series for weight at age of haddock in the stock. SSB is estimated lower in the new run, probably due to the lower mean weight at age used in 2017.

Saithe

The two saithe assessments are quite similar; despite the saithe assessment has been benchmarked since the last key run (Figure 5.2.4).

Herring

The two herring assessments are quite similar (Figure 5.2.5), however with slightly higher F and lower SSB in the 2017. There has been little change in the predation mortality of herring of age 1 and 2 between the 2014 and 2017 key runs. Predation mortality of age 0 has increased further, while that of age 3 and 4 has decreased. The changes to predation mortality at age 0 seems to be linked to the changes in mackerel and horse mackerel biomass, consumption and diet, as mackerel now feeds less on sandeel and hence more on alternative prey. Predation mortality of age 3 and older herring has decreased compared to earlier key runs as the mean weight and consumption of larger cod and saithe has decreased due to the change of the assumption of constant mean weights and rations at age of the predators.

Northern sandeel

There is a substantial difference in the recruitment, F and SSB for the two assessments (Figure 5.2.6). The predation mortalities of older northern sandeel has become more variable as the mean weight of sandeel now varies from year to year and furthermore exhibits trends over the time-series. Values for older age groups have increased somewhat, likely as a result of the lower mean weight at age in the second half of the time period. Re-estimation of single species data (new stock definition) may also influence the results.

Southern sandeel

The predation mortalities of older northern sandeel has become more variable as the mean weight of sandeel now varies from year to year and furthermore exhibits trends over the time-series (Figure 5.2.7). Values for older age groups have increased somewhat, likely as a result of the lower mean weight at age in the second half of the time period which has increased the predation of grey gurnards and whiting on older sandeel. Re-estimation of single species data (new stock definition) may also influence the results.

Norway pout

The assessment of Norway pout has changed considerably between the two key runs (Figure 5.2.8), probably linked to the benchmark and inter-benchmark for this stock in the period. Predation mortality of Norway pout is very similar in the 2017 key run to those of the 2014 key run. At the end of the time period, hake becomes an important predator and is responsible for the increase in recent years (see section 5.3).

Sprat

The sprat assessment has changed (benchmark) which is also reflected in the stock summary (Figure 5.2.9) for the two key-runs. $M2$ from the 2017-run are more variable than in the previous key-run, but the trend in the two time-series is the same. The higher variability in the 2017 is probably due the variable mean weight in the sea used in the 2017 run.

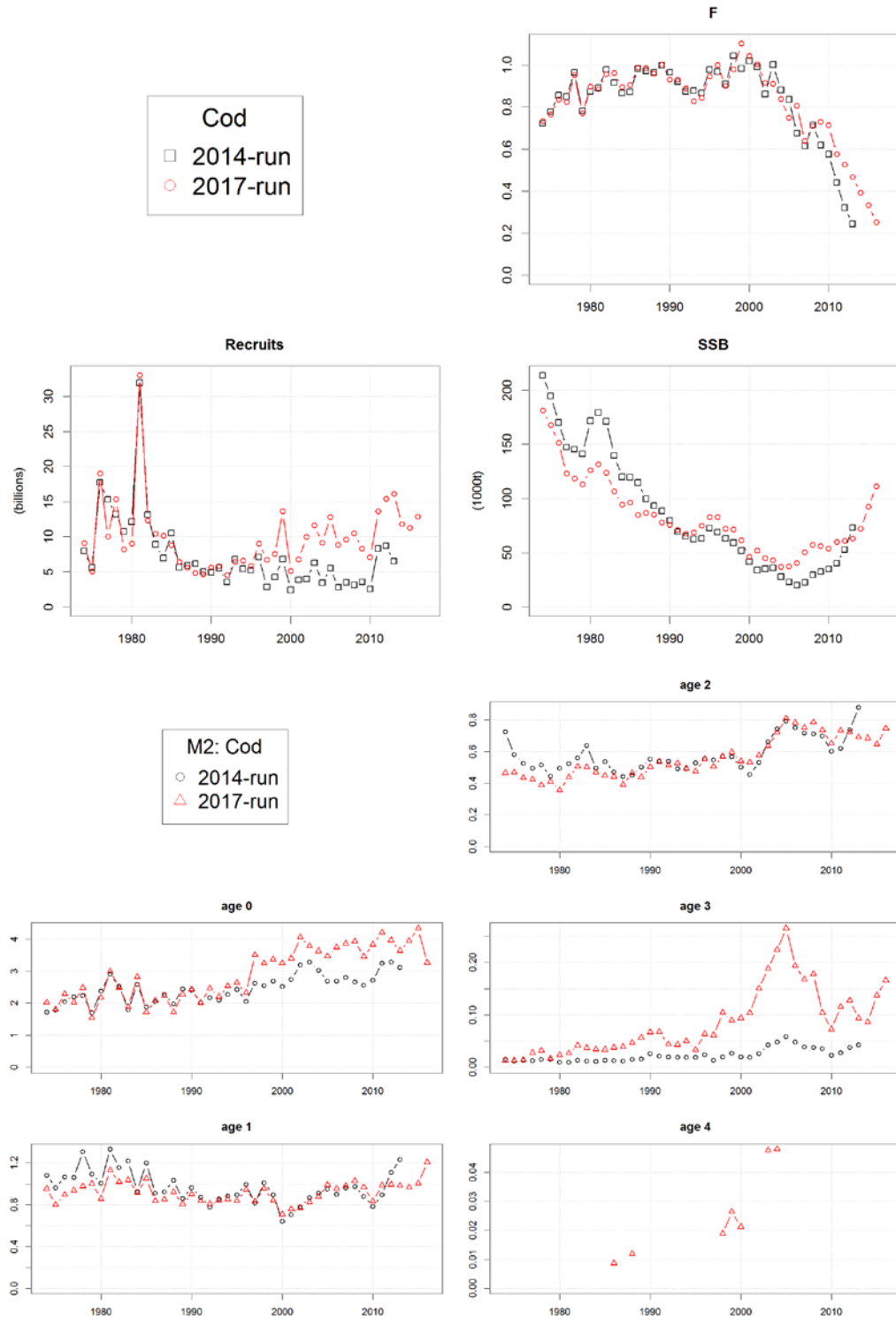


Figure 5.2.30. Comparison of estimated recruitment, mean F , SSB and predation mortality (M2) of cod from the 2014 and 2017 key runs.

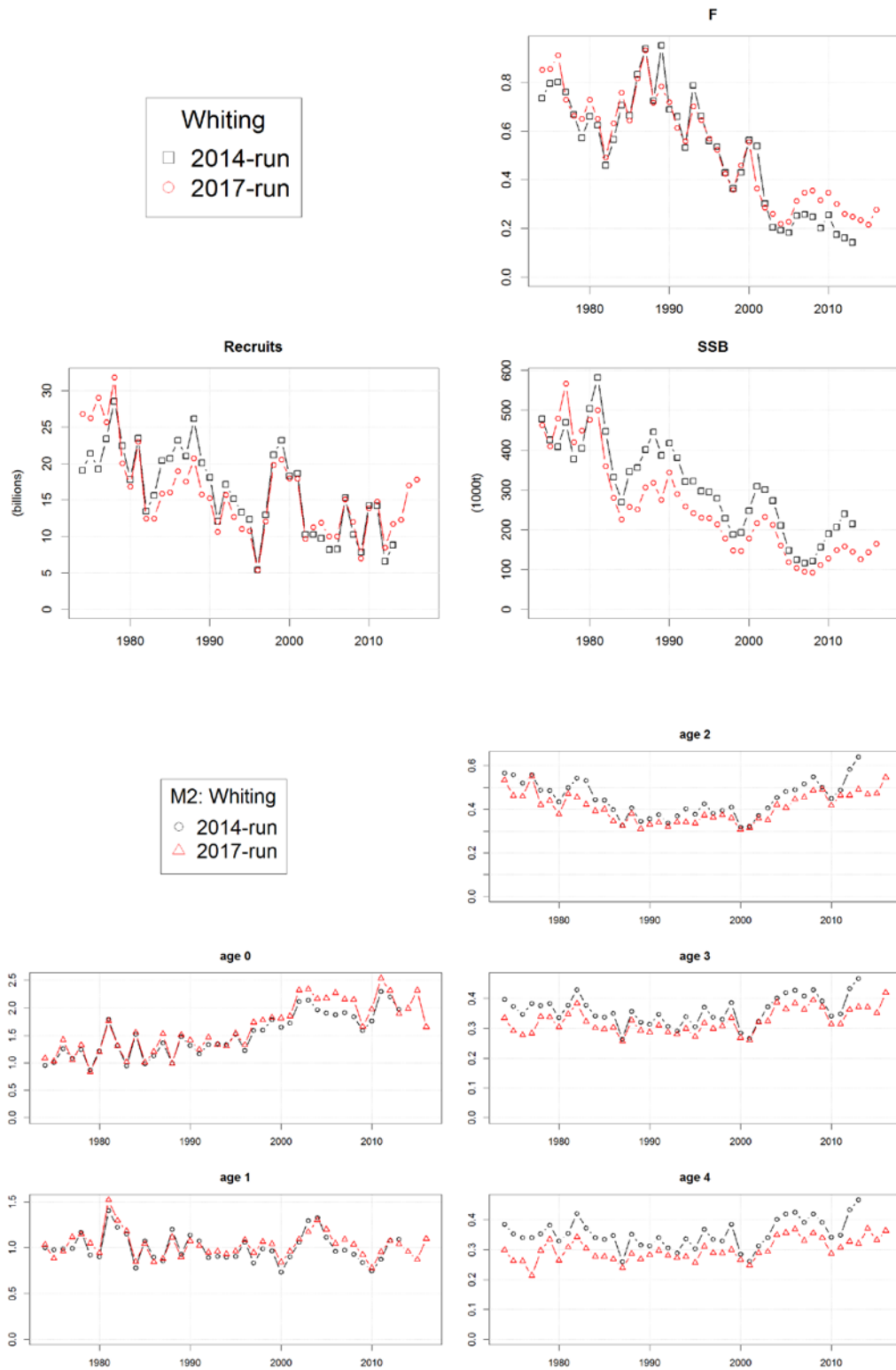


Figure 5.2.31. Comparison of estimated recruitment, mean F , SSB and predation mortality (M_2) of whiting from the 2015 and 2017 key runs.

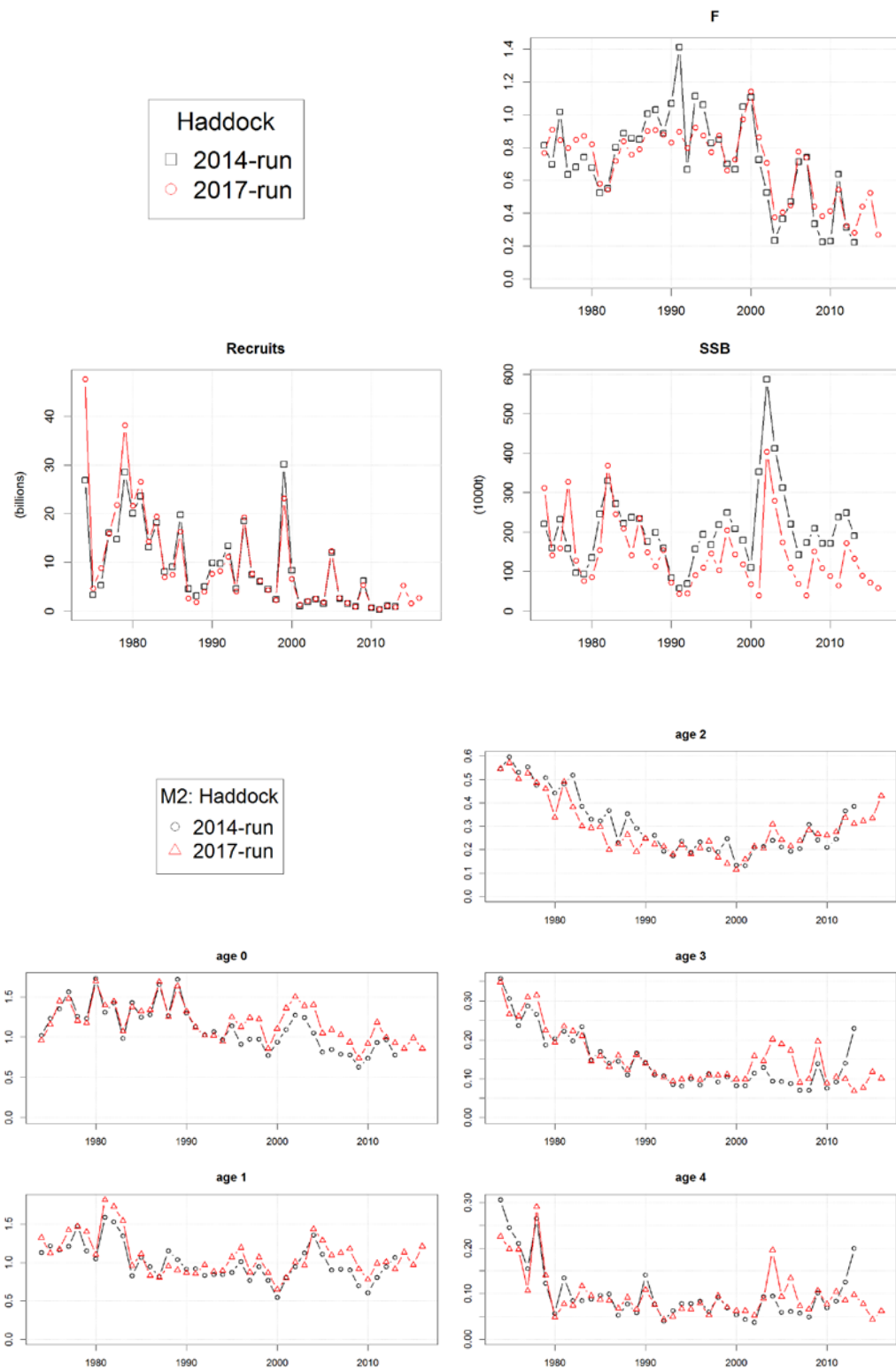


Figure 5.2.32. Comparison of estimated recruitment, mean F , SSB and predation mortality (M2) of haddock from the 2014 and 2017 key runs.

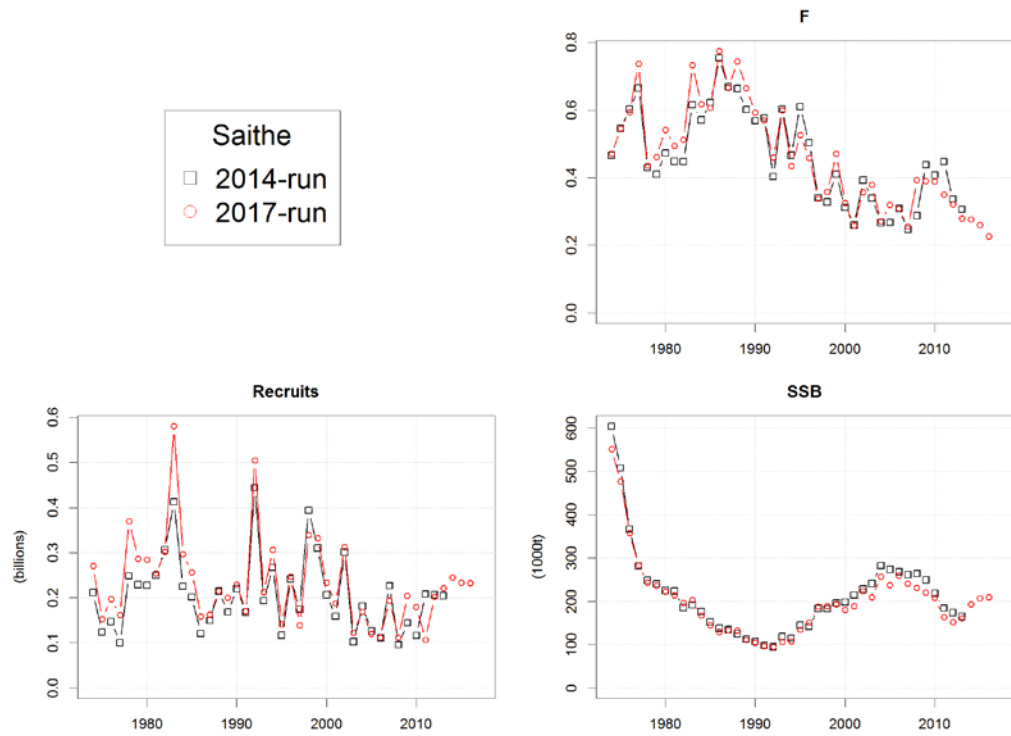


Figure 5.2.33. Comparison of estimated recruitment, mean F and SSB of Saithe from the 2014 and 2017 key runs.

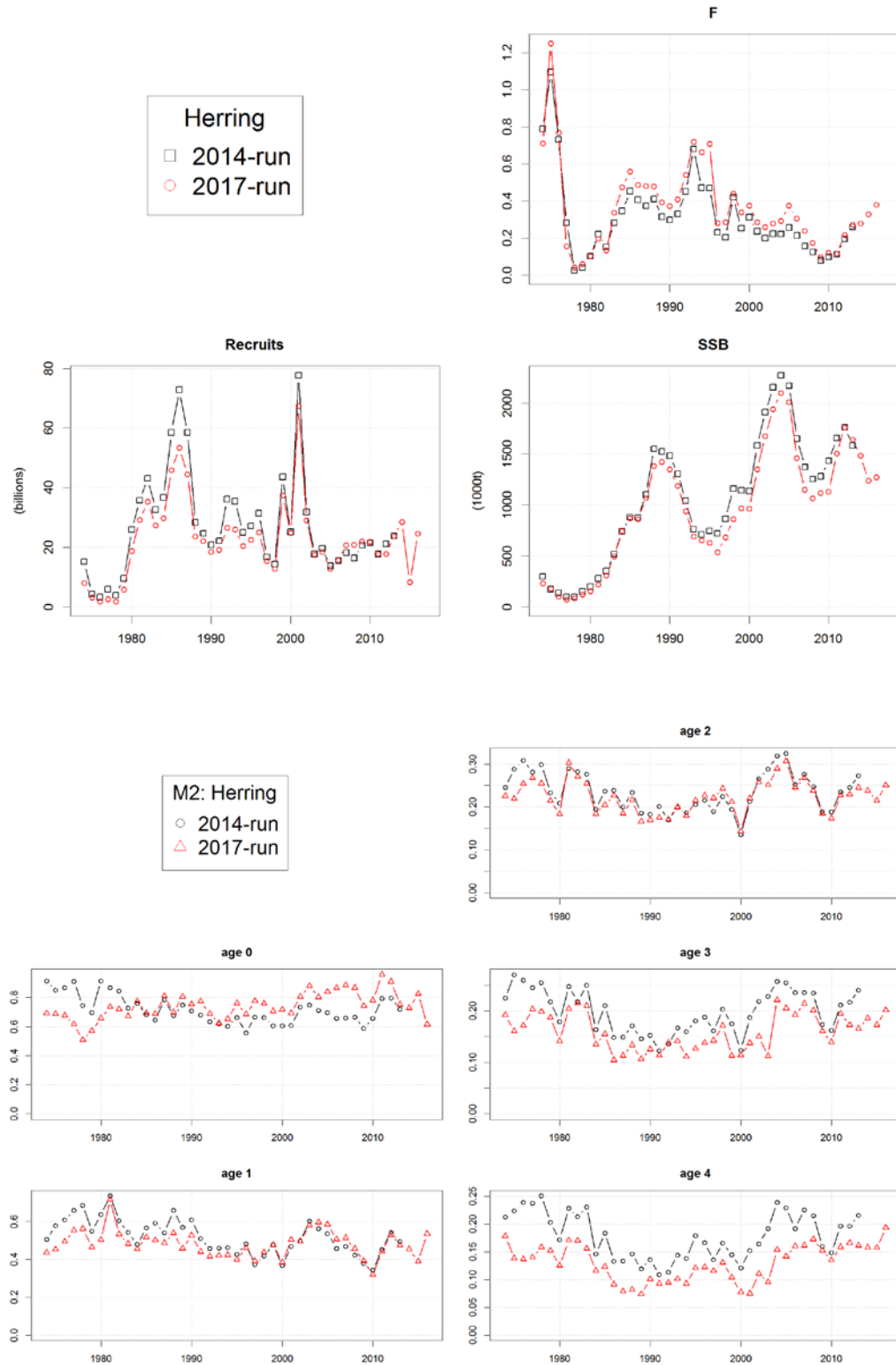


Figure 5.2.34. Comparison of estimated recruitment, mean F, SSB and predation mortality (M2) of herring from the 2015 and 2017 key runs.

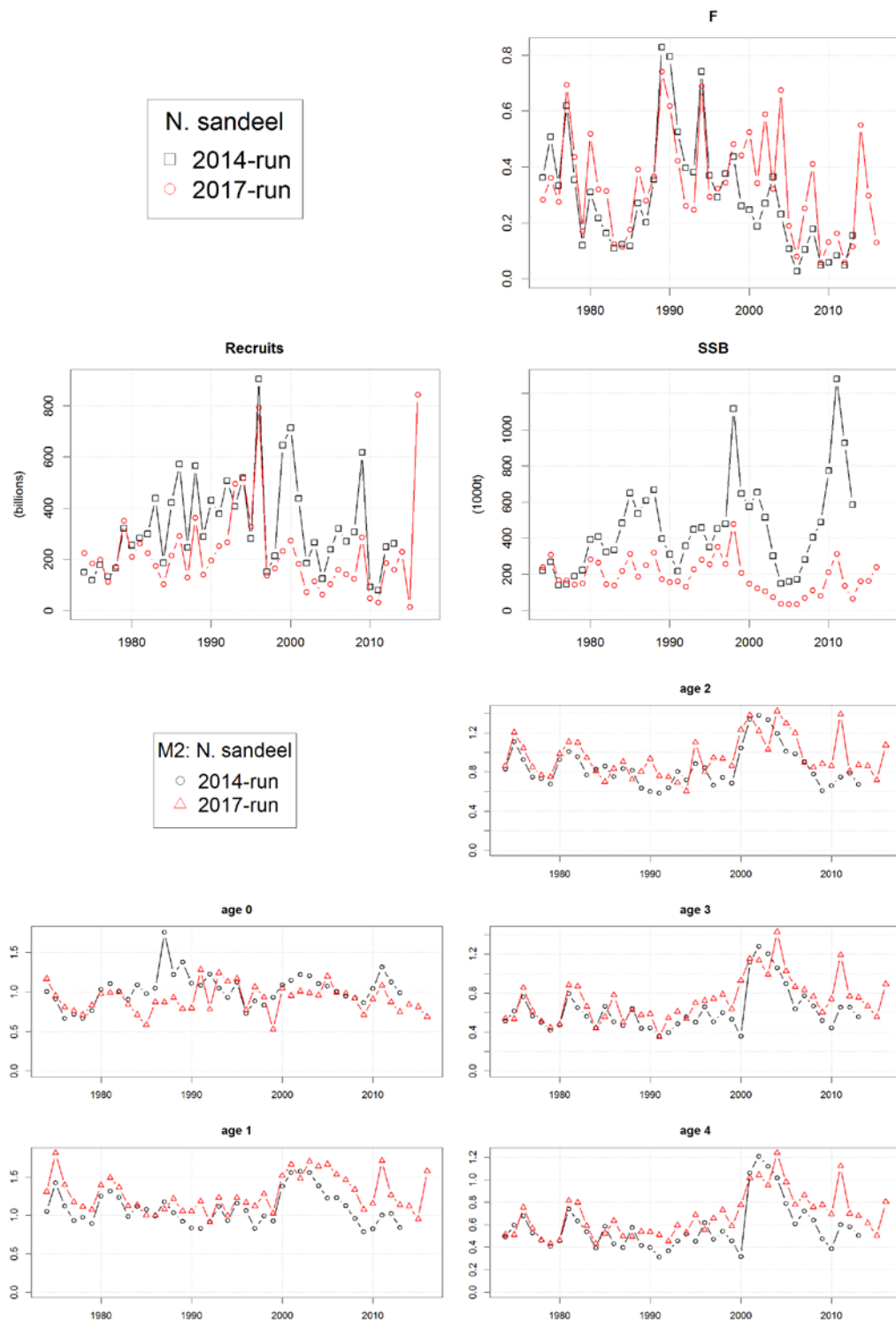


Figure 5.2.35. Comparison of estimated recruitment, mean F, SSB and predation mortality (M2) of northern sandeel from the 2015 and 2017 key runs.

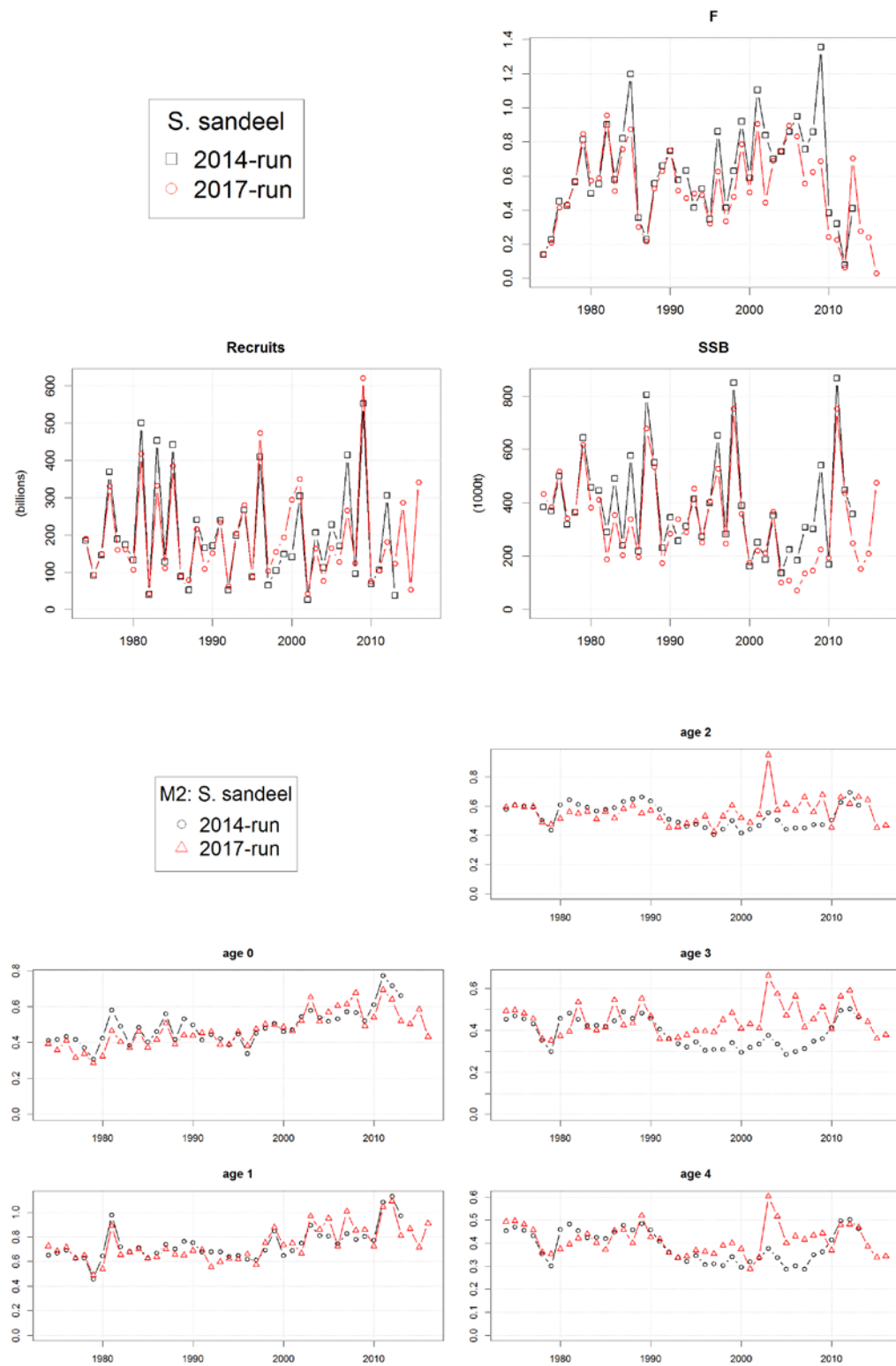


Figure 5.2.36. Comparison of estimates recruitment, mean F, SSB and predation mortality (M2) of southern sandeel from the 2014 and 2017 key runs.

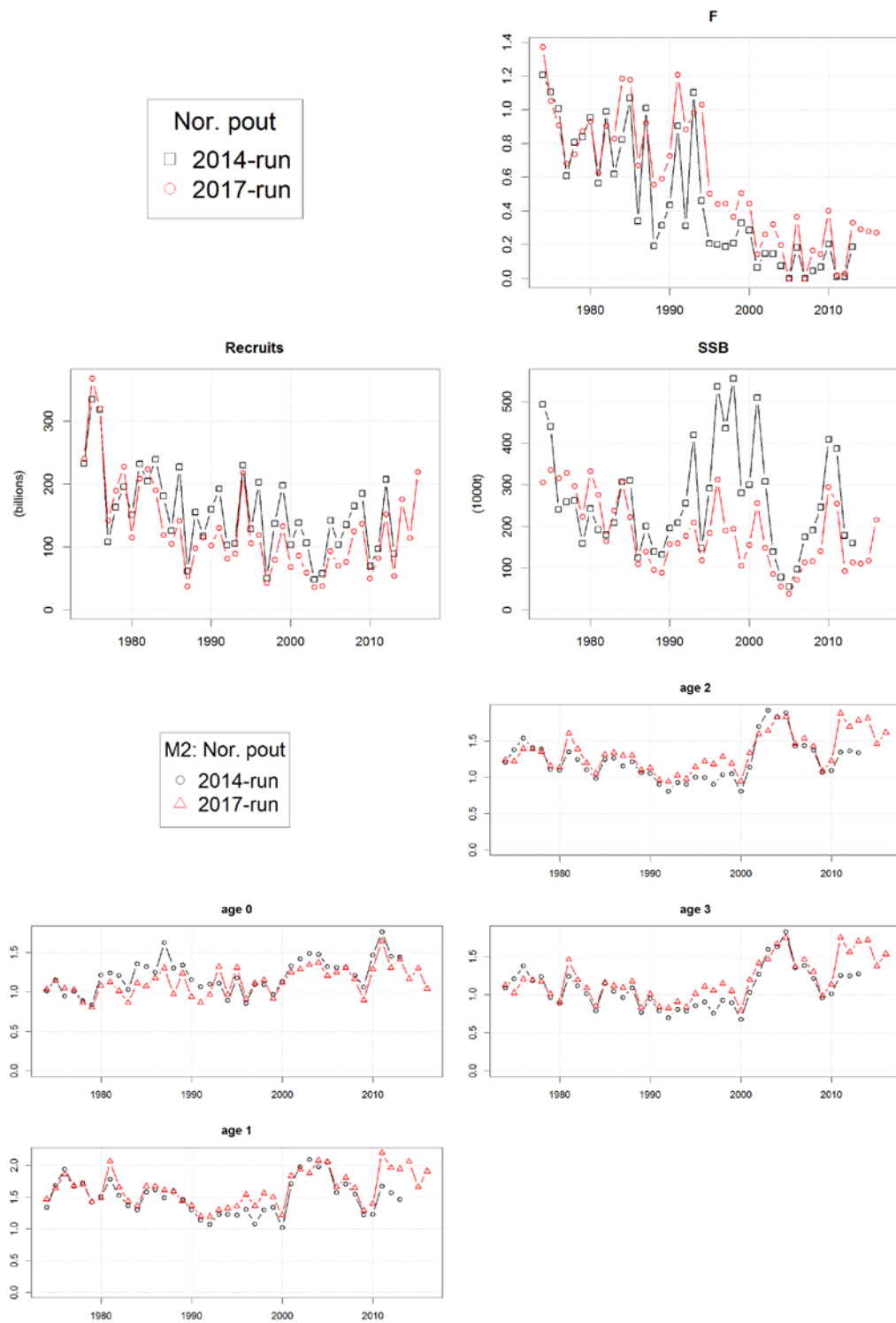


Figure 5.2.37. Comparison of estimates recruitment, mean F , SSB and predation mortality (M2) of Norway pout from the 2014 and 2017 key runs.

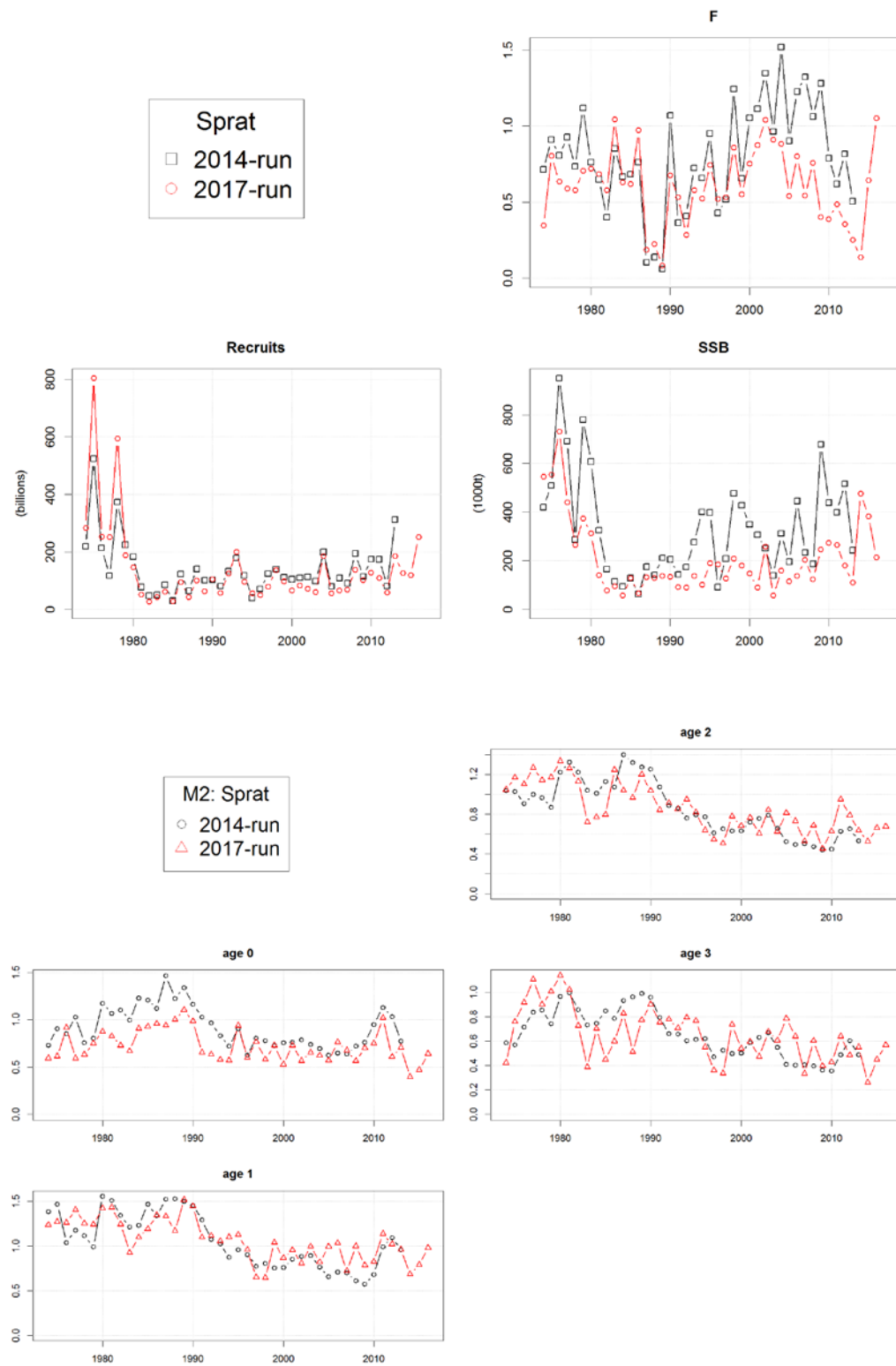


Figure 5.2.38. Comparison of estimates recruitment, mean F, SSB and predation mortality (M2) of sprat from the 2014 and 2017 runs.

5.3 Conclusion, 2017 key run

WGSAM 2017 discussed the changes in input data and the results in detail and concluded that:

- The new time-series is seen as more accurate than the previous time-series as the change in input data is based on the best available knowledge
- M2 seems consistently estimated between key-runs and shows a very limited retrospective pattern using the last key run and excluding 1–4 years of data. Changes in ration and diet data had also a rather limited effect on M2 values.
- Some ICES assessments make use of the estimated natural mortalities (M1+M2) from SMS and update those in benchmark. If used, WGSAM does not recommend updating existing data series of natural mortality by simply adding the latest three new years. The time-series as a whole shows patterns which are not retained by this procedure. For example, herring shows an increased natural mortality over the past decade, but adding only the latest three years will give the impression that natural mortality has decreased over the last five years.

5.4 Identified areas of priority research

WGSAM 2017 considers that the following topics should be priority areas of study prior to the next North Sea key run:

- Estimating the proportion of hake, mackerel and horse mackerel stocks present in the North Sea and their distribution in northern and southern areas for a better estimation of M2 for the two sandeel stocks.
- Estimating distributions of seabirds in southern and northern North Sea.
- Reviewing the method used to estimate grey gurnard and starry ray abundance to identify the reference period and sizes to which the average biomass estimates apply. Consider if the SMS model by its likelihood statistics can estimate a likely mean biomass over a given period.
- Update the number of seabirds, grey seals and harbour porpoise with the most recent information.
- Update the diet and consumption data for grey seal with the most recent data.
- Assigning prey to length groups for the 2013 mackerel stomach data.
- Establishing quarterly catch histories for the all predator species (cod, whiting, haddock, saithe, mackerel) as initiated with data from InterCatch.
- Investigate changes to modelling performance when including overwintering mortality of sandeel (M1, possible condition or weight at age dependent).
- Investigate the most appropriate species and size selection of different predators.

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APPENDIX 1: SMS, a stochastic age-length-structured multispecies model applied to North Sea and Baltic Sea stocks

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1. Overview

SMS (Stochastic Multi Species model) is a fish stock assessment model in which includes estimation of predation mortalities from observation of catches, survey indices and stomach contents. Estimation of predation mortality is based on the theory for predation mortality as defined by Andersen and Ursin (1977) and Gislason and Helgason (1985). SMS is a “forward running” model that operates with a chosen number of time steps (e.g. quarters of the year). The default SMS is a one-area model, but the model has options for spatial explicit predation mortality given a known stock distribution.

Model parameters are estimated using maximum likelihood (ML) technique. Uncertainties of the model parameters are estimated from the Hessian matrix and confidence limits of derived quantities like historical fishing mortalities and stock abundances are estimated from the parameter estimates and the delta-method. SMS can be used to for forecast scenarios and Management Strategy Evaluations, where fishing mortalities are estimated dynamically from Harvest Control Rules.

This document describes the model structure and the statistical models used for parameter estimation.

2. Model Structure

2.1 Survival of the stocks

The survival of the stocks is described by the standard exponential decay equation of stock numbers (N).

$$N_{s,a,y,q+1} = N_{s,a,y,q} e^{-Z_{s,a,y,q}} \quad \text{Eq. 1}$$

or

$$N_{s,a+1,y+1,q=1} = N_{s,a,y,q=\text{last season}} e^{-Z_{s,a,y,q=\text{last season}}} \quad \text{Eq. 2}$$

The instantaneous rate of total mortality, $Z_{s,a,y,q}$ by species s , age-group a , year y and season q , is divided into three components; predation mortality ($M2$), fixed residual natural mortality ($M1$) and fishing mortality (F):

$$Z_{s,a,y,q} = M1_{s,a,q} + M2_{s,a,y,q} + F_{s,a,y,q}$$

For non-assessment species which act as predators (e.g. grey seal and horse mackerel) stock numbers are assumed known and must be given as input.

2.2 Fishing mortality

Fishing mortality, $F_{s,a,y,q}$ is modelled from an extended separable model including age, year and season effects. However, as these effects may change over time a more flexible structure is assumed allowing for such changes for specified periods. For convenience the species index is left out in the following:

$$F_{a,y,q} = F_{Y,A1}^1 F_Y^2 F_{Y,A2,q}^3 \quad \text{Eq. 3}$$

where indices $A1$ and $A2$ are grouping of ages, (e.g. ages 1–3, 4–7 and 8–9) and Y is grouping of years (e.g. 1975–1989, 1990–2011).

Eq. 3 defines that the years included in the model can be grouped into a number of period clusters (Y), in which the age selection (F^1) and seasonal selection (F^3) are assumed constant. F^2 is the year effect, specifying the overall level of F for a particular year. The grouping of ages for age selection, $A1$, and season selection, $A2$, can be defined independently.

2.2.1 Options for year effect

Given a good relationship between F and effort the fishing mortality can be calculated from the observed effort.

$$F_{a,y,q} = F_{Y,A1}^1 \text{EFFORT}_y F_{Y,A2,q}^1$$

2.3 Natural Mortality

Natural mortality is divided into two components, predation mortality ($M2$) caused by the predators included in the model and a residual natural mortality ($M1$), which is assumed to be known and is given as input.

$M2$ of a prey species, prey , with size group l_{prey} due to a predator species, pred , with size group l_{pred} is calculated as suggested by Andersen and Ursin (1977) and Gislason and Helgason (1985).

$$M2_{\text{prey},l_{\text{prey}},y,q} = \sum_{\text{pred}} \sum_{l_{\text{pred}}} \frac{\bar{N}_{\text{pred},l_{\text{pred}},y,q} \text{RA}_{\text{pred},l_{\text{pred}},y,q} S_{\text{prey},\text{pred},q}(l_{\text{prey}},l_{\text{pred}})}{AB_{\text{pred},l_{\text{pred}},y,q}} \quad \text{Eq. 4}$$

where RA denotes the total food ration (weight) of one individual predator per time unit, where S denotes the food suitability defined in section 0 and where AB is the total available (suitable) biomass. AB is defined as the sum of the biomass of preys weighted by their suitability. This total prey biomass includes also the so-called “other food” (OF) which includes all prey items not explicitly modelled, e.g. species of invertebrates and non-commercial fish species. Other food species are combined into one group, such that the total available prey biomass becomes:

$$AB_{\text{pred},l_{\text{pred}},y,q} = \sum_{\text{prey}} \sum_{l_{\text{prey}}} (\bar{N}_{\text{prey},l_{\text{prey}},y,q} W_{\text{prey},l_{\text{prey}},y,q} S_{\text{prey},\text{pred},q}(l_{\text{prey}},l_{\text{pred}})) + OF_{\text{pred}} S_{OF,\text{pred},q}(l_{\text{pred}}) \quad \text{Eq. 5}$$

$M2$ cannot directly be calculated from Eq. 4 because $M2$ also is included in the right hand term in Eq. 6 to calculate \bar{N} .

$$\bar{N} = \frac{N (1 - e^{-(M1+M2+F)})}{M1 + M2 + F} \quad \text{Eq. 6}$$

As no analytical solution for $M2$ exists, $M2$ has to be found numerically. If the time step considered is sufficiently small, for instance a quarter, $M2$ becomes small and can option-

ally be approximated by replacing the average number during the season, \bar{N} , on the right hand side of Eq. 4 by the stock at the beginning of the season, N . As the right hand side of equation now is independent of $M2$ this quantity can be calculated directly from Eq. 4 where AB (Eq. 5) is modified correspondingly.

2.3.1 Use of size distribution by age

The equations outlined in the section above provide $M2$ at size groups. However, predation mortality by age is needed as well because F and catches are age-structured. If just one size group per age group of predators and preys is assumed Eq. 4 can be used directly where the age index substitutes the size group index in stock numbers

$$(\bar{N}_{prey,a,y,q} = \bar{N}_{prey,l_{prey},y,q})$$

Given more size groups per age, the calculation of $M2$ at age requires age-length-keys to split N at age to N at size group.

$$N_{s,l_s,y,q} = \sum_a N_{s,a,y,q} ALK_{s,a,l_s,y,q} \quad \text{Eq. 7}$$

where $ALK_{s,a,l_s,y,q}$ denotes the observed proportion of size group l_s for a given species and age group, i.e. $\sum_{l_s} ALK_{s,a,l_s,y,q} = 1$

Assuming that F and $M1$ depends only of the age and that $M2$ only depends of the length, $M2$ at age is estimated by: (leaving out the species, year and quarter indices)

$$M2_a = Z_a \frac{\sum_l \bar{N}_{a,l} M2_{a,l}}{D_a} = \log\left(\frac{N_a}{N_a - D_a}\right) \frac{\sum_l \bar{N}_{a,l} M2_l}{D_a}$$

where

$$\bar{N}_{a,l} = N_{a,l} \frac{1 - e^{-(F_{a,l} + M1_{a,l} + M2_{a,l})}}{F_{a,l} + M1_{a,l} + M2_{a,l}} = N_{a,l} \frac{1 - e^{-(F_a + M1_a + M2_l)}}{F_a + M1_a + M2_l}$$

and where

$$D_a = \sum_l \bar{N}_{a,l} (F_a + M1_a + M2_l)$$

denotes the number of individuals at age died within a season.

2.3.2 Food suitability

As suggested by Andersen and Ursin (1977) and Gislason and Helgason (1985) the size dependent food suitability of prey entity j for predator entity i is defined as the product of a species dependent vulnerability coefficient, $\rho_{i,j}$, a size preference coefficient $q_{i,j}(l_i, l_j)$, and an overlap index $o_{i,j,q}$. Suitability is then defined as:

$$S_{pred,prey,q}(l_{pred}, l_{prey}) = \rho_{pred,prey} q_{pred,prey}(l_{pred}, l_{prey}) o_{pred,prey,q} \quad \text{Eq. 8}$$

For the “other food” part suitability is defined as

$$S_{OF,pred,q}(l_{pred}) = \rho_{OF,pred} o_{OF,pred,q} \exp(u_{pred} \log(W_{pred,l_{pred,q}} / \bar{W}_{pred})) \quad \text{Eq. 9}$$

Where \bar{W}_{pred} is the average size of the predator species. Eq. 9 extends the original equation, to allow size dependent suitability for other food, for values of u_{pred} different from zero. The overlap index may change between seasons, but is assumed independent of year and sizes.

2.3.2.1 log-normal distributed size selection

Several functions can be used for size preference of a prey. Andersen and Ursin (1977) assumed that a predator has a preferred prey size ratio and that a prey twice as big as the preferred size is as attractive as another half the prey size. This was formulated as a log-normal distribution:

$$q_{pred,prey}(l_{pred}, l_{prey}) = \exp\left(-\frac{\left(\log\left(\frac{W_{l_{pred}}}{W_{l_{prey}}}\right) - \eta_{PREF\ pred}\right)^2}{2\sigma_{PREF\ pred}^2}\right); 0 < q \leq 1 \quad \text{Eq. 10}$$

Where η_{PREF} is the natural logarithm of the preferred size ratio, σ_{PREF}^2 is the “variance” of relative preferred size ration, expressing how selective a predator is with respect to the size of a prey and where W_{l_i} is the mean weight for a species size group.

The basic size selection equation (Eq. 10) has been extended by modifying the preferred size ratio parameter.

$$Q_{pred,prey}(l_{pred}, l_{prey}) = \exp \left(- \frac{\left(\log \left(\frac{W_{l_{pred}}}{W_{l_{prey}}} \right) - (\eta_{PREF\ pred} + \xi_{prey} + \varpi_{pred} \log(W_{l_{pred}})) \right)^2}{2 \sigma_{PREF\ pred}^2} \right) \quad \text{Eq. 11}$$

Where ξ_{prey} specify a prey specific adjustment term for the preferred size ratio, and where ϖ_{pred} specifies how the preferred size range can change by predator size.

2.3.2.2 Uniform size selection

Alternatively, a uniform size preference can be assumed within the range of the observed size ratio and zero size selection outside that ratio:

$$Q_{pred,prey}(l_{pred}, l_{prey}) = \begin{cases} 1 & \text{for } \eta_{MIN\ pred,prey} \leq \frac{W_{l_{pred}}}{W_{l_{prey}}} \leq \eta_{MAX\ pred,prey} \\ 0 & \text{for values outside observed range} \end{cases} \quad \text{Eq. 12}$$

where η_{MIN} and η_{MAX} are the observed minimum and maximum predator/prey size ratios.

2.3.2.2.1 Constraint uniform size selection

The uniform size preference does not take into account that the preferred predator/prey size ratio might change by size, such that larger individuals select relatively smaller preys (Floeter and Temming, 2005; Sharft *et al.*, 2000). A way to account for that is to assume that the fixed minimum and maximum constants, η_{MIN} and η_{MAX} , depend on the predator size:

$$Q_{pred,prey}(l_{pred}, l_{prey}) = \begin{cases} 1 & \text{for } U1_{pred,prey} + U2_{pred,prey} \log(W_{l_{pred}}) \leq \log \left(\frac{W_{l_{pred}}}{W_{l_{prey}}} \right) \leq U3_{pred,prey} + U4_{pred,prey} \log(W_{l_{pred}}) \\ 0 & \text{for values outside regression range} \end{cases} \quad \text{Eq. 13}$$

The regression parameters are estimated externally by quantile regression (e.g. Koenker and Bassett 1978) using e.g. the 2.5% and 97.5% percentiles of stomach content data. Figure 1 shows an example of such regression.

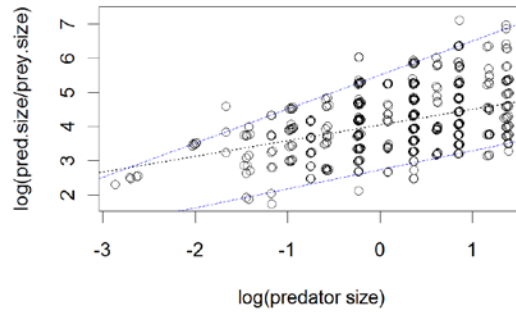


Figure 1. Quantile regression of stomach contents observations (Baltic cod eating cod), with 2.5%, 50% and 97.5 % lines shown. Predator and prey size in weight.

2.4 Adjustment of age-size keys

For the North Sea configuration, age length keys were obtained from the IBTS surveys where the same gear (i.e. the GOV trawl) has been used in the period considered. This allows an adjustment of the observed ALK's to account for mesh size selection. Using a logistic length dependent selection function, selection is defined as:

$$SL_s(l) = 1 / (1 + e^{(S1_s - S2_s * l)})$$

Where $S1_s$ and $S2_s$ are species specific gear selection parameters.

The adjusted ALK can then be derived from the observed ALK by:

$$ALK_{s,l_s,a,y,q} = \text{Observed}ALK_{s,l_s,a,y,q} / SL_{s,l_s}$$

which finally has to be standardised to 1 for each age before used in Eq. 7.

2.5 Growth

Not implemented yet!

2.6 Food ration

Food ration, RA , pr. time step is given as input or estimated from mean weight by size group assuming an exponential relationship between ration and body weight W

$$RA_{pred,l_{pred},q} = \gamma_{pred,q} W_{pred,l_{pred}}^{c_{pred}}$$

Eq. 14

where the coefficient γ and ψ are assumed to be known.

Body weight at size group l_{pred} is estimated from mean length within the size group and a length weight relation.

2.7 Area based SMS

SMS has three area explicit options:

1. Default one area model. Both F and M2 are calculated for the entire stock area
2. M2 by area. M2 is calculated by sub-areas, but F is assumed global
3. M2 and F by area. Both M2 and F are calculated by area (forecast only)

2.7.1 Stock distribution

For the area based models the stock is assumed redistributed between areas between each seasonal time step.

$$N_{a,y,q}^{area} = N_{a,y,q} \text{ DIST}_{a,y,q,area}$$

Where DIST is a stock distribution key that sums up to 1

$$\sum_{area} \text{DIST}_{a,y,q,area} = 1$$

The calculation of M2 for Option 1) is provided in the previous section

The method for option 3) is very similar, but the calculations must be done by each sub-area separately.

$$Z_a^{area} = F_a^{area} + M1_a^{area} + M2_a^{area}$$

where $M2_a^{area}$ is calculated as given in Eq. 4

Option 2) is the hybrid, where F is global but M is calculated by area.

$$Z_a^{area} = F_a + M1_a^{area} + M2_a^{area}$$

\bar{N} in an area is calculate in the usual way

$$\bar{N}_a^{area} = N_a^{area} \frac{1 - e^{-Z_a^{area}}}{Z_a^{area}}$$

The total number of individuals died due to predation mortality (DM2) then becomes

$$DM2_a = \sum_{area} M2_a^{area} \bar{N}_a^{area} \quad \text{Eq. 15}$$

M2 for the whole stock can be estimated from

$$M2_a = \log\left(\frac{N_a}{N_a - D_a}\right) \frac{DM2_a}{D_a}$$

where

$$D_a = \sum_{area} DF_a^{area} + DM1_a^{area} + DM2_a^{area}$$

and DF and DM1 are the number died due to fishery and residual mortality (M1) and are calculated in similar ways as specified for DM2 (Eq. 15).

2.7.2 Area based suitability parameters

For the "one area" SMS suitability is defined by Eq. 8.

The area based version of suitability uses an area specific vulnerability and overlap index, while the size preference (θ) is assumed independent of area.

$$S_{pred,prey,q}^{area} (l_{pred}, l_{prey}) = \rho_{pred,prey}^{area} Q_{pred,prey} (l_{pred}, l_{prey}) \phi_{pred,prey,q}^{area}$$

3. Statistical models

Three types of observations are considered: Total international catch at age; survey abundance indices and relative stomach content. For each type a stochastic model is formulated and the likelihood function is calculated. As the three types of observations are

independent the total log likelihood is the sum of the contributions from three types of observations. A stock-recruitment (penalty) function is added as a fourth contribution.

3.1 Catch-at-age

Catch-at-age observations are considered stochastic variables subject to sampling and process variation. The probability model for these observations is modelled along the lines described by Lewy and Nielsen (2003):

Catch at age is assumed to be lognormal distributed with log mean equal to log of the standard catch equation. The variance is assumed to depend on age and season and to be constant over years. To reduce the number of parameters, ages and seasons can be grouped, e.g. assuming the same variance for age 3 and age 4 in one or all seasons. Thus, the likelihood function, L_{CATCH} , associated with the catches is

$$L_{CATCH} = \prod_{s,a,y,q} \frac{1}{\sigma_{CATCH\ s,a,q} \sqrt{2\pi}} \exp\left(-\frac{(\log(C_{s,a,y,q}) - E(\log(C_{s,a,y,q})))^2}{2 \sigma_{CATCH\ s,a,q}^2}\right) \quad \text{Eq. 16}$$

Where

$$E(\log(C_{s,a,y,q})) = \log(F_{s,a,y,q} \bar{N}_{s,a,y,q})$$

Leaving out the constant term, the negative log-likelihood of catches then becomes:

$$\begin{aligned} l_{CATCH} = -\log(L_{CATCH}) \\ \propto \text{NOY} \sum_{s,a,q} \log(\sigma_{CATCH\ s,a,q}) \\ + \sum_{s,a,y,q} (\log(C_{s,a,y,q}) - E(\log(C_{s,a,y,q})))^2 / 2\sigma_{CATCH\ s,a,q}^2 \end{aligned} \quad \begin{array}{l} \text{Eq.} \\ 17 \end{array}$$

Where NOY is the number of years in the time-series.

3.1.1 Annual catches

Catch at age numbers by quarter have not been available for some of the demersal North Sea stocks in recent years. For use in the default SMS configuration of the North Sea,

where quarterly time step is used, it is assumed that the seasonal distribution (the F^q parameter in Eq. 3) is known and given as input. The likelihood function is modified to make use of the observed annual catches.

$$E(\log(C_{s,a,y})) = \log\left(\sum_q F_{s,a,y,q} \bar{N}_{s,a,y,q}\right)$$

$$L_{CATCH} = \prod_{s,a,y} \frac{1}{\sigma_{CATCH\ s,a} \sqrt{2\pi}} \exp\left(-\frac{(\log(C_{s,a,y}) - E(\log(C_{s,a,y})))^2}{2\sigma_{CATCH\ s,a}^2}\right) \quad \text{Eq. 18}$$

3.2 Survey indices

Similarly to the catch observations, survey indices, $CPUE_{survey,s,a,y,q}$ are assumed to be log-normally distributed with mean

$$E(\log(CPUE_{survey,s,a,y,q})) = \log(Q_{survey,a} \bar{N}_{SURVEY\ s,a,y,q}) \quad \text{Eq. 19}$$

where Q denotes catchability by survey and \bar{N}_{SURVEY} is mean stock number during the survey period. Catchability may depend on a single age or groups of ages. Similarly, the variance of $\log CPUE$, σ_{SURVEY}^2 may be estimated individually by age or by clusters of age groups. The negative log likelihood is on the same form as Eq. 16.

$$\begin{aligned} l_{SURVEY} &= -\log(L_{SURVEY}) \\ &\propto NOY_{survey,s} \sum_{survey,s,a} \log(\sigma_{SURVEY\ survey,s,a}) \\ &\quad + \sum_{survey,s,a,y} (\log(CPUE_{survey,s,a,y}) - E(\log(CPUE_{survey,s,a,y})))^2 / 2\sigma_{SURVEY\ s,a}^2 \end{aligned} \quad \begin{array}{l} \text{Eq.} \\ 20 \end{array}$$

3.3 Stomach contents

The stomach contents observations, which are the basis for modelling predator food preference, consist of the average proportions by weight of the stomach content averaged over the stomach samples in the North Sea. The model observations, $STOM_{pred,i,pred,prey,j,prey,y,q}$ are given for combinations of prey and predator species and size classes. In the following we use entity \bar{i} for a combination of predator species and predator size class (e.g. saithe 50–60 cm) and entity \bar{j} for the combination of prey species and prey size class eaten by entity \bar{i} . Model observations therefore becomes $STOM_{\bar{i},\bar{j},y,q}$

STOM is assumed to be stochastic variables subject to sampling and process variations.

For a given predator entity the observations across prey entities \bar{i} are continuous variables which sum to one. Thus, the probability distribution of the stomach observations for a given predator including all prey/length groups needs to be a multivariate distribution defined on the simplex. As far as the authors know the Dirichlet distribution is the only distribution fulfilling this requirement. Leaving out the year and season index, the Dirichlet density function for a predator entity \bar{i} with \bar{k} observed diet proportions $STOM_{i,1}, \dots, STOM_{i,k-1} > 0$ and the parameters $p_1, \dots, p_k > 0$ has the probability density given by

$$f_i = f(STOM_{i,1}, \dots, STOM_{i,k-1} | p_{i,1}, \dots, p_{i,k}) = \frac{\Gamma(p_i)}{\prod_{j=1}^k \Gamma(p_{i,j})} \prod_{j=1}^k STOM_{i,j}^{p_{i,j}-1} \quad \text{Eq. 21}$$

Where

$$STOM_{i,k} = 1 - \sum_{j=1}^{k-1} STOM_{i,j}$$

and

$$p_i = \sum_{j=1}^k p_{i,j}$$

The mean and variance of the observations in the Dirichlet distribution are:

$$E(STOM_{i,j}) = \frac{p_{i,j}}{p_i}$$

$$Var(STOM_{i,j}) = \frac{E(STOM_{i,j}) (1 - E(STOM_{i,j}))}{p_i + 1} \quad \text{Eq. 22}$$

The expected value of the stomach contents observations is modelled using the theory developed by Andersen and Ursin (1977):

$$E(STOM_{i,j}) = \frac{\bar{N}_j W_j S_{ij}(l_i, l_j)}{\sum_j (\bar{N}_j W_j S_{ij}(l_i, l_j)) + OF_i S_{OF,i}(l_i)} = \frac{P_{i,j}}{P_i} \quad \text{Eq. 23}$$

where the food suitability function, S , is defined by Eq. 8 and Eq. 9. We make the same assumption as made for the calculation of M2 (Eq. 4) that the small time steps used in the model, allows a replacement of \bar{N}_j by N_j in Eq. 23.

Regarding the variance of stomach contents observations unpublished analyses of the present authors of data from the North Sea stomach sampling project 1991 (ICES, 1997) indicate that the relationship between the variance and the mean of the stomach contents may be formulated in the following way:

$$Var(STOM_{i,j,y,q}) = \frac{E(STOM_{i,j,y,q}) (1 - E(STOM_{i,j,y,q}))}{V_{pred} U_{i,y,q}} \quad \text{Eq. 24}$$

where $U_{i,y,q}$ is a known quantity reflecting the sampling level of a predator entity, e.g. the number of hauls containing with stomach samples of a given predator and size class.

V_{pred} is a predator species dependent parameter linking the sampling level and variance. Equating Eq. 22 and Eq. 24 implies that

$$P_{i,j,y,q} = V_{pred} U_{i,y,q} - 1 \quad \text{Eq. 25}$$

Insertion of Eq. 25 into Eq. 23 results in that

$$P_{i,j,y,q} = (V_{pred} U_{i,y,q} - 1) \frac{\bar{N}_j W_j S_{ij}(l_i, l_j)}{\sum_j (\bar{N}_j W_j S_{ij}(l_i, l_j)) + OF_i S_{OF,i}(l_i)}$$

The parameters, $P_{i,j,y,q}$ are uniquely determined through stock numbers, total mortality, suitability parameters and V_{pred} .

Assuming that the diet observations for the predator/length groups are independent the negative log likelihood function including all predators/length groups are derived from Eq. 21:

$$l_{STOM} = -\log(L_{STOM}) = -\sum_{i,j,y,q} \log(f_{i,j,y,q}) \quad \text{Eq. 26}$$

3.3.1. Modification of the stomach contents model

The stomach contents observations, $STOM_{prey,i,prey,pred,i,pred,y,q}$ are given for combinations of prey and predator species and size classes. For a diet consisting of a large proportion “other food” and several species and prey size classes, the proportion of the individual combination of species and size becomes small (less than 0.1%) for several prey entities. Very small proportions, in combination with a modest sampling size per stratum, make the estimation of parameters impossible in some cases. To overcome the problem SMS has an option to let the likelihood use proportion summed overall size classes for a given prey species such that the prey entity equals the species.

The same grouping of all sizes from a prey is applied when the uniform size selection option (Eq. 12 and Eq. 13) is used, as The likelihood function is the same as used for stomach observations which include prey size.

3.4 Stock-recruitment

In order to enable estimation of recruitment in the last year for cases where survey indices catch from the recruitment age is missing (e.g. saithe), and to estimate parameters for forecast use, a stock-recruitment relationship $R_{s,y} = R(SSB_{s,y} | \alpha_s, \beta_s)$ penalty function is included in the likelihood function.

Recruitment to the model takes place in the same season ($recq$) and at the same age (fa) for all species. It is estimated from the Spawning Stock Biomass (SSB) in the first season (fq) of the year, and a stock recruitment relation. SSB is calculated from stock numbers, proportion mature (PM) and mean weight in the sea.

$$SSB_{s,y} = \sum_a N_{s,y,a,q=recq} PM_{s,y,a,q=recq} W_{s,y,a,q=recq} \quad \text{Eq. 27}$$

At present the Ricker (Eq. 28), the Beverton & Holt (Eq. 29), segmented regression (Eq. 30) and geometric mean are implemented.

$$R_{s,y} = \alpha_s SSB_{s,y-fa,fq} e^{(\beta_s SSB_{s,y-fa,fq})} \quad \text{Eq. 28}$$

$$R_{s,y} = \frac{\alpha_s SSB_{s,y-fa,fq}}{1 + \beta_s SSB_{s,y-fa,fq}} \quad \text{Eq. 29}$$

$$R_{s,y} = \begin{cases} \alpha_s SSB_{s,y-fa,fq} & \text{for } SSB_{s,y-fa,fq} < \beta_s \\ \alpha_s \beta_s & \text{for } SSB_{s,y-fa,fq} \geq \beta_s \end{cases} \quad \text{Eq. 30}$$

Assuming that recruitment is lognormal distributed, the negative log likelihood, l_{SR} , equals

$$l_{SR} = -\log(L_{SR}) \propto NOY \sum_s \log(\sigma_{SR,s}) + \sum_{s,a,y} (\log(N_{ss,a=fa,y,q=recq}) - E(\log(R_{s,y})))^2 / 2\sigma_{SR,s}^2 \quad \text{Eq. 31}$$

Where *NOY* gives the number of years selected and where Eq. 32 gives the expected recruitment for the Ricker case.

$$E(\log(R_s)) = \log(\alpha_s SSB_{s,y-fa,fq} e^{(\beta_s SSB_{s,y-fa,fq})}) \quad \text{Eq. 32}$$

4. Total likelihood function and parameterisation

The total negative log likelihood function, l_{TOTAL} , is found as the sum of the four terms:

$$l_{TOTAL} = l_{CATCH} + l_{SURVEY} + l_{STOM} + l_{SR}$$

To ensure uniquely determined parameters it is necessary to fix part of them. For the F at age model (Eq. 3) the year selection in the beginning of each year range (Y) has been fixed to one ($F_{y=\text{first year in each group of years}}^2 = 1$). The season effect in the last season of all years and ages is also fixed ($F_{y,a,q=\text{last season}}^2 = 1/\text{number of seasons}$).

Eq. 4 and Eq. 8 indicate that it is only possible to determine relative vulnerability parameters, $P_{pred,prey}$. We have chosen to fix the vulnerability of other food for all predators to 1.0. Similarly the biomass of other food OF_{pred} has arbitrarily been set (e.g. at 1 million tonnes) for each predators. The actual value by predator was chosen to obtain estimates of vulnerability parameters for the fish prey at around 1. Other parameters than suitability are practically unaffected of the actual choice of biomass of other food.

In the food suitability function (Eq. 8 and Eq. 9) vulnerability and overlap effects cannot be distinguished. Hence the overlap parameters were must be fixed for at least one season. In practice, several combinations of overlap have however to be fixed (at e.g. 1).

Initial stock size, i.e. the stock numbers in the first year and recruitment over years are used as parameters in the model while the remaining stock sizes are considered as functions of the parameters determined by Eq. 1 and Eq. 2.

The year effect ($F_{y,s}^2$) in the separable model for fishery mortality (Eq. 3) takes one parameter per species for each year in the time-series which sum up to a considerable number of parameters. To reduce this high number of parameters, the year effect can optionally be model from a cubic spline function which requires fewer parameters. The number of knots must be specified if this option is used.

Another way to reduce the number of parameters is to substitute the parameters σ_{CATCH} , σ_{SURVEY} and σ_{SR} used in the likelihood functions by their empirical estimates. This optional substitution has practically no effect on the model output and the associated uncertainty.

Appendage 1 gives an overview of parameters and variables in the model.

The parameters are estimated using maximum likelihood (ML) i.e. by minimizing the negative log likelihood, l_{TOTAL} . The variance/covariance matrix is approximated by the inverse Hessian matrix. Uncertainties of functions of the estimated parameters (such as biomass and mean fishing mortality) are calculated using the delta method.

5. SMS forecast

SMS is a forward running model and can as such easily be used for forecast scenarios and Management Strategy Evaluation (MSE). SMS used the estimated parameters to calculate the initial stock numbers and exploitation pattern used in the forecast. Exploitation pattern are assumed constant in the forecast period, but is scaled to a specified average F , derived dynamically from Harvest Control Rules (HCR). Recruits are produced from the stock/recruitment relation, input parameters and a noise term.

5.1 Recruitment

Recruitment is estimated from the available stock recruitment relationships, $f(SSB)$, (see section 0) and optionally a log normal distributed noise term with standard deviation std .

$$R = f(SSB) e^{(std \cdot NORM(0,1))} \quad \text{Eq. 33}$$

Where $NORM(0,1)$ is a random number drawn from a normal distribution with mean=0 and standard deviation 1. A default value for std can be obtained from the estimated variance of stock recruitment relationship, $\sigma_{SR_s}^2$ (Eq. 31)

Application of the noise function for the lognormal distributed recruitment gives on average a median recruitment as specified by $f(SSB)$. Optionally, recruitment can be adjusted with half of the variance, to obtain, on average, a mean recruitment given by $f(SSB)$.

$$R = f(SSB) e^{(std \cdot NORM(0,1))} e^{(-(std^2/2))} \quad \text{Eq. 34}$$

5.2 Harvest Control Rules

Several HCR have been implemented, e.g. constant F and the ICES interpretation of management according to MSY for both short and long-lived species. Selected, more complex management plans in force for the North Sea and Baltic Sea species have also been implemented.

6. Model validation

Model validation (in the years 2004–2009) was focused on the performance of the model using simulated data from an independent model and simulated data produced by the SMS model itself. The independent model was implemented using the R-package (R Development Core Team. 2011) and include a medium complex North Sea configuration (9 species, of which 4 are predators and 8 species preys). The simulation model follows the SMS model specification with an addition of von Bertalanffy growth curves to model mean length at age. Variance around mean length at age was assumed to increase by increasing age. This combined age-length approach made it possible to simulate all the data needed for model verification. Test data set from the simulation model included 20 years of catch data, one survey times series per species covering all years and ages, and 4 quarterly stomach samples in year 10 including stomach observations for all predator length groups. Data from the independent simulation model was used to verify that the SMS model actually works as intended and to investigate model sensitivity with respect to observation errors on catch, survey CPUE and stomach data.

To test if model parameters were identifiable when uncertainties estimated from real data were applied, the SMS model was modified to produce observations with the estimated observation noise of catch, survey and stomach data. The experiment consists of the following steps:

1. Estimate model parameters using the SMS model and available North Sea data.
2. Generate 100 set of input data from SMS output (expected catch numbers, survey indices and stomach observations) and their associated variance of these values).
3. Let SMS estimate 100 sets of parameters from the 100 sets of input data.

This procedure results in one set of “true parameters”, $\theta = (\theta_1, \dots, \theta_k)$ and 100 sets of estimated parameters, $\hat{\theta}_j = (\hat{\theta}_{1,j}, \dots, \hat{\theta}_{k,j})$, $j = 1, \dots, k$. Based on the 100 repetitions and for each of the k parameters the mean and the standard deviation of the mean $\bar{\theta}_i$ and σ_i and hence the 95% confidence limits, was calculated. Finally the proportion of the parameters was calculated for which θ_i lies in the 95% confidence interval of $\bar{\theta}_i$.

The test showed that parameters are identifiable for most “real” North Sea configurations. For some species with relatively few diet observations, size selection parameters (Eq. 11) and the variance parameter (V) linking the stomach sampling level to the vari-

ance of Dirichlet distribution (Eq. 24 and Eq. 25) , were outside the 95% confidence interval of $\bar{\theta}_i$.

A more informal testing of the model has been done by simply using the model. SMS has been applied to produce the so-called key-run for both the species rich North Sea system (10 species with stock number estimation including 7 prey species, and 16 species of “other predators”) (ICES WGSAM 2011) and the species poor Baltic Sea (cod, herring and sprat, one predator and three prey species) (WGSAM, 2008, WKMAMPEL 2009). In addition the model has been used in single species mode for the ICES advice of blue whiting in the North East Atlantic (WGWIDE, 2011) since 2005 and several sandeel stocks in the North Sea since 2009 (WGNSSK, 2011). For MSE purposes the model has been applied for sandeel and Norway pout in the North Sea (AGSANNOP 2007), blue whiting and pelagic stocks in the Baltic (WKMAMPEL 2009) in both single and multi-species mode.

SMS is essentially an extension of the statistical models normally used for single species stock assessment. This allows the use the long list of available diagnostics tools, e.g. residuals plots, and retrospective analysis, developed for model testing of sub-models for catch at age and survey indices. For stomach observations however, fewer established methods are available. To apply reliable residual plots for stomach observations residuals need to be independent, which are not the case for the stomach contents model as the observations with respect to prey entity sum to one. Instead we do the following: Let the predator entity, year and quarter be given and consider the stomach contents observations following the Dirichlet distribution:

$$STOM_r = (STOM_{r,1}, \dots, STOM_{r,k-1}) \sim Dir(p_{r,1}, \dots, p_{r,k})$$

Where r is the combined entity of predator entity, year and quarter and where $p_{r,j}, j = 1, \dots, k$ are the Dirichlet parameters estimated. Instead of considering the weight proportions, $STOM$, we consider absolute weight in the stomachs, $W_{r,j}, j = 1, \dots, k$, where

$$STOM_{r,j} = \frac{W_{r,j}}{\sum_j W_{r,j}}$$

If we assume that $W_{r,j}, j = 1, \dots, k$ are independent and follow gamma distributions with the same scale parameter, θ_r , i.e.

$$W_{r,j} \sim \Gamma(p_{r,j}, \theta_r) \quad j = 1, \dots, k$$

it is well known that $STOM_{r,j}$ follows the Dirichlet distribution. We now assume that opposite is the case (we have to prove that!) and hence assume that the absolute weights, $W_{r,j}$ are independent gamma distributed variables. We then transform these observations to obtain normal distributed residuals: Leaving out the indices we get that $U = pgamma(W, p, \theta)$, where $pgamma$ is the distribution function of the gamma distribution, is uniform distributed. To obtain normal distributed variables U is finally transformed to $V = qnorm(U)$, where $qnorm$ is the inverse of the distribution function of the standardized normal distribution. This mean that V is our new residuals for stomach contents observations.

To obtain the absolute weight of the prey entities from the relative stomach content, $STOM_{r,j}$, we have to know the total stomach weight for the predator entity. We have not extracted those from the basic observations, but simply assumed that the total weight in the stomach is proportional to the number of stomachs sampled for a given predator entity.

7. Implementation

The SMS has been implemented using the AD Model Builder (Fournier *et al.*, 2011), which is freely available from ADMB Foundation (www.admb-project.org). ADMB is an efficient tool including automatic differentiation for Maximum likelihood estimation of many parameters in nonlinear models.

SMS configurations may contain more than 1000 parameters of which less than 5% are related to predation mortality. It is not possible to estimate all parameters simultaneously without sensible initial parameter values. Such values are obtained in three phases:

1. Estimate “single species” stock numbers, fishing mortality and survey catchability parameters assuming that natural mortality ($M1+M2$) are fixed and known (i.e. as used by the ICES single species assessments).
2. Fix all the “single species” parameters estimated in step 1 and use the fixed stock numbers to estimate initial parameter values for the predation parameters.
3. Use the parameter values from step 1 and 2 as initial parameter values and re-estimate all parameters simultaneously in the full model including estimation of predation mortality $M2$.

Optimisation might potentially be dependent on the initial parameter values, however the same final result was obtained using the three steps above or using a configuration where step two is omitted. Using step two however in general makes the estimation process more robust as extreme values and system crash are avoided.

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Appendage 1. Notation, parameters and variables.

Indices

a	age
$area$	area with specific predation mortality
A_1, A_2	group of ages
Fa	first age group in the model
i	prey entity, combination of prey species and prey size group
j	predator entity, combination of predator group and predator size group
l	species size class
l_{pred}	predator size class
l_{prey}	prey size class
$other$	other food “species”
$pred$	predator species
$prey$	prey species
q	season of the year, e.g. quarter
$recq$	recruitment season
s	species
$survey$	survey identifier
y	year
Y	group of years

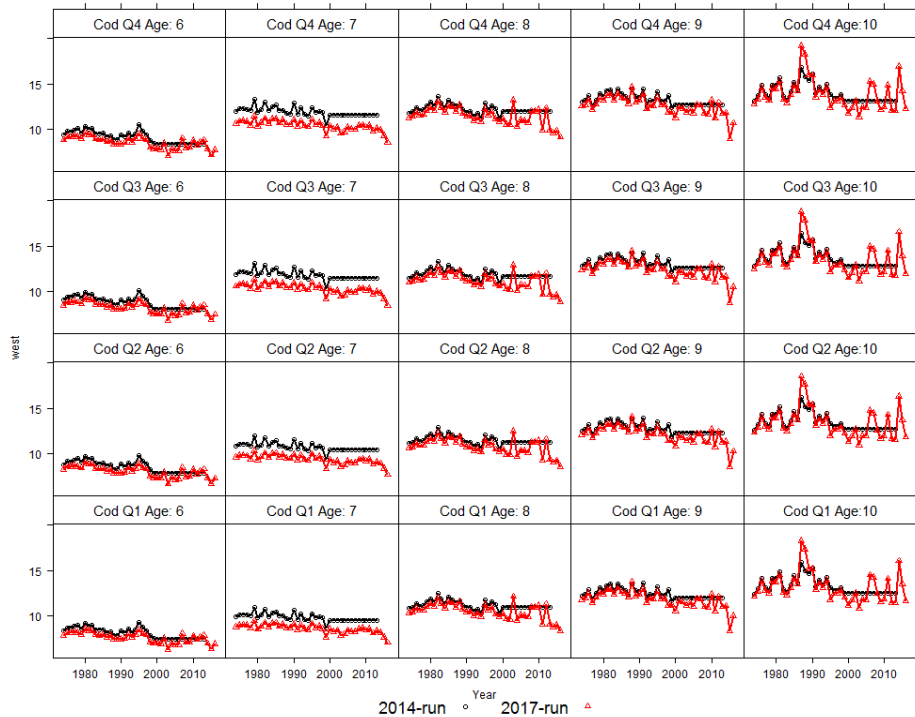
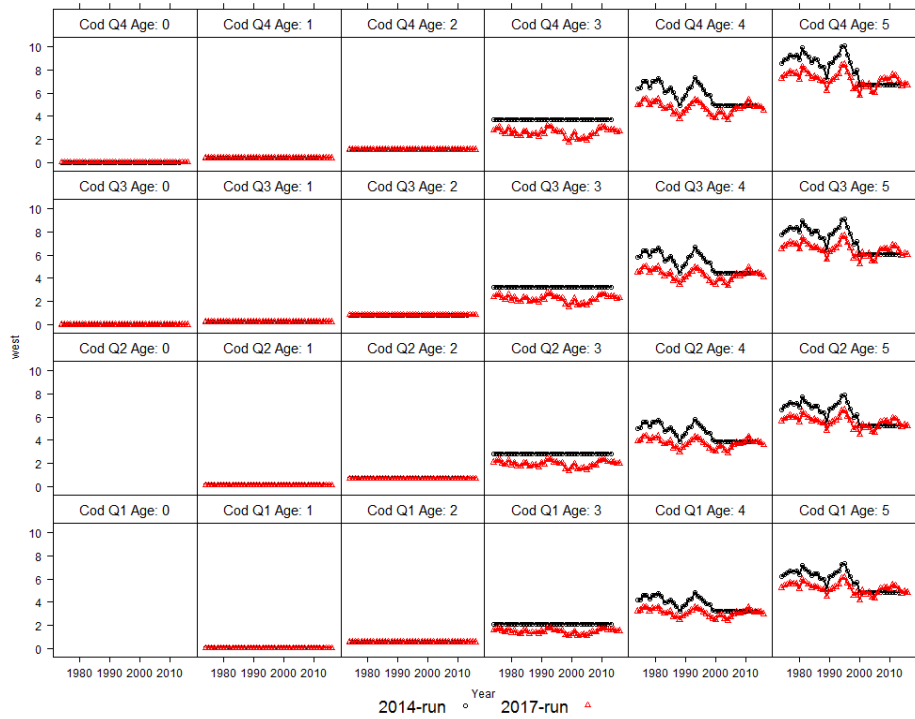
Parameters and variables

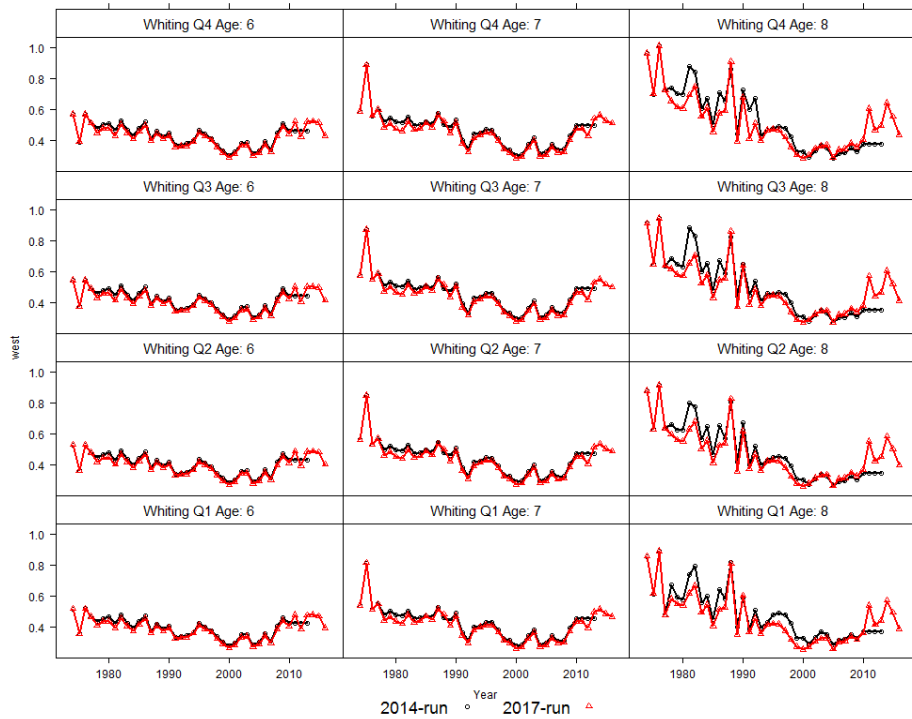
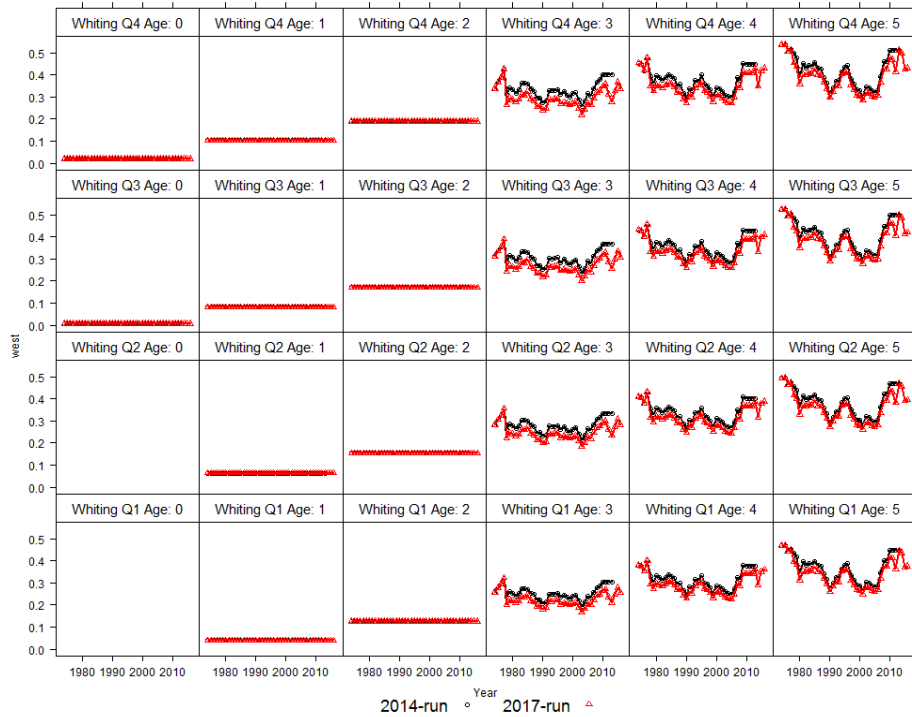
AB	available (suitable) prey biomass for a predator
ALK	proportion at size for a given age group. Input
C	catch in numbers. Observations
$CPUE$	catch in numbers per unit of effort. Observations
D	number died
$DM1$	number died due to M1
$DM2$	number died due to M2

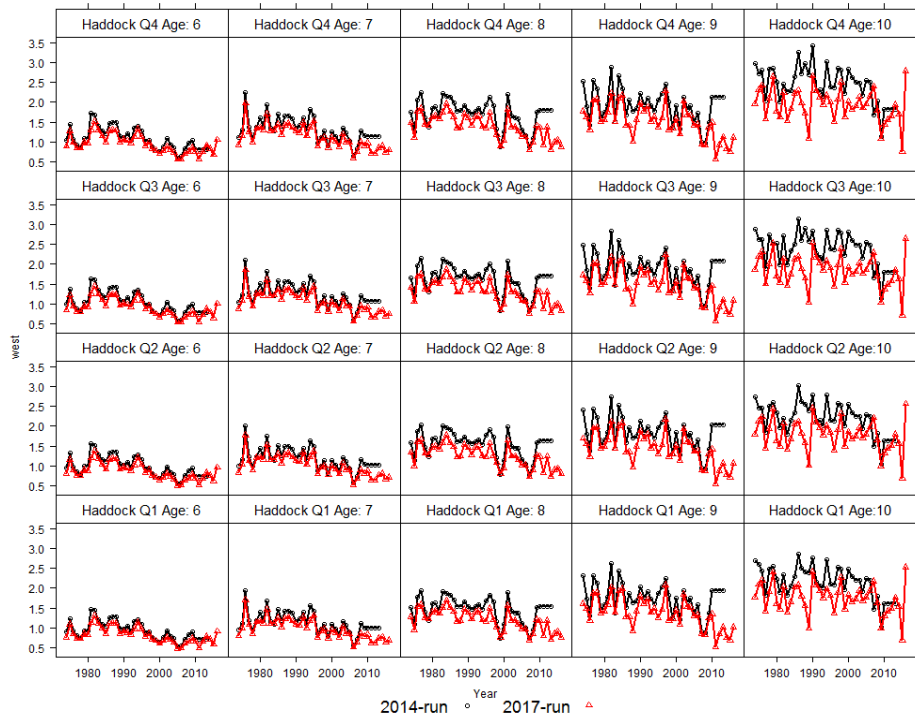
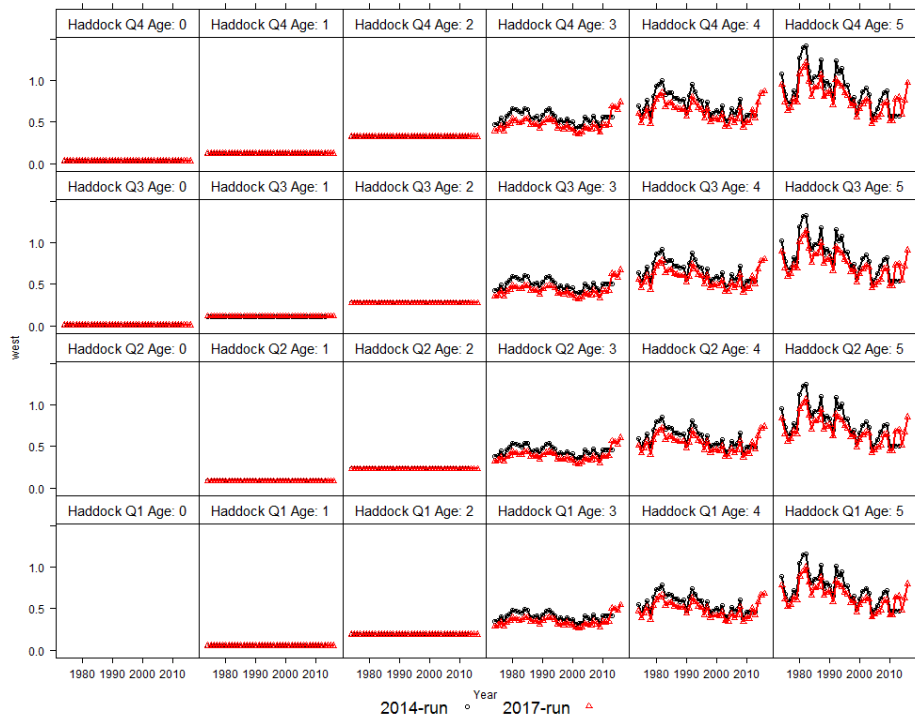
DF	number died due to F
F	instantaneous rate of fishing mortality.
F^1	age effect in separable model for fishing mortality. Estimated parameter
F^2	year effect in separable model for fishing mortality. Estimated parameter
F^3	season effect in separable model for fishing mortality. Estimated parameter
$M1$	instantaneous rate of residual natural mortality. Input
$M2$	instantaneous rate of predation mortality estimated in the model
N	stock number
$N_{s,a,y=first\ year,q=1}$	Stock number in the first year of the model. Estimated parameters.
$N_{s,a=fa,q=recq}$	Stock numbers at youngest age (recruitment). Estimated parameter.
OF	Biomass of other food for a predator. Input
Q	catchability, proportion of the population caught by one effort unit. Estimated
$R_{s,y}$	recruitment calculated from stock recruitment model
RA	food ration, biomass consumed by a predator. Input
S	suitability of a prey entity as food for a predator entity
$S1, S2$	mesh selection parameters. Estimated
SSB	spawning stock biomass
$STOM$	weight proportion of prey i found in the stomach of predator j . Observations
U	sampling intensity of stomachs. Observation
V	variance of diet observations in relation to sampling intensity. Estimated Parameter
W	body weight. Input
Z	instantaneous rate of total mortality
α	stock recruitment parameter. Estimated
β	stock recruitment parameter. Estimated
ϱ	prey size preference of a predator. Estimated parameter
γ	food ration coefficients. Input

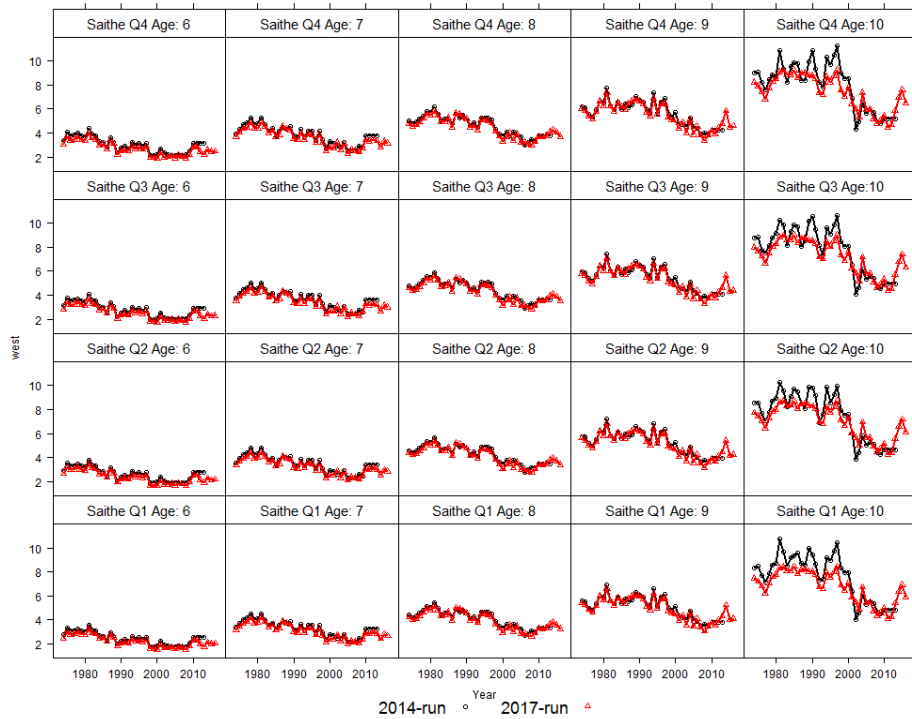
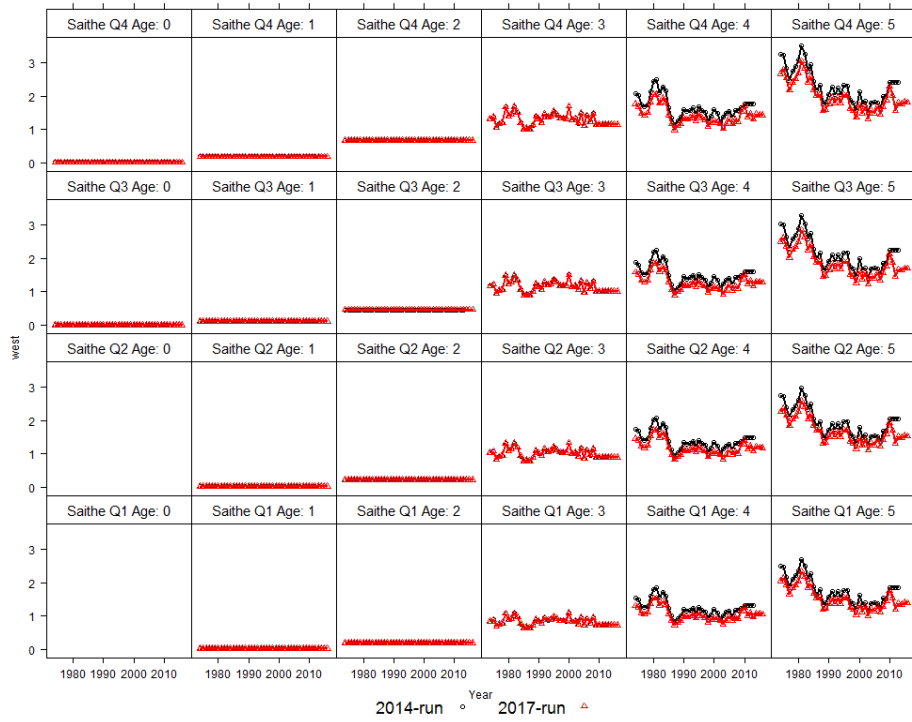
ξ	food ration exponent. Input
v	parameter for size dependent preference for other food. Estimated parameter
η_{PREF}	natural logarithm of the preferred predator prey size ratio. Estimated parameter
η_{MIN}	observed minimum relative prey size for a predator species. Input
η_{MAX}	observed maximum relative prey size for a predator species. Input
o	spatial overlap between predator and prey species. Estimated parameter
ρ	coefficient of species vulnerability. Estimated parameter
σ_{CATCH}	standard deviation of catch observations. Estimated parameter
σ_{PREF}	parameter expressing how particular a predator is about the size of its prey. Parameter
σ_{SR}	standard deviation of stock recruitment estimate. Estimated parameter
σ_{STOM}	standard deviation of stomach content observations (used with log normal distribution)
σ_{SURVEY}	standard deviation of survey cpue observations. Estimated parameter

APPENDIX 2: Mean weight at age in the sea

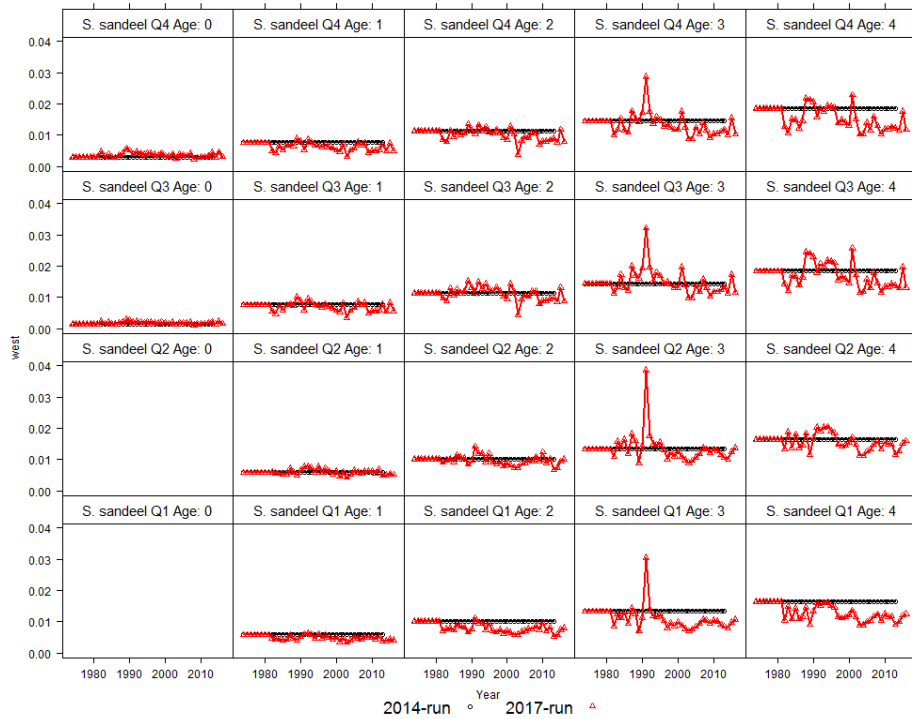
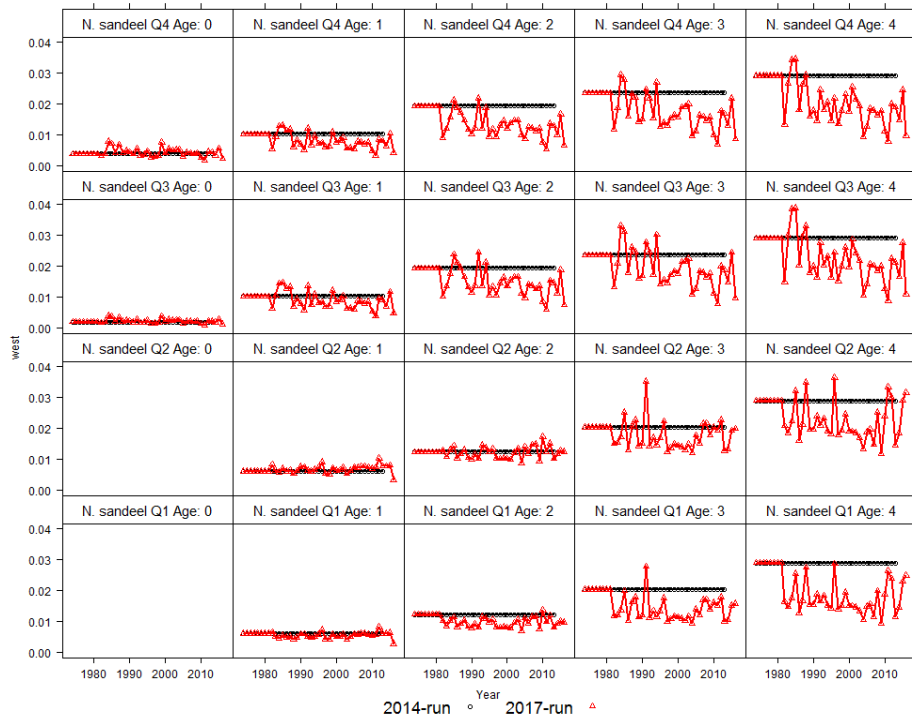


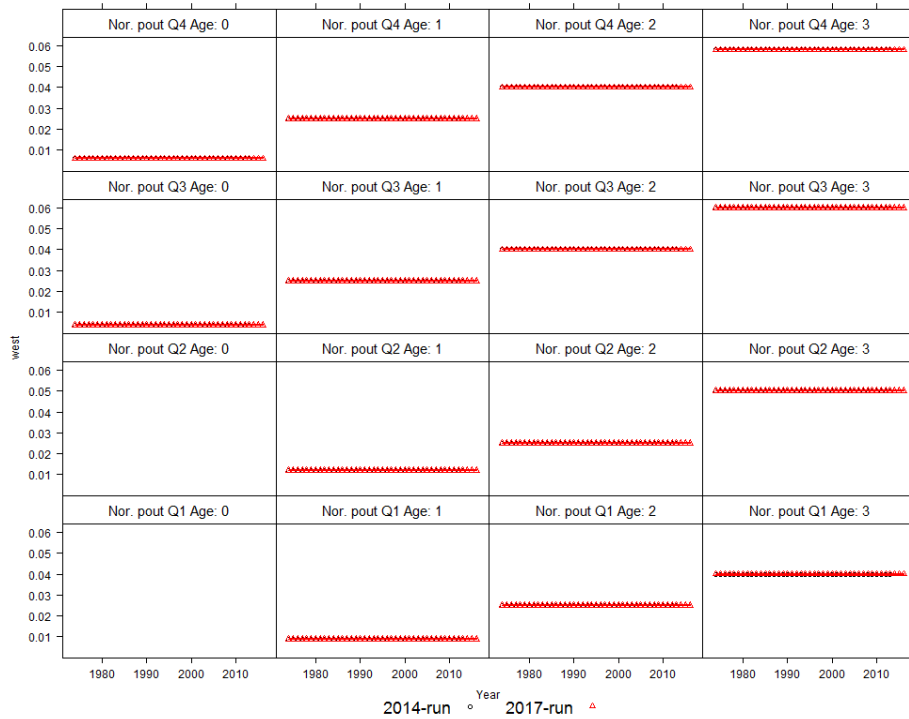
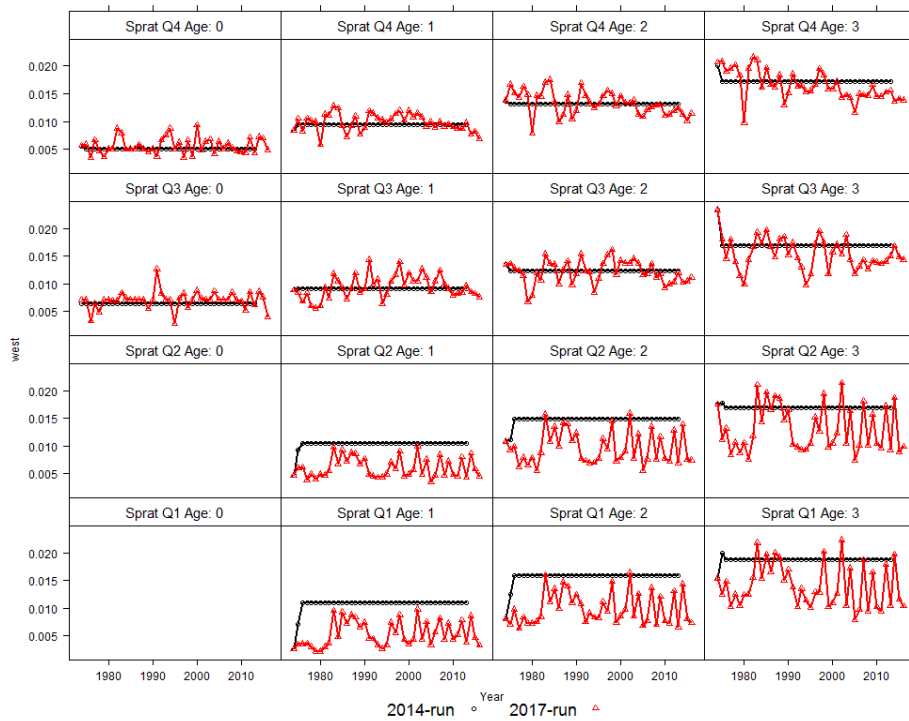


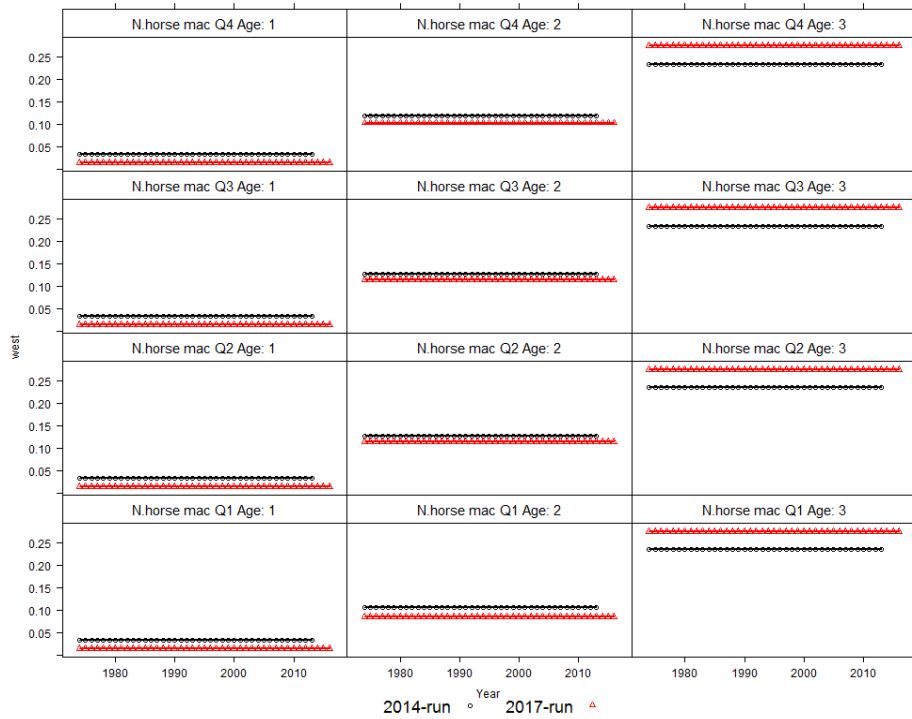
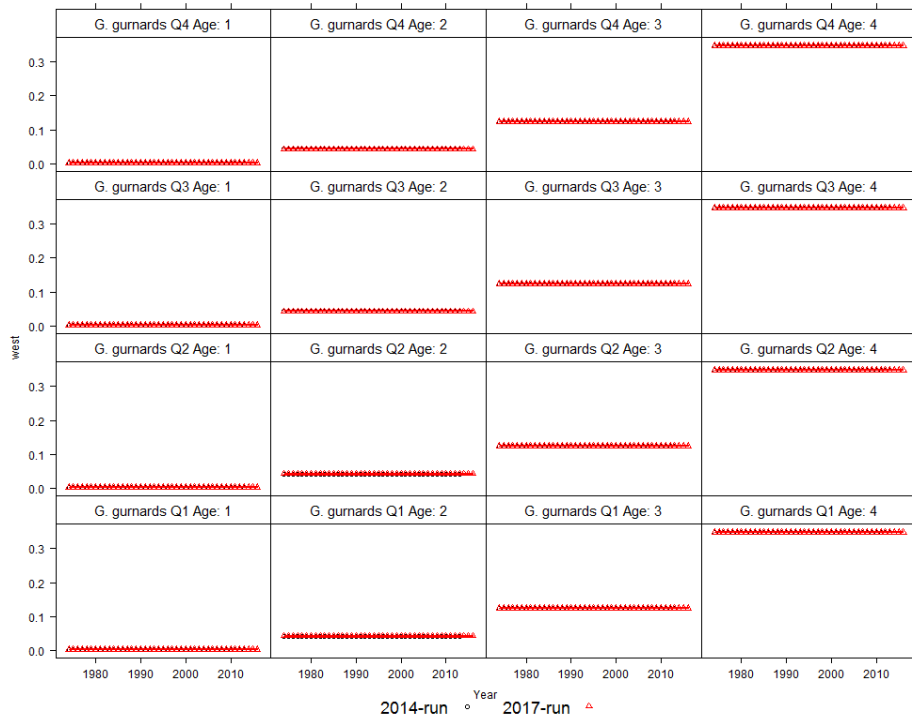


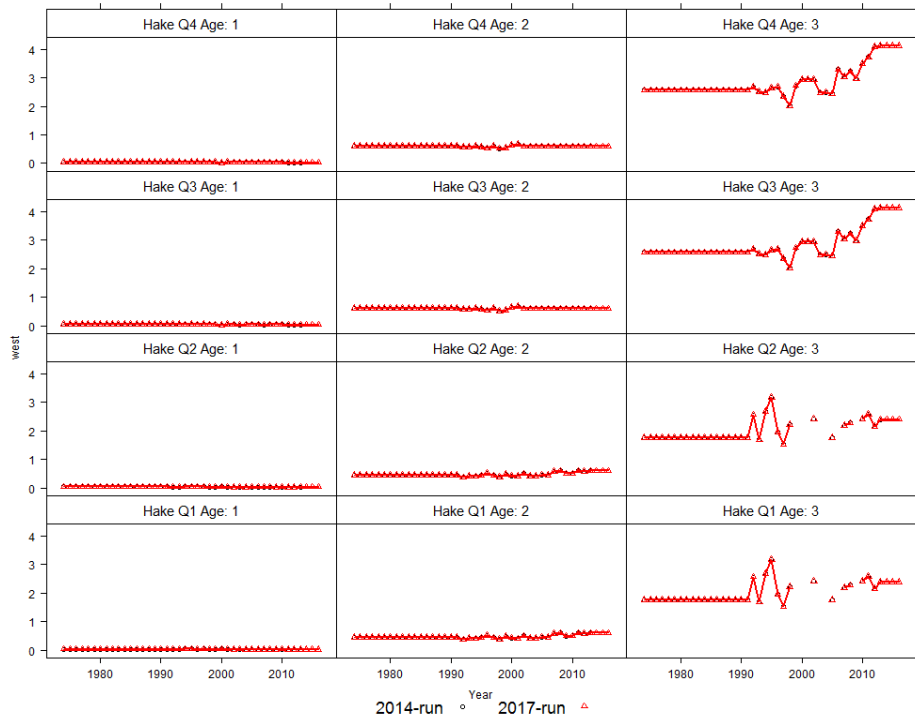






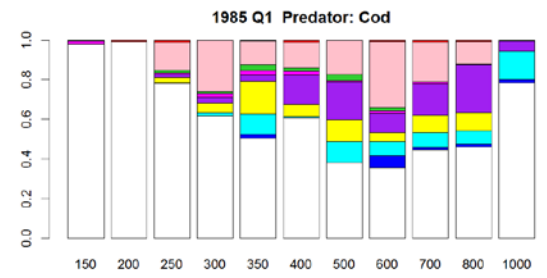
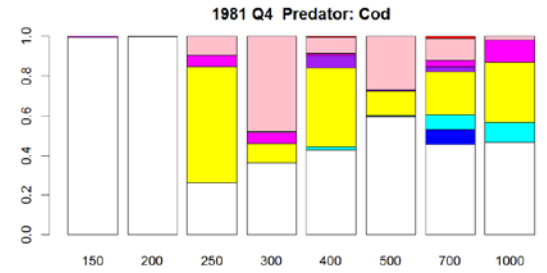
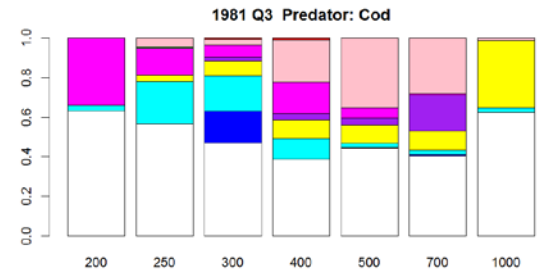
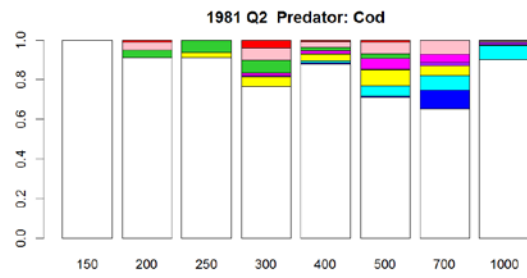
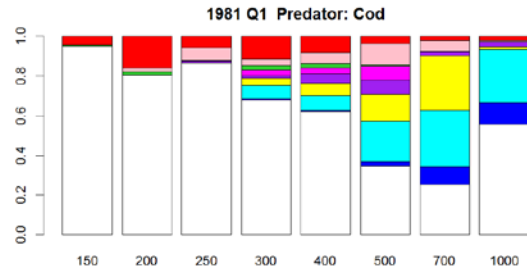
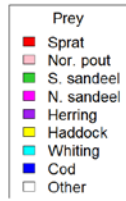


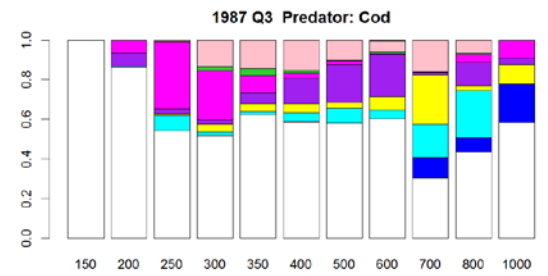
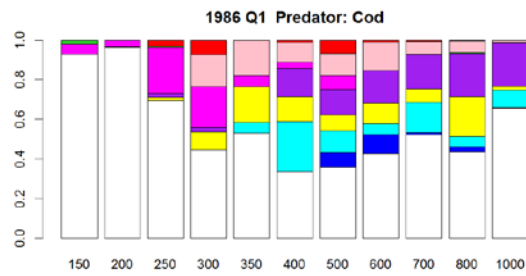
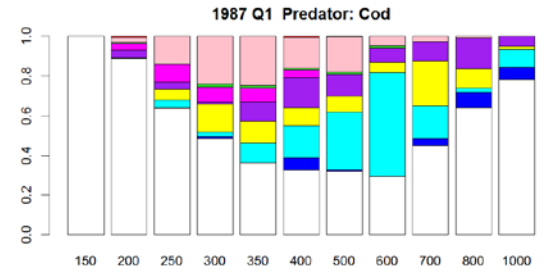
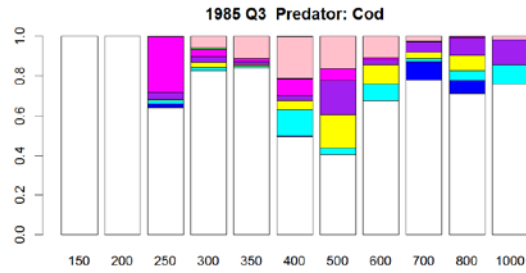
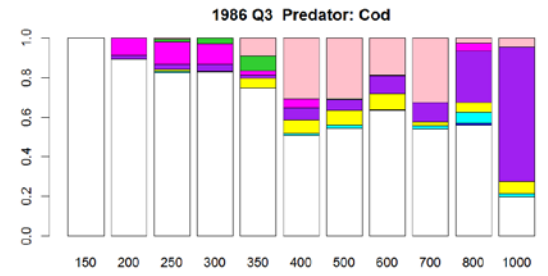
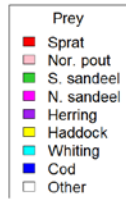


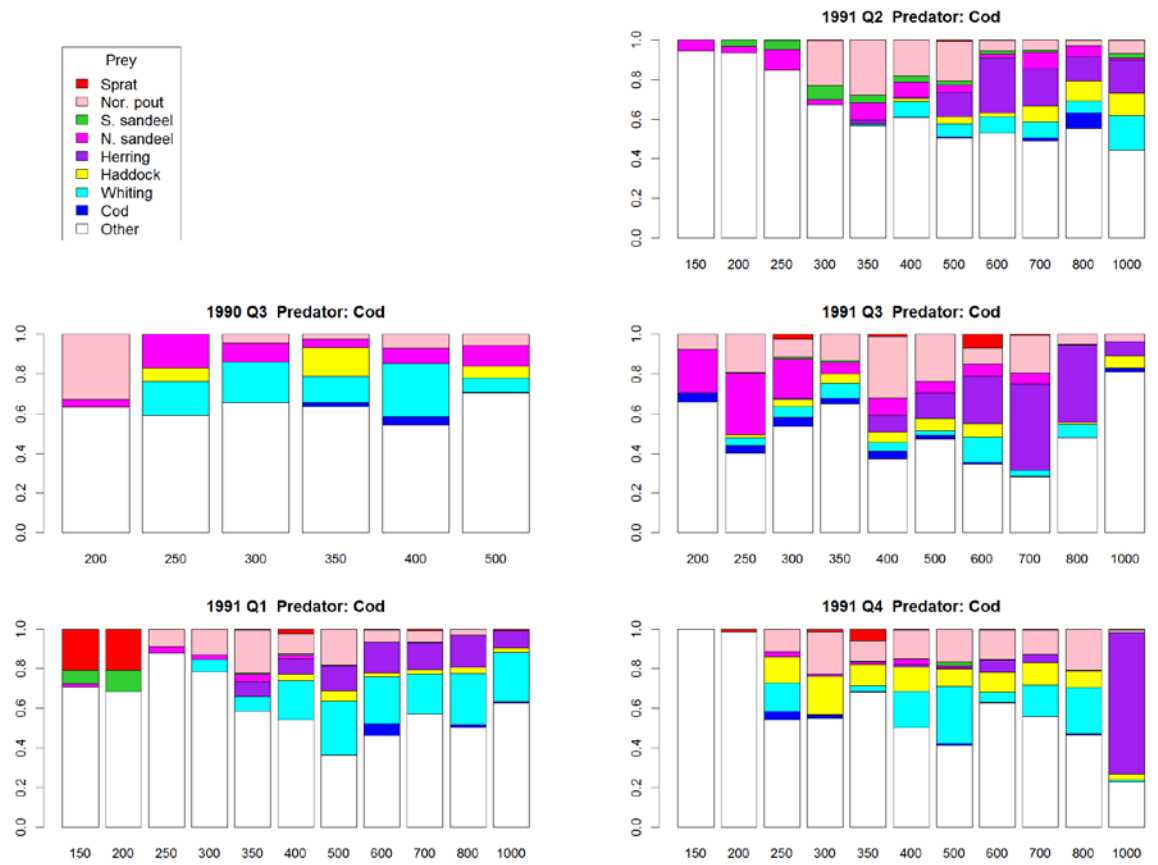


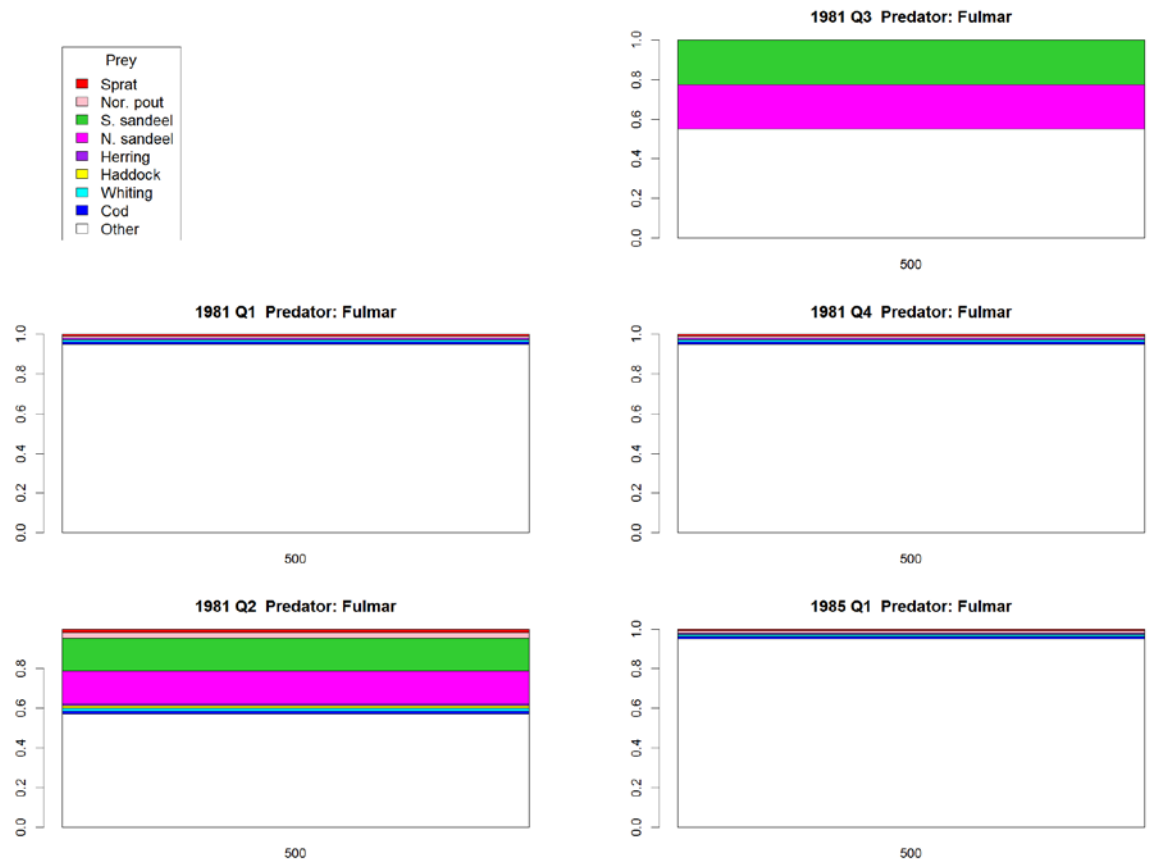
APPENDIX 3: Diet composition used in the model

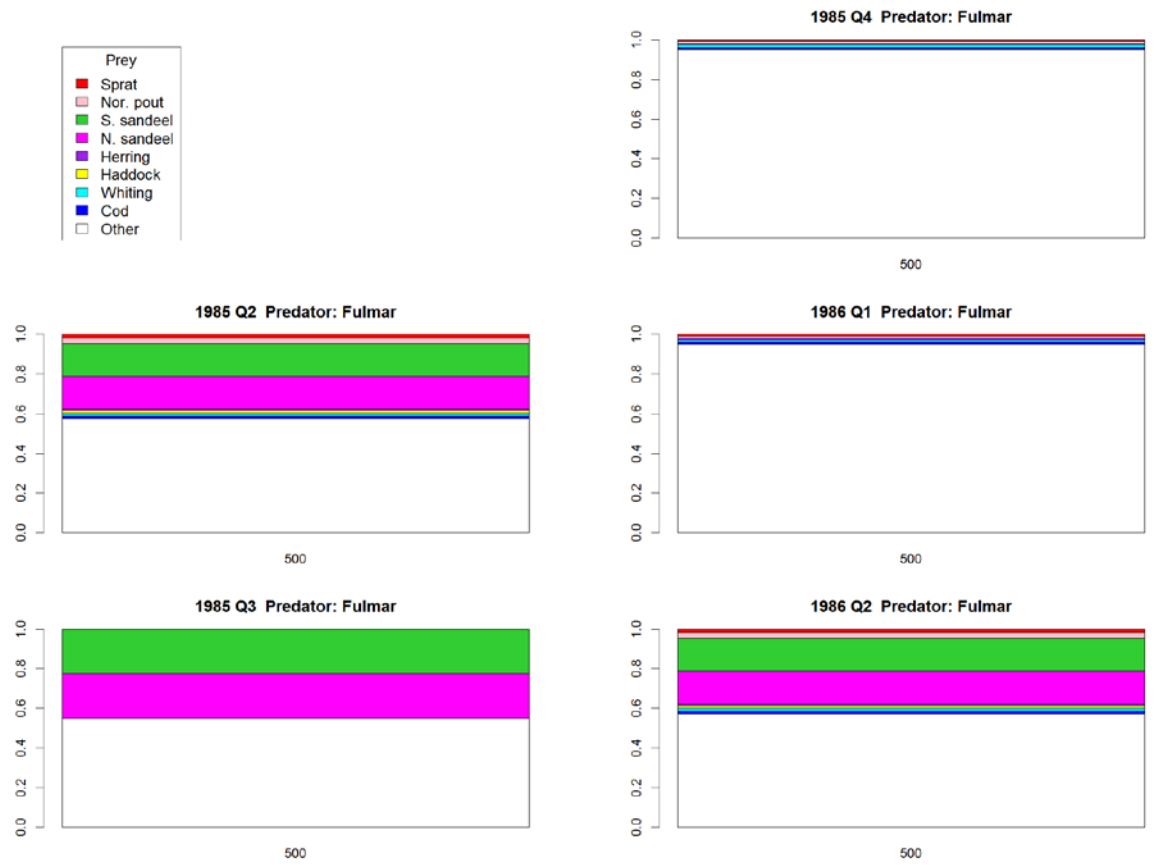
The following figures show the stomach content composition of fish and the diet composition (after correction of stomach contents for evacuation rate differences) for mammals. For each predator the stomach content is shown by observed predator size classes (showing the lower length in mm for the size class) or by dummy size class (birds and marine mammals). On the figures, all length classes of preys are merged. An example of stomach content, including prey size classes are shown in the table in the end of this appendix.

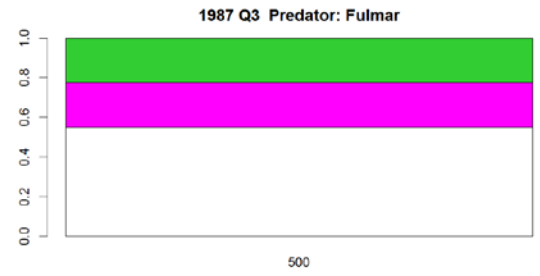
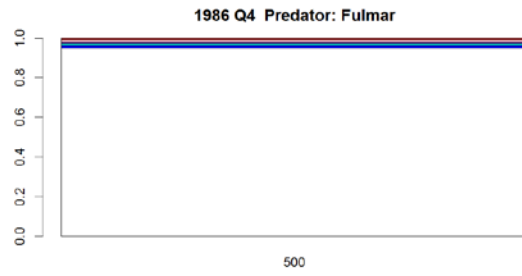
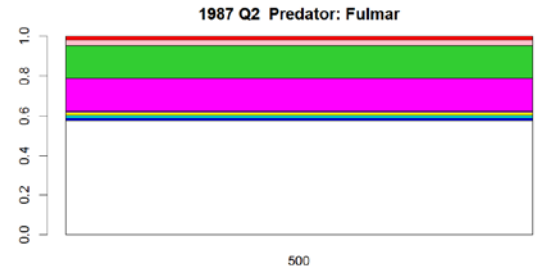
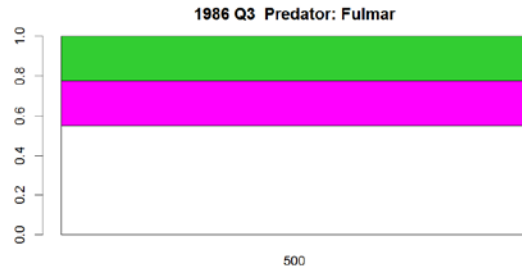
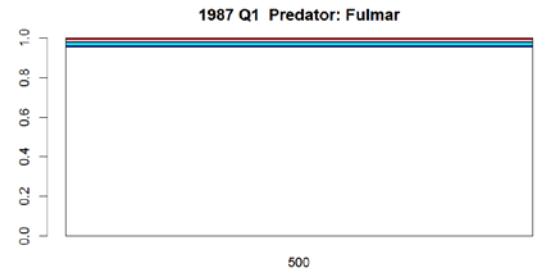
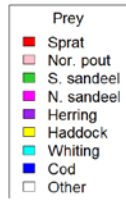


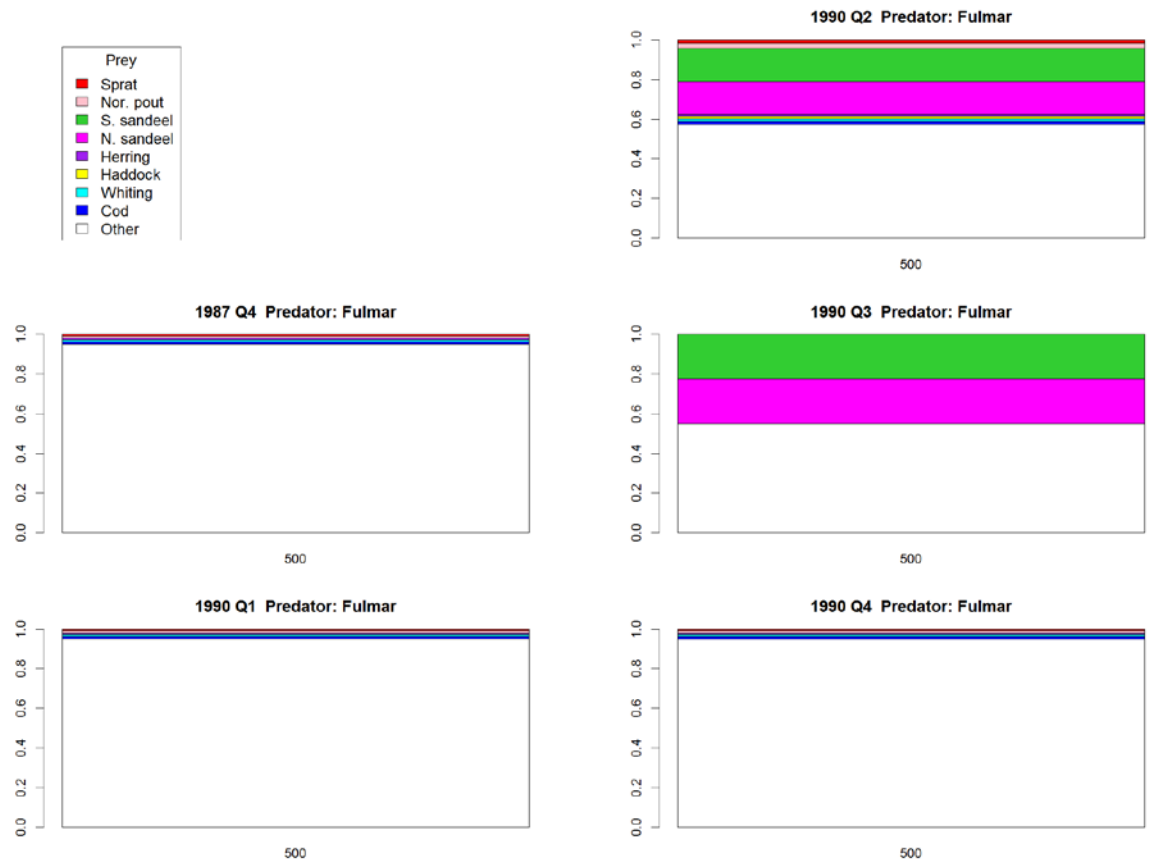


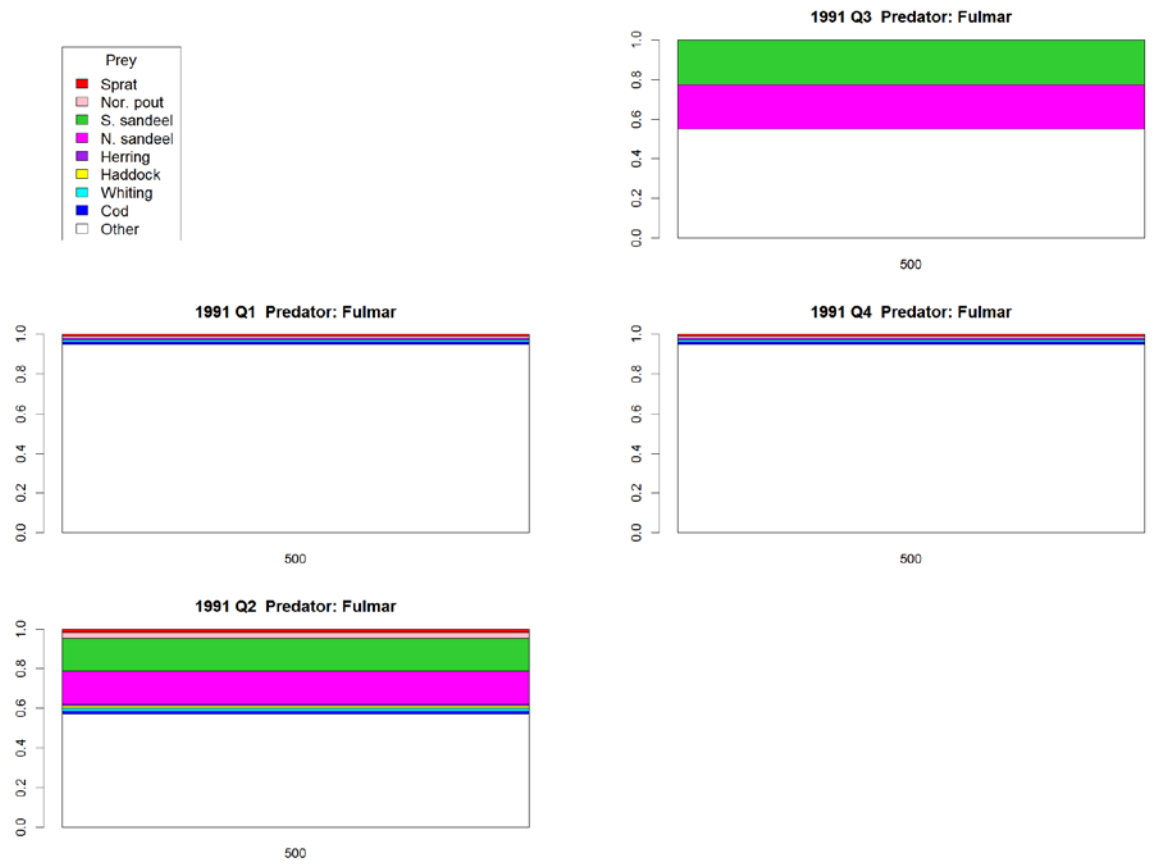


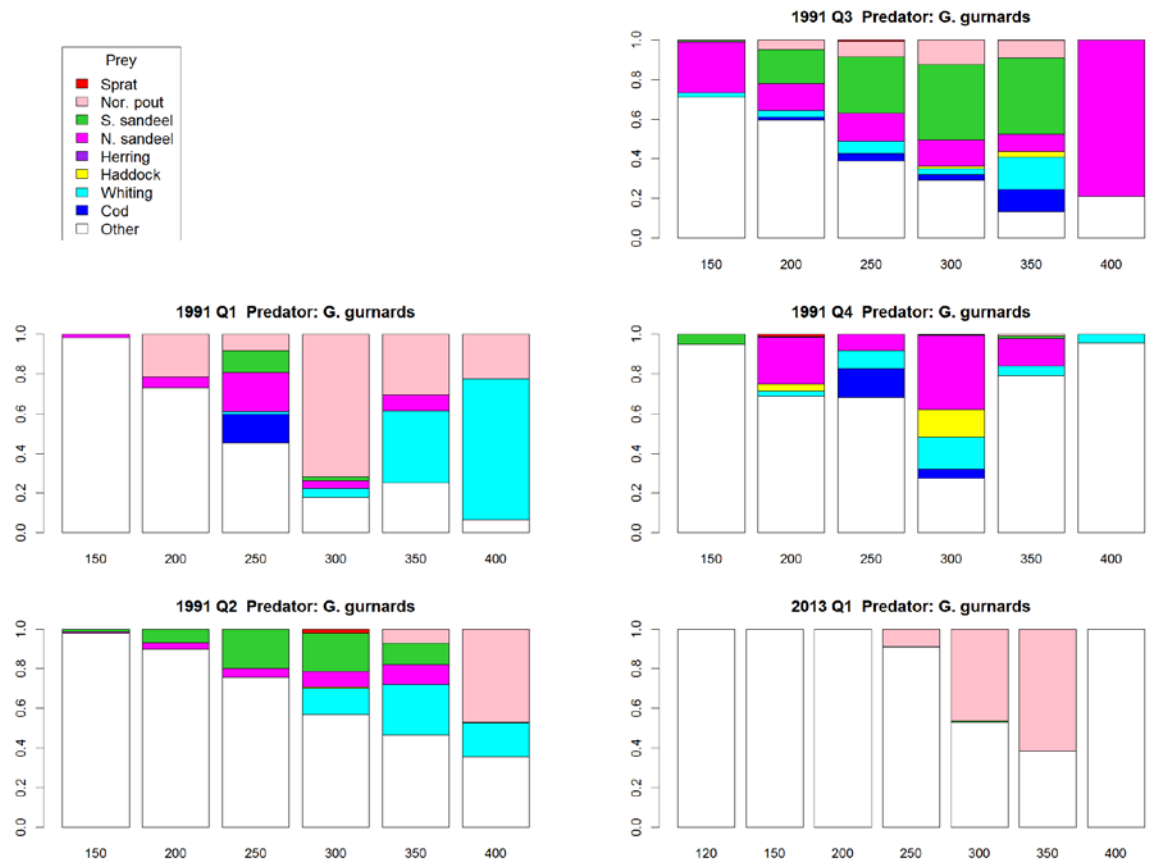


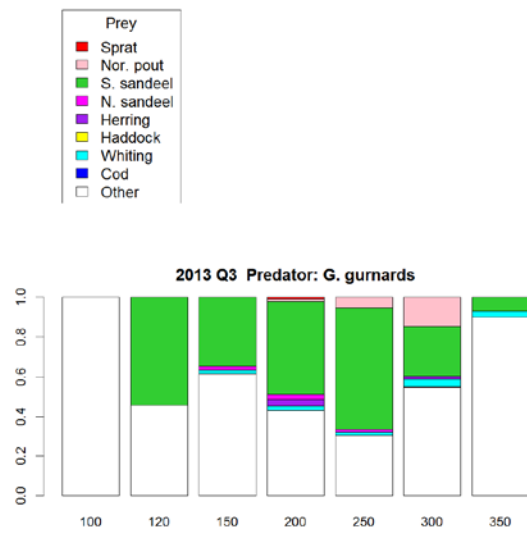


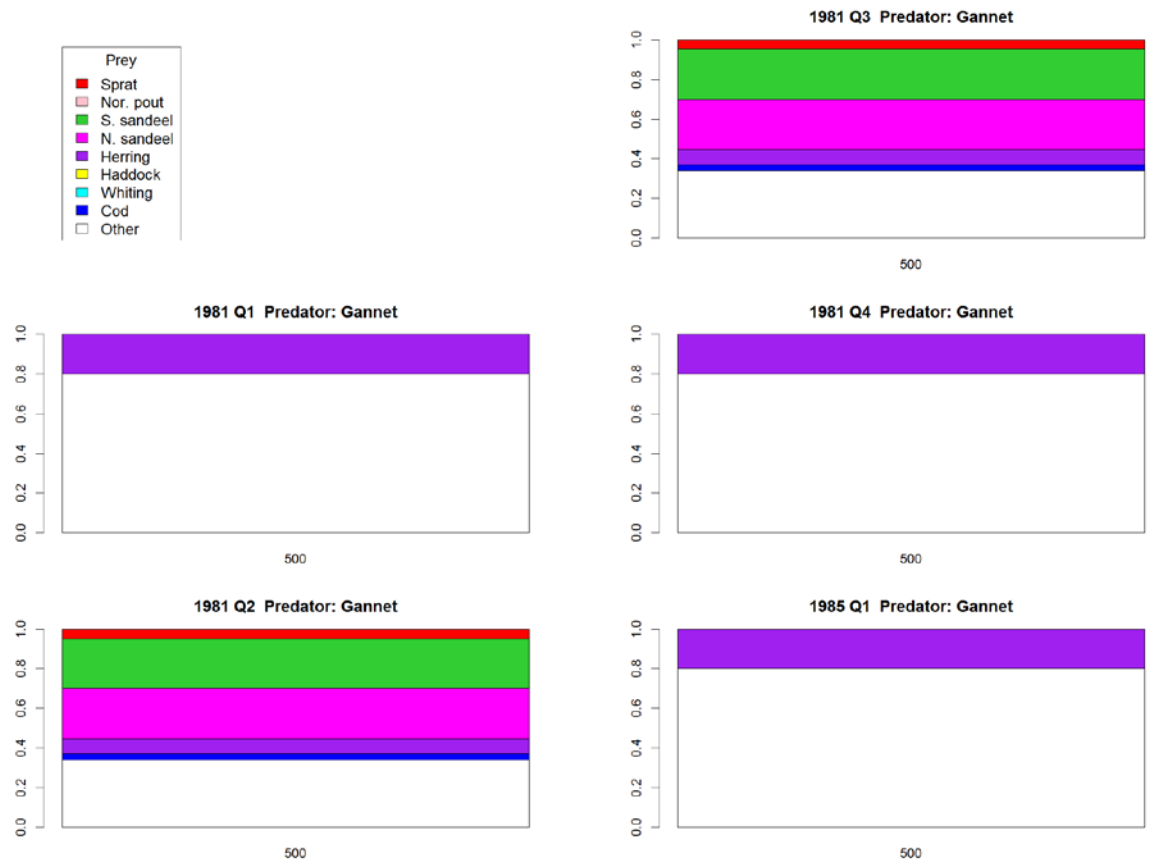


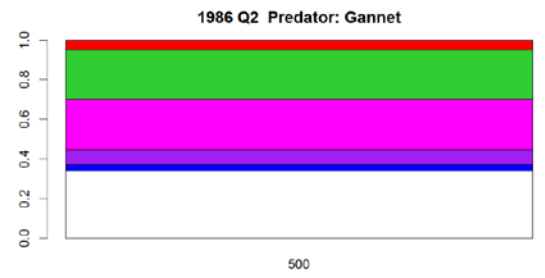
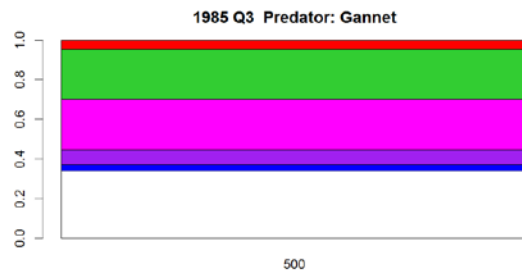
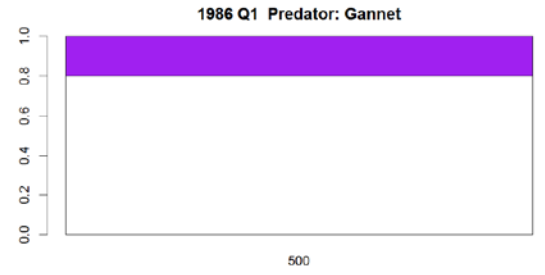
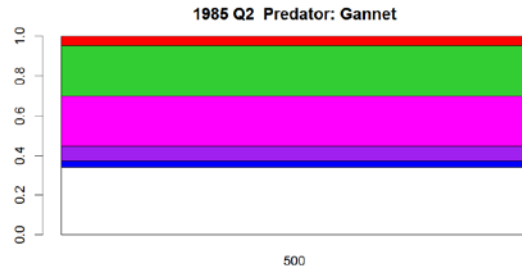
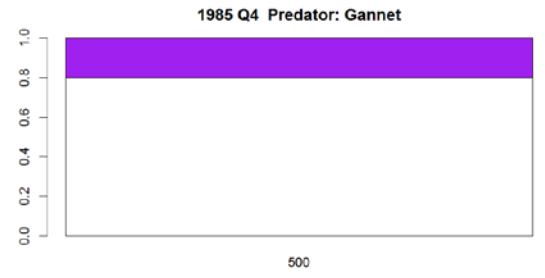
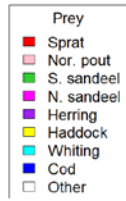


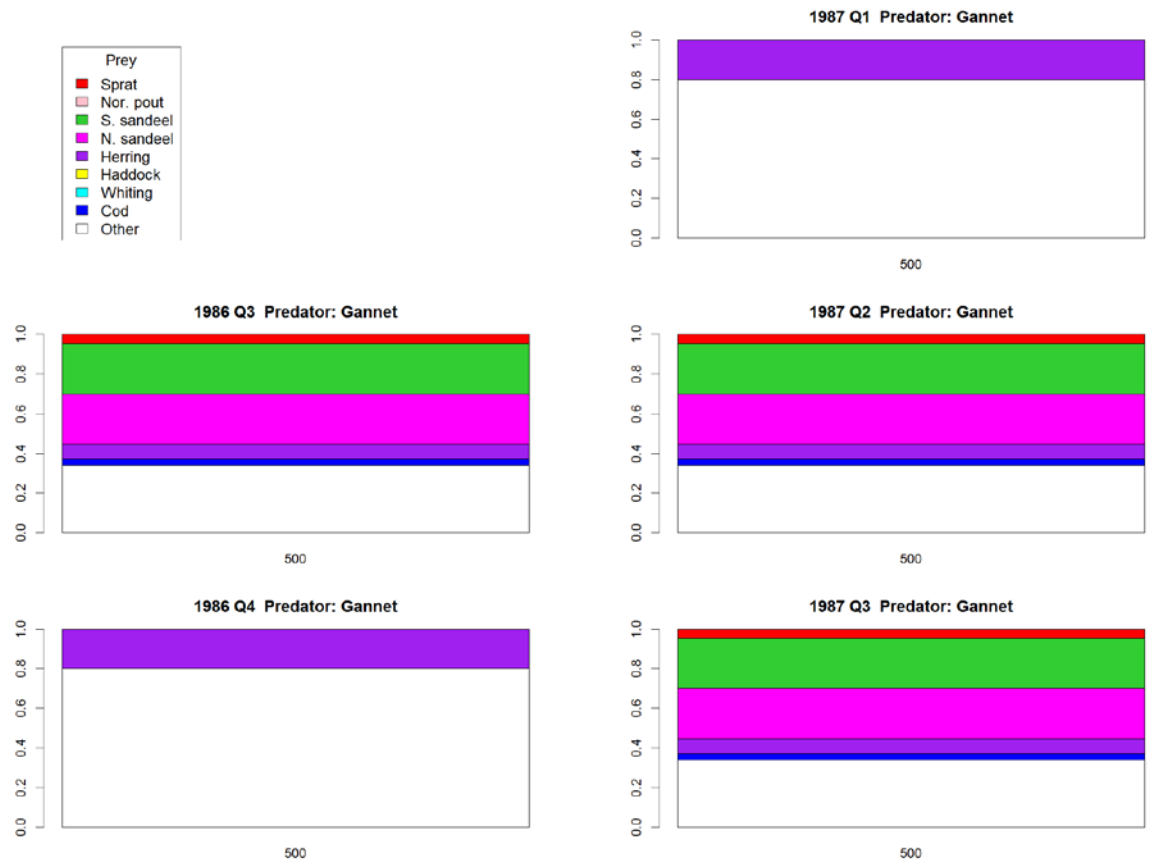


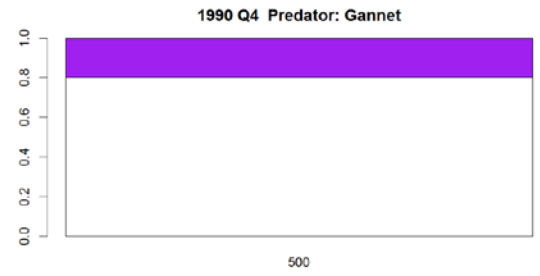
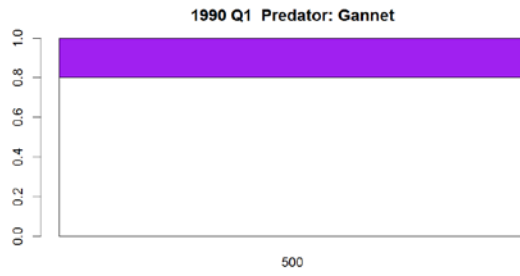
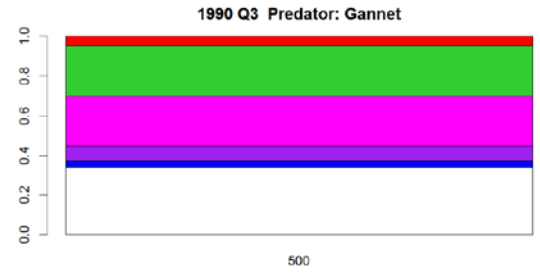
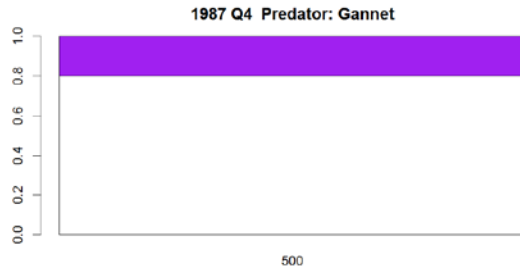
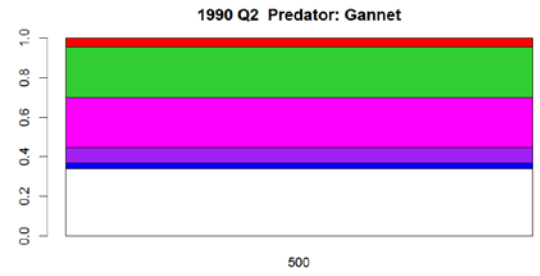
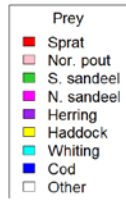


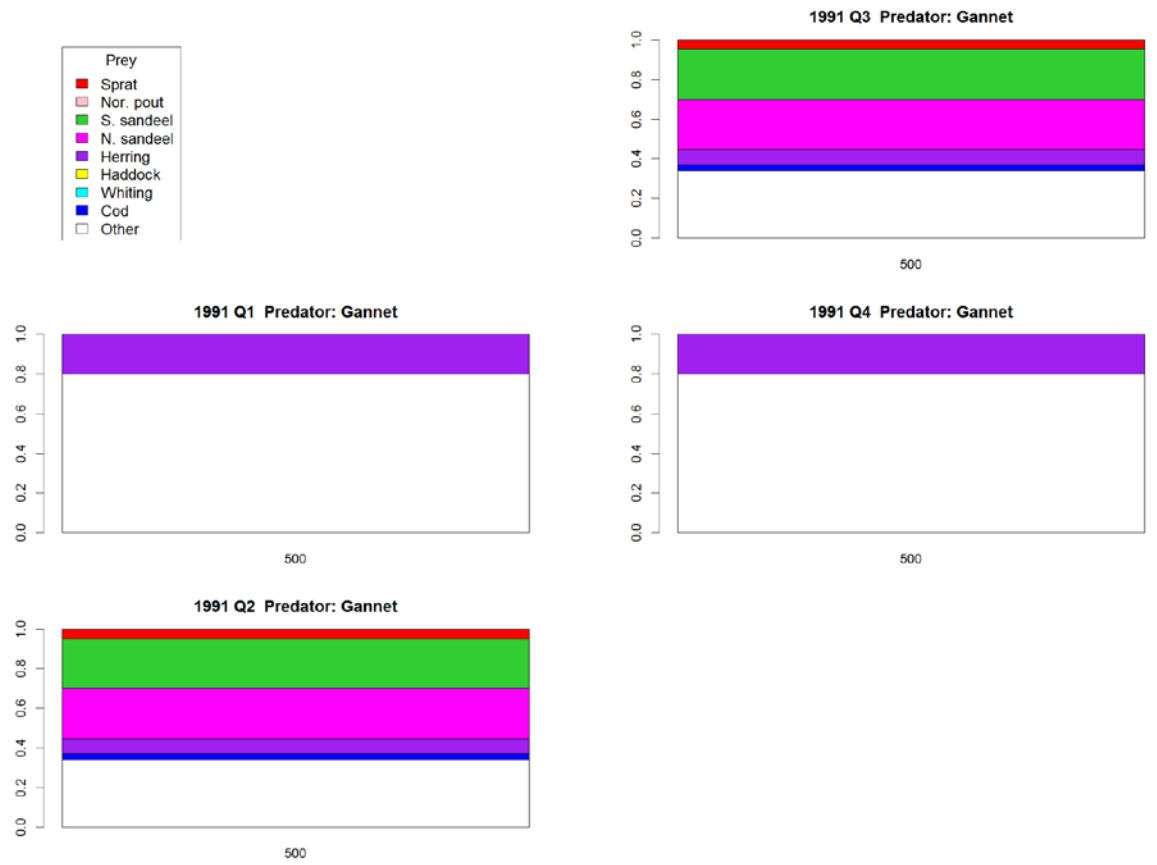


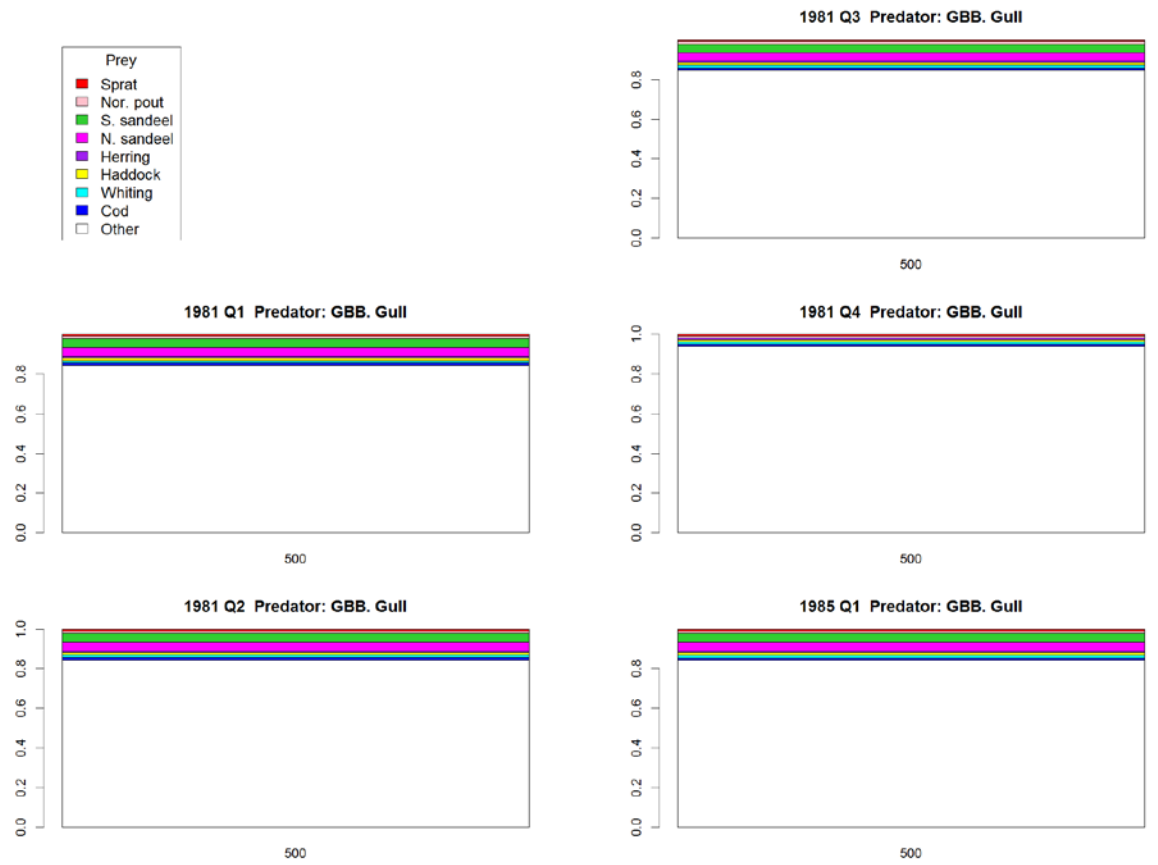


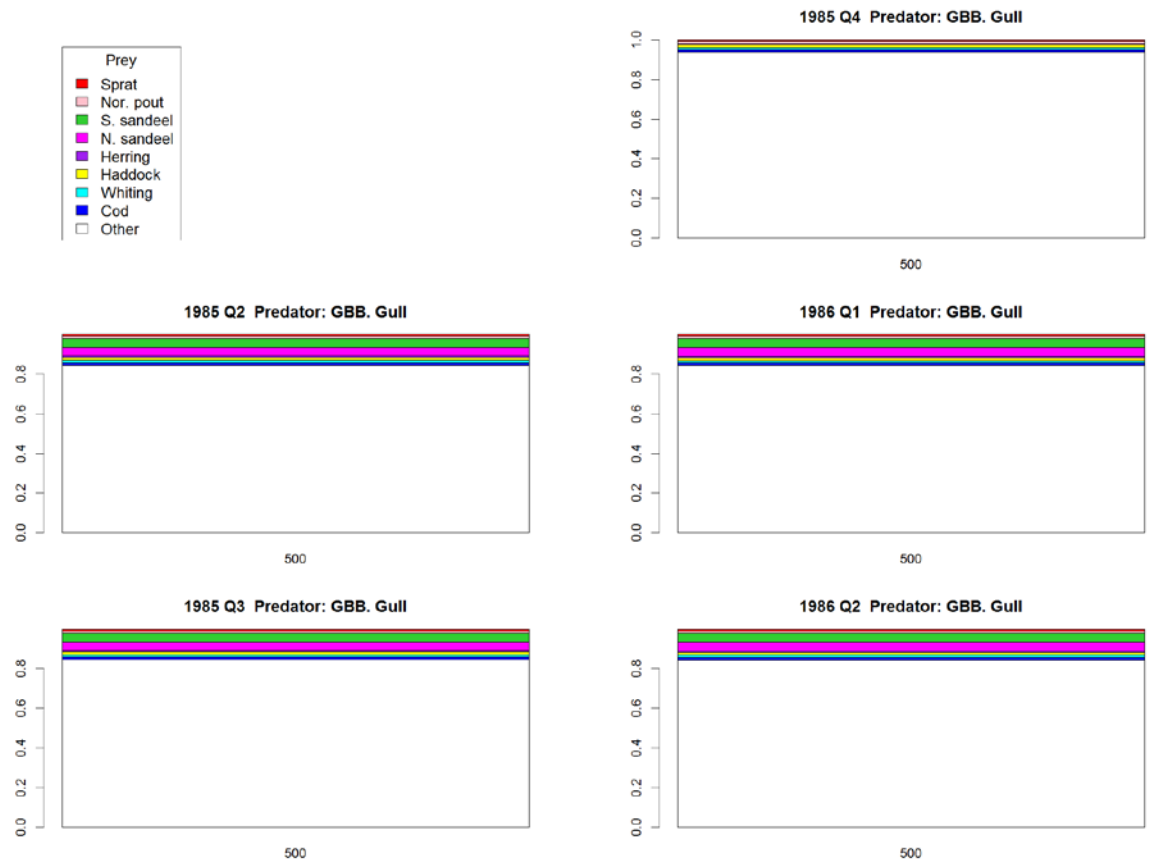


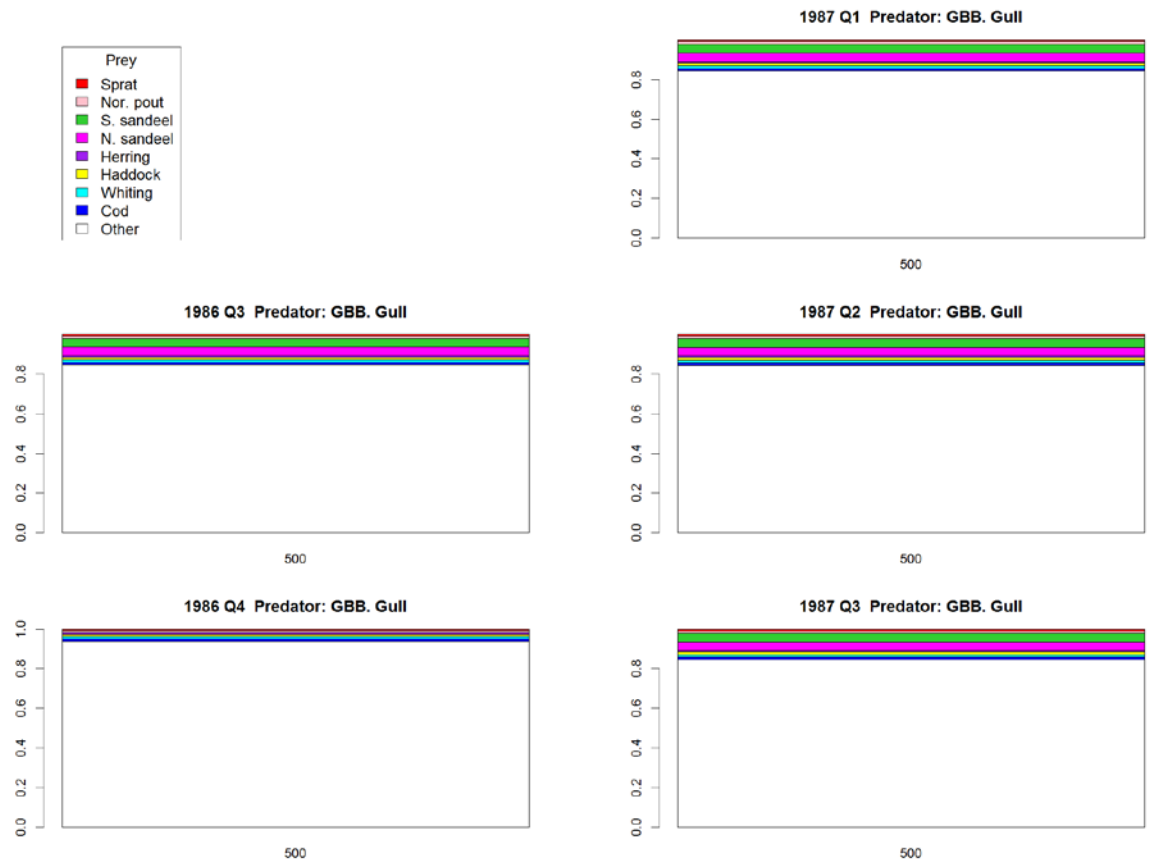


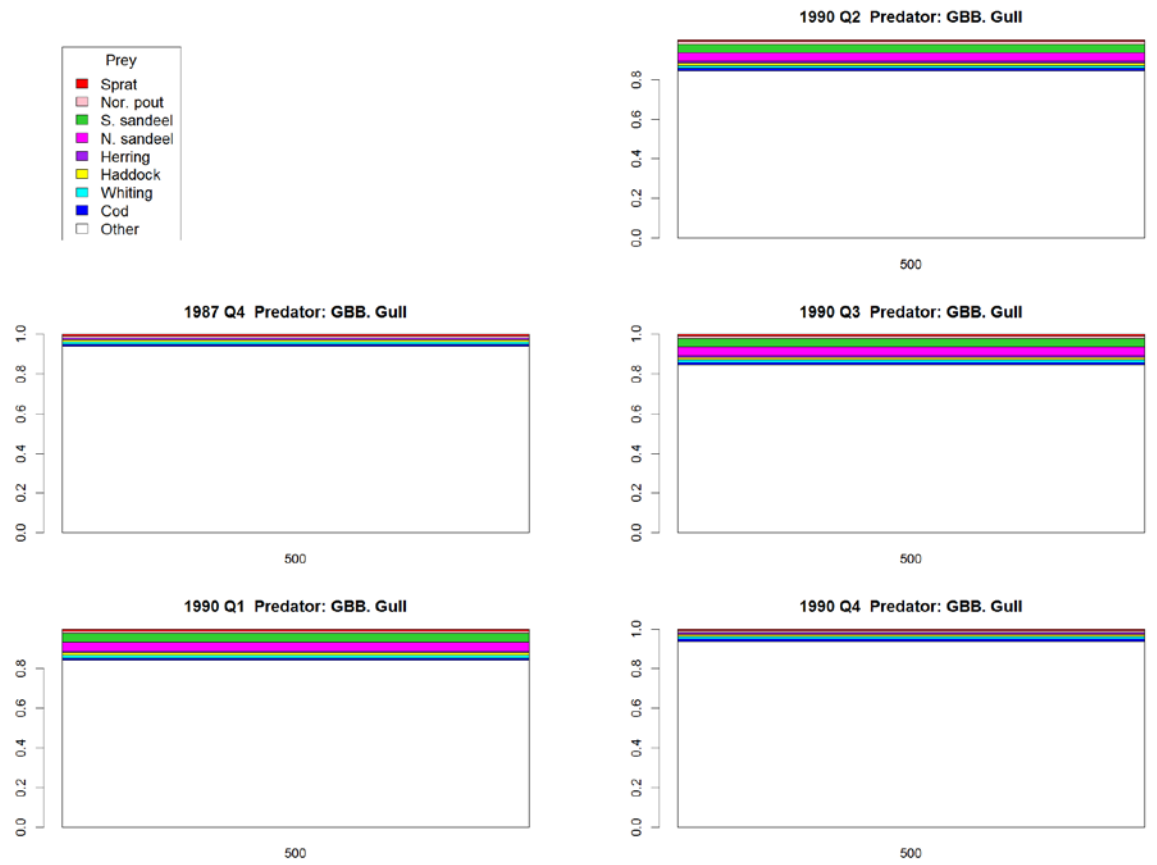


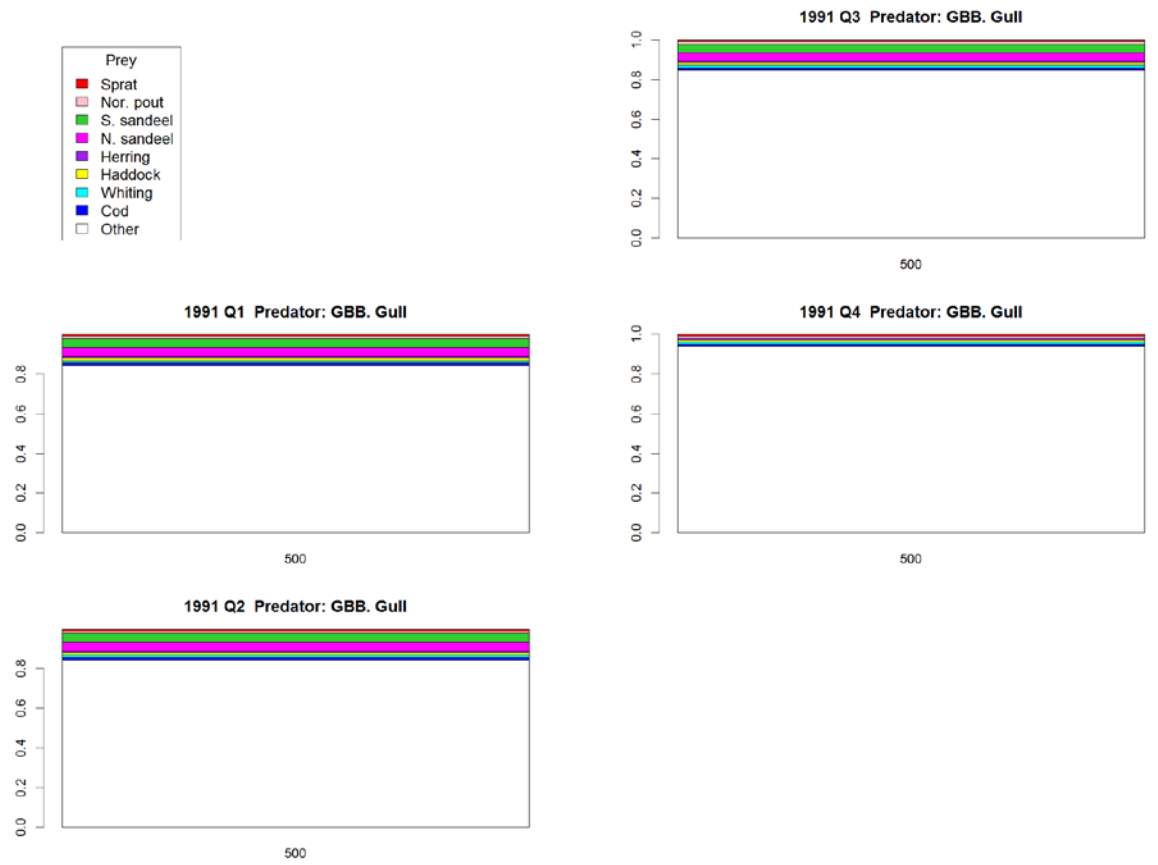


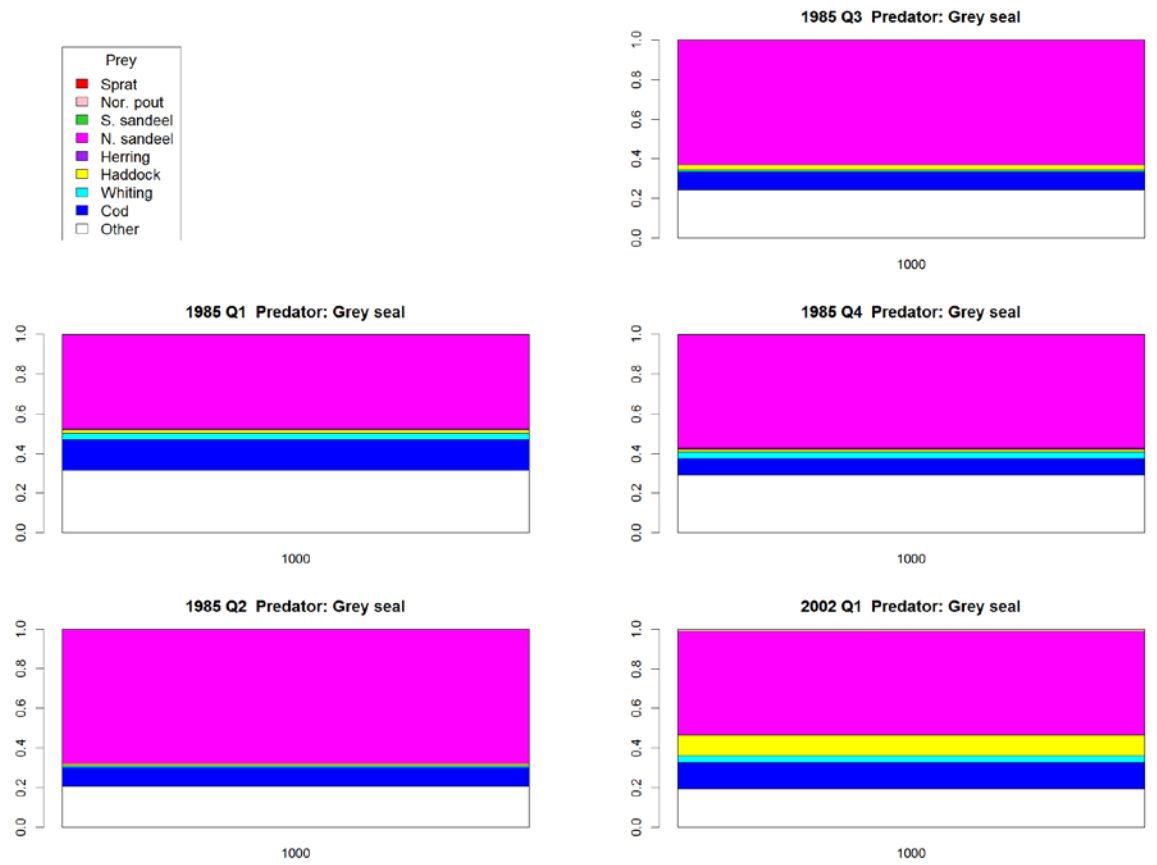


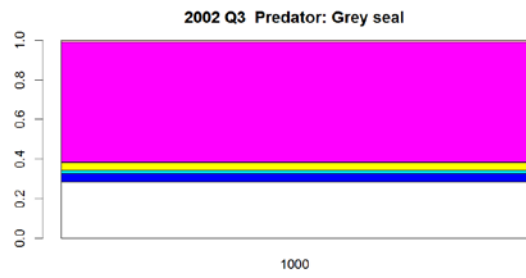
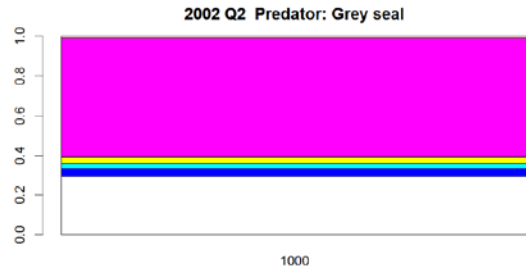
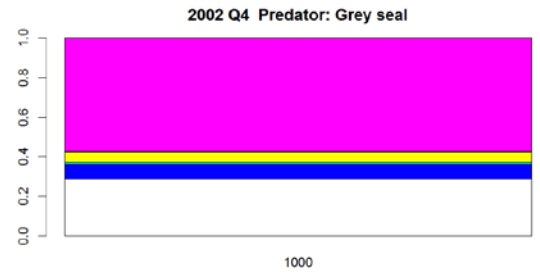
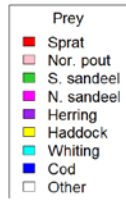


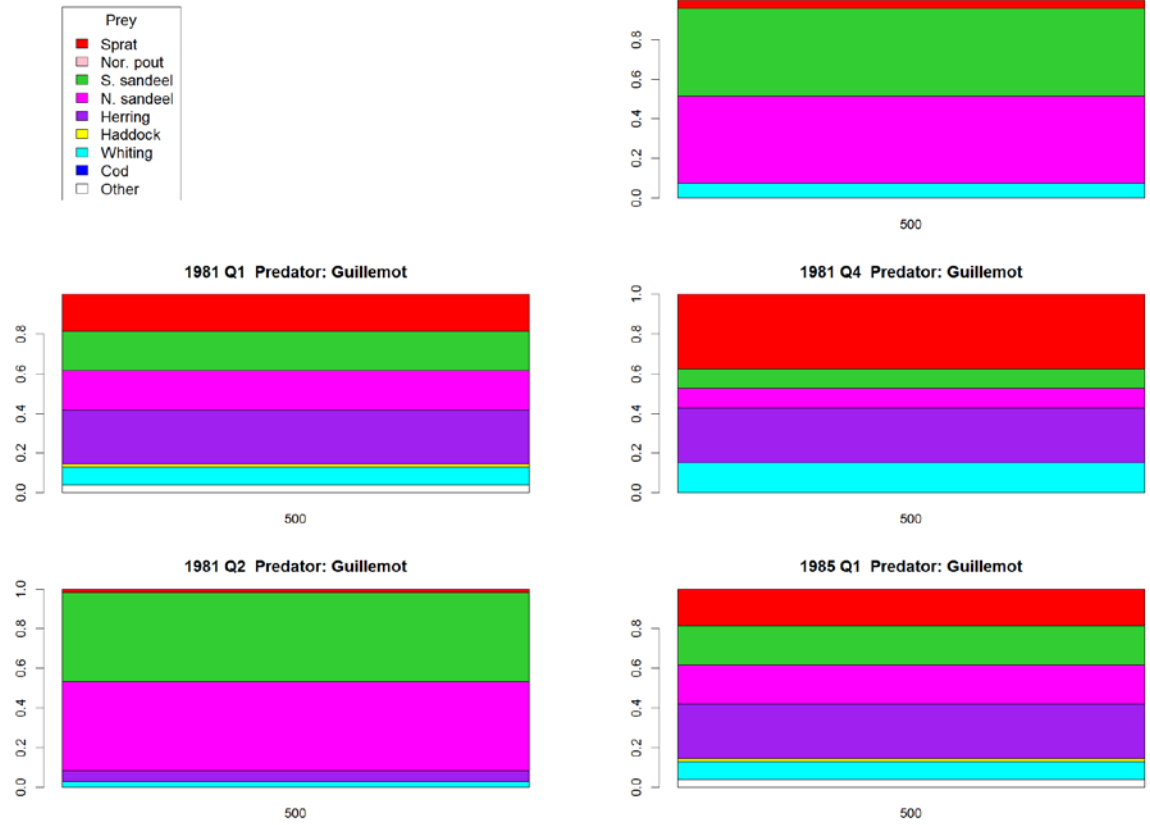


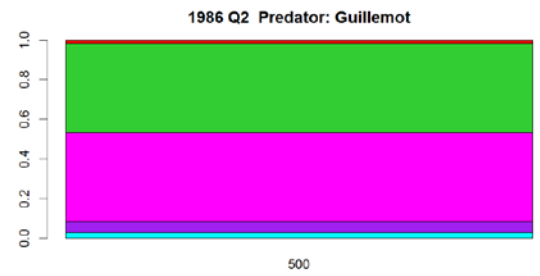
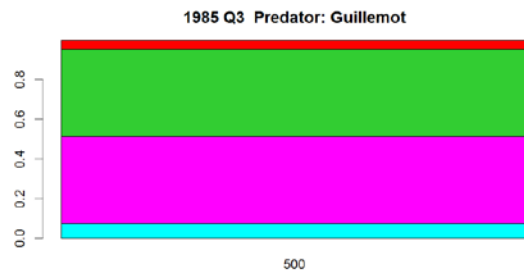
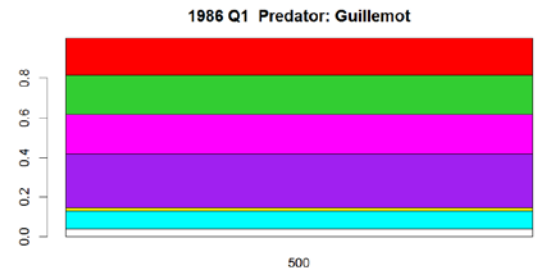
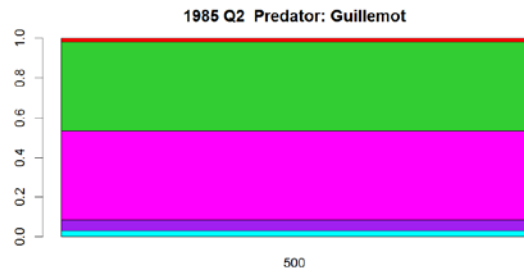
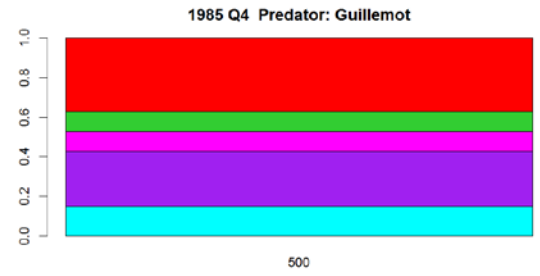
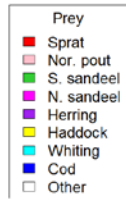


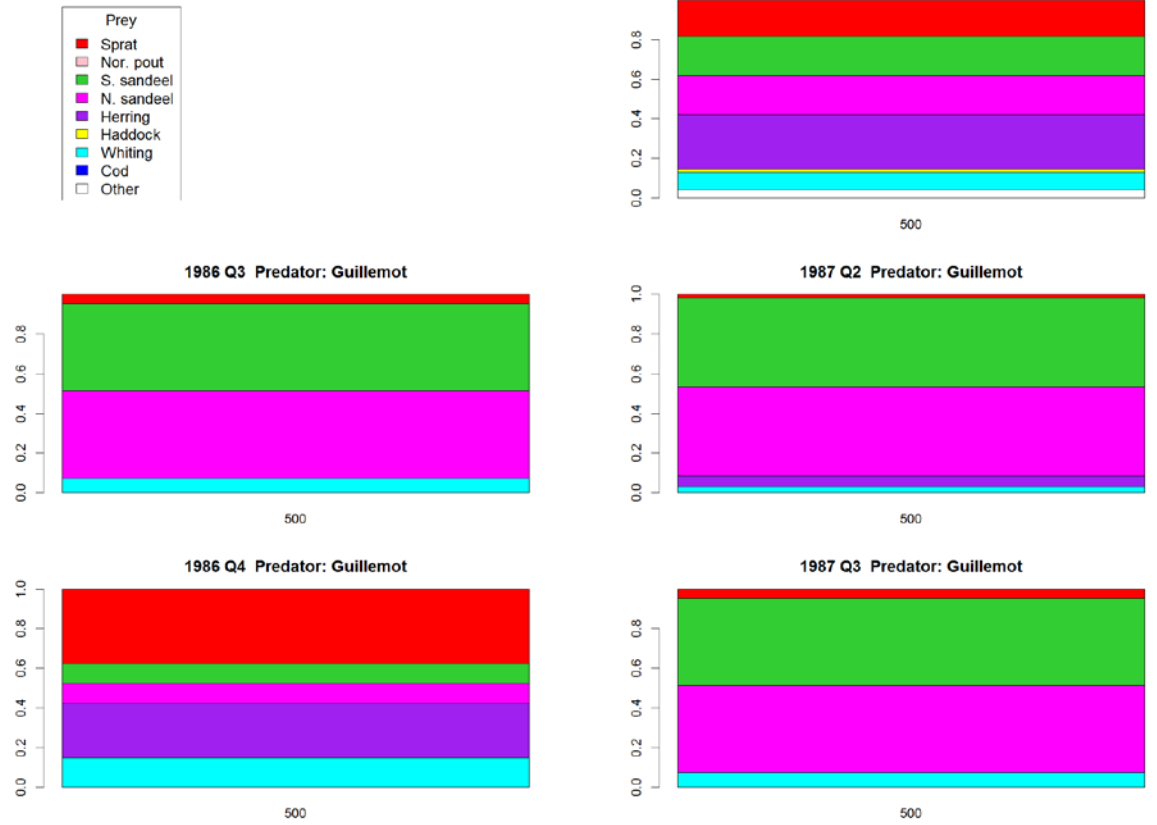


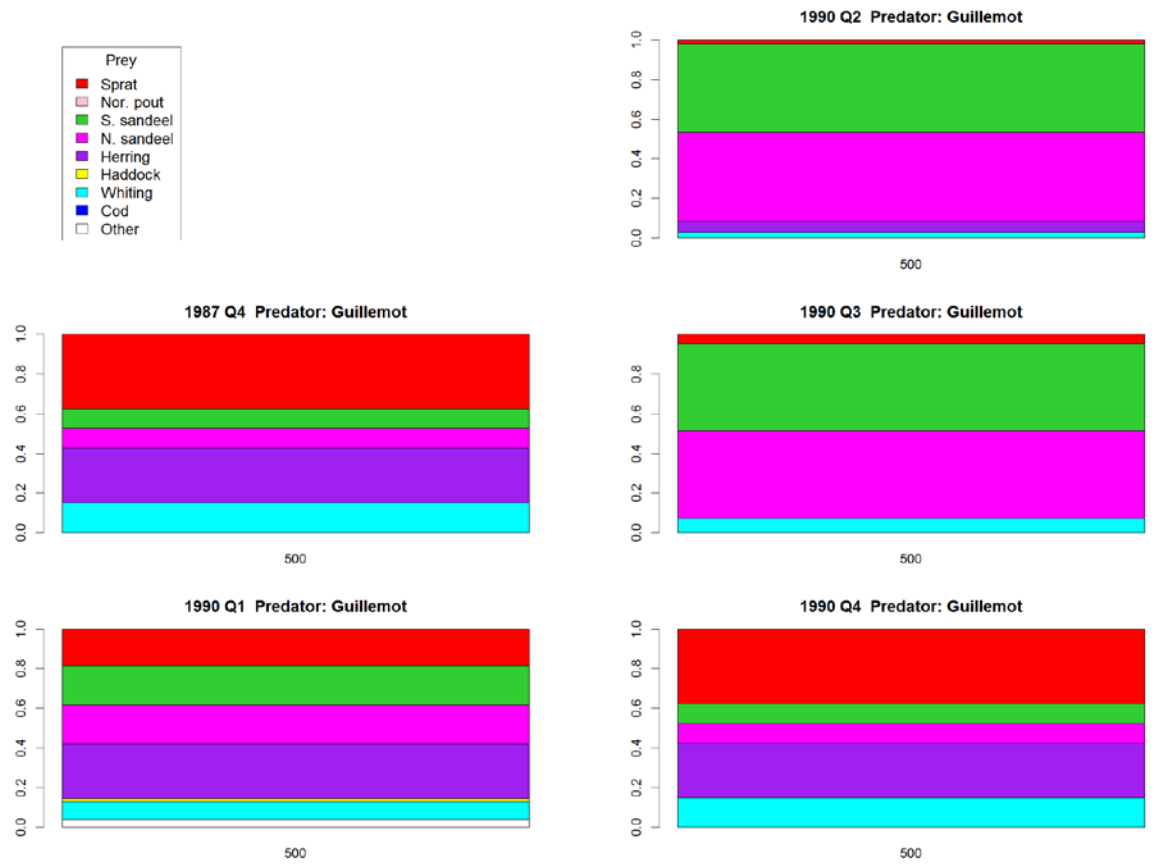


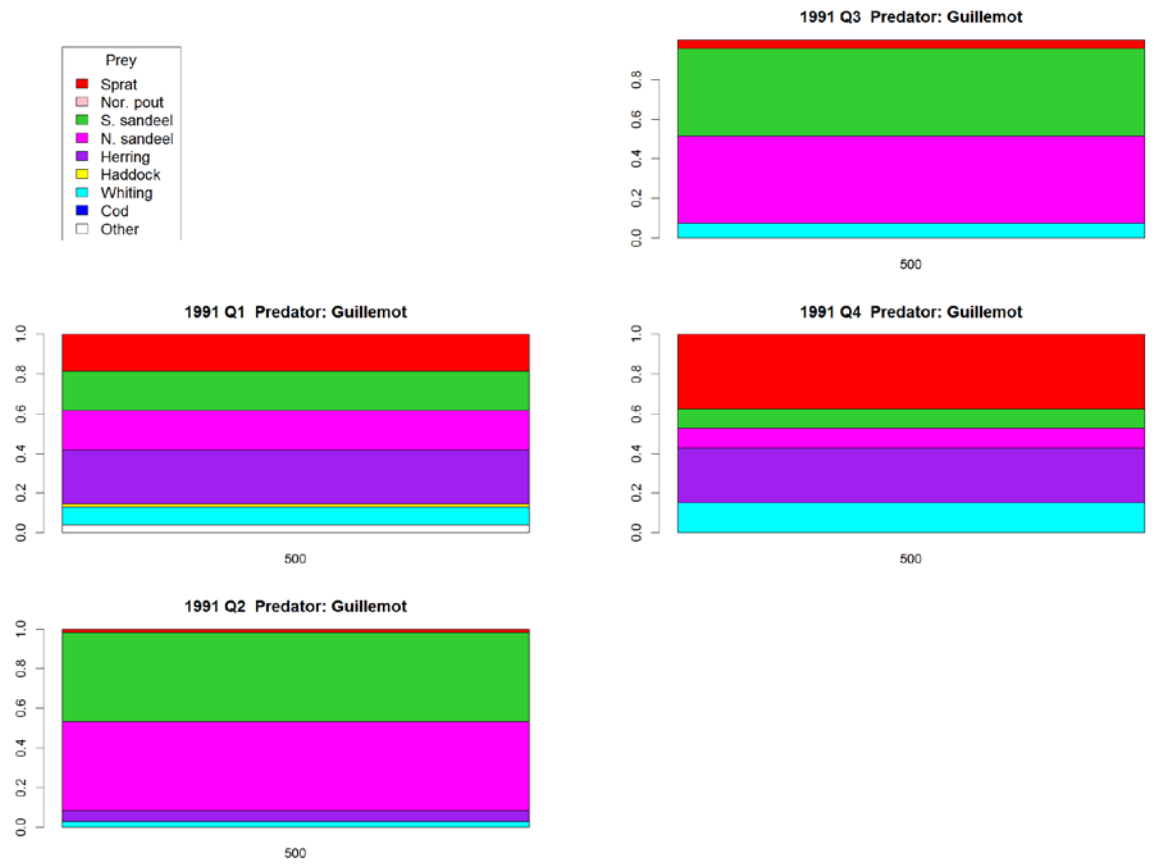


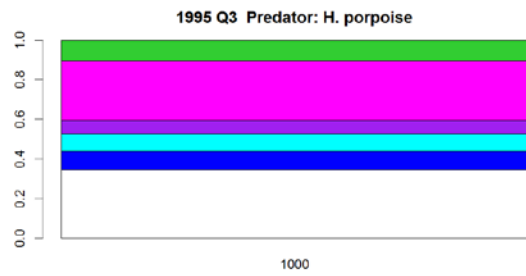
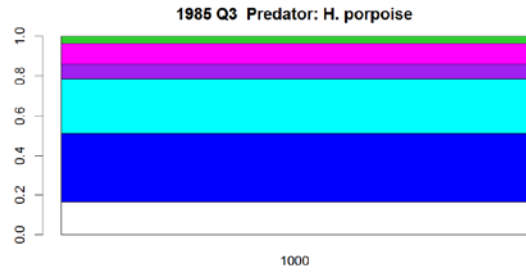
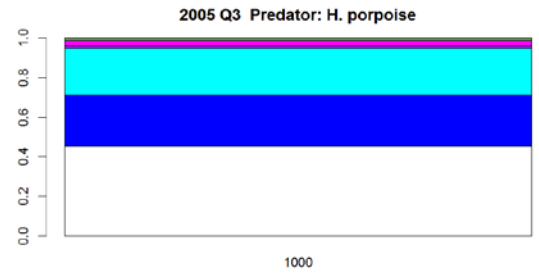
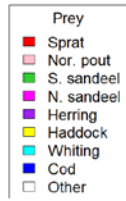


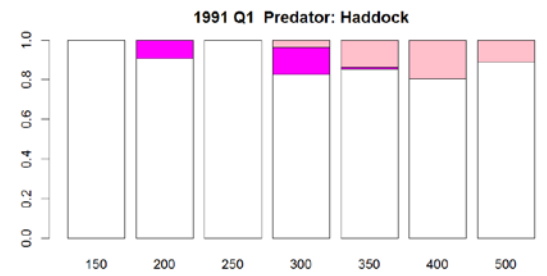
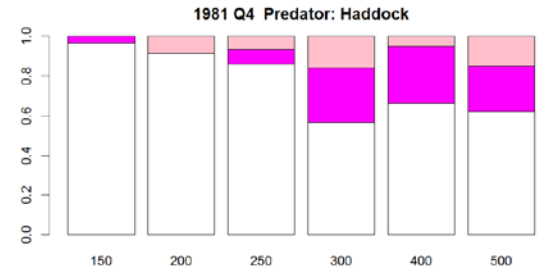
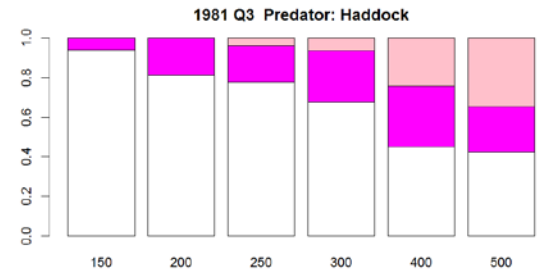
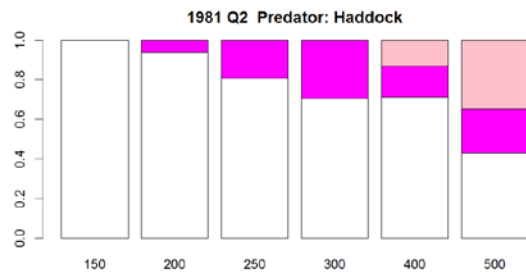
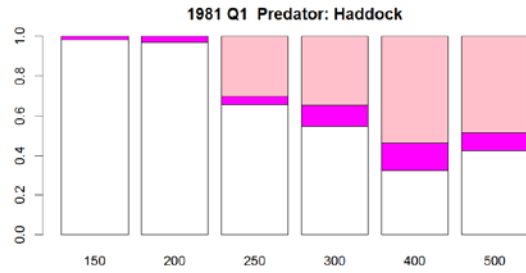
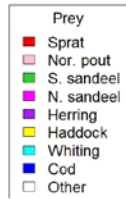


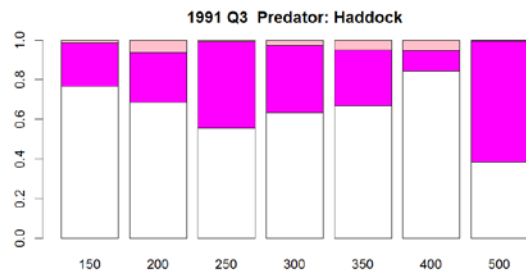
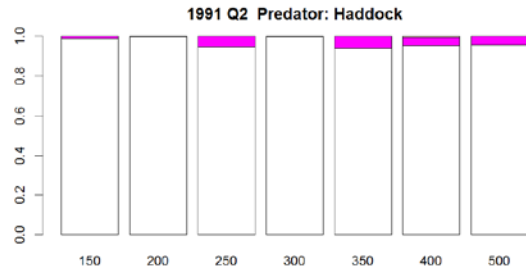
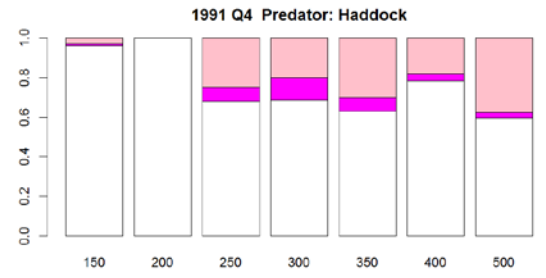
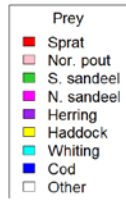


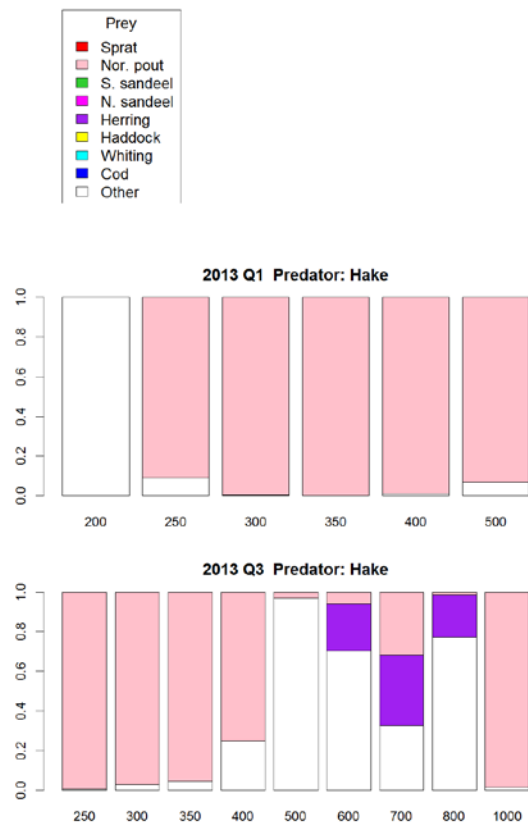


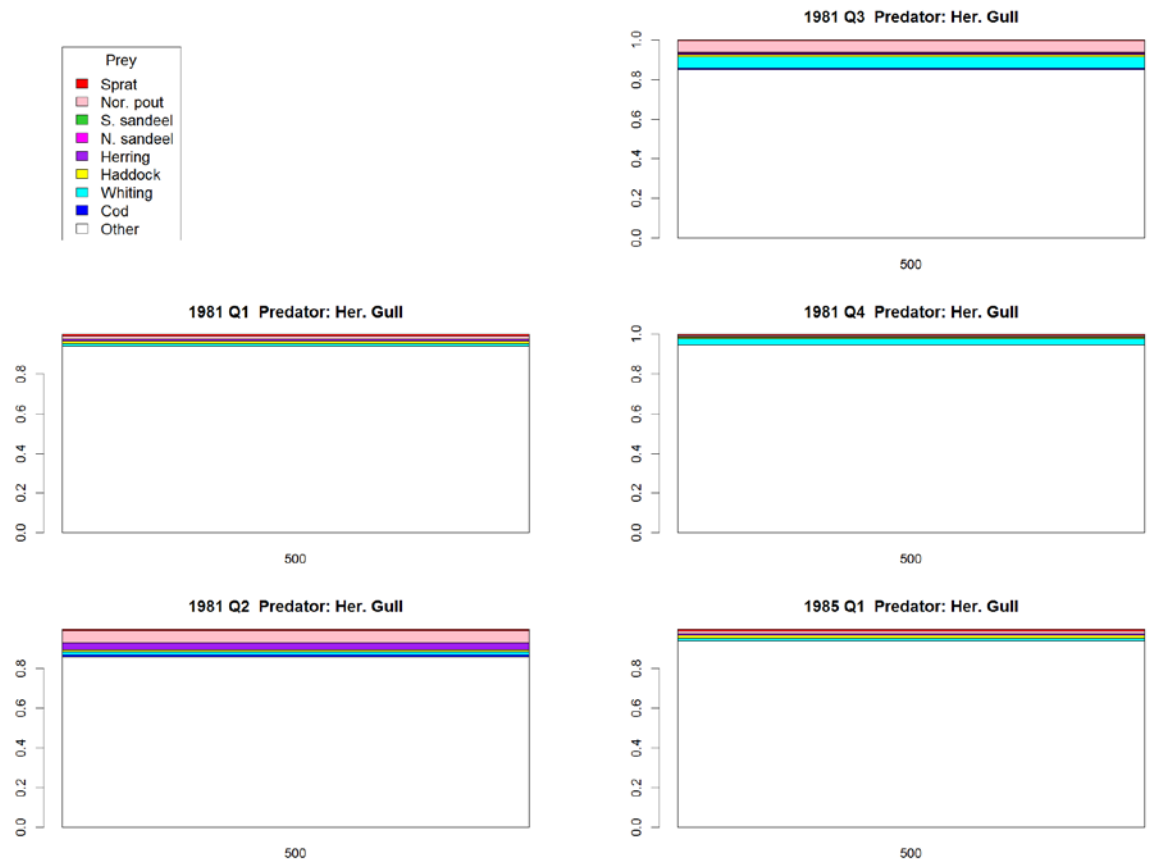


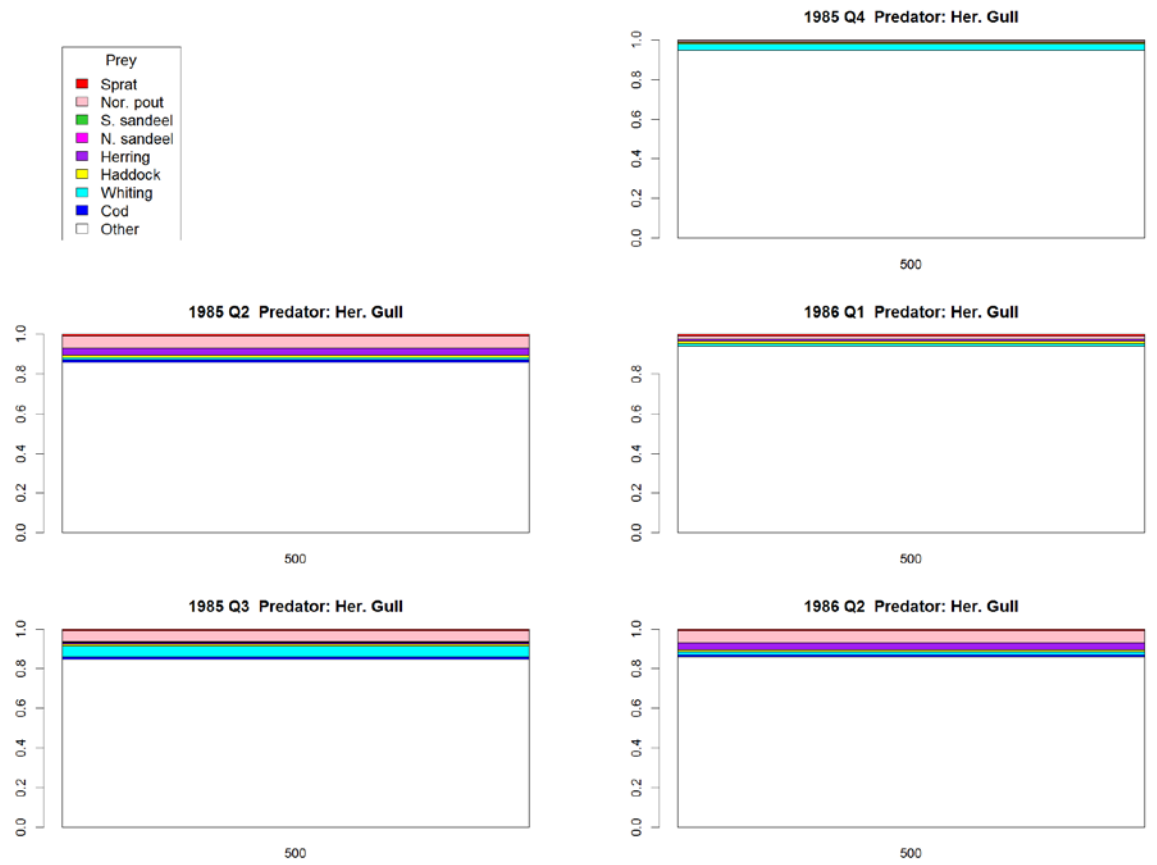


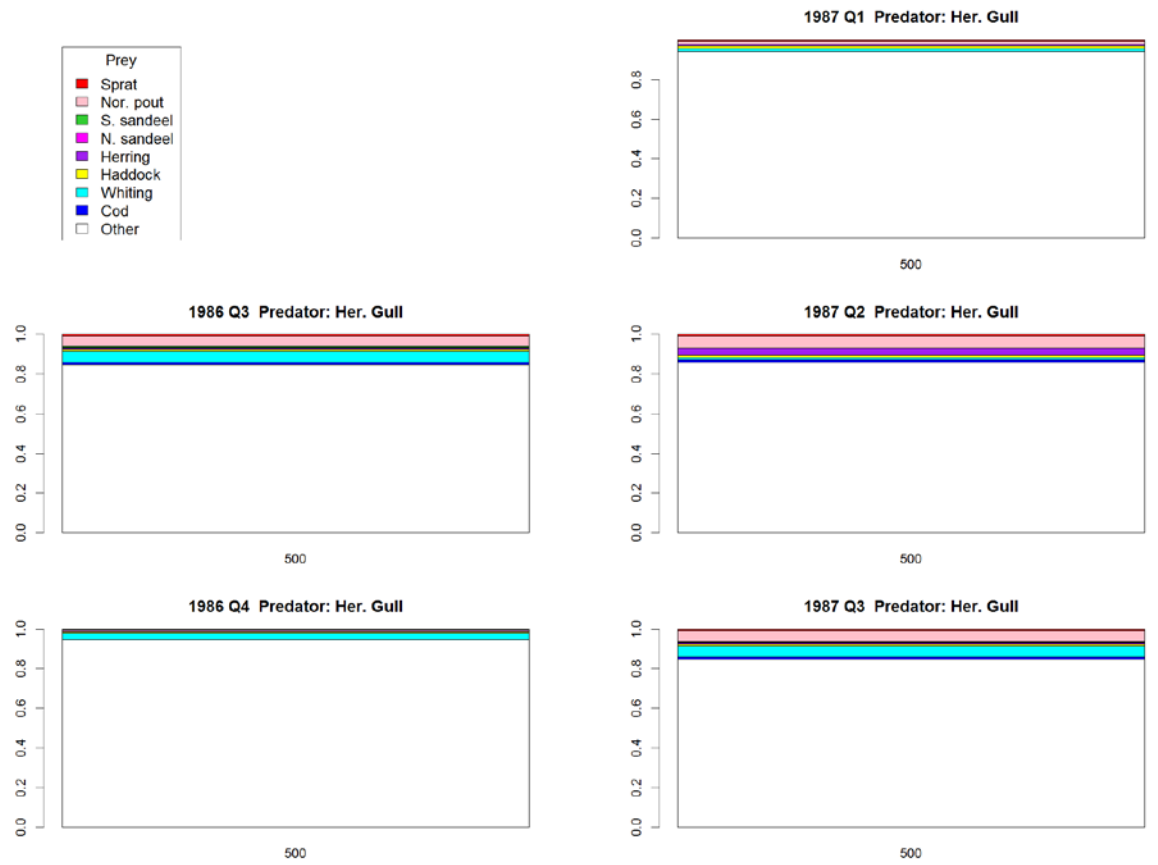


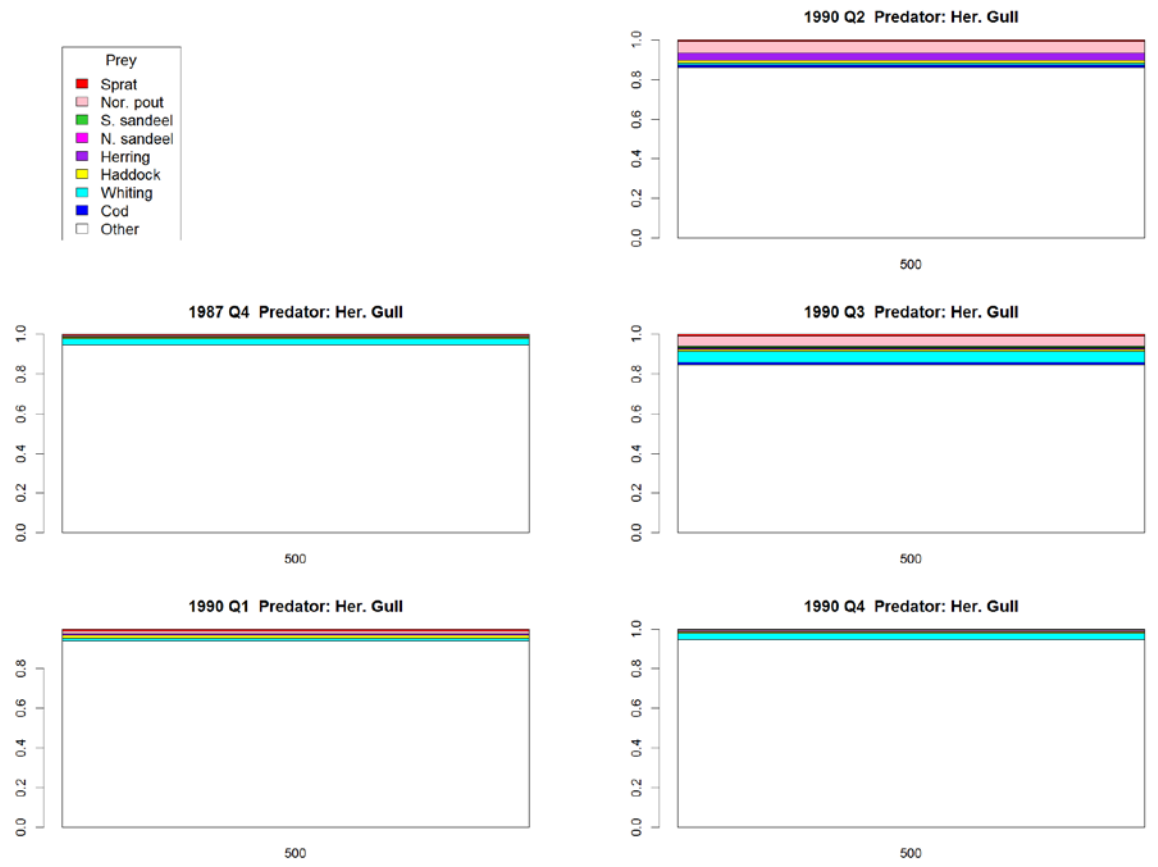


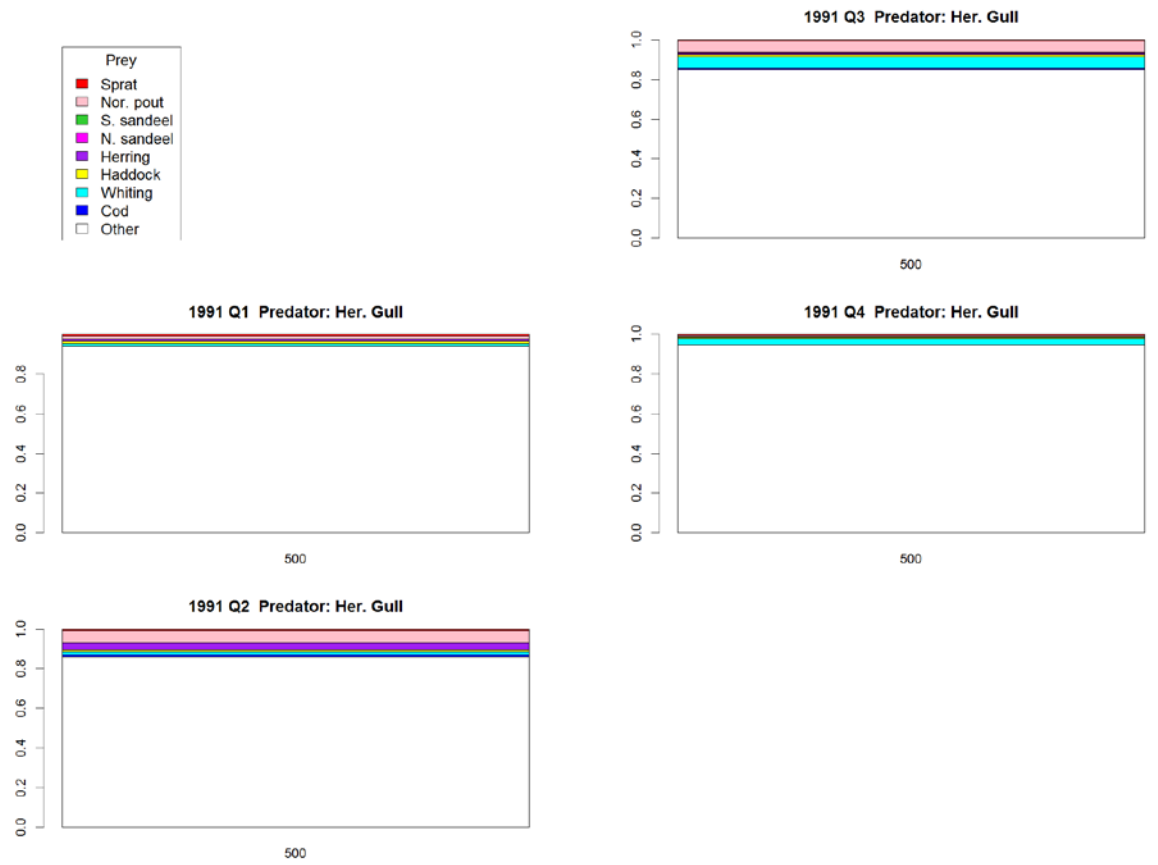


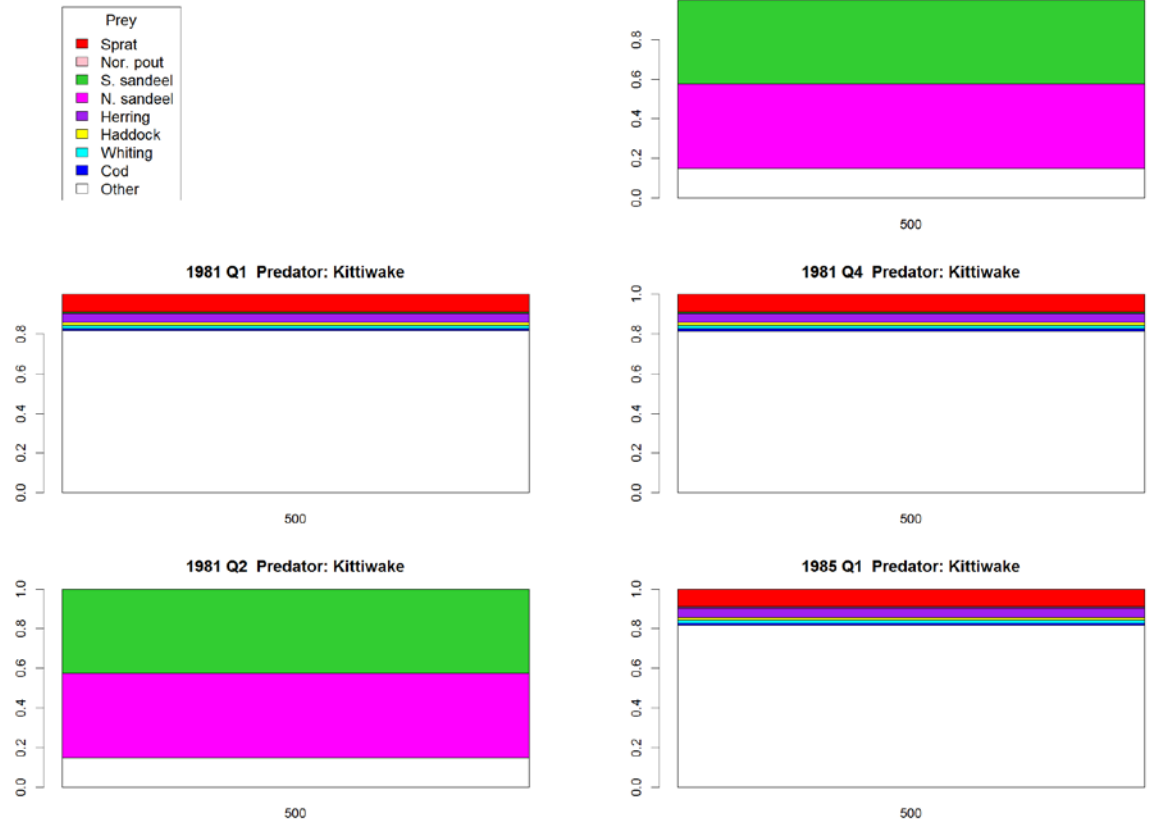


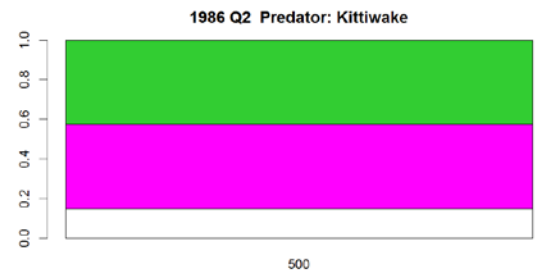
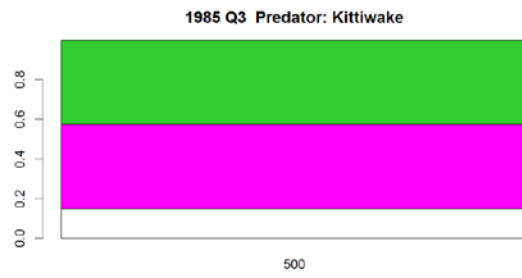
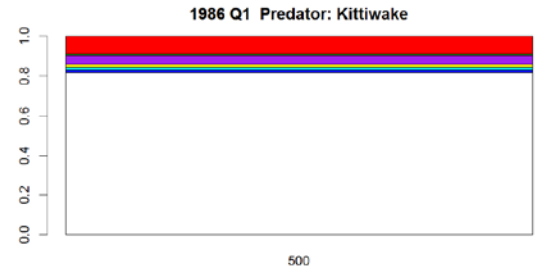
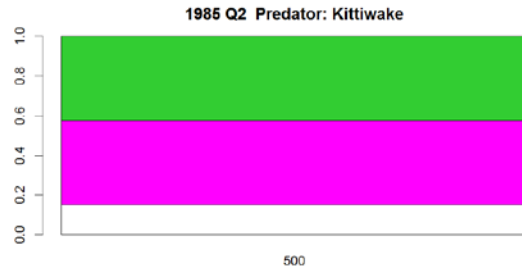
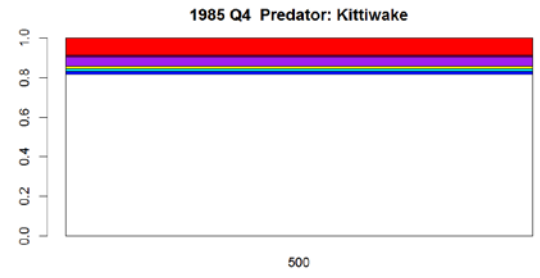
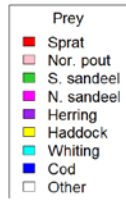


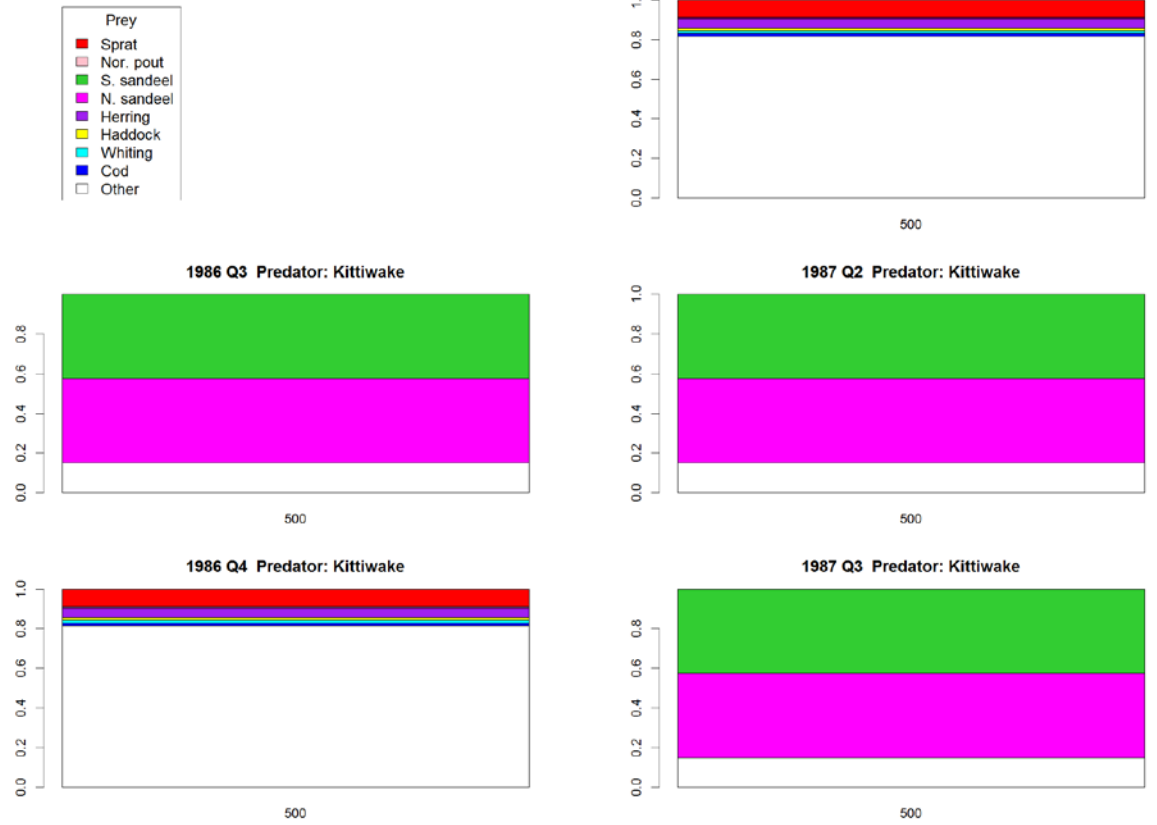


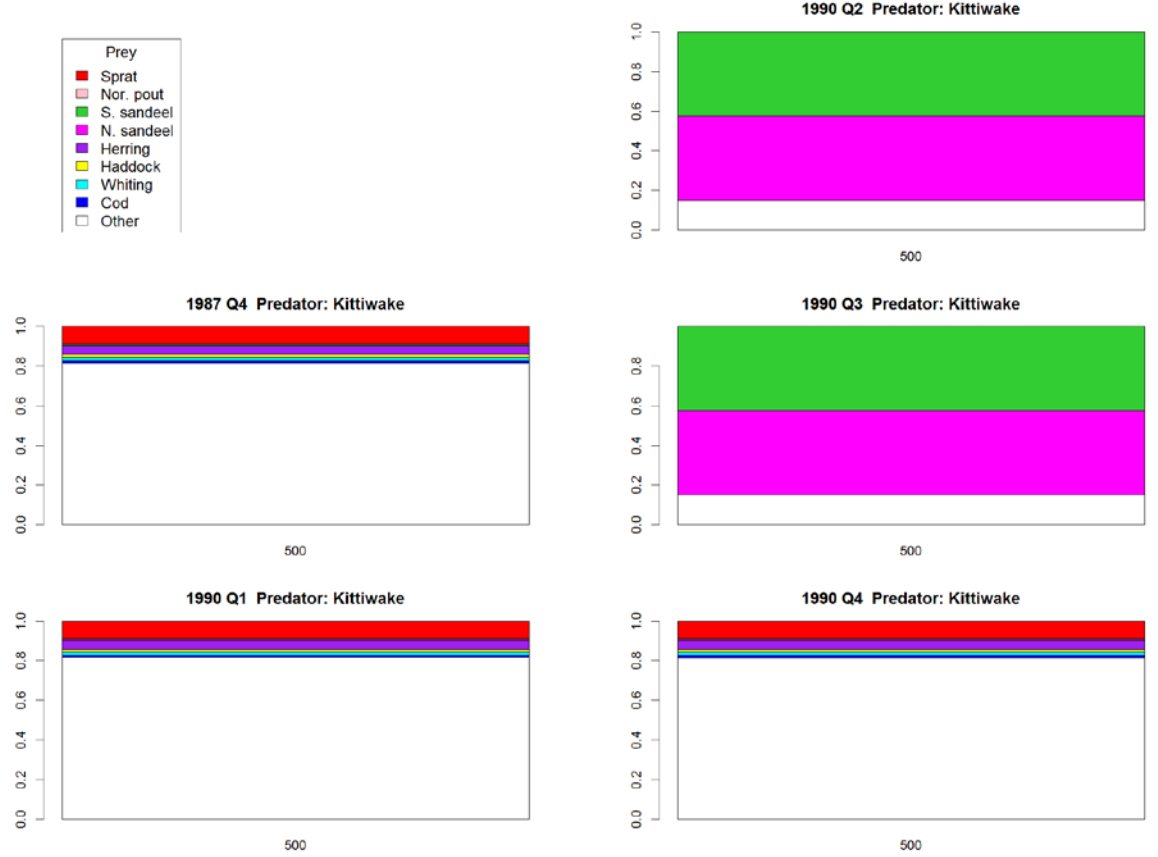


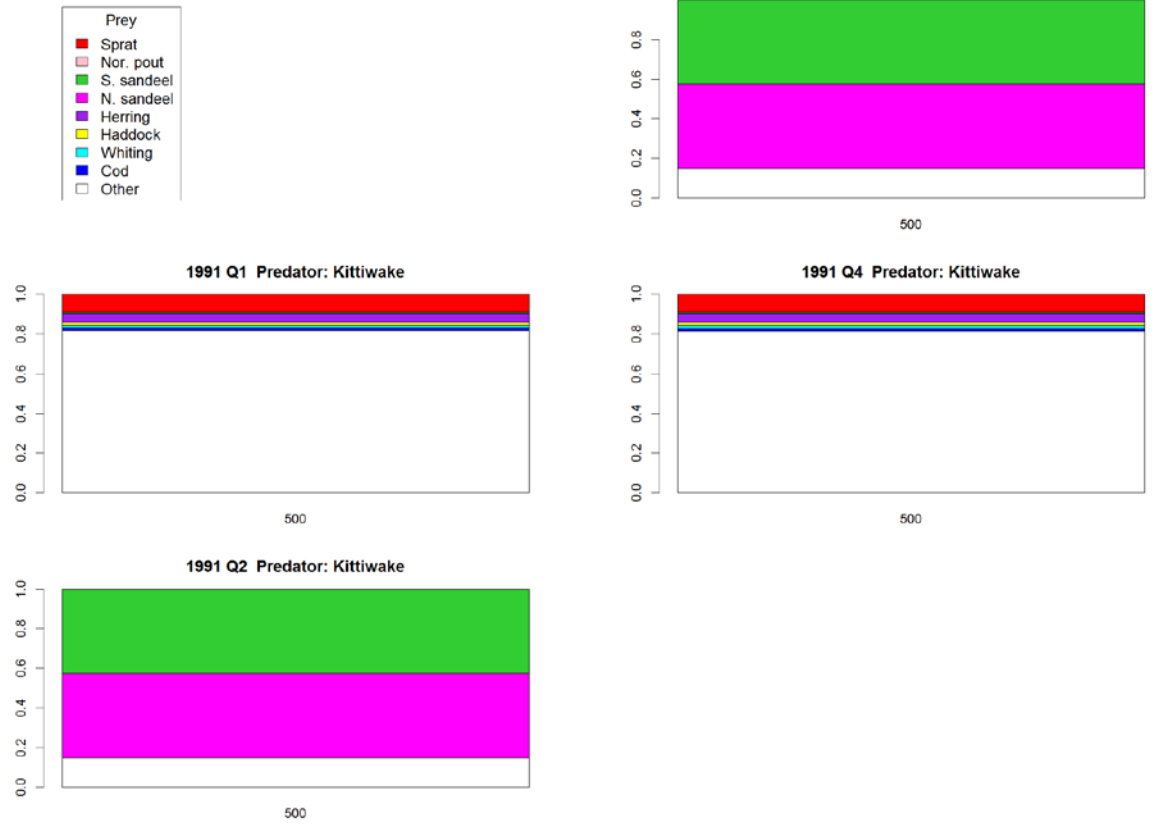


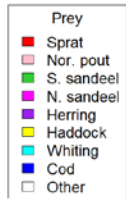


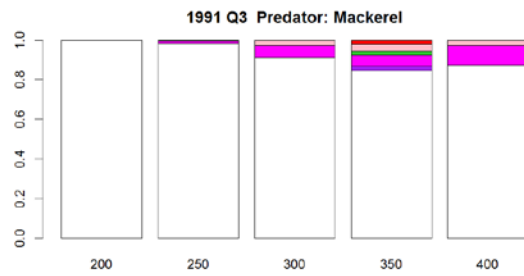
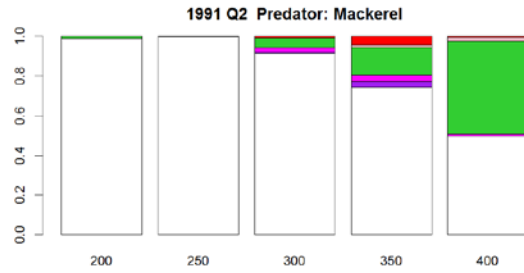
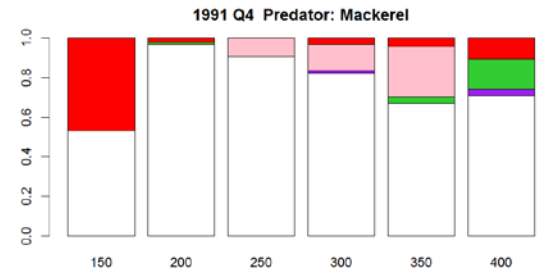
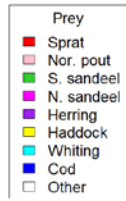


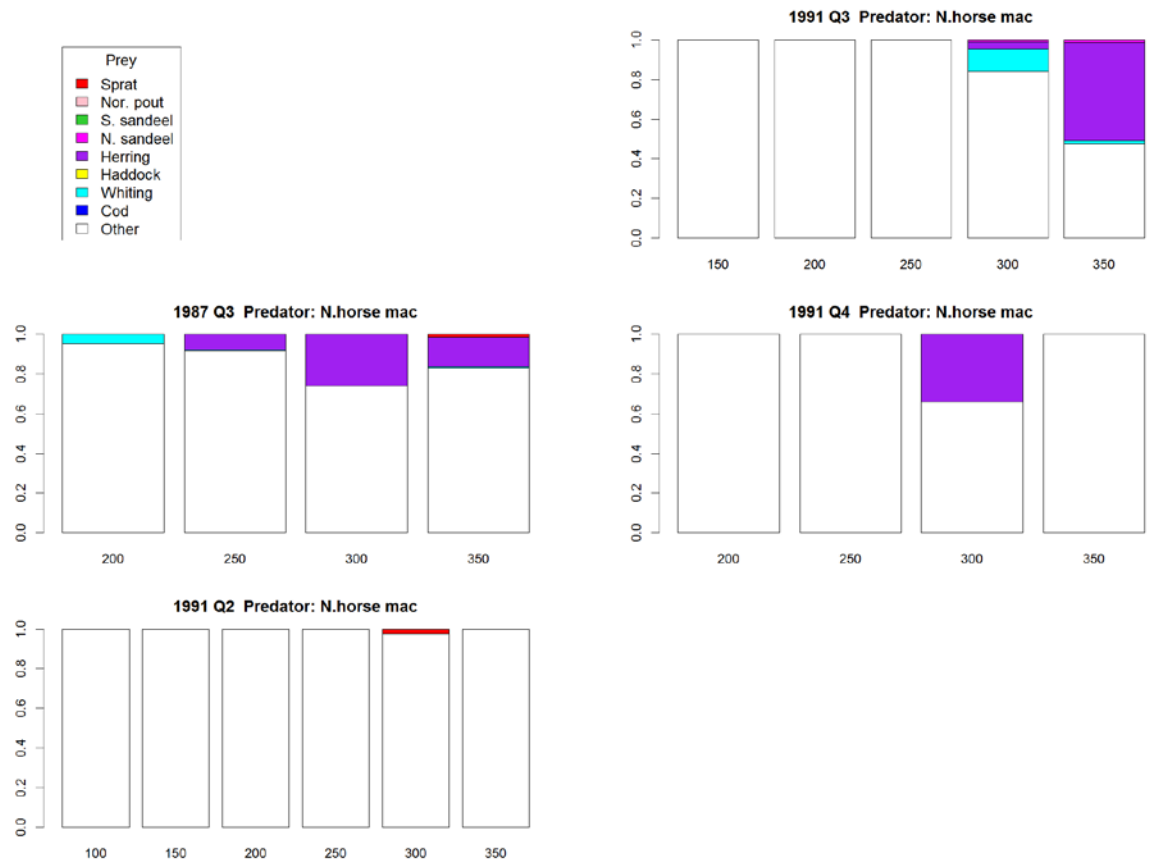


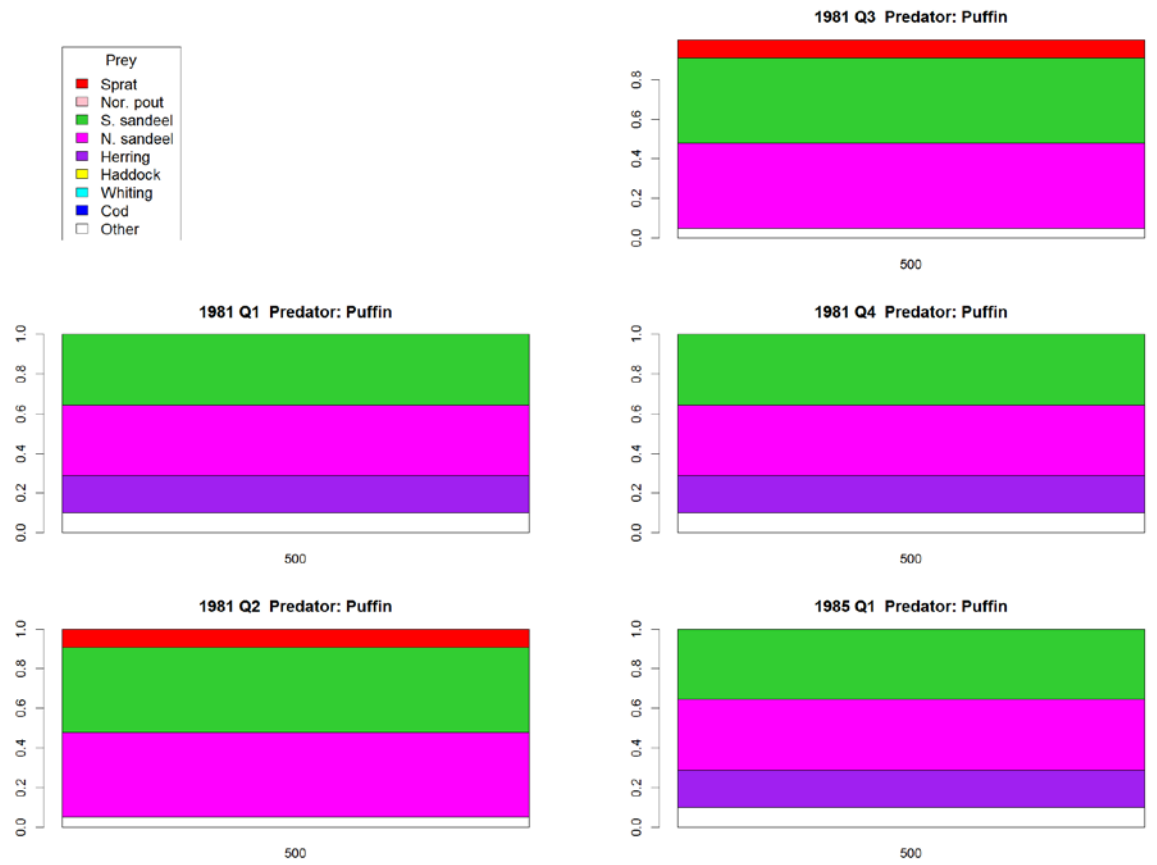


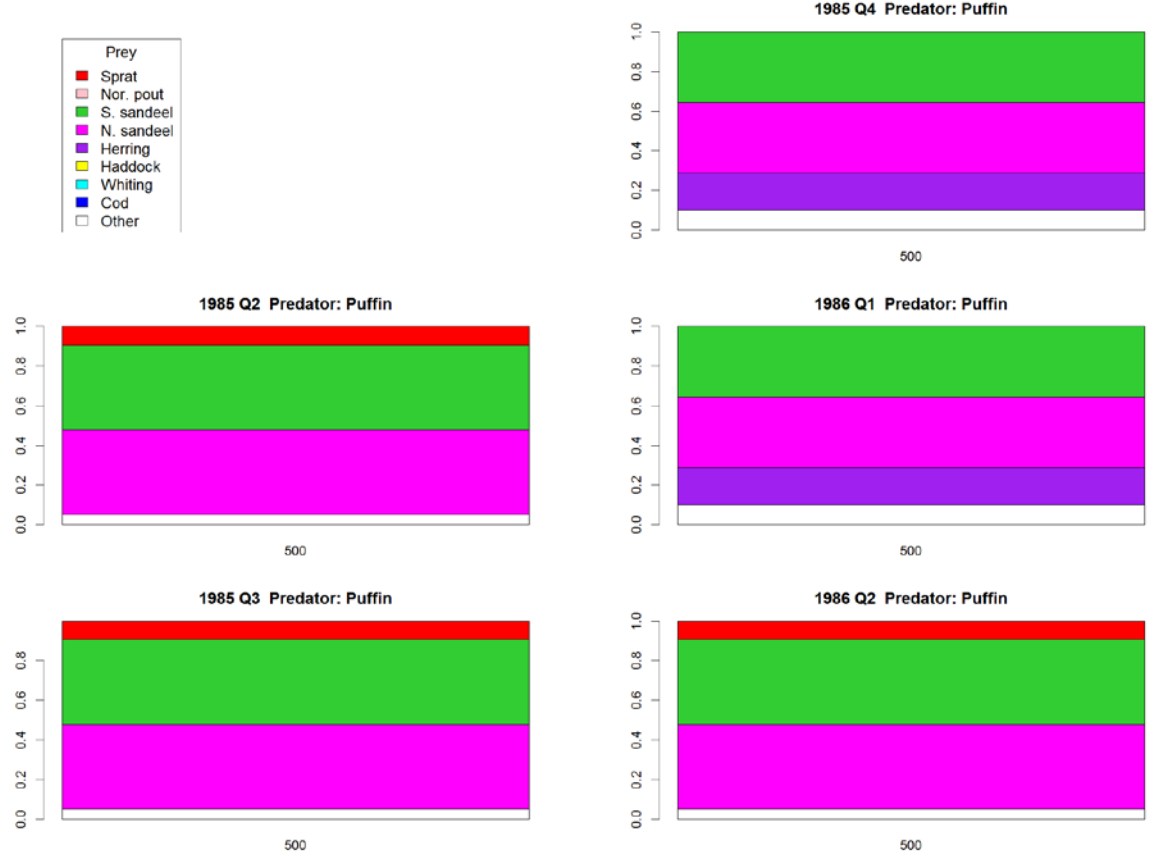


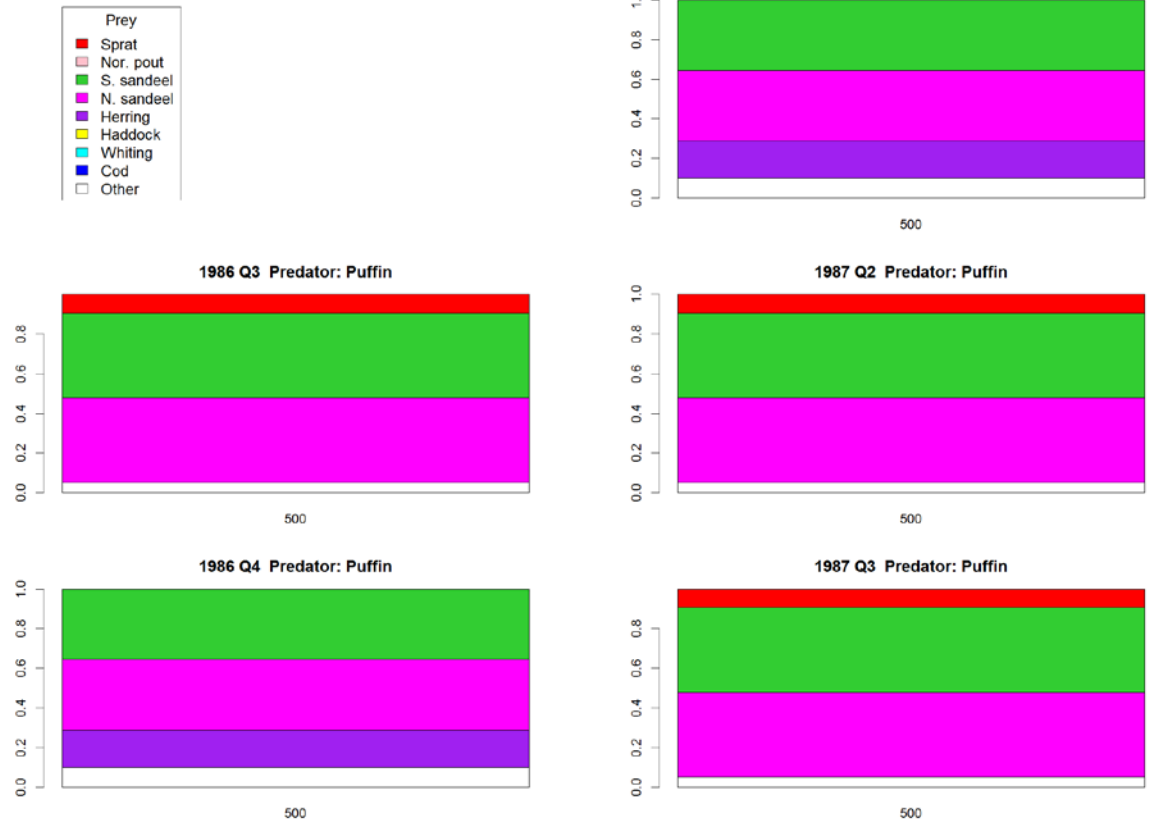


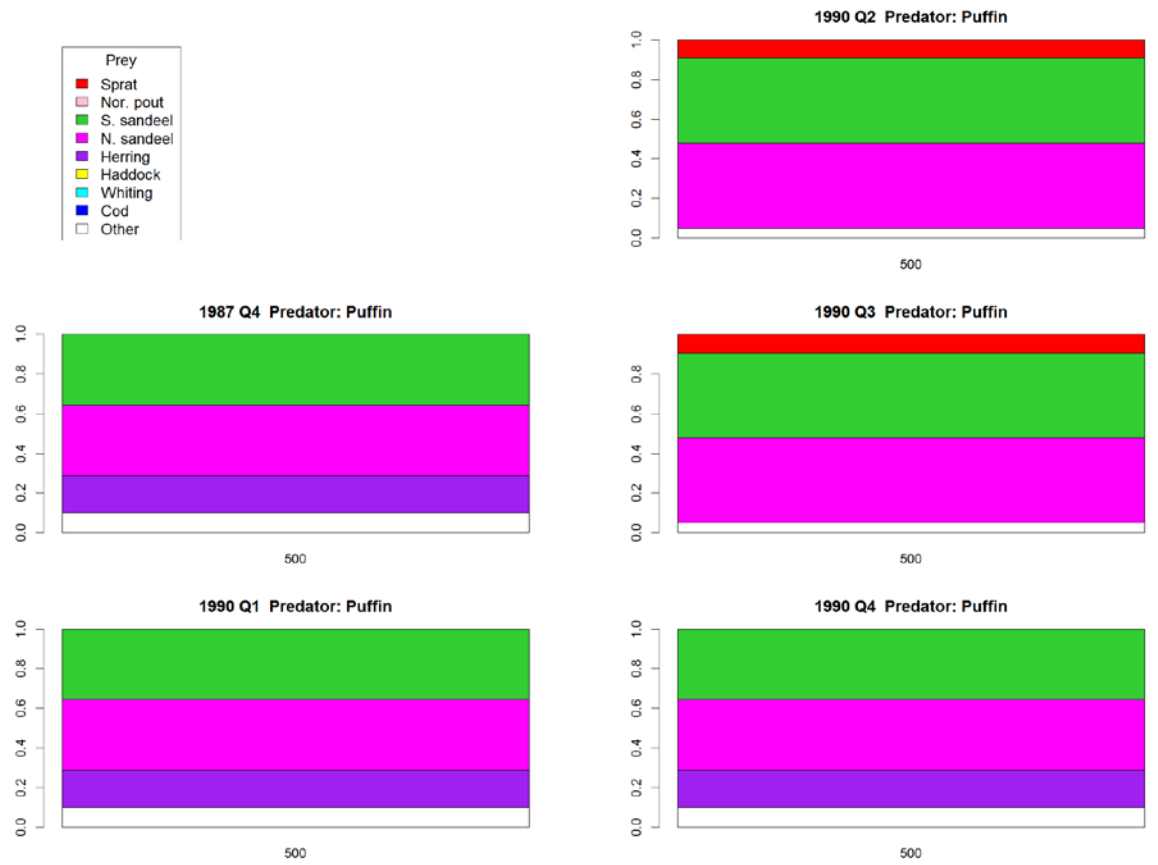


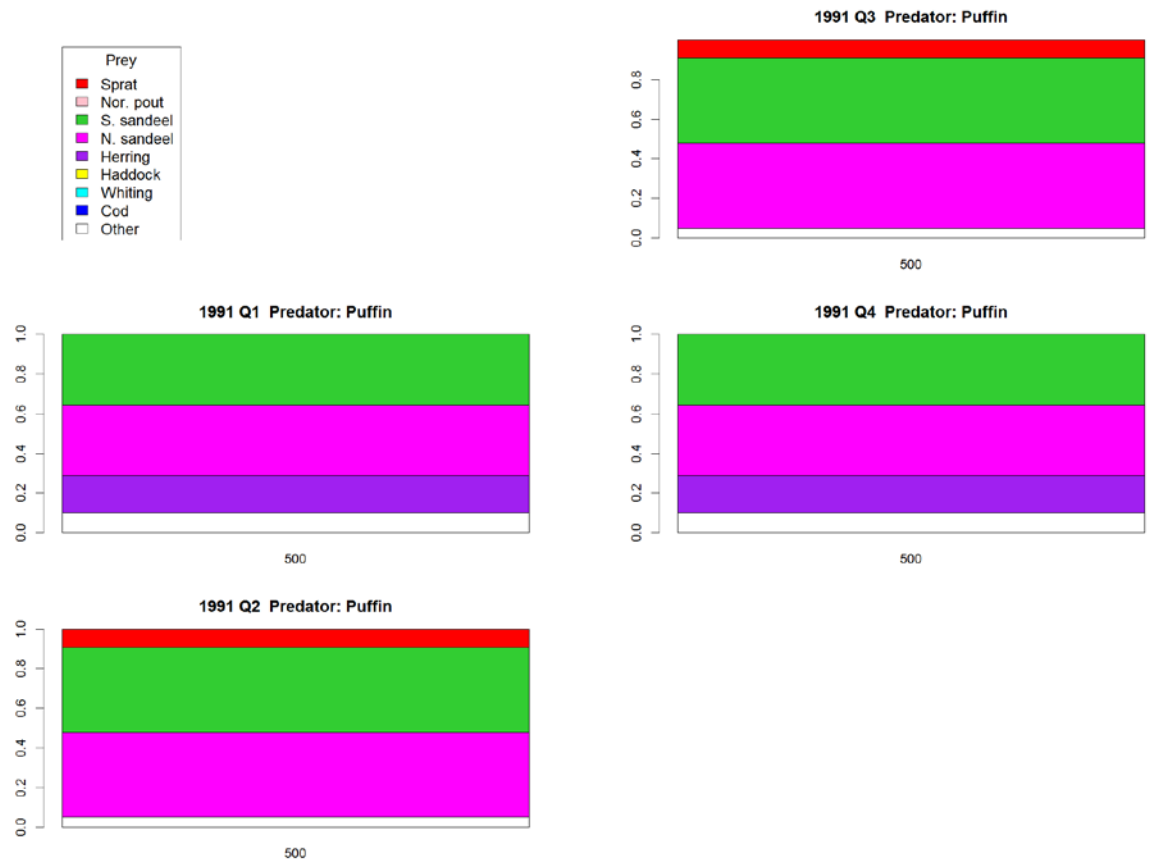


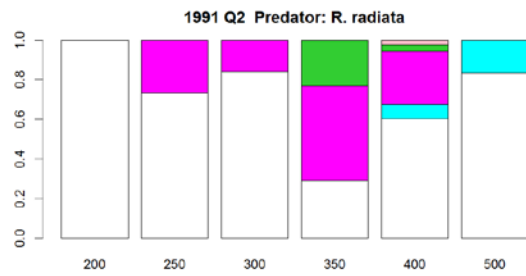
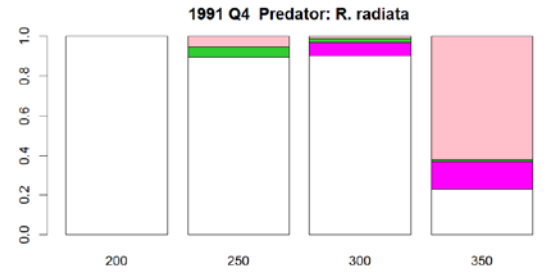
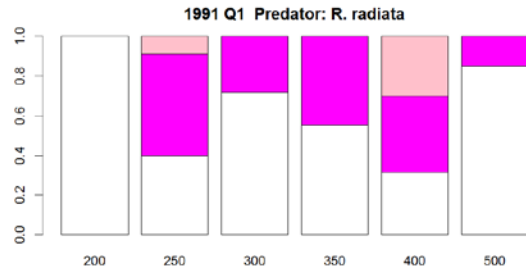
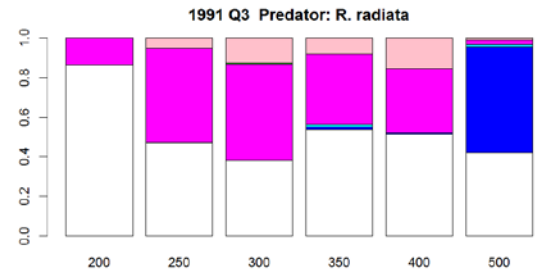
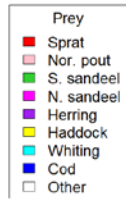


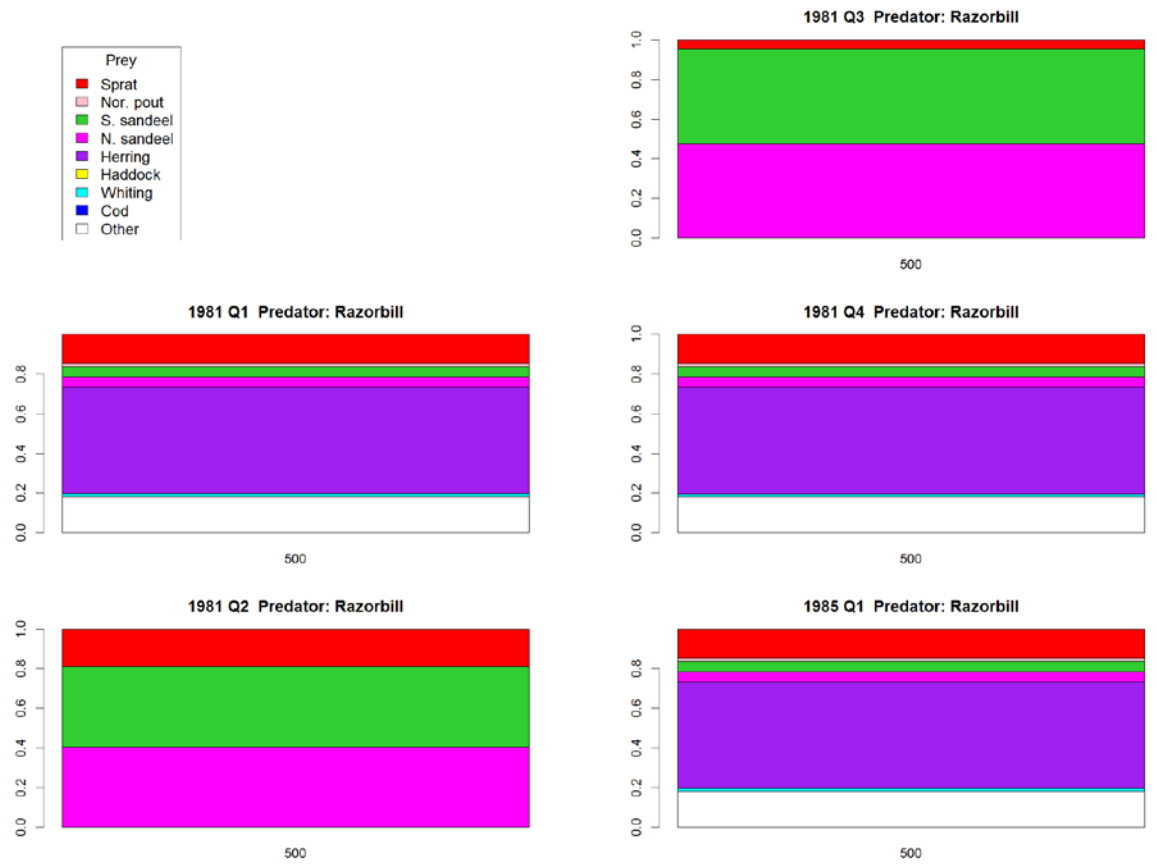


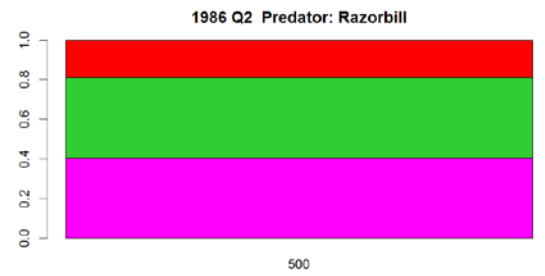
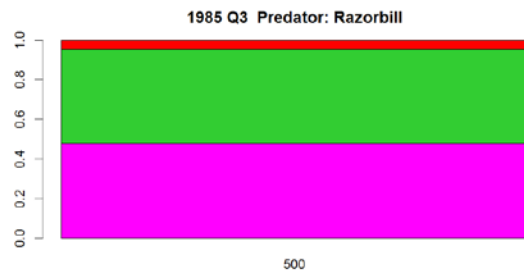
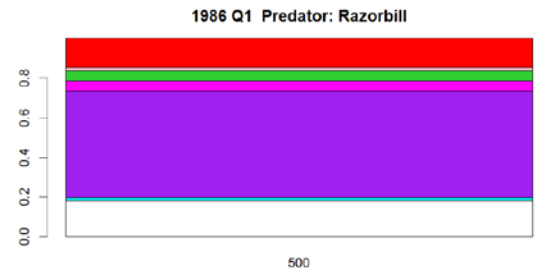
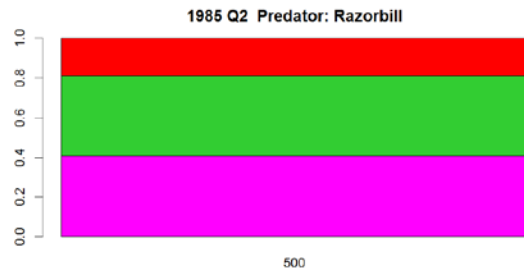
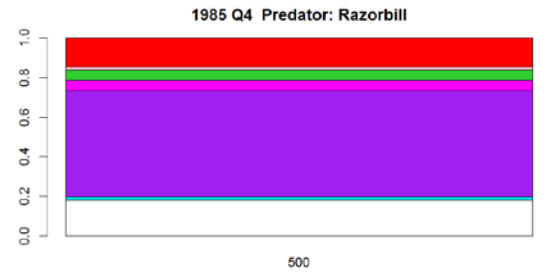
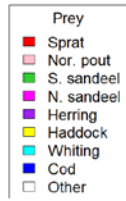


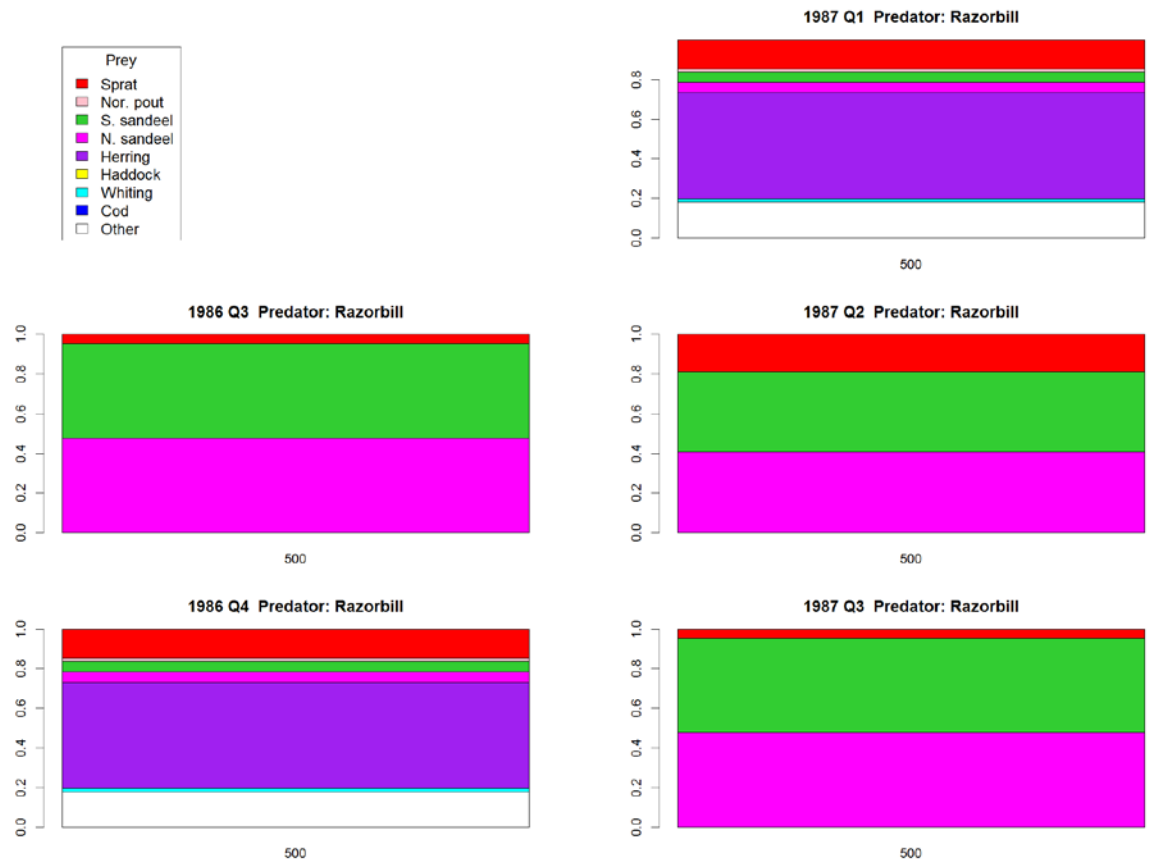


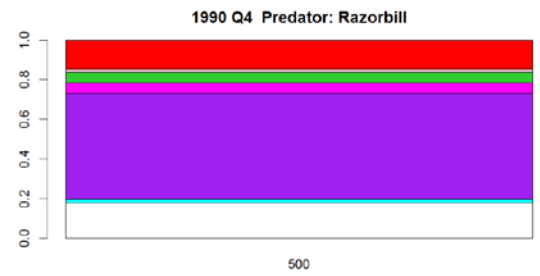
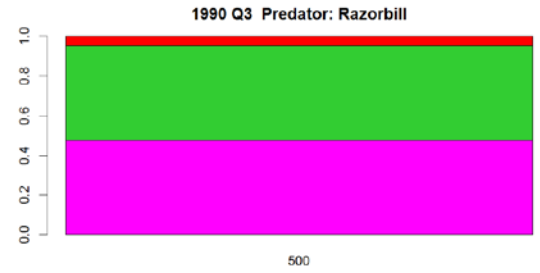
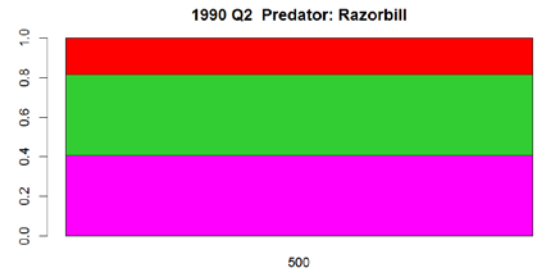
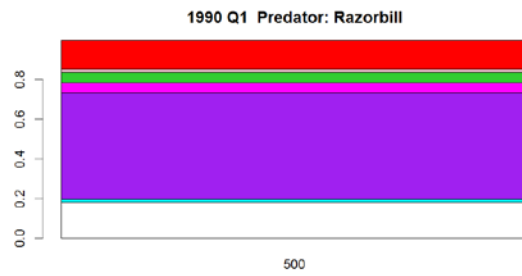
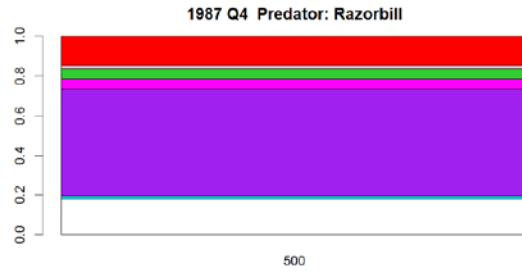
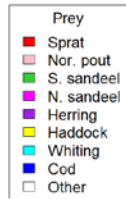


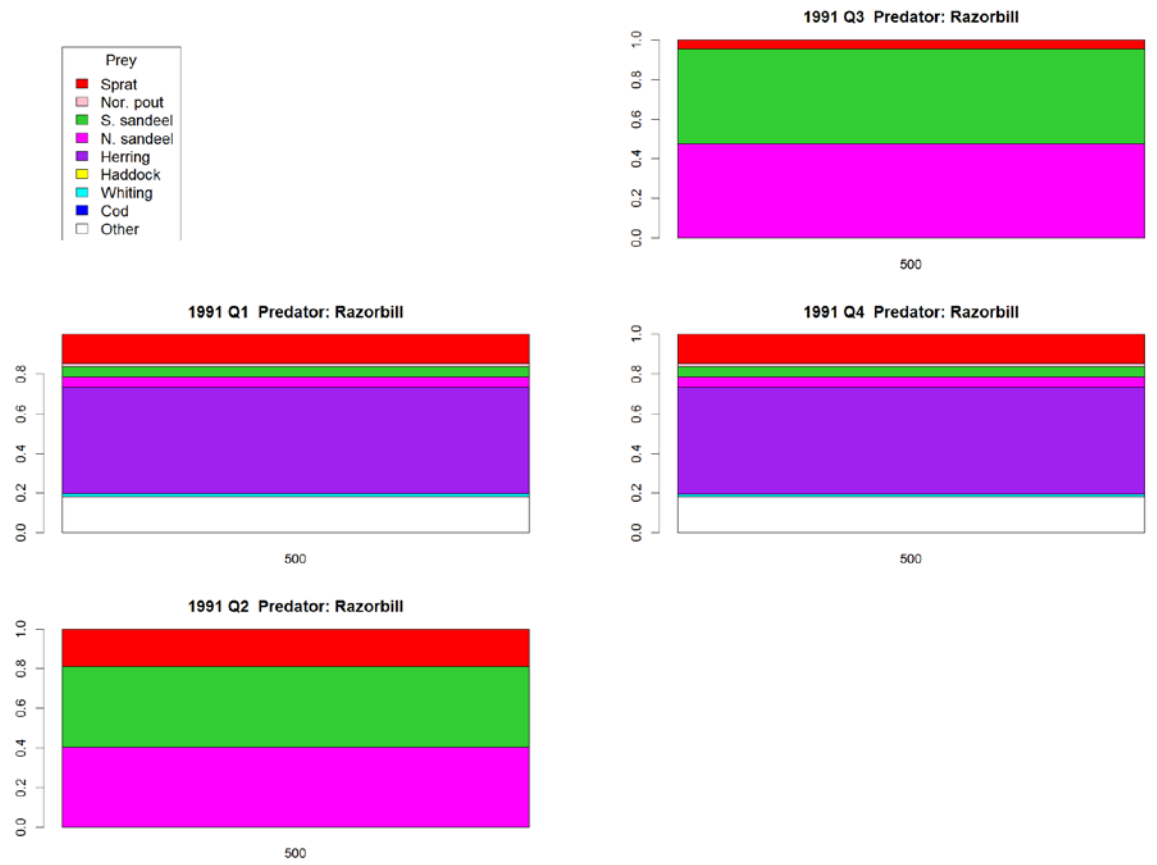


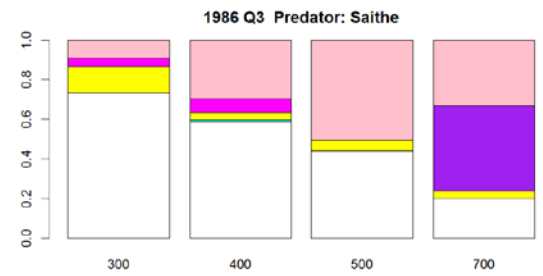
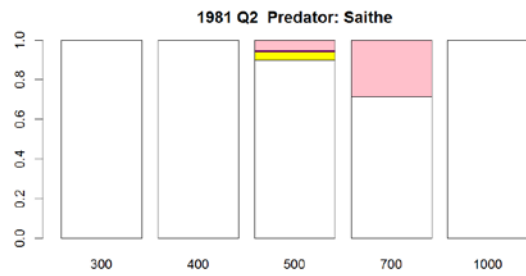
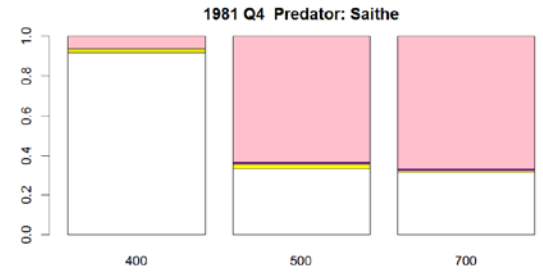
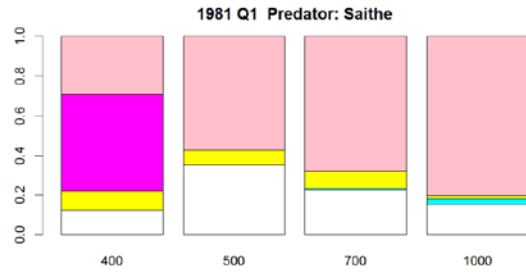
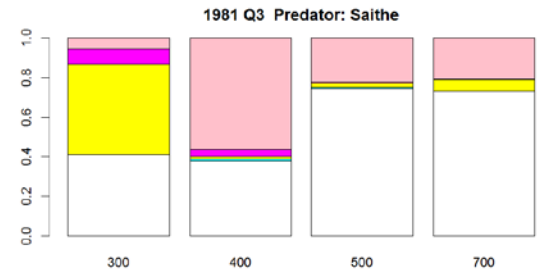
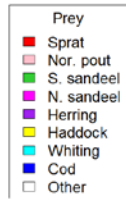


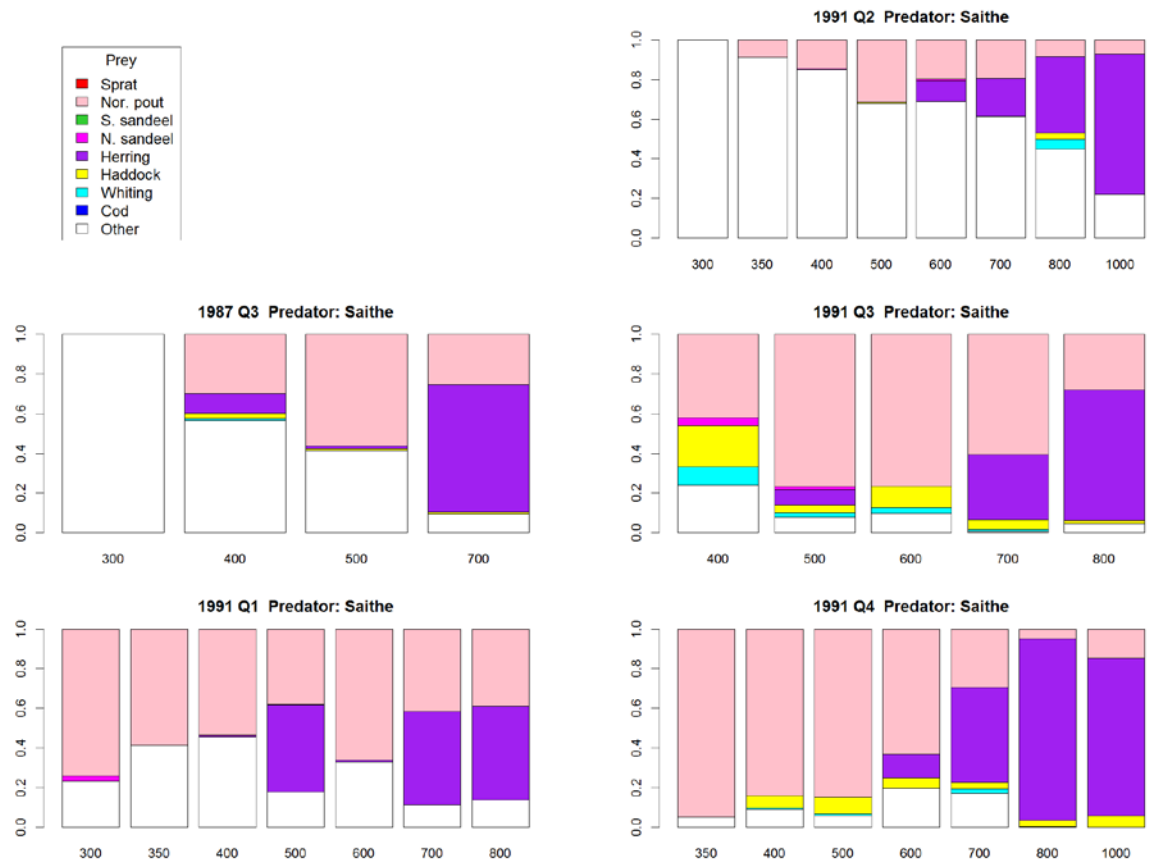


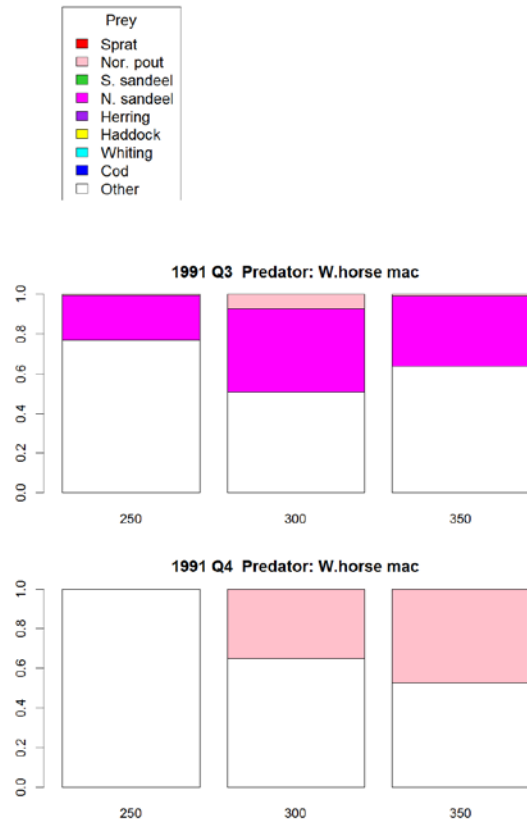


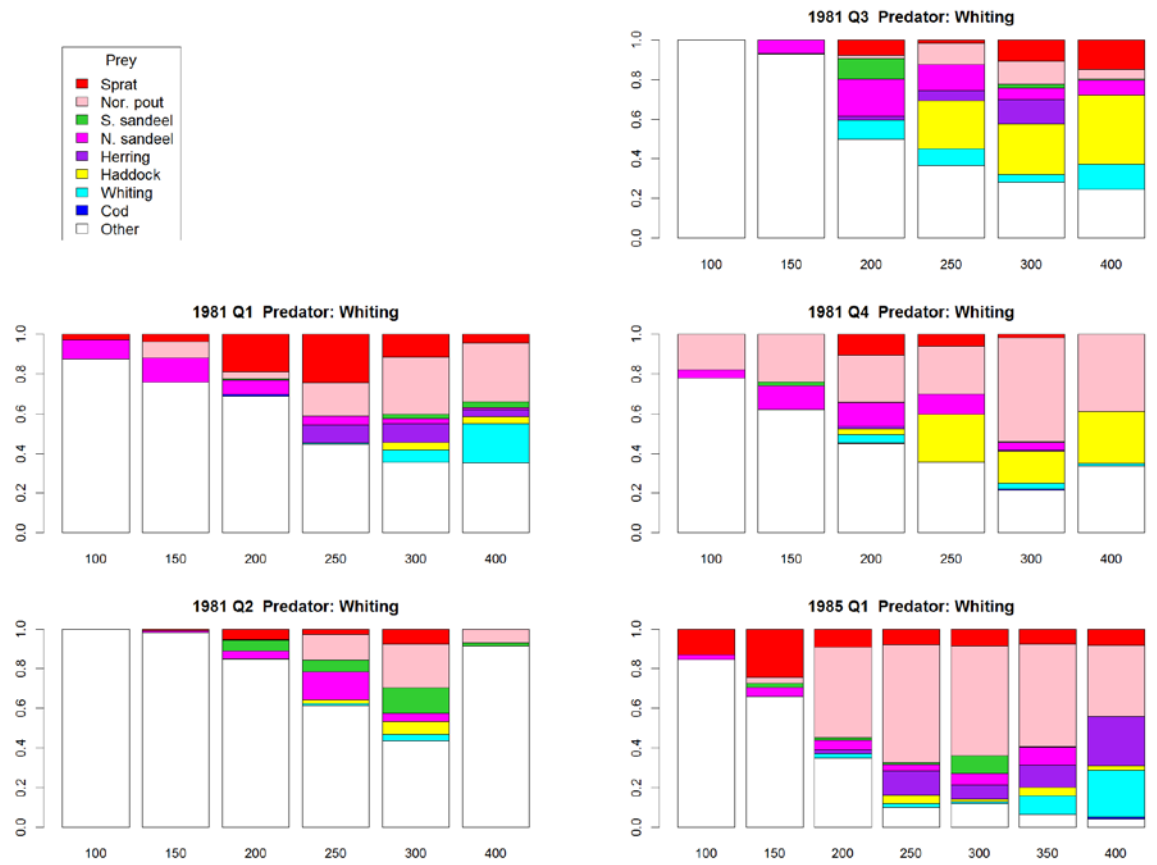


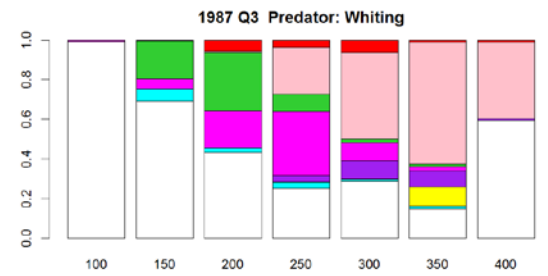
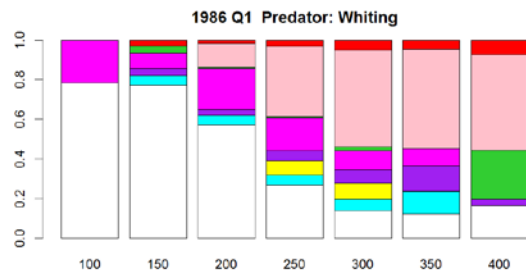
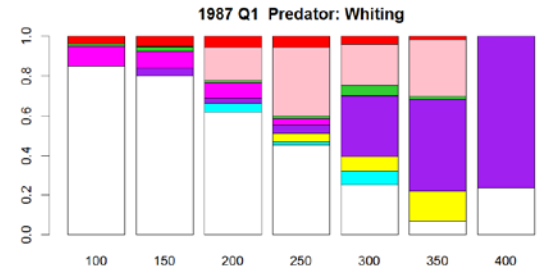
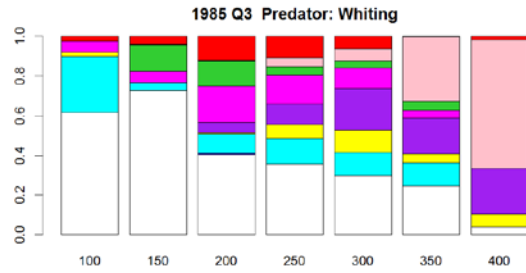
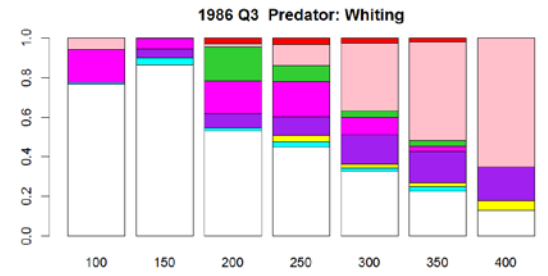
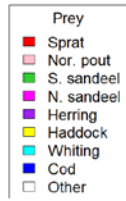












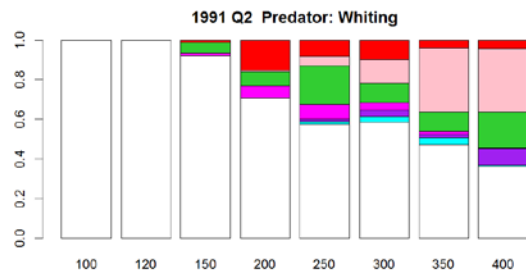
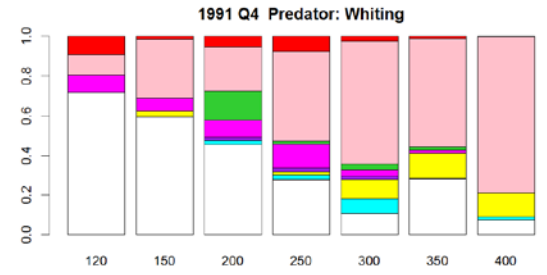
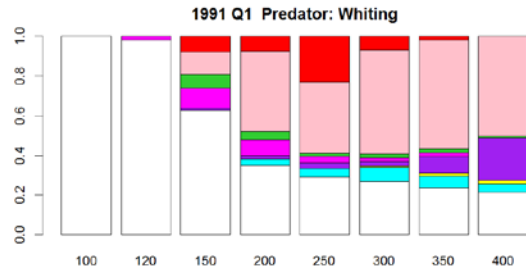
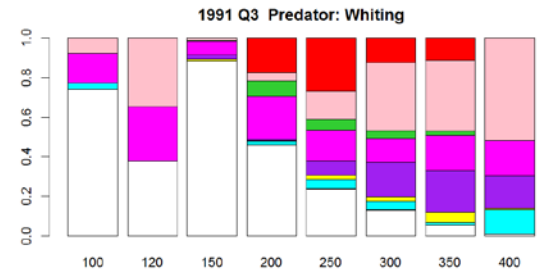
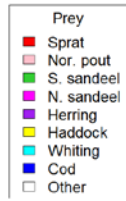


Table A3.1. Example of relative observed stomach contents by predator and prey length classes for Cod in 1991 quarter 1.

		Predator length class										
		150	200	250	300	350	400	500	600	700	800	1000
Prey	length											
COD	120	0.000	.	.
	150	0.003	0.003	.	.	0.007
	200	0.000	.	.
	250	0.003	.	0.014	.
	350	0.053	.	.	.
	All	0.003	0.058	0.000	0.014	0.007
HAD	length											
	100	0.001
	120	0.015	0.040	0.011	0.002	.	.
	150	0.020	0.014	0.005	0.021	0.005	.
	200	0.005	0.000	.	0.006
	250	0.015
	400	0.025	.
	All	0.035	0.055	0.021	0.022	0.031	0.021
HER	length											
	70	0.000	.
	80	0.009	.	.	.	0.000	0.002	.
	100	0.002	0.002	0.000	0.001	0.002
	120	0.002	0.009	0.013	0.001	0.01	0.013
	150	0.049	0.059	0.003	0.016	0.081	0.008	0.047
	200	0.016	0.017	0.079	0.105	0.04	0.076	0.028
	250	0.031	0.018	0.016	0.064	.

	All	0.074	0.077	0.125	0.154	0.137	0.161	0.090
NOP	length											
	80	0.004	0.003	0.002	0.001	0.001	.
	100	.	.	0.087	0.106	0.032	0.052	0.05	0.019	0.005	0.011	.
	120	.	.	.	0.024	0.184	0.045	0.075	0.031	0.053	0.009	.
	150	0.053	0.010	.	0.007	.
	All	.	.	0.087	0.129	0.217	0.101	0.181	0.062	0.058	0.028	.
NSA	length											
	70	.	.	.	0.007	0.005	0.001	.	.	.	0.000	.
	80	0.012	.	0.034	0.015	0.01	0.002	0.001	.	.	0.000	.
	100	.	.	.	0.002	0.021	0.009	.	.	.	0.000	0.000
	120	0.002	0.006	.	.	.	0.001	.
	150	0.001	0.001	.
	All	0.012	.	0.034	0.024	0.038	0.018	0.001	.	0.001	0.002	0.000
SPR	length											
	50	0.026
	70	0.181	0.000	.	0.000
	80	.	0.208	.	.	0.003	0.000	0.000	0.001	0.005	0.001	0.005
	100	0.001	.	.	0.000	.	.
	120	0.022	.	0.002	0.002	.	.
	All	0.207	0.208	.	.	0.003	0.023	0.000	0.003	0.007	0.001	0.005
SSA	length											
	70	0.000
	80	0.001	.
	100	.	0.031	.	.	0.000	0.000	.	.	0.001	0.001	.
	120	.	0.076	.	.	0.007	0.003	0.002	0.000	0.000	.	.
	150	0.071	.	.	0.001	.	0.003	0.001	0.000	.	.	.

[illegible]

Table A3.2 Number of stomach sampled by predator, year, quarter and predator size class (lower limit in mm)

Predator Cod

	Year														All
	1981				1985		1986		1987		1991				
	Quarter				Quarter		Quarter		Quarter		Quarter				
	1	2	3	4	1	3	1	3	1	3	1	2	3	4	
100	.	.	355	189	.	70	.	21	.	3	.	.	193	212	1043
120	42	6	55	165	268
150	251	176	232	199	91	6	639	204	209	89	117	216	4	335	2768
200	531	328	87	199	254	91	311	825	314	477	123	498	149	102	4289
250	601	370	185	233	449	217	194	935	483	655	61	331	392	80	5186
300	837	538	370	424	484	528	93	644	486	703	172	248	320	256	6103
350	353	420	128	333	357	746	207	334	158	230	3266
400	455	391	337	404	378	484	315	243	246	691	327	564	263	205	5303
500	556	392	367	453	253	311	198	232	85	230	320	428	165	119	4109
600	157	186	244	114	53	87	281	245	99	107	1573
700	684	180	257	357	105	120	171	84	50	61	186	112	41	73	2481
800	110	79	146	70	84	53	258	96	36	33	965
1000	117	19	49	54	30	15	64	15	41	13	81	29	9	9	545
All	4032	2394	2239	2512	2664	2527	2503	3720	2408	3808	2175	3107	1884	1926	37899

Predator Whiting

	Year														All
	1981				1985		1986		1987		1991				
	Quarter				Quarter		Quarter		Quarter		Quarter				
	1	2	3	4	1	3	1	3	1	3	1	2	3	4	
100	1455	435	229	522	1084	303	1414	936	1766	300	292	92	883	548	10259
120	891	495	754	673	2813
150	1604	758	317	518	1394	767	1667	1060	2232	1121	1341	2148	1061	1756	17744
200	1587	963	807	704	1691	1846	1400	1955	1666	1466	1284	3010	2387	1915	22681
250	1515	1246	1075	795	1360	1896	1243	2209	1161	1763	1262	3422	3084	2148	24179
300	1215	1024	944	711	712	1129	631	1467	619	1174	789	1742	2084	1616	15857
350	315	290	150	390	158	388	205	331	344	556	3127

400	156	64	152	107	91	68	29	83	9	53	37	81	24	68	1022
500	3	1	5	4	1	1	.	.	1	1	1	9	.	.	27
All	7535	4491	3530	3361	6648	6300	6534	8100	7612	6266	6102	11330	10621	9280	97710

Table A3.2 (continued) Number of stomach sampled by predator, year, quarter and predator size class (lower limit in mm)

Predator Haddock

	Year								All
	1981				1991				
	Quarter				Quarter				
	1	2	3	4	1	2	3	4	
100	238	.	772	692	19	.	590	180	2491
120	289	34	602	299	1224
150	444	576	679	812	529	482	379	413	4314
200	572	719	1049	919	445	555	763	359	5381
250	629	802	1333	947	340	526	866	527	5970
300	690	871	1451	1012	341	464	624	535	5988
350	262	350	423	304	1339
400	195	387	455	503	170	270	241	185	2406
500	42	39	82	80	45	54	46	66	454
600	1	14	5	17	37
All	2810	3394	5821	4965	2441	2749	4539	2885	29604

Predator Saithe

	Year										All
	1981				1986	1987	1991				
	Quarter				Quarter	Quarter	Quarter				
	1	2	3	4	3	3	1	2	3	4	
300	90	14	68	10	727	91	98	12	4	4	1118
350	179	258	56	73	566
400	70	7	171	62	695	361	375	455	198	499	2893
500	279	45	363	156	577	400	71	204	70	194	2359
600	38	96	27	50	211
700	324	113	278	147	97	66	20	75	15	13	1148
800	12	72	29	17	130
1000	34	6	15	174	4	4	3	10	.	6	256
All	797	185	895	549	2100	922	796	1182	399	856	8681

Table A3.2 (Continued). Number of stomach sampled by predator, year, quarter and predator size class (lower limit in mm)

Predator Mackerel

	Year								All
	1981				1991				
	Quarter				Quarter				
	1	2	3	4	1	2	3	4	
50	1	.	1
150	3	3	.	.	71	2	.	22	101
200	68	39	58	4	134	207	66	50	626
250	71	188	621	101	48	554	616	100	2299
300	83	466	1212	406	33	972	1359	274	4805
350	5	468	629	225	1327
400	16	358	307	145	1	129	126	34	1116
All	241	1054	2198	656	292	2332	2797	705	10275

Predator Grey gurnard

	Year												All
	1980	1982	1983	1987	1989	1990			1991				
	Quarter	Quarter	Quarter	Quarter	Quarter	Quarter			Quarter				
	3	3	2	3	2	1	2	3	1	2	3	4	
80	2	2	.	.	.	17	21
100	.	.	26	.	5	58	5	25	.	43	20	105	287
120	19	51	20	68	158
150	10	10	35	.	24	99	99	169	605	1682	1234	465	4432
200	10	10	136	10	53	64	92	175	587	1524	1469	485	4615
250	10	10	101	.	45	27	69	83	358	510	737	326	2276
300	10	2	2	.	21	2	42	38	248	214	356	166	1101
350	7	.	13	17	85	97	157	59	435
400	1	.	1	.	14	7	8	10	41
All	40	32	300	10	156	250	323	509	1916	4128	4001	1701	13366

Table A3.2 (continued) Number of stomach sampled by predator, year, quarter and predator size class (lower limit in mm)

Predator Horse Mackerel

	Year					All
	1987	1991				
	Quarter	Quarter				
	3	1	2	3	4	
100	.	.	35	.	.	35
120	.	.	12	.	2	14
150	28	.	47	119	.	194
200	100	.	180	188	19	487
250	320	1	269	495	265	1350
300	242	6	291	591	380	1510
350	15	4	83	93	89	284
400	.	.	3	3	4	10
All	705	11	920	1489	759	3884

Predator Amblyraja radiata

	Year				All
	1991				
	Quarter				
	1	2	3	4	
100	.	.	1	.	1
120	.	.	1	2	3
150	19	12	40	8	79
200	33	35	121	17	206
250	111	51	217	53	432
300	99	75	267	76	517
350	114	85	297	86	582
400	185	257	336	152	930
500	28	34	49	15	126
All	589	549	1330	409	2877

Table A3.3 Number of stomach sampled by predator and year.

	Year								All
	1981	1983	1985	1986	1987	1989	1990	1991	
Predator									
Cod	11177	.	5191	6223	6216	.	.	9092	37899
Grey gurnard	.	300	.	.	10	156	1082	11746	13366
Haddock	16990	12614	29604
Horse Mackerel	705	.	.	3179	3884
Mackerel	4149	6126	10275
Amblyraja radi- ata	2877	2877
Saithe	2426	.	.	2100	922	.	.	3233	8681
Whiting	18917	.	12948	14634	13878	.	.	37333	97710
All	53659	300	18139	22957	21731	156	1082	86200	204296

APPENDIX 4: Option file for SMS-key-runs

Key-run 2017

```
# sms.dat option file
# the character "#" is used as comment character, such that all text and
# numbers
# after # are skipped by the SMS program
#
#####
# Produce test output (option test.output)
# 0 no test output
# 1 output file sms.dat and file fleet.info.dat as read in
# 2 output all single species input files as read in
# 3 output all multi species input files as read in
# 4 output option overview
#
# 11 output between phases output
# 12 output iteration (obj function) output
# 13 output stomach parameters
# 19 Both 11, 12 and 13
#
# Forecast options
# 51 output hcr_option.dat file as read in
# 52 output prediction output summary
# 53 output prediction output detailed
0
#####
# Produce output for SMS-OP program. 0=no, 1=yes
0
#####
# Single/Multispecies mode (option VPA.mode)
# 0=single species mode
# 1=multi species mode, but  $Z=F+M$  (used for initial food suitability
# parm. est.)
# 2=multi species mode,  $Z=F+M1+M2$ 
0
#####
```



```
# Number of areas for multispecies run (default=1)
1
#
#####
#
# single species parameters
#
#####
#
## first year of input data (option first.year)
1974
#####
## first year used in the model (option first.year.model)
1974
#####
## last year of input data (option last.year)
2016
#####
## last year used in the model (option last.year.model)
2016
#####
## number of seasons (option last.season). Use 1 for annual data
4
#####
## last season last year (option last.season.last.year). Use 1 for annual
data
42
#####
## number of species (option no.species)
27
#####
## first age all species (option first.age)
0
#####
## recruitment season (option rec.season). Use 1 for annual data
3
#####
## maximum age for any species(max.age.all)
10
#####
```

```

## various information by species
# 1. last age
# 2. first age where catch data are used (else F=0 assumed)
# 3. last age with age dependent fishing selection
# 4. Estimate F year effect from effort data. 0=no, 1=yes
# 5. Last age included in the catch at age likelihood (normally last age)
# 6. plus group, 0=no plus group, 1=plus group
# 7. predator species, 0=no, 1=VPA predator, 2=Other predator
# 8. prey species, 0=no, 1=yes
# 9. Stock Recruit relation
#     1=Ricker, 2=Beverton & Holt, 3=Geom mean,
#     4= Hockey stick, 5=hockey stick with smoother,
#     51=Ricker with estimated temp effect,
#     52=Ricker with known temp effect,
#     >100= hockey stick with known breakpoint (given as input)
# 10. Additional data for Stock Recruit relation
# 11. Additional data for Stock Recruit relation
##
1 0 0 0 0 0 2 0 0 0 0 # 1 Fulmar
1 0 0 0 0 0 2 0 0 0 0 # 2 Guillemot
1 0 0 0 0 0 2 0 0 0 0 # 3 Her. Gull
1 0 0 0 0 0 2 0 0 0 0 # 4 Kittiwake
1 0 0 0 0 0 2 0 0 0 0 # 5 GBB. Gull
1 0 0 0 0 0 2 0 0 0 0 # 6 Gannet
1 0 0 0 0 0 2 0 0 0 0 # 7 Puffin
1 0 0 0 0 0 2 0 0 0 0 # 8 Razorbill
3 0 0 0 0 0 2 0 0 0 0 # 9 A. radiata
4 0 0 0 0 0 2 0 0 0 0 # 10 G. gurnards
2 0 0 0 0 0 2 0 0 0 0 # 11 W.horse mac
3 0 0 0 0 0 2 0 0 0 0 # 12 N.horse mac
1 0 0 0 0 0 2 0 0 0 0 # 13 Grey seal
1 0 0 0 0 0 2 0 0 0 0 # 14 H. porpoise
3 0 0 0 0 0 2 0 0 0 0 # 15 Hake
10 1 9 0 10 1 1 1 118000 0 0 # 16 Cod
8 0 6 0 8 1 1 1 184000 0 0 # 17 Whiting
10 0 7 0 10 1 1 1 1e+05 0 0 # 18 Haddock
10 3 8 0 10 1 1 0 1 0 0 # 19 Saithe
10 1 7 0 10 1 1 0 3 0 0 # 20 Mackerel
9 0 5 0 9 1 0 1 1 0 0 # 21 Herring

```

```

4 0 3 0 4 1 0 1 1 0 0 # 22 N. sandeel
4 0 3 0 4 1 0 1 1 0 0 # 23 S. sandeel
3 0 2 0 3 0 0 1 1 0 0 # 24 Nor. pout
3 1 2 0 3 1 0 1 90000 0 0 # 25 Sprat
10 1 7 0 10 1 0 0 1 0 0 # 26 Plaice
10 2 7 0 10 1 0 0 1 0 0 # 27 Sole
#####
## use input recruitment estimate (option use.known.rec)
# 0=estimate all recruitments
# 1=yes use input recruitment from file known_recruitment.in
0
#####
## adjustment factor to bring the beta parameter close to one (option
beta.cor)
1e+06 # Cod
1e+06 # Whiting
1e+05 # Haddock
1e+05 # Saithe
1e+06 # Mackerel
1e+05 # Herring
1e+05 # N. sandeel
1e+06 # S. sandeel
1e+06 # Nor. pout
1e+06 # Sprat
1e+06 # Plaice
1e+05 # Sole
#####
## year range for data included to fit the R-SSB relation (option
SSB.R.year.range)
# first (option SSB.R.year.first) and last (option SSB.R.year.last) year
to consider.
# the value -1 indicates the use of the first (and last) available year
in time-series
# first year by species
1988 # Cod
1982 # Whiting
1988 # Haddock
-1 # Saithe
1980 # Mackerel
-1 # Herring
-1 # N. sandeel

```

```

-1 # S. sandeel
1977 # Nor. pout
1981 # Sprat
-1 # Plaice
-1 # Sole
# last year by species
-1 # Cod
-1 # Whiting
-1 # Haddock
-1 # Saithe
-1 # Mackerel
-1 # Herring
-1 # N. sandeel
-1 # S. sandeel
-1 # Nor. pout
-1 # Sprat
-1 # Plaice
-1 # Sole

#####
## Objective function weighting by species (option objec-
tive.function.weight)
# first=catch observations,
# second=CPUE observations,
# third=SSB/R relations
# fourth=stomach observations, weight proportions
# fifth=stomach observations, number at length
##
0 0 0 0.1 1 # 1 Fulmar
0 0 0 0.1 1 # 2 Guillemot
0 0 0 0.1 1 # 3 Her. Gull
0 0 0 0.1 1 # 4 Kittiwake
0 0 0 0.1 1 # 5 GBB. Gull
0 0 0 0.1 1 # 6 Gannet
0 0 0 0.1 1 # 7 Puffin
0 0 0 0.1 1 # 8 Razorbill
0 0 0 1 1 # 9 A. radiata
0 0 0 1 1 # 10 G. gurnards
0 0 0 1 1 # 11 W.horse mac
0 0 0 1 1 # 12 N.horse mac
0 0 0 1 1 # 13 Grey seal

```

```

0 0 0 1 1 # 14 H. porpoise
0 0 0 1 1 # 15 Hake
1 1 0.1 1 0 # 16 Cod
1 1 0.1 1 0 # 17 Whiting
1 1 0.1 1 0 # 18 Haddock
1 1 0.3 1 0 # 19 Saithe
1 1 0.1 1 0 # 20 Mackerel
1 1 0.1 0 0 # 21 Herring
1 1 0.1 0 0 # 22 N. sandeel
1 1 0.1 0 0 # 23 S. sandeel
1 1 0.1 0 0 # 24 Nor. pout
1 1 0.1 0 0 # 25 Sprat
1 1 0.1 0 0 # 26 Plaice
1 1 0.1 0 0 # 27 Sole
#####
## parameter estimation phases for single species parameters
# phase.rec (stock numbers, first age) (default=1)
1
# phase.rec.older (stock numbers, first year and all ages) (default=1)
1
# phase.F.y (year effect in F model) (default=1)
1
# phase.F.y.spline (year effect in F model, implemented as spline func-
tion)
-1
# phase.F.q (season effect in F model) (default=1)
1
# phase.F.a (age effect in F model) (default=1)
1
# phase.catchability (survey catchability) (default=1)
1
# phase.SSB.R.alfa (alfa parameter in SSB-recruitment relation) (de-
fault=1)
1
# phase.SSB.R.beta (beta parameter in SSB-recruitment relation) (de-
fault=1)
1
#####
## minimum CV of catch observation used in ML-estimation (option
min.catch.CV)
0.1

```

```
#####
## minimum CV of catch SSB-recruitment relation used in ML-estimation
(option min.SR.CV)
0.2
#####
## Use proportion landed information in calculation of yield (option
calc.discard)
# 0=all catches are included in yield
# 1=yield is calculated from proportion landed (file propor-
tion_landed.in)
1 # Cod
1 # Whiting
1 # Haddock
1 # Saithe
0 # Mackerel
0 # Herring
0 # N. sandeel
0 # S. sandeel
0 # Nor. pout
0 # Sprat
1 # Plaice
1 # Sole
#####
## use seasonal or annual catches in the objective function (option com-
bined.catches)
# do not change this options from default=0, without looking in the manu-
al
#0=annual catches with annual time steps or seasonal catches with season-
al time steps
#1=annual catches with seasonal time steps, read seasonal relative F from
file F_q_ini.in (default=0)
1 # Cod
1 # Whiting
1 # Haddock
1 # Saithe
1 # Mackerel
0 # Herring
0 # N. sandeel
0 # S. sandeel
0 # Nor. pout
0 # Sprat
1 # Plaice
```

```

1 # Sole
#####
## use seasonal or common combined variances for catch observation
# seasonal=0, common=1 (use 1 for annual data)

1 # Cod
1 # Whiting
1 # Haddock
1 # Saithe
1 # Mackerel
0 # Herring
0 # N. sandeel
0 # S. sandeel
0 # Nor. pout
0 # Sprat
1 # Plaice
1 # Sole
#####
##
# catch observations: number of separate catch variance groups by species

4 # Cod
4 # Whiting
5 # Haddock
3 # Saithe
3 # Mackerel
3 # Herring
3 # N. sandeel
2 # S. sandeel
3 # Nor. pout
3 # Sprat
3 # Plaice
2 # Sole

# first age group in each catch variance group
1 2 7 9 # Cod
0 1 2 5 # Whiting
0 1 2 6 8 # Haddock
3 5 8 # Saithe
1 2 3 # Mackerel
0 1 8 # Herring

```

```

0 1 4 # N. sandeel
0 1   # S. sandeel
0 1 3 # Nor. pout
1 2 3 # Sprat
1 2 3 # Plaice
2 3   # Sole

#####

##

# catch observations: number of separate catch seasonal component groups
# by species

      4 #      Cod
      4 #      Whiting
      3 #      Haddock
      2 #      Saithe
      3 #      Mackerel
      2 #      Herring
      3 #      N. sandeel
      3 #      S. sandeel
      3 #      Nor. pout
      3 #      Sprat
      2 #      Plaice
      1 #      Sole

# first ages in each seasonal component group by species
1 2 3 5      # Cod
0 1 2 3      # Whiting
0 1 2 # Haddock
3 4   # Saithe
1 2 4 # Mackerel
0 1   # Herring
0 1 2 # N. sandeel
0 1 2 # S. sandeel
0 1 3 # Nor. pout
1 2 3 # Sprat
1 2   # Plaice
2     # Sole

#####

## first and last age in calculation of average F by species (option
## avg.F.ages)
2 4 # Cod

```



```

2 6 # Whiting
2 6 # Haddock
3 6 # Saithe
4 8 # Mackerel
2 6 # Herring
1 2 # N. sandeel
1 2 # S. sandeel
1 2 # Nor. pout
1 2 # Sprat
3 6 # Plaice
2 6 # Sole
#####
## minimum 'observed' catch, (option min.catch). You cannot log zero
## catch at age!
#
# 0 ignore observation in likelihood
#
# negative value gives percentage (e.g. -10 ~ 10%) of average catch in
# age-group for input catch=0
# negative value less than -100 substitute all catches by the option/100
# /100 *average catch in the age group for catches less than (average
# catch*-option/10000
#
# if option>0 then will zero catches be replaced by catch=option
#
# else if option<0 and option >-100 and catch=0 then catches will be re-
# placed by catch=average(catch at age)*(-option)/100
# else if option<-100 and catch < average(catch at age)*(-option)/10000
# then catches will be replaced by catch=average(catch at age)*(-
# option)/10000
0 # Cod
0 # Whiting
0 # Haddock
0 # Saithe
0 # Mackerel
0 # Herring
0 # N. sandeel
0 # S. sandeel
0 # Nor. pout
0 # Sprat
0 # Plaice
0 # Sole

```

```
#####
```

```
##
```

```
# catch observations: number of year groups with the same age and season-
al selection
```

```

3   #      Cod
3   #      Whiting
3   #      Haddock
2   #      Saithe
3   #      Mackerel
4   #      Herring
2   #      N. sandeel
2   #      S. sandeel
2   #      Nor. pout
2   #      Sprat
3   #      Plaice
2   #      Sole
```

```
# first year in each group (please note #1 will always be changed to
first model year)
```

```

1974 1993 2007      # Cod
1974 1991 2007      # Whiting
1974 1985 2000      # Haddock
1974 1992      # Saithe
1974 1980 2004      # Mackerel
1974 1978 1983 1998      # Herring
1974 2005      # N. sandeel
1974 2005      # S. sandeel
1974 2003      # Nor. pout
1974 1996      # Sprat
1974 1990 2003      # Plaice
1974 1990      # Sole
```

```
#####
```

```
##
```

```
# number of nodes for year effect Fishing mortality spline
```

```
# 1=no spline (use one Fy for each year), >1 number of nodes
```

```

1   #      Cod
1   #      Whiting
1   #      Haddock
1   #      Saithe
1   #      Mackerel
```

```

1 # Herring
1 # N. sandeel
1 # S. sandeel
1 # Nor. pout
1 # Sprat
1 # Plaice
1 # Sole

# first year in each group
1975 # Cod
1975 # Whiting
1975 # Haddock
1975 # Saithe
1975 # Mackerel
1975 # Herring
1975 # N. sandeel
1975 # S. sandeel
1975 # Nor. pout
1975 # Sprat
1975 # Plaice
1975 # Sole
#####
## year season combinations with zero catch (F=0) (option ze-
ro.catch.year.season)
# 0=no, all year-seasons have catches,
# 1=yes there are year-season combinations with no catch.
# Read from file zero_catch_seasons_ages.in
# default=0
1
#####
## season age combinations with zero catch (F=0) (option ze-
ro.catch.season.ages)
# 0=no, all seasons have catches,
# 1=yes there are seasons with no catch. Read from file ze-
ro_catch_season_ages.in
# default=0
1
#####
## Factor for fixing last season effect in F-model (default=1)
(fix.F.factor))
1 # Cod

```



```

## N in the beginning of the period or N bar for calculation of M2 (option use.Nbar)
# 0=use N in the beginning of the time step (default)
# 1=use N bar
0

#####
## Maximum M2 iterations (option M2.iterations) in case of use.Nbar=1
3

#####
## convergence criteria (option max.M2.sum2) in case of use.Nbar=1
# use max.M2.sum2=0.0 and M2.iterations=7 (or another high number) to make Hessian
3

#####
## likelihood model for stomach content observations (option stom.likelihood)
# 1 =likelihood from prey weight proportions only (see option below)
# 2 =likelihood from prey weight proportions and from prey numbers to estimate size selection
# 3 =Gamma distribution for prey absolute weight and size selection from prey numbers
1

#####
# Variance used in likelihood model for stomach contents as prey weight proportion
# (option stomach.variance)
# 0 =not relevant,
# 1 =log normal distribution,
# 2 =normal distribution,
# 3 =Dirichlet distribution
3

#####
## Usage of age-length-keys for calc of M2 (option simple.ALK)
# 0=Use only one sizegroup per age (file lsea.in or west.in)
# 1=Use size distribution per age (file ALK_all.in)
0

#####
## Usage of food-rations from input values or from size and regression parameters (option consum)
# 0=Use input values by age (file consum.in)
# 1=use weight at age (file west.in) and regression parameters (file consum_ab.in)

```

```

# 2=use length at age (file lsea.in), l-w relation and regression param-
eters (file consum_ab.in)

1

#####

## Size selection model based on (option size.select.model)

# l=length:

#     M2 calculation:

#         Size preference:

#             Predator length at age from file: lsea.in

#             Prey     length at age from file: lsea.in

#             Prey mean weight is weight in the sea from file: west.in

#     Likelihood:

#         Size preference:

#             Predator mean length per length group (file:
stom_pred_length_at_sizecl.in)

#             Prey mean length per ength group (file stomlen_at_length.in

#             Prey mean weight from mean weight per prey length group (file:
stomweight_at_length.in

# 2=weight:

#     M2 calculation:

#         Size preference:

#             Predator weight at age from file: west.in

#             Prey     weight at age from file: west.in

#             Prey mean weight is weight in the sea from file: west.in

#     Likelihood:

#         Size preference

#             Predator mean weight is based on mean length per predator
length group (file: stom_pred_length_at_sizecl.in)

#             and l-w relation (file: length_weight_relations.in),

#             Prey mean weight per prey length group (file: stom-
weight_at_length.in)

#             Prey mean weight from mean weight per prey length group (file:
stomweight_at_length.in

# 3=weight:

#     M2 calculation: Same as option 2

#     Likelihood:

#         Size preference:

#             Predator mean weight is based on mean length per predator
length group (file: stom_pred_length_at_sizecl.in)

#             and l-w relation (file: length_weight_relations.in),

#             Prey mean weight per prey length group (file: stom-
len_at_length.in)

```

```

#           and l-w relation (file:length_weight_relations.in)
#           Prey mean weight from prey mean length per prey length group
#           (file: stomlen_at_length.in) and
#           l-w relation (file: length_weight_relations.in)
# 4=weight:
#           M2 calculation:
#           Size preference:
#           Predator mean weight from file lsea.in (length in the sea)
and
#           l-w relation (file: length_weight_relations.in)
#           Prey mean weight from file lsea.in (length in the sea) and
#           l-w relation (file: length_weight_relations.in)
#           Likelihood: Same as option 3
# 5=weight in combination with simple.ALK=1:
#           M2 calculation:
#           Size preference:
#           Predator weight based on length from file ALK_all.in,
#           (length distribution at age) and
#           l-w relation (file: length_weight_relations.in)
#           Prey weight based on length from file ALK_all.in (length
distribution at age) and l-w relation (file: length_weight_relations.in)
#           Prey mean weight based on length from file ALK_all.in (length
distribution at age) and l-w relation (file: length_weight_relations.in)
#           Likelihood: Same as for option 2
# 6=weight in combination with simple.ALK=1:
#           M2 calculation: Same as option 5
#           Likelihood: Same as option 3
2
#####
# Adjust Length at Age distribution by a mesh selection function (option
L50.mesh)
# Please note that options simple.ALK should be 1 and option
size.select.model should be 5
# L50 (mm) is optional given as input. Selection Range is estimated by
the model
# L50= -1 do not adjust
# L50=0, estimate L50 and selection range
# L50>0, input L50 (mm) and estimate selection range
# by VPA species
-1 # Cod
-1 # Whiting
-1 # Haddock

```

```

-1 #      Saithe
-1 #      Mackerel
-1 #      Herring
-1 #      N. sandeel
-1 #      S. sandeel
-1 #      Nor. pout
-1 #      Sprat
-1 #      Plaice
-1 #      Sole

#####

## spread of size selection (option size.selection)
#  0=no size selection, predator/preys size range defined from observa-
#  tions
#  1=normal distribution size selection
#  3=Gamma distribution size distribution
#  4=no size selection, but range defined by input min and max regres-
#  sion parameters (file pred_preys_size_range_param.in)
#  5=Beta distributed size distribution, within observed size range
#  6=log-Beta size distributed, within observed size range
#
# by predator
    0 #      Fulmar
    0 #      Guillemot
    0 #      Her. Gull
    0 #      Kittiwake
    0 #      GBB. Gull
    0 #      Gannet
    0 #      Puffin
    0 #      Razorbill
    0 #      A. radiata
    0 #      G. gurnards
    0 #      W.horse mac
    0 #      N.horse mac
    0 #      Grey seal
    4 #      H. porpoise
    0 #      Hake
    0 #      Cod
    0 #      Whiting
    0 #      Haddock
    0 #      Saithe

```



```

0 #      Mackerel
#####

## sum stomach contents over prey size for use in likelihood for prey
weight proportions (option sum.stom.like)

# 0=no, use observations as they are; 1=yes, sum observed and predicted
stomach contents before used in likelihood for prey weight proportions
#
# by predator
1 #      Fulmar
1 #      Guillemot
1 #      Her. Gull
1 #      Kittiwake
1 #      GBB. Gull
1 #      Gannet
1 #      Puffin
1 #      Razorbill
1 #      A. radiata
1 #      G. gurnards
1 #      W.horse mac
1 #      N.horse mac
1 #      Grey seal
1 #      H. porpoise
1 #      Hake
1 #      Cod
1 #      Whiting
1 #      Haddock
1 #      Saithe
1 #      Mackerel
#####

## # Use estimated scaling factor to link number of observation to vari-
ance for stomach observation likelihood (option stom_obs_var)

# 0=no, do not estimate factor (assumed=1); 1=yes, estimate the fac-
tor; 2=equal weight (1) for all samples
#
# by predator
1 #      Fulmar
1 #      Guillemot
1 #      Her. Gull
1 #      Kittiwake
1 #      GBB. Gull
1 #      Gannet

```

```

1 #      Puffin
1 #      Razorbill
1 #      A. radiata
1 #      G. gurnards
1 #      W.horse mac
1 #      N.horse mac
1 #      Grey seal
1 #      H. porpoise
1 #      Hake
1 #      Cod
1 #      Whiting
1 #      Haddock
1 #      Saithe
1 #      Mackerel

#####

## # Upper limit for Dirichlet sumP. A low value (e.g. 10) limits the
risk of overfitting. A high value (e.g. 100) allows a full fit. (option
stom_max_sumP)

# by predator

100 #      Fulmar
100 #      Guillemot
100 #      Her. Gull
100 #      Kittiwake
100 #      GBB. Gull
100 #      Gannet
100 #      Puffin
100 #      Razorbill
100 #      A. radiata
100 #      G. gurnards
100 #      W.horse mac
100 #      N.horse mac
100 #      Grey seal
100 #      H. porpoise
100 #      Hake
100 #      Cod
100 #      Whiting
100 #      Haddock
100 #      Saithe
100 #      Mackerel

#####

```

```
## Scaling factor (to bring parameters close to one) for relation between
no of stomachs sampling and variance
```

```
# value=0: use default values i.e. 1.00 for no size selection and other-
wise 0.1 (option var.scale.stom)
```

```
1 # Fulmar
1 # Guillemot
1 # Her. Gull
1 # Kittiwake
1 # GBB. Gull
1 # Gannet
1 # Puffin
1 # Razorbill
1 # A. radiata
1 # G. gurnards
1 # W.horse mac
1 # N.horse mac
1 # Grey seal
1 # H. porpoise
100 # Hake
1 # Cod
1 # Whiting
1 # Haddock
1 # Saithe
1 # Mackerel
```

```
#####
```

```
## other food suitability size dependency (option size.other.food.suit)
```

```
# 0=no size dependency
```

```
# 1=yes, other food suitability is different for different size classes
```

```
0 # Fulmar
0 # Guillemot
0 # Her. Gull
0 # Kittiwake
0 # GBB. Gull
0 # Gannet
0 # Puffin
0 # Razorbill
1 # A. radiata
0 # G. gurnards
0 # W.horse mac
0 # N.horse mac
```

```

0 # Grey seal
0 # H. porpoise
0 # Hake
0 # Cod
1 # Whiting
1 # Haddock
1 # Saithe
1 # Mackerel

```

```
#####
```

```
## Minimum observed relative stomach contents weight for inclusion in ML
estimation (option min.stom.cont)
```

```

9e-05 # Fulmar
9e-05 # Guillemot
9e-05 # Her. Gull
9e-05 # Kittiwake
9e-05 # GBB. Gull
9e-05 # Gannet
9e-05 # Puffin
9e-05 # Razorbill
9e-05 # A. radiata
9e-05 # G. gurnards
9e-05 # W.horse mac
9e-05 # N.horse mac
9e-05 # Grey seal
9e-05 # H. porpoise
9e-09 # Hake
9e-09 # Cod
9e-09 # Whiting
9e-09 # Haddock
9e-05 # Saithe
9e-05 # Mackerel

```

```
#####
```

```
## Upper limit for no of samples used for calculation of stomach observa-
tion variance (option max.stom.sampl)
```

```

1000 # Fulmar
1000 # Guillemot
1000 # Her. Gull
1000 # Kittiwake
1000 # GBB. Gull
1000 # Gannet

```

```

1000 #      Puffin
1000 #      Razorbill
1000 #      A. radiata
1000 #      G. gurnards
1000 #      W.horse mac
1000 #      N.horse mac
1000 #      Grey seal
1000 #      H. porpoise
1000 #      Hake
1000 #      Cod
1000 #      Whiting
1000 #      Haddock
1000 #      Saithe
1000 #      Mackerel
#####
## Max prey size/ pred size factor for inclusion in M2 calc (option
max.prey.pred.size.fac)
      5 #      Fulmar
      5 #      Guillemot
      5 #      Her. Gull
      5 #      Kittiwake
      5 #      GBB. Gull
      5 #      Gannet
      5 #      Puffin
      5 #      Razorbill
      0.5 #      A. radiata
      0.5 #      G. gurnards
      0.5 #      W.horse mac
      0.5 #      N.horse mac
      50 #      Grey seal
      50 #      H. porpoise
      0.9 #      Hake
      0.5 #      Cod
      0.9 #      Whiting
      0.5 #      Haddock
      0.5 #      Saithe
      0.5 #      Mackerel
#####
## inclsion of individual stomach contents observations in ML for weight
proportions (option stom.type.include)

```

```

# 1=Observed data
# 2= + (not observed) data within the observed size range (=fill in)
# 3= + (not observed) data outside an observed size range. One obs below
and one above (=tails)
# 4= + (not observed) data for the full size range of a prey species ir-
respective of predator size (=expansion)

2 # Fulmar
2 # Guillemot
2 # Her. Gull
2 # Kittiwake
2 # GBB. Gull
2 # Gannet
2 # Puffin
2 # Razorbill
2 # A. radiata
2 # G. gurnards
2 # W.horse mac
2 # N.horse mac
2 # Grey seal
2 # H. porpoise
2 # Hake
2 # Cod
2 # Whiting
2 # Haddock
2 # Saithe
2 # Mackerel

#####

## use overlap input values by year and season (use.overlap)
# 0: overlap assumed constant or estimated within the model
# 1: overlap index from file overlap.in (assessment only, use overlap
from last year in forecast)
# 2: overlap index from file overlap.in (assessment and forecast)
0

#####

## parameter estimation phases for predation parameters
# the number gives the phase, -1 means no estimation
#
# vulnerability (default=2) (phase phase.vulnera)
2
# other food suitability slope (default=-1) (option
phase.other.suit.slope)

```

```
2
# preferred size ratio (default=2) (option phase.pref.size.ratio)
-1
# predator size ratio adjustment factor (default=-1)
# (option phase.pref.size.ratio.correction)
-1
# prey species size adjustment factor (default=-1)
# (option phase.prey.size.adjustment)
-1
# variance of preferred size ratio (default=2) (option
phase.var.size.ratio)
-1
# season overlap (default=-1) (option phase.season.overlap)
2
# Stomach variance parameter (default=2) (option phase.Stom.var)
2
# Mesh size selection of stomach age length key (default=-1)
# (option phase.mesh.adjust)
-1
#####
```

APPENDIX 5: Comparison of ICES assessment and SMS assessment using fixed M

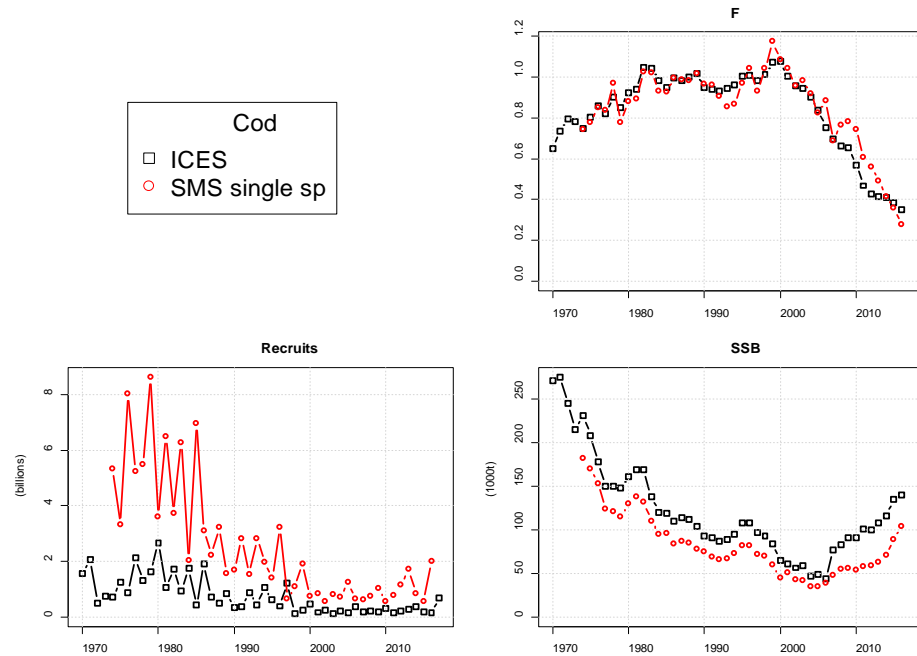


Figure A5.1. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

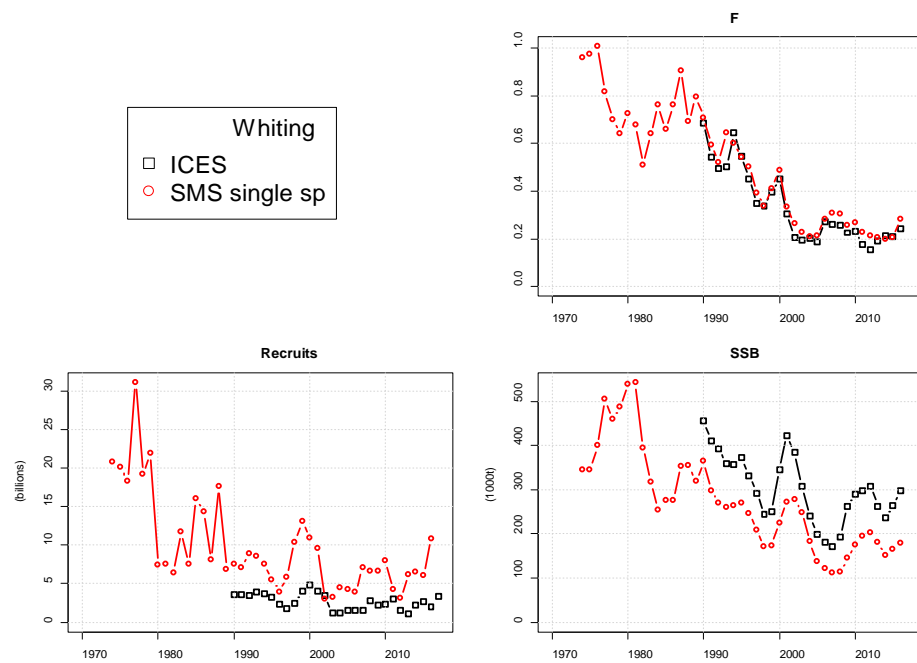


Figure A5.2. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

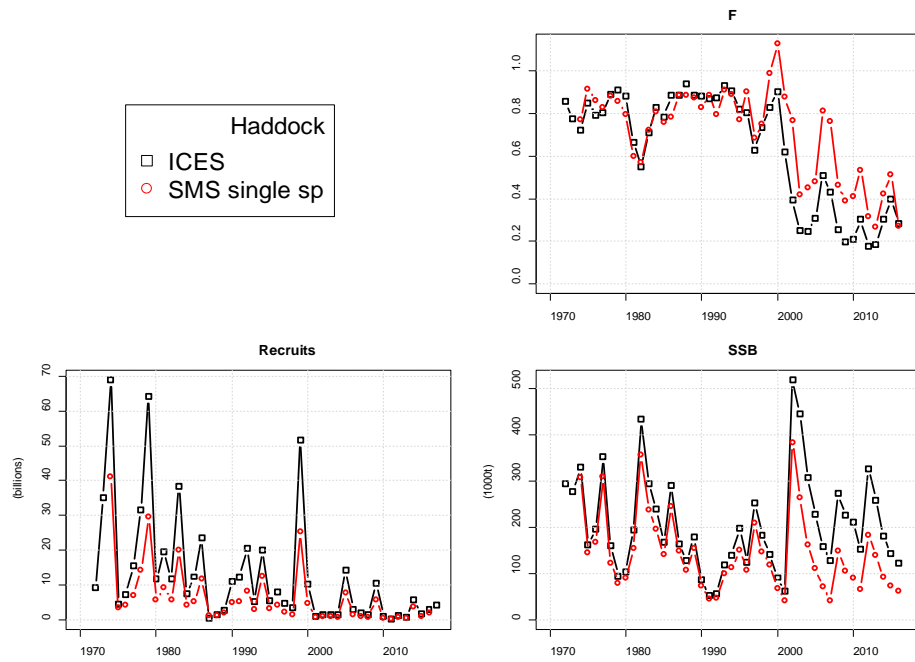


Figure A5.3. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

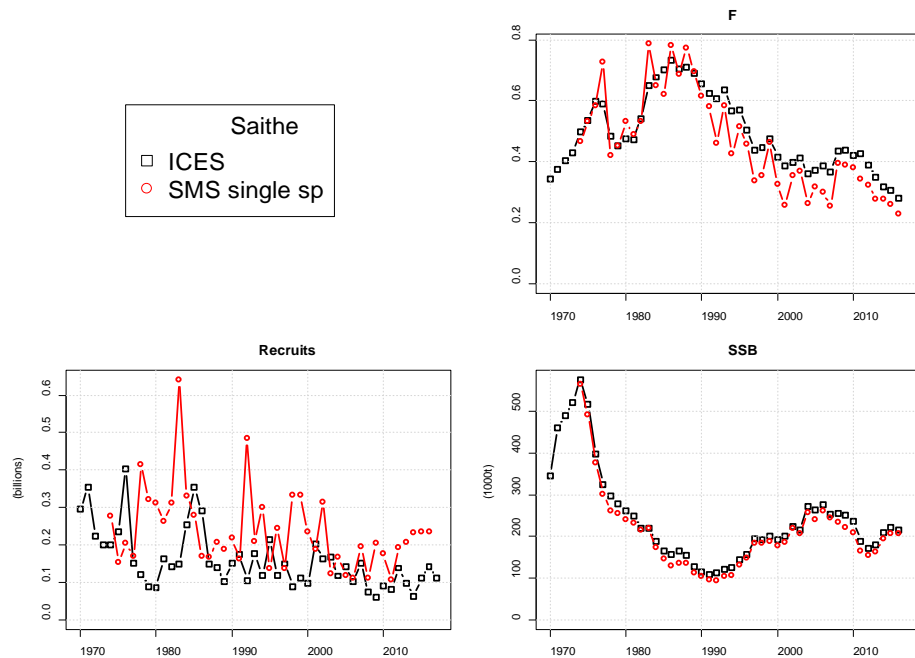


Figure A5.4. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

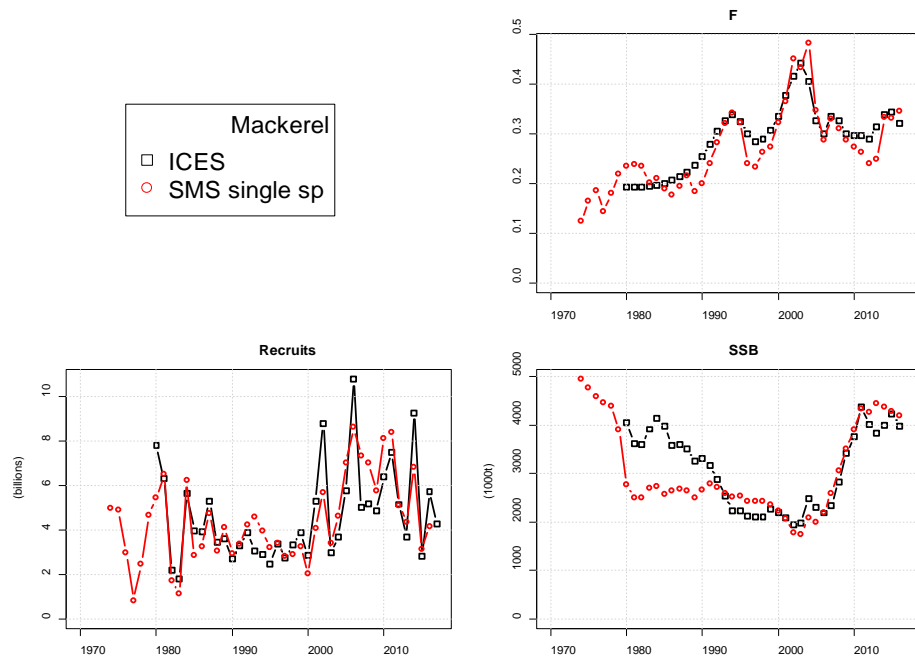


Figure A5.5. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

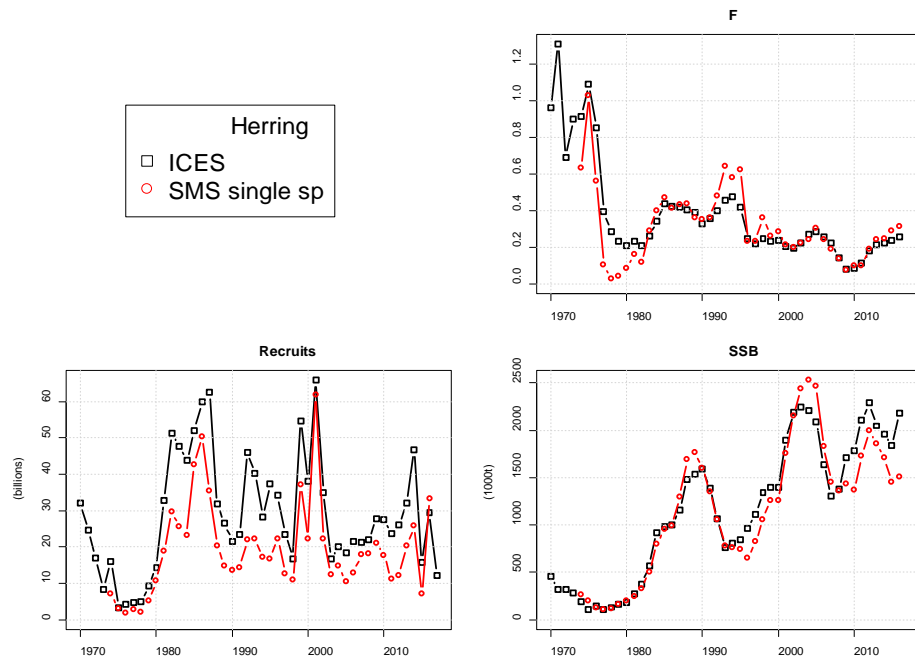


Figure A5.6. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

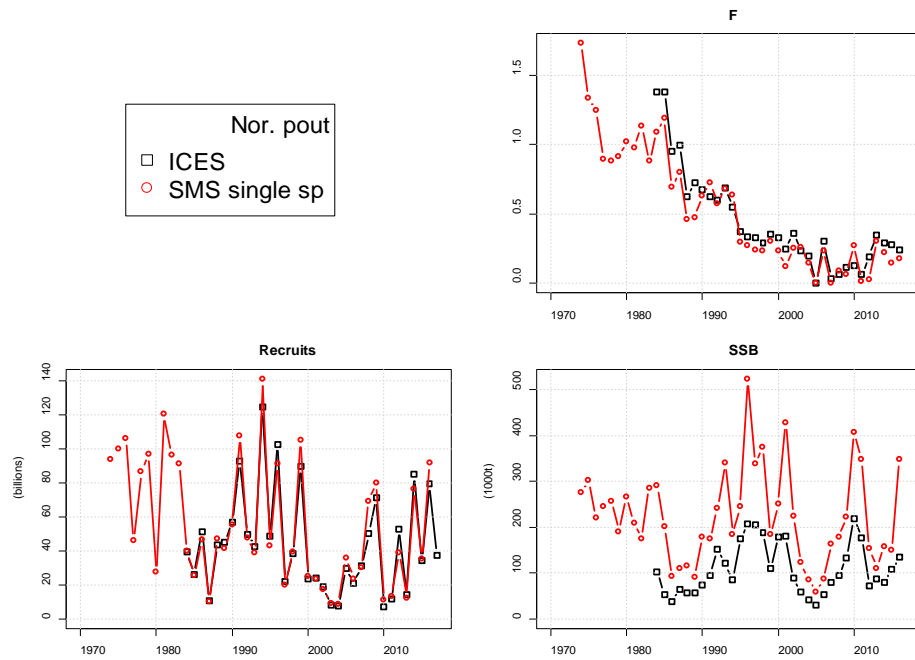


Figure A5.7. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

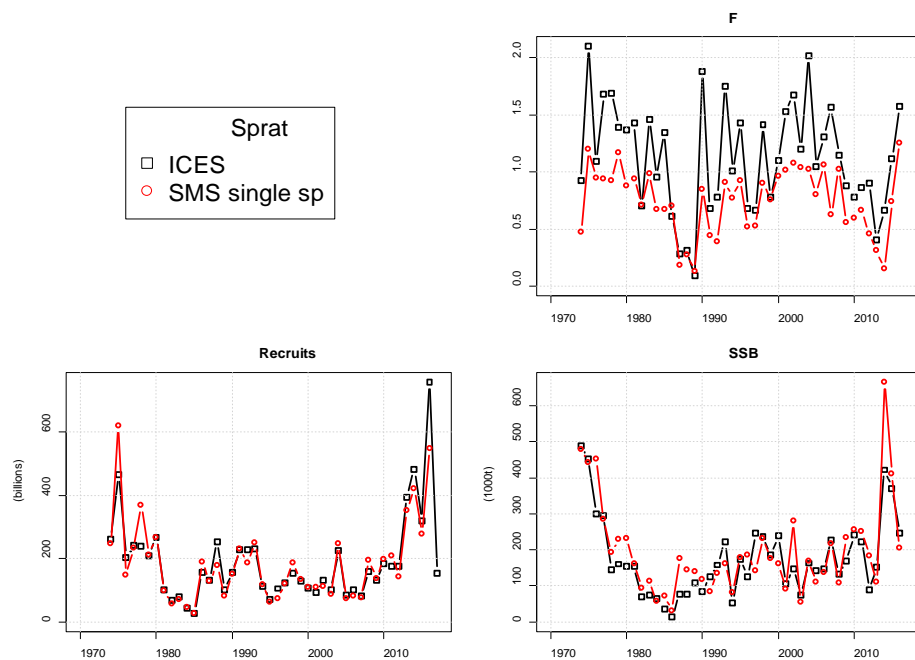


Figure A5.8. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

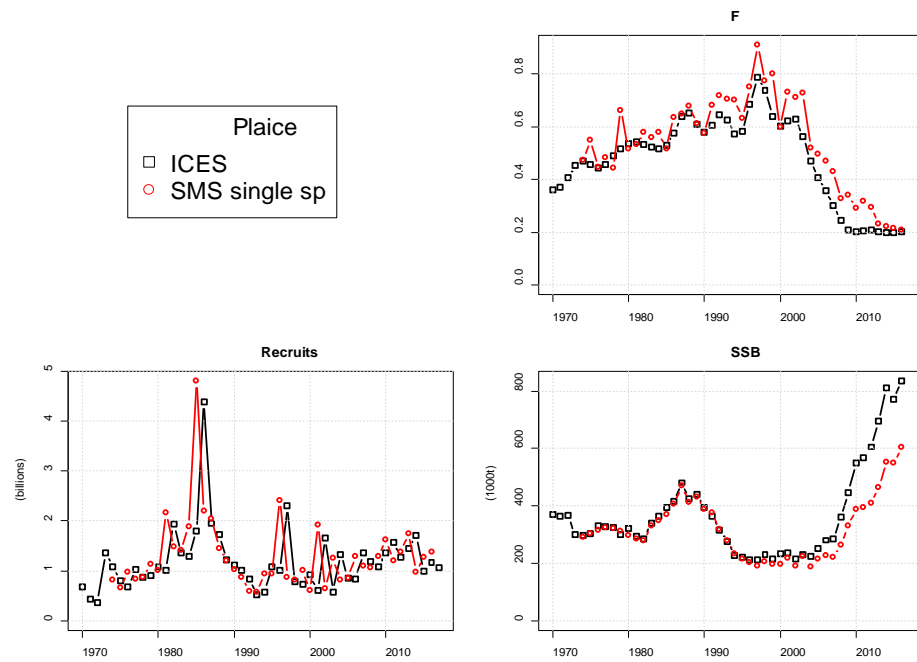


Figure A5.9. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

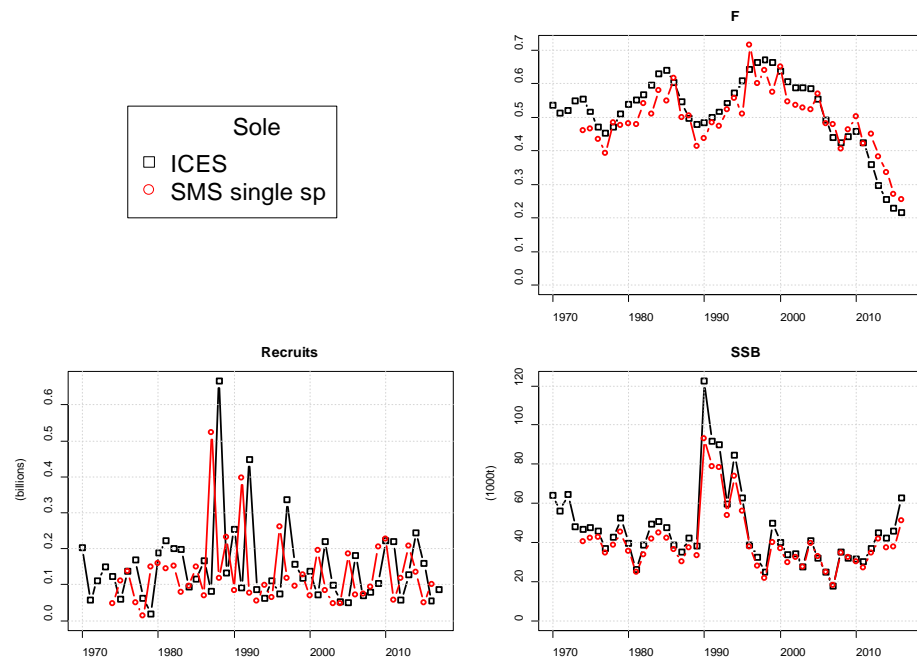
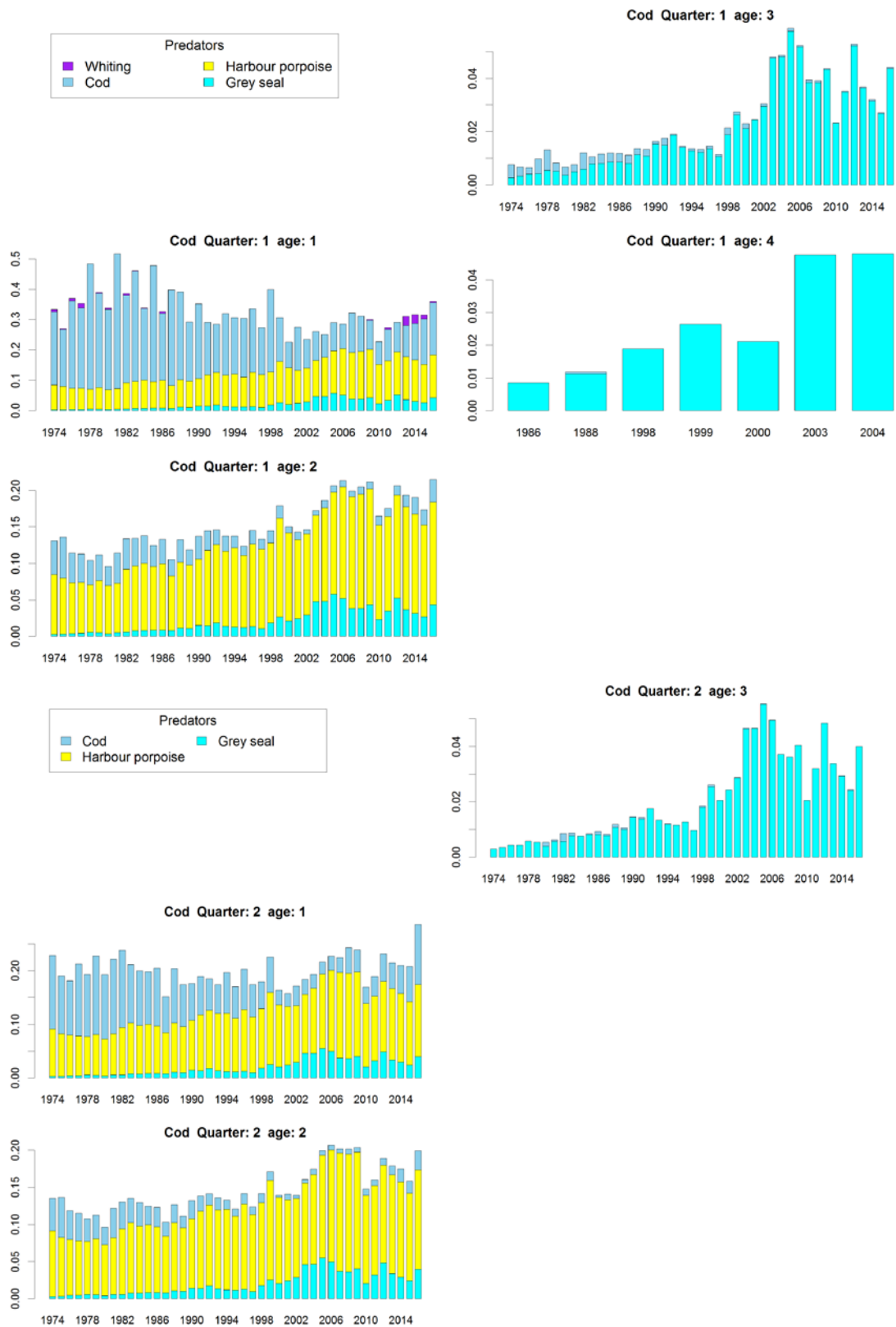
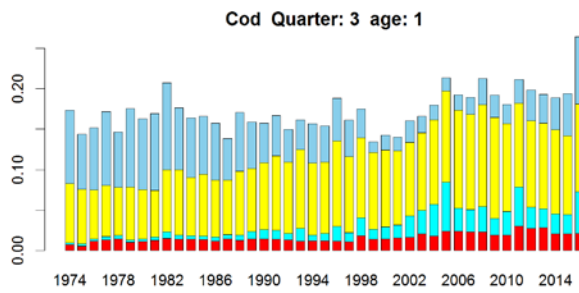
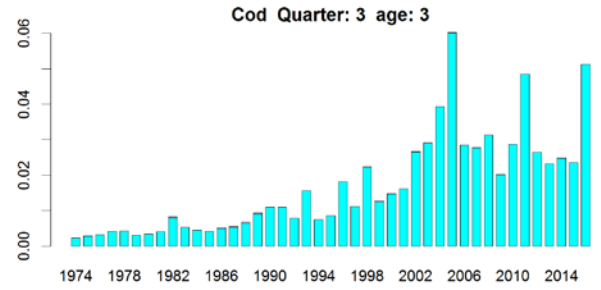
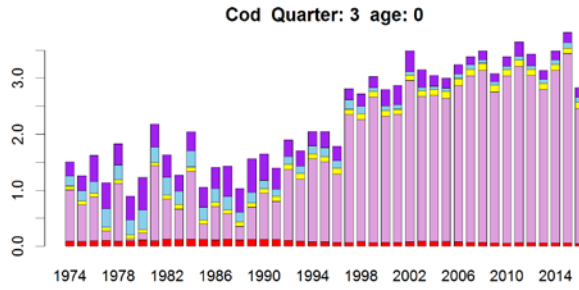
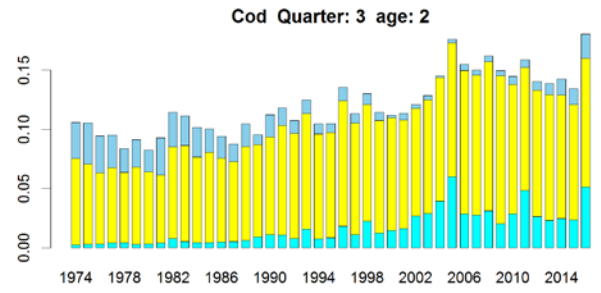
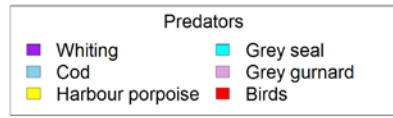
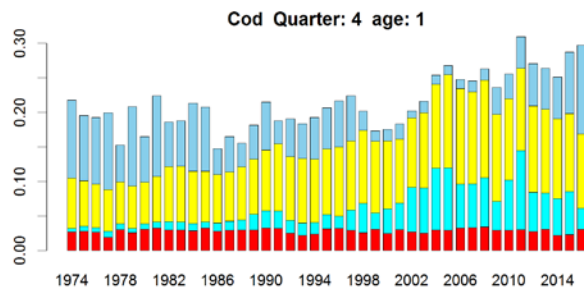
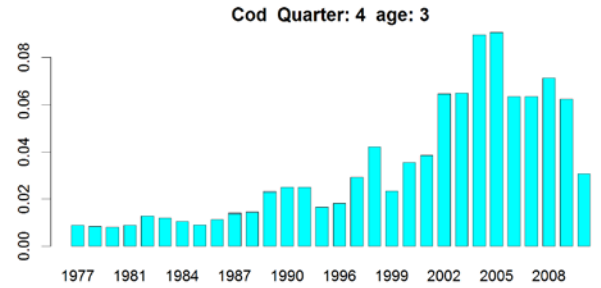
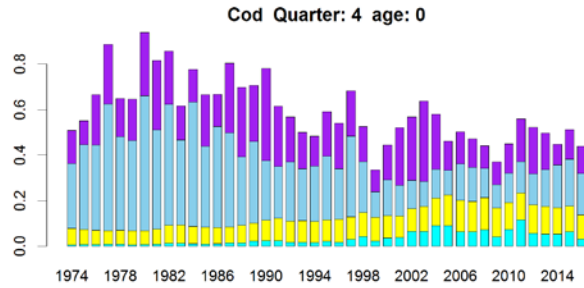
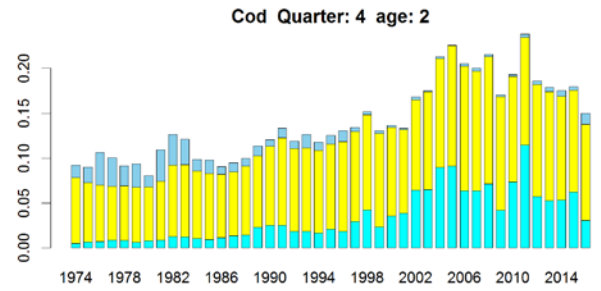


Figure A5.10. Stock summary comparison, ICES single species assessment and SMS in single species mode (constant M)

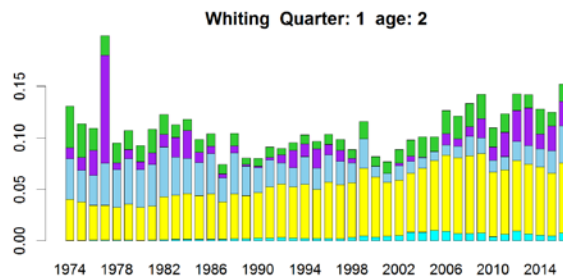
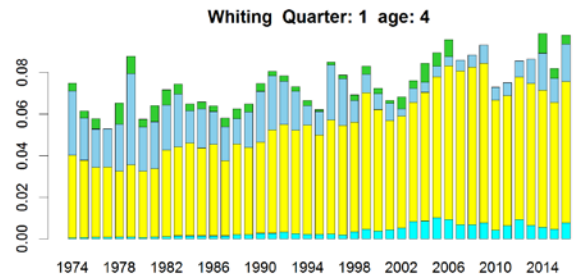
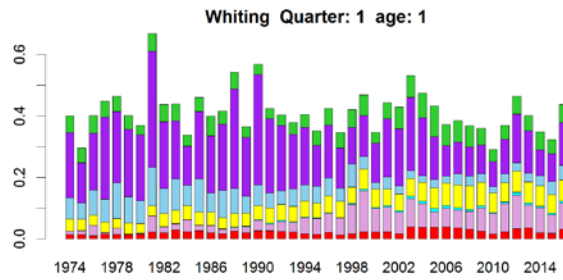
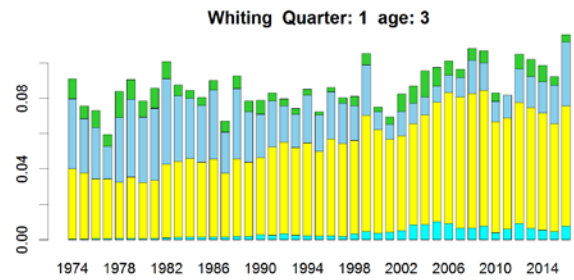
APPENDIX 6: Quarterly predation mortality by prey species and age.**Predation mortality for Cod**

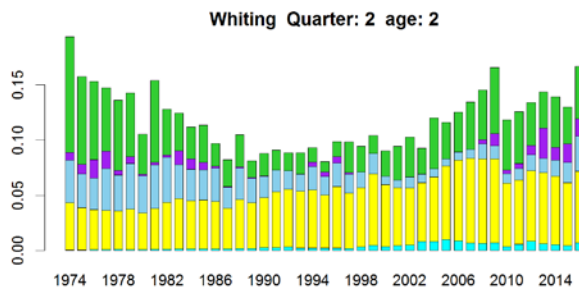
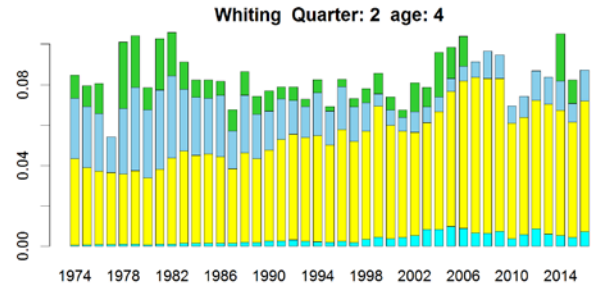
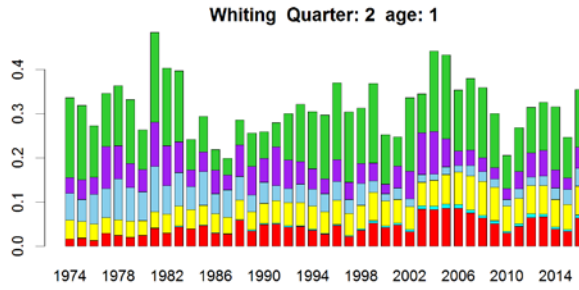
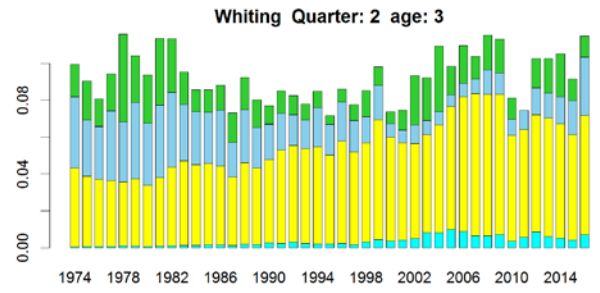
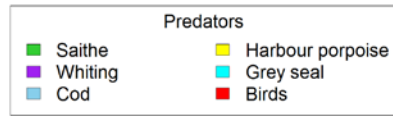






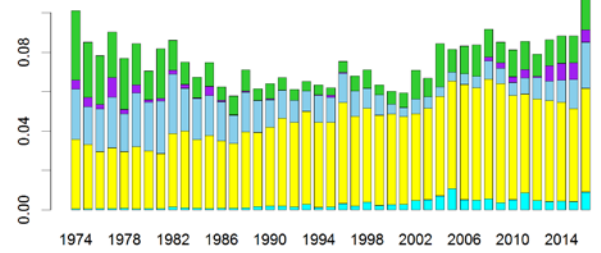
Predation mortality for Whiting



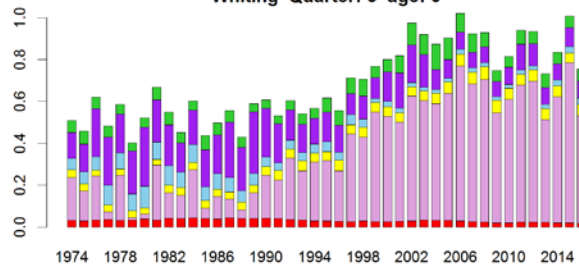




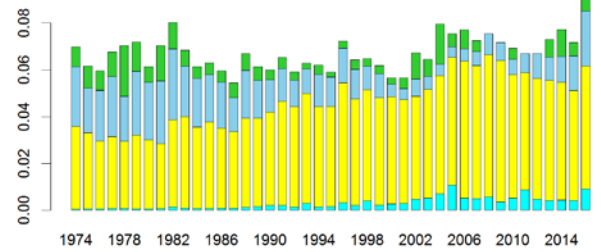
Whiting Quarter: 3 age: 2



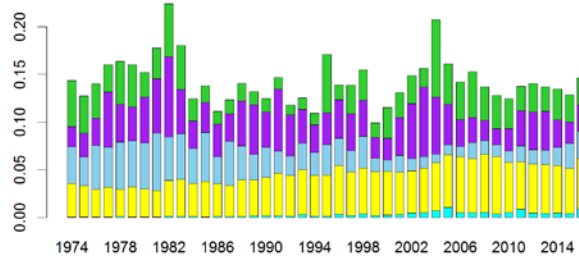
Whiting Quarter: 3 age: 0



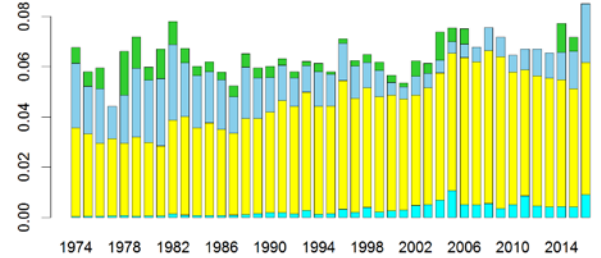
Whiting Quarter: 3 age: 3



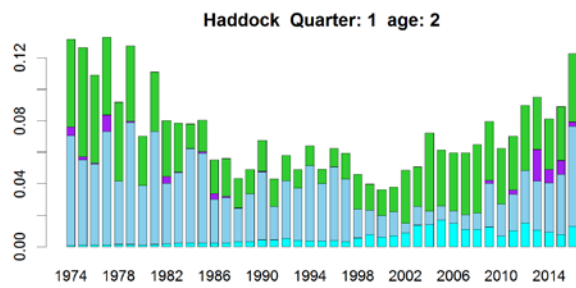
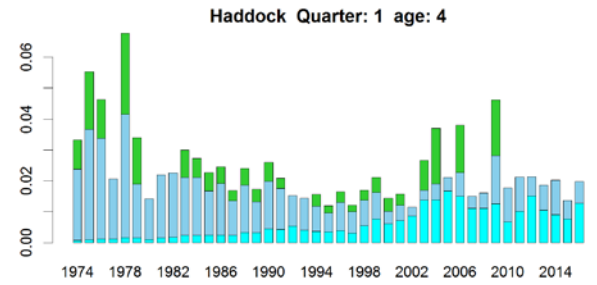
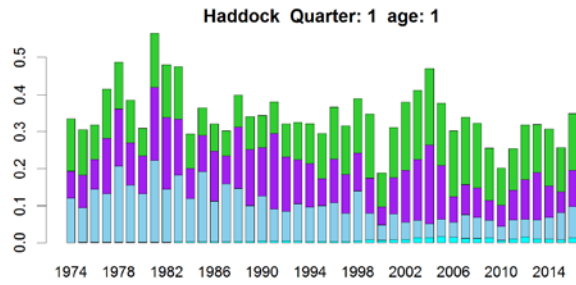
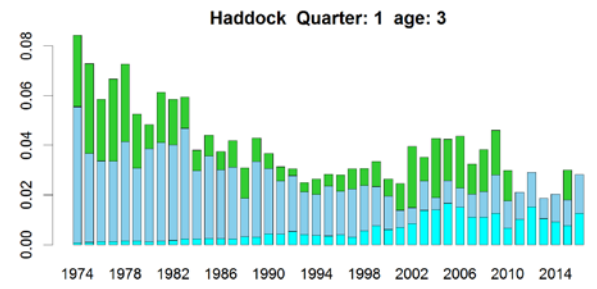
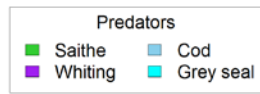
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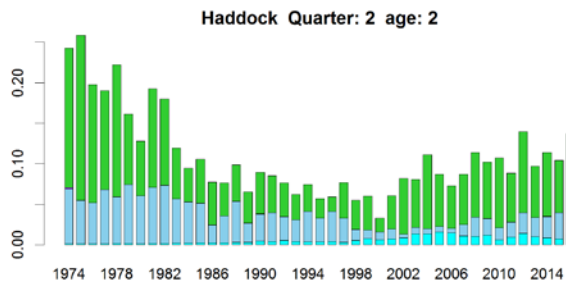
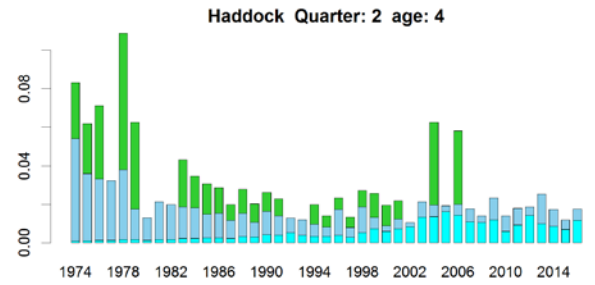
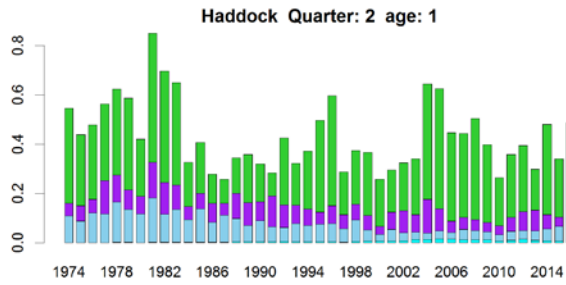
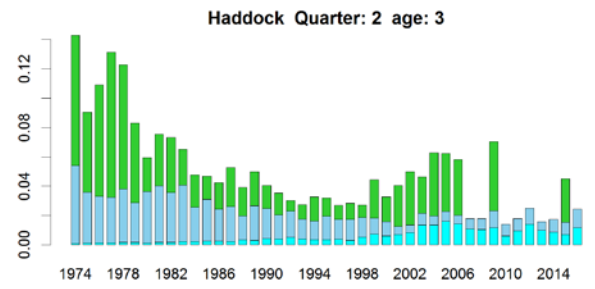
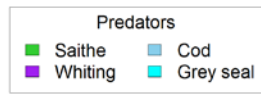


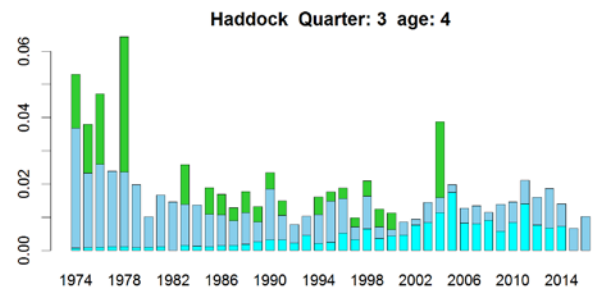
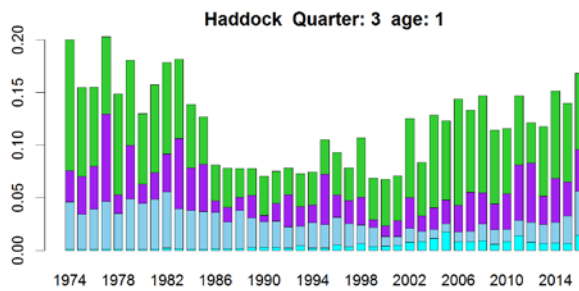
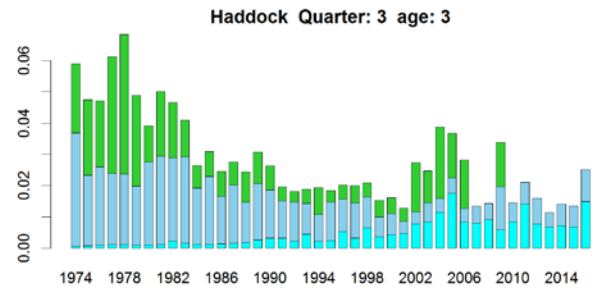
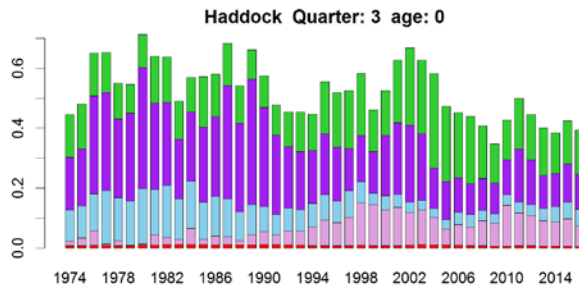
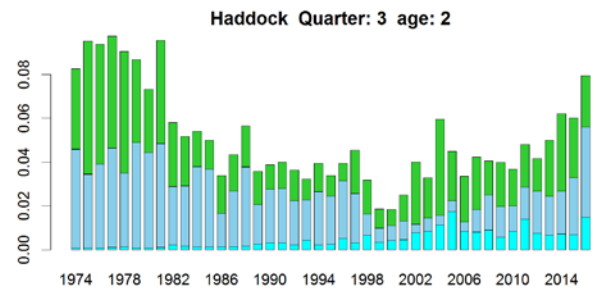
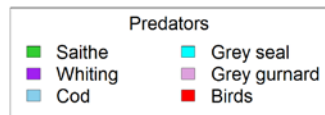
Whiting Quarter: 3 age: 4

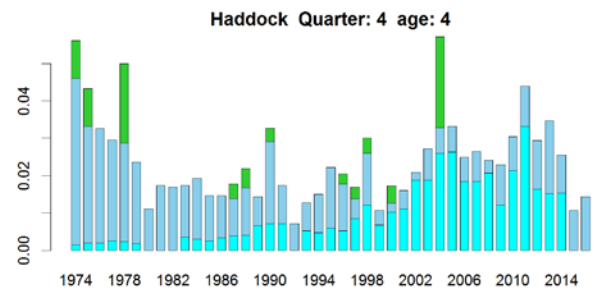
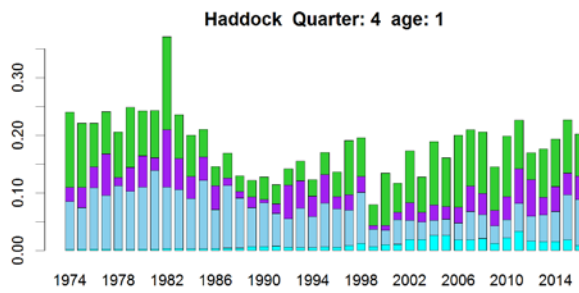
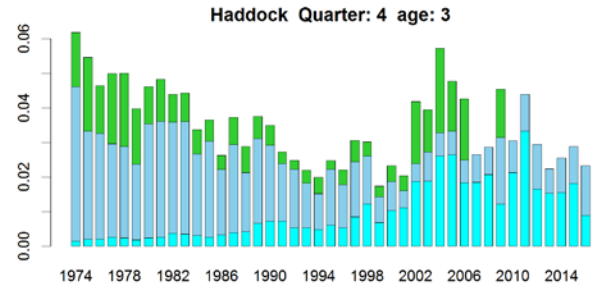
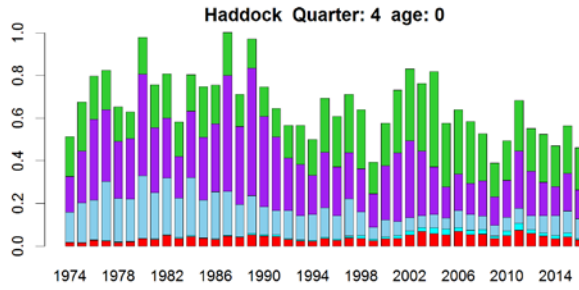
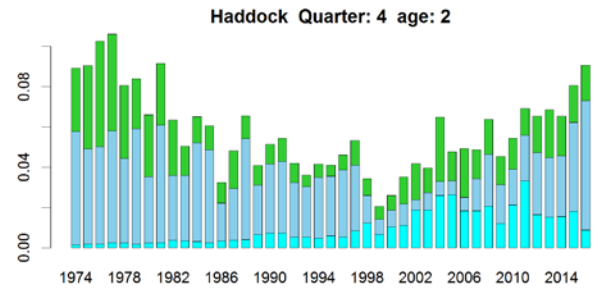
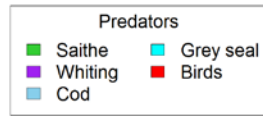




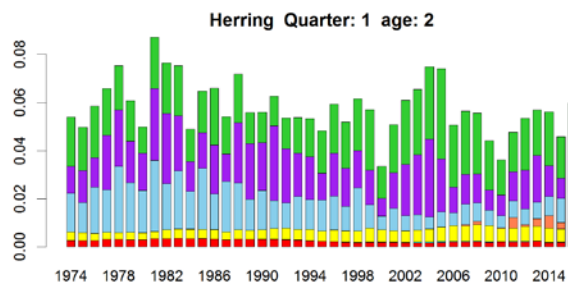
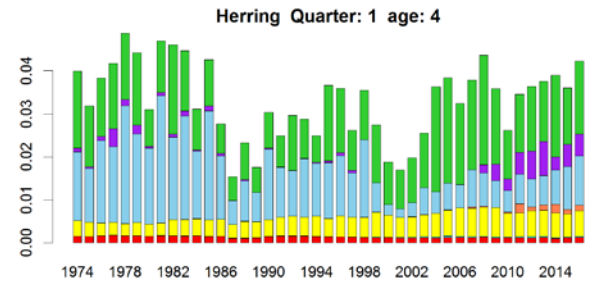
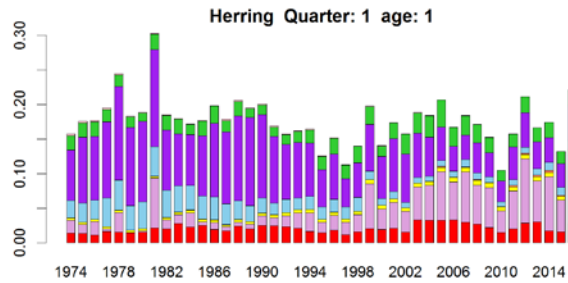
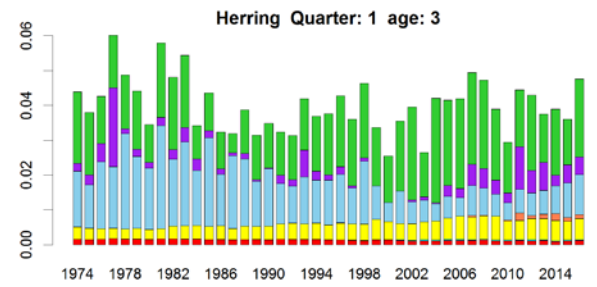
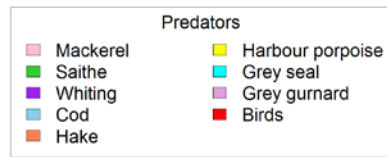


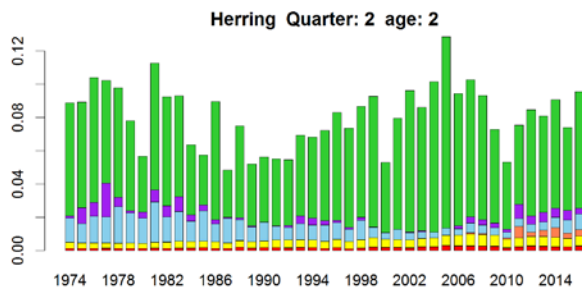
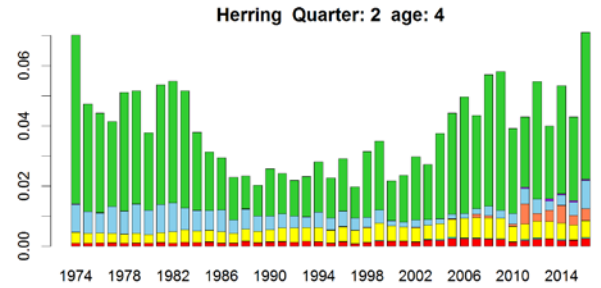
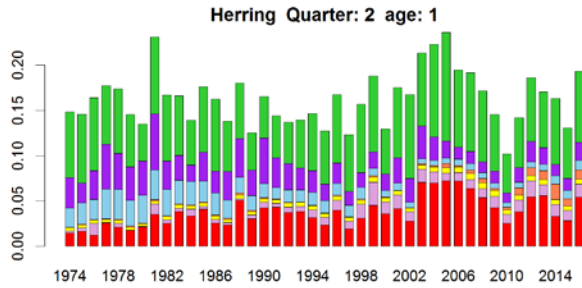
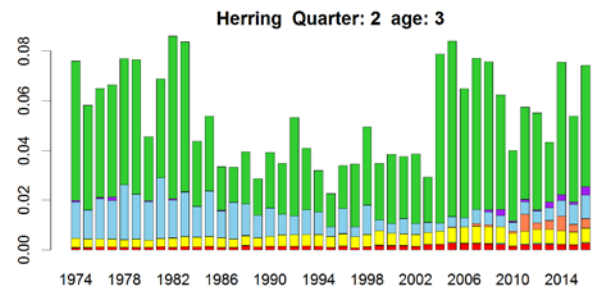
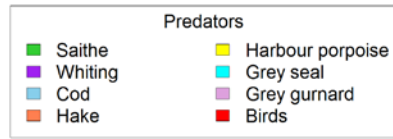


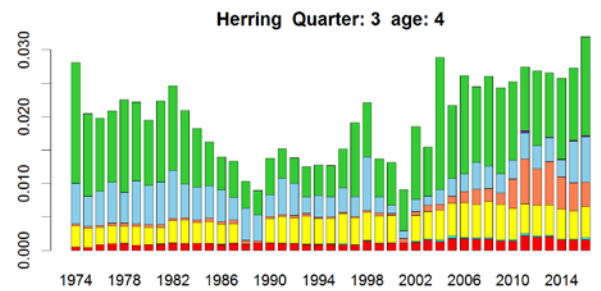
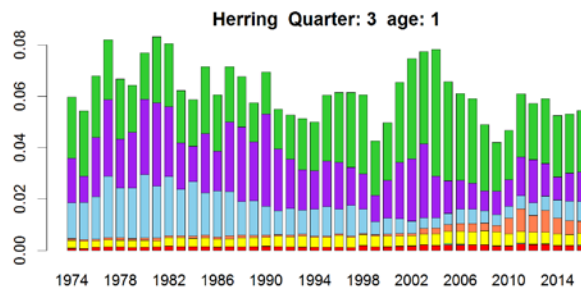
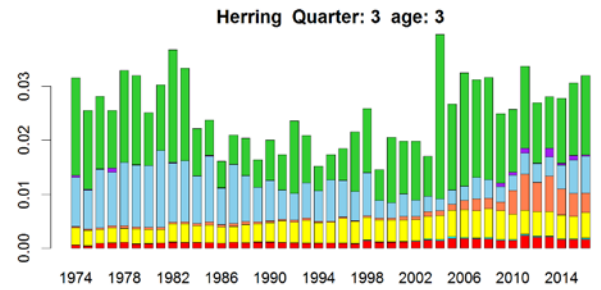
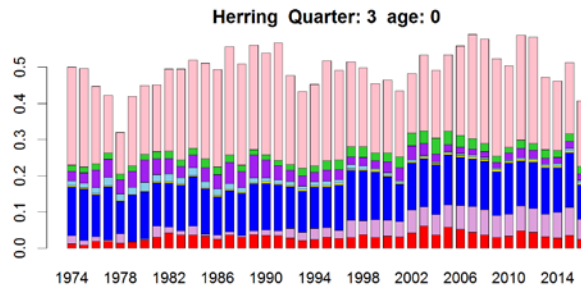
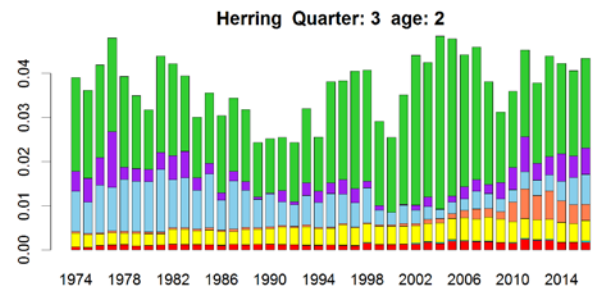
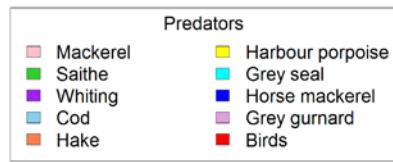


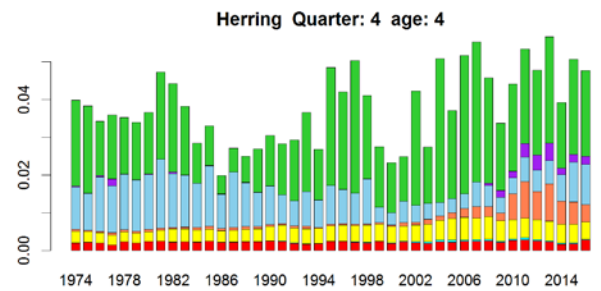
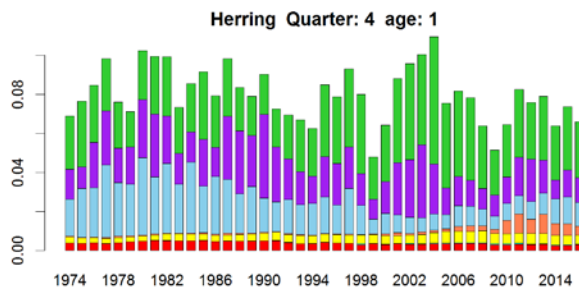
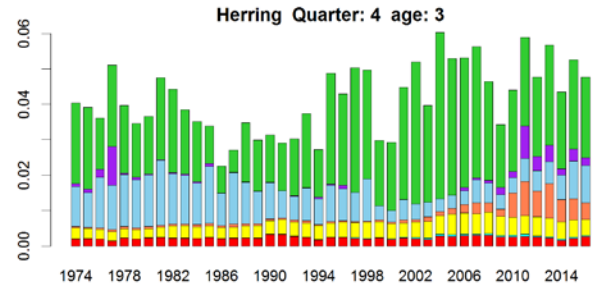
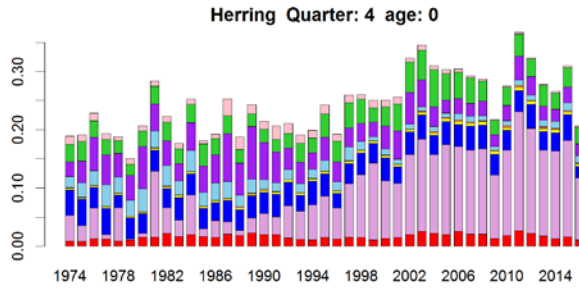
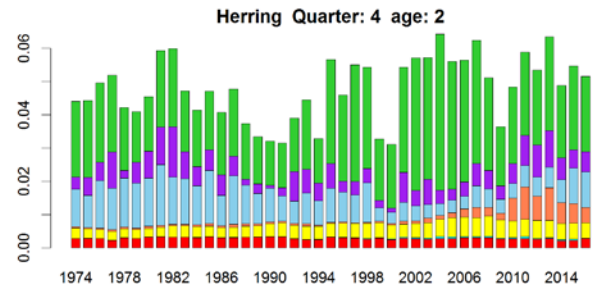
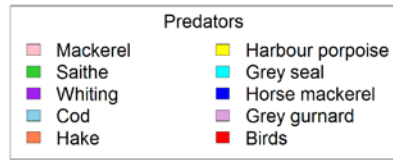


Predation mortality for herring

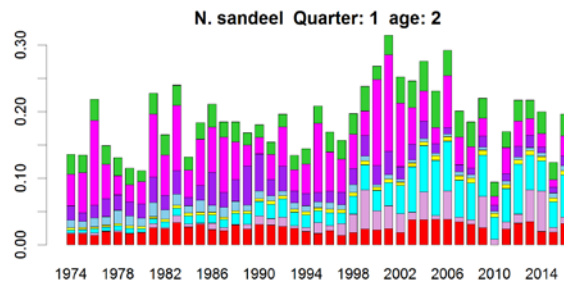
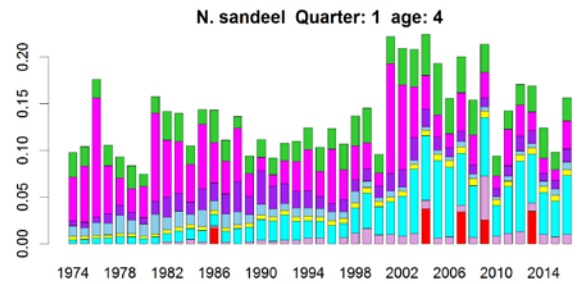
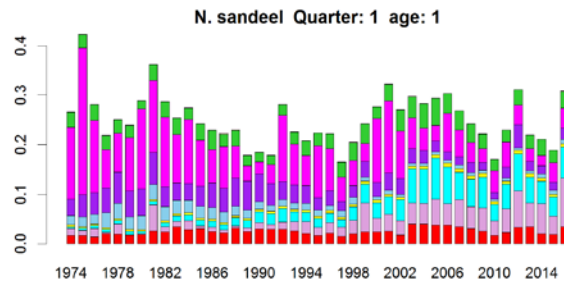
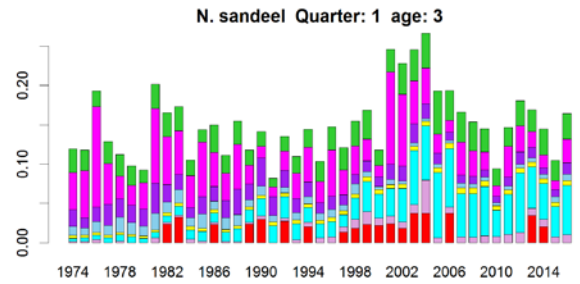
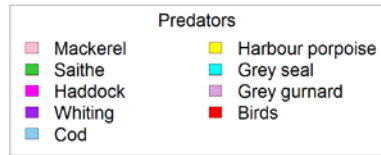


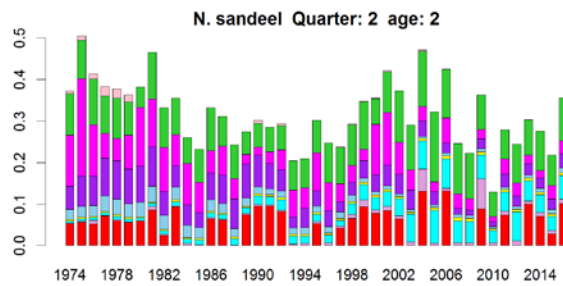
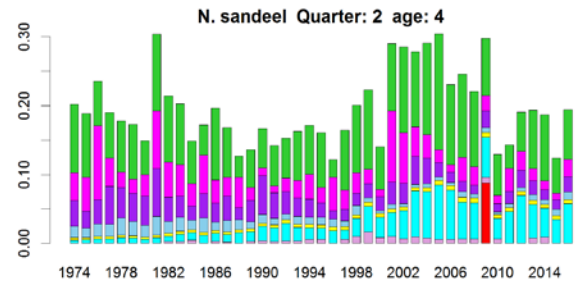
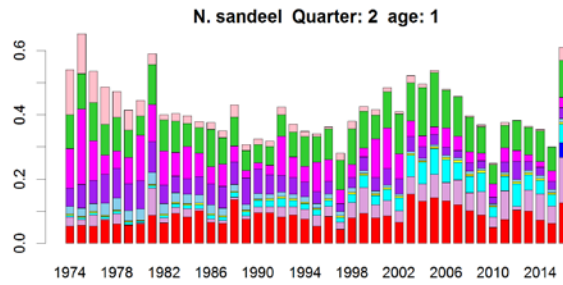
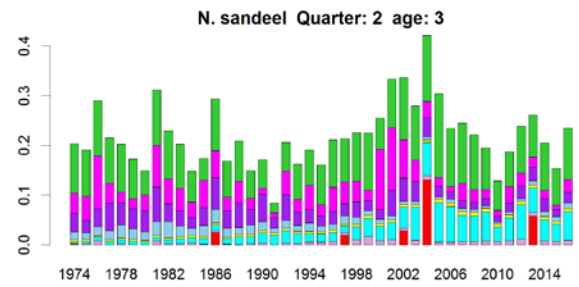
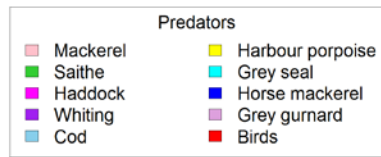


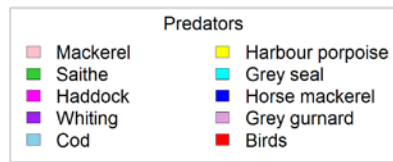




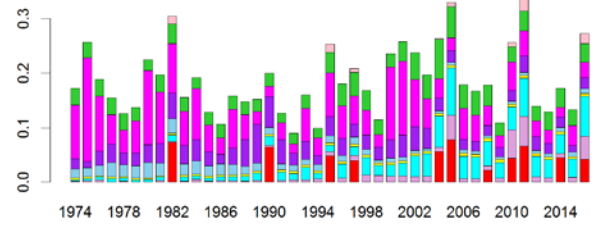
Predation mortality for northern Sandeel



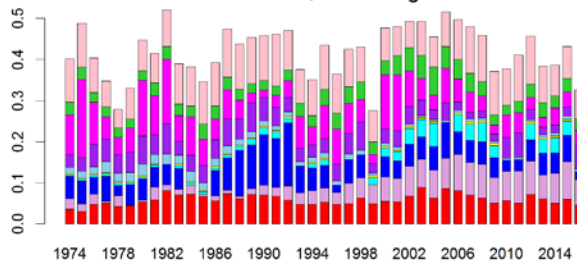




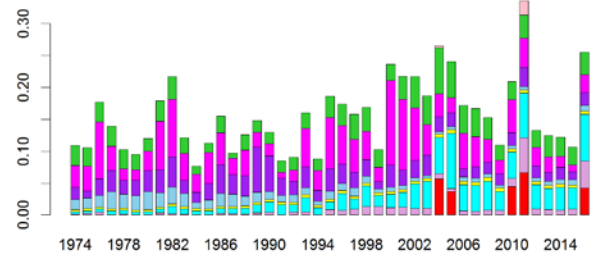
N. sandeel Quarter: 3 age: 2



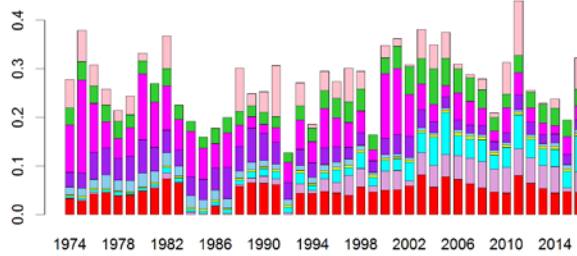
N. sandeel Quarter: 3 age: 0



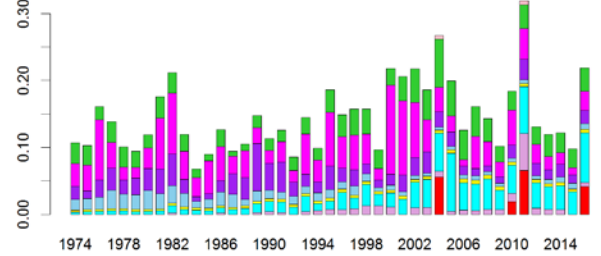
N. sandeel Quarter: 3 age: 3

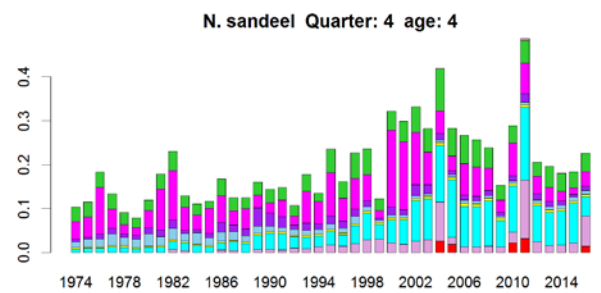
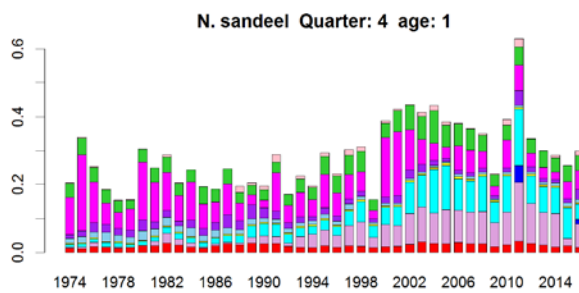
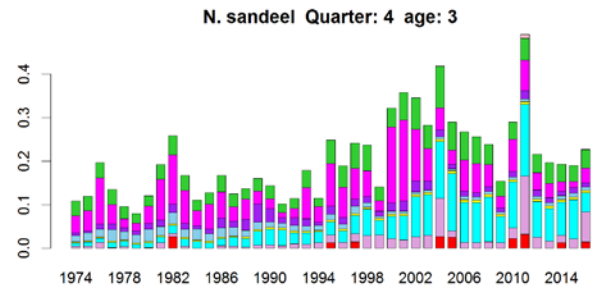
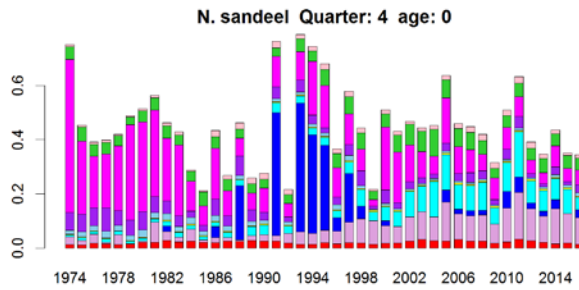
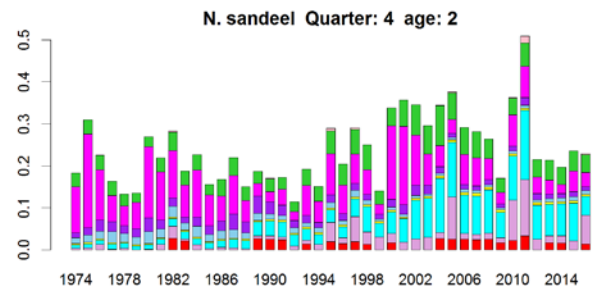
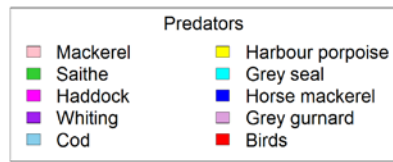


N. sandeel Quarter: 3 age: 1



N. sandeel Quarter: 3 age: 4



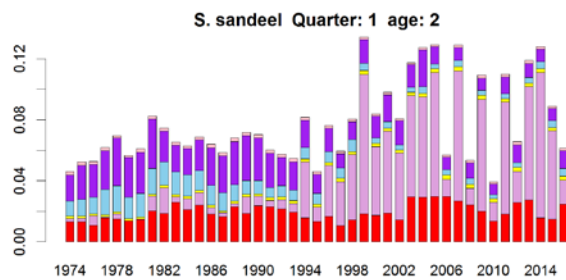
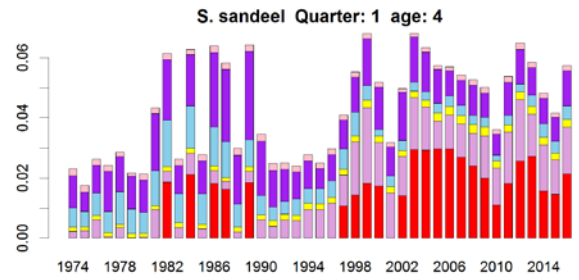
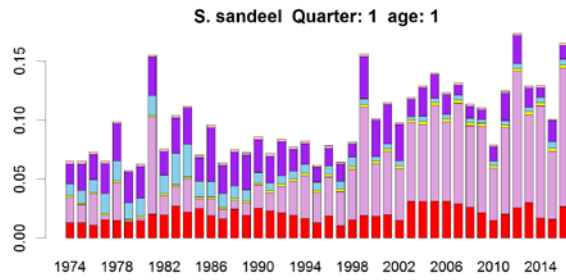
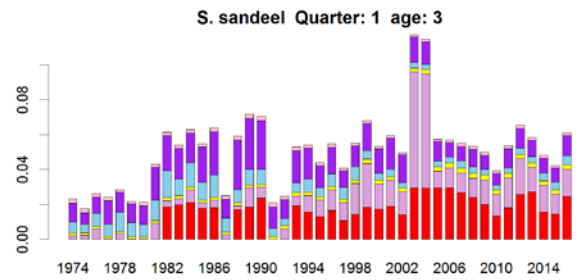
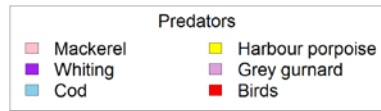


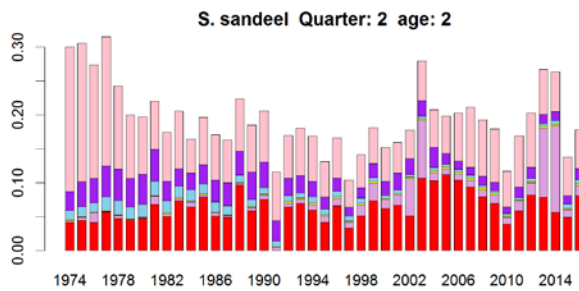
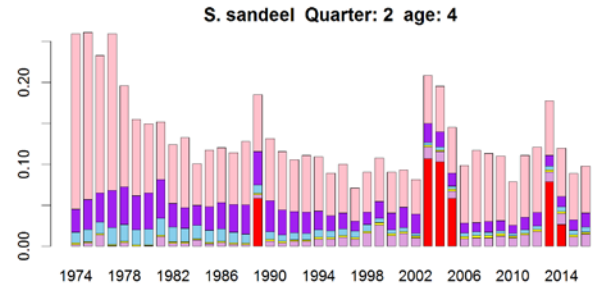
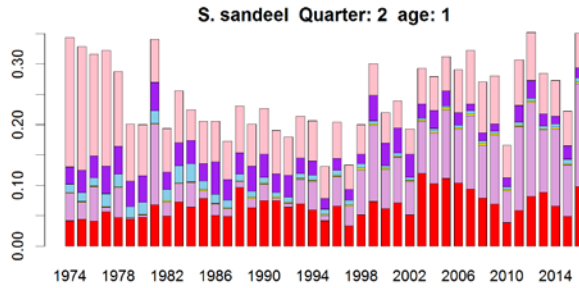
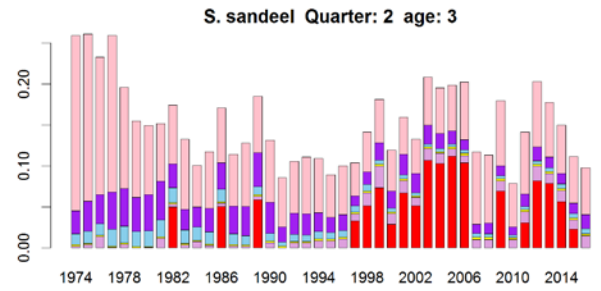
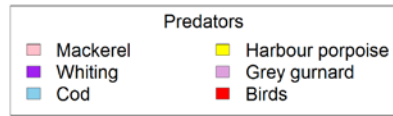
Predation
Sandeel

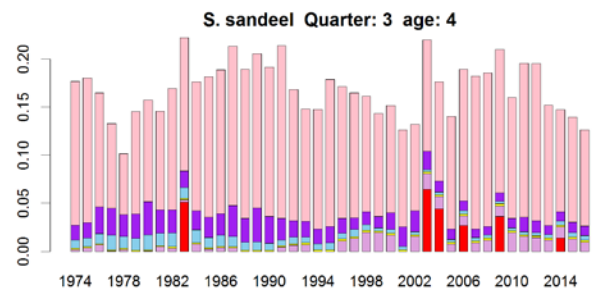
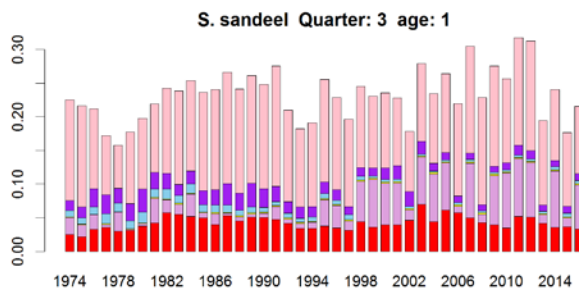
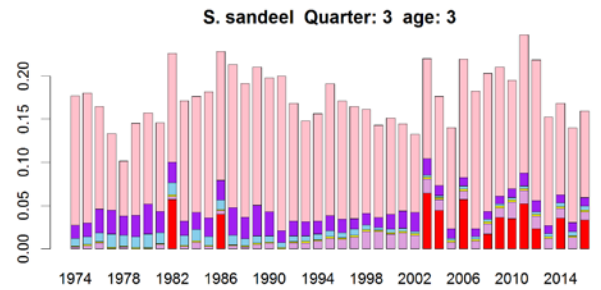
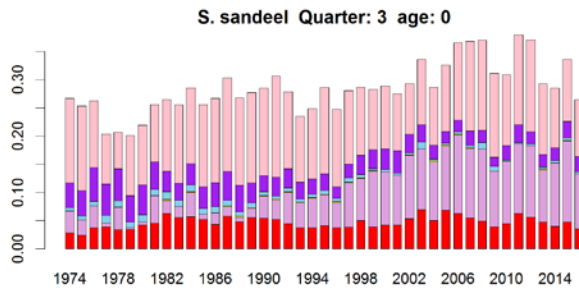
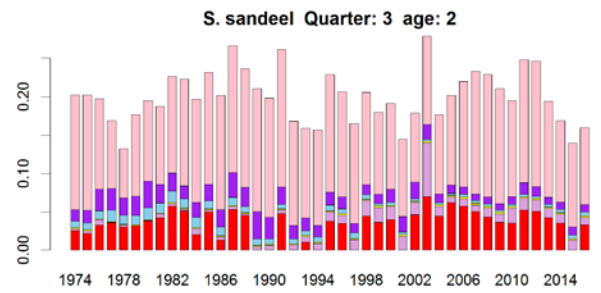
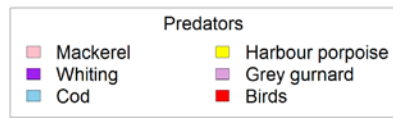
mortality

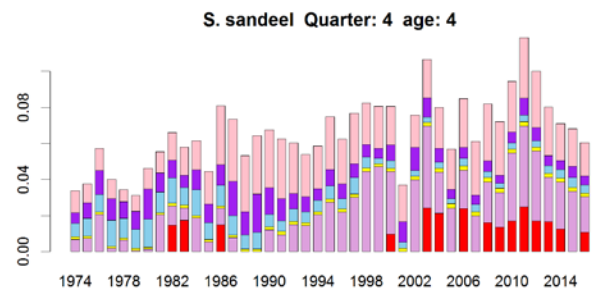
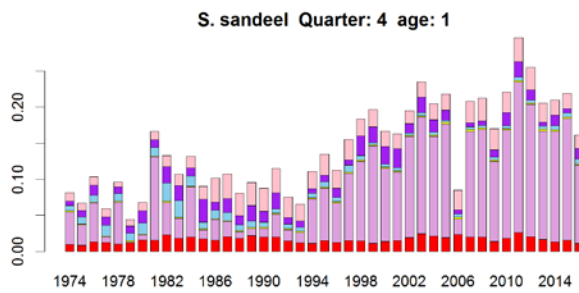
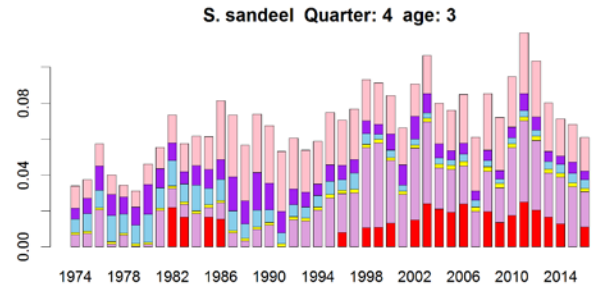
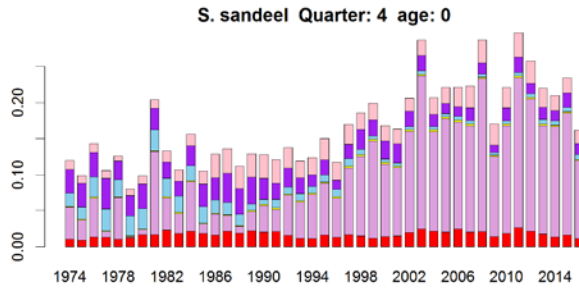
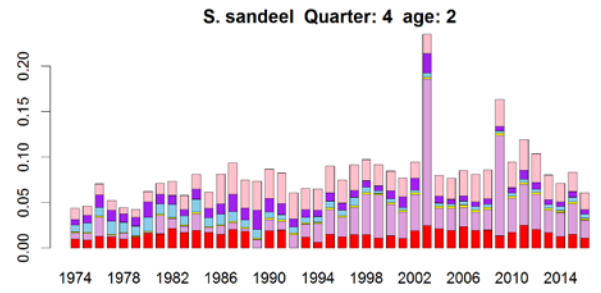
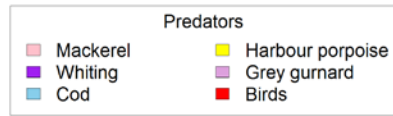
for

Southern

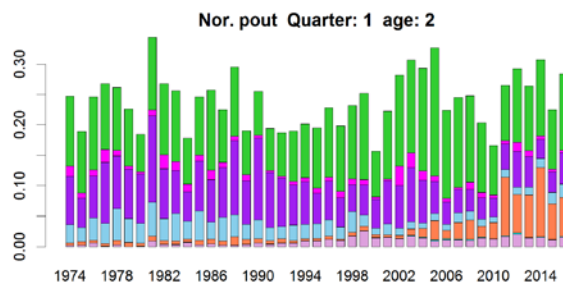
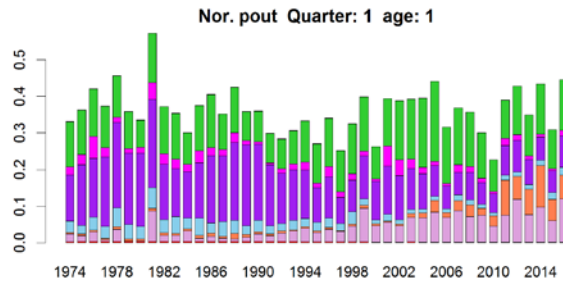
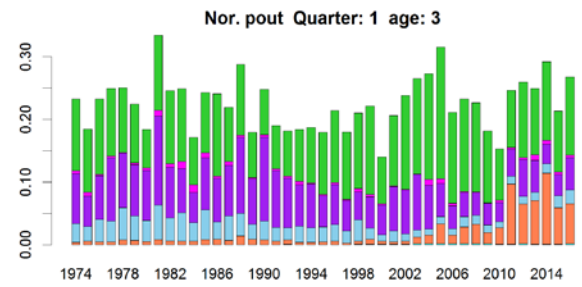
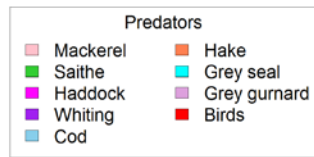


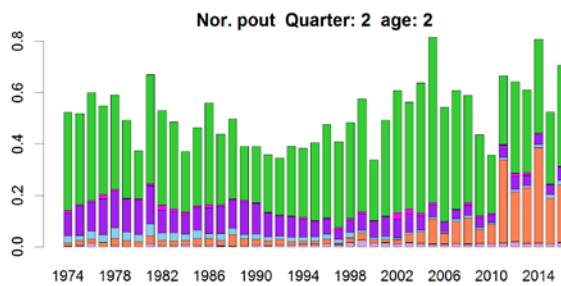
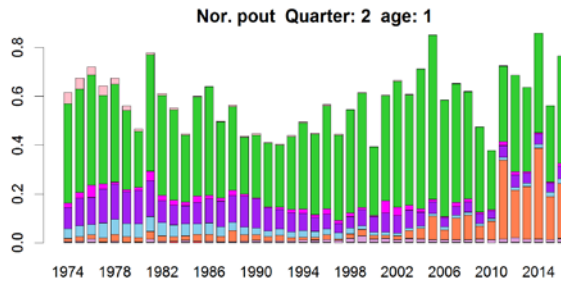
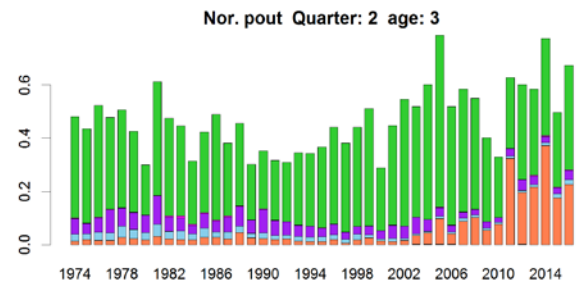
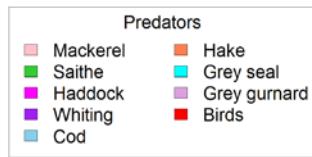


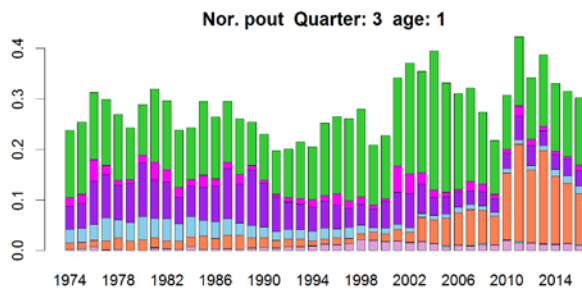
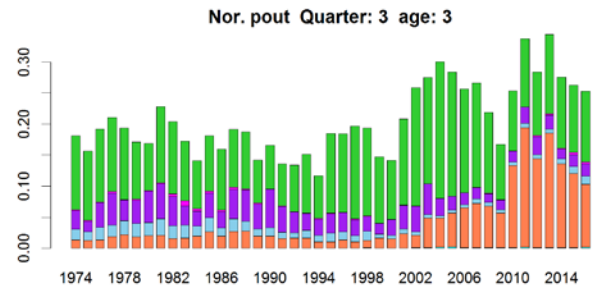
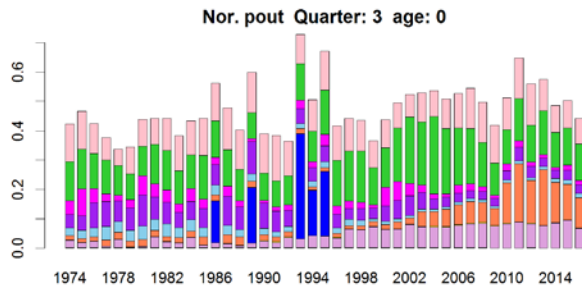
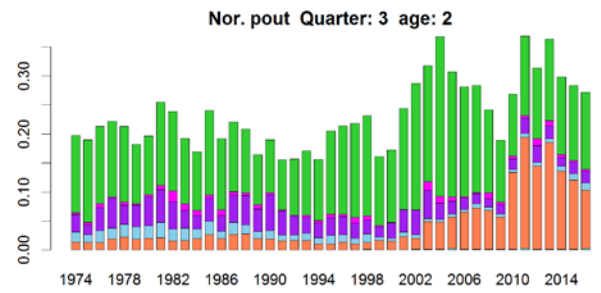
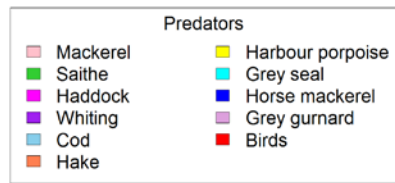


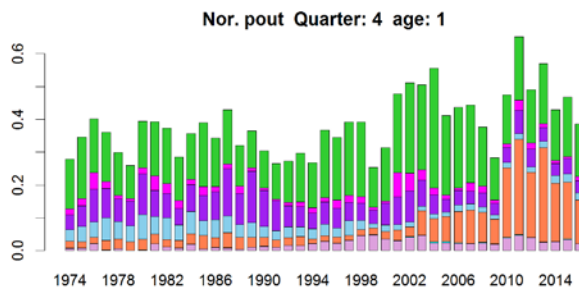
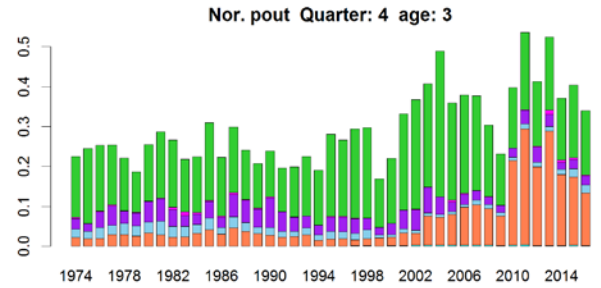
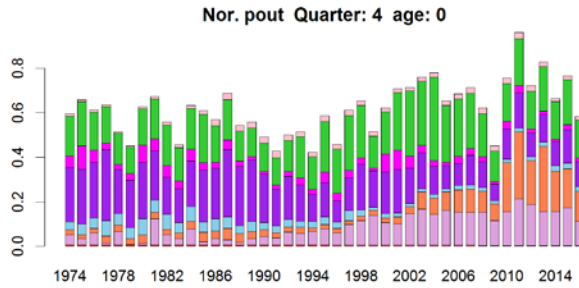
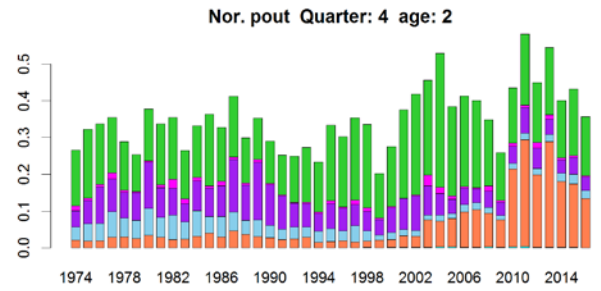
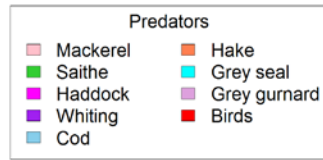


Predation mortality for Norway pout

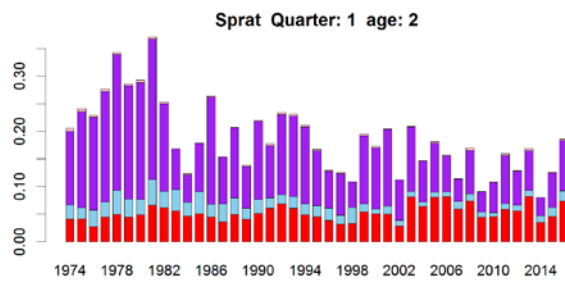
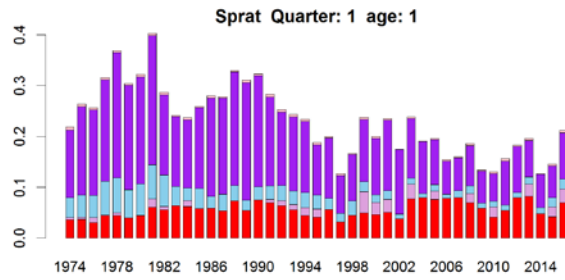
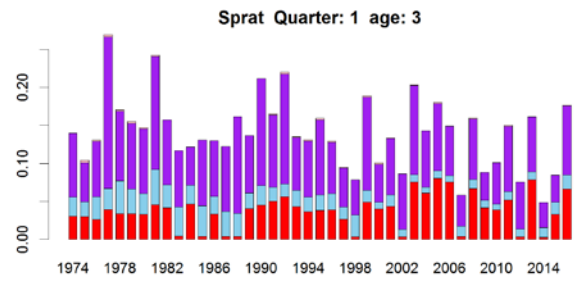
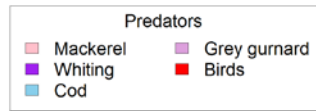


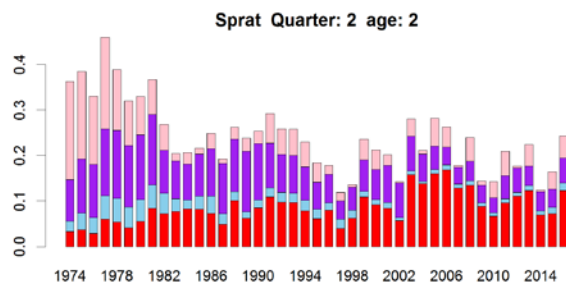
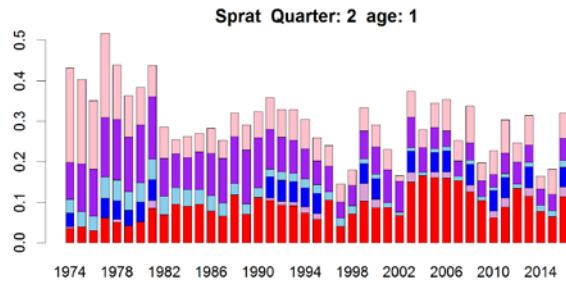
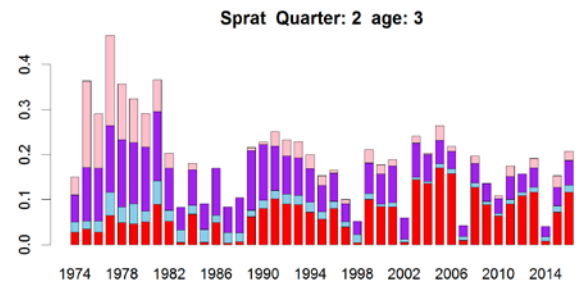
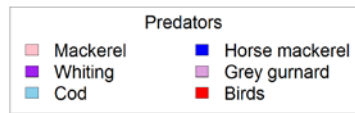


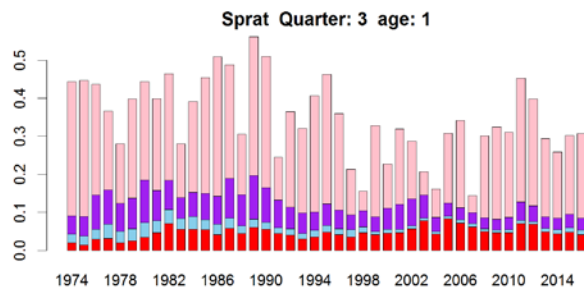
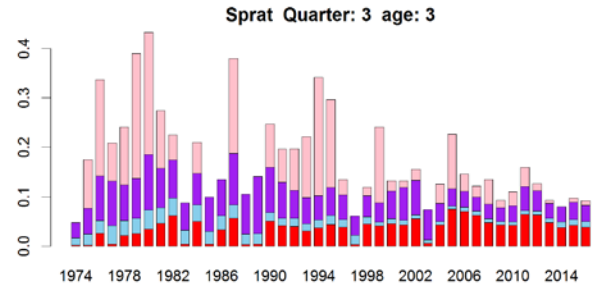
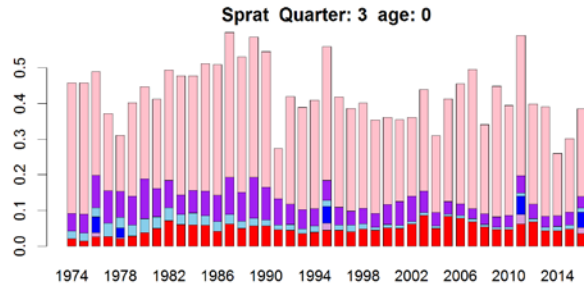
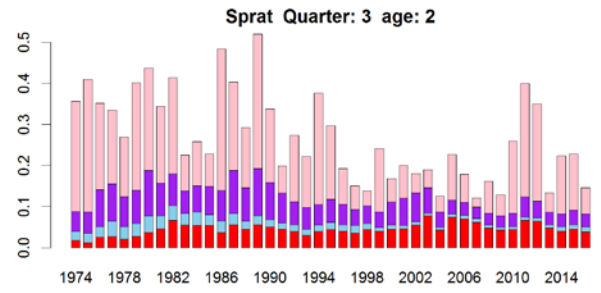
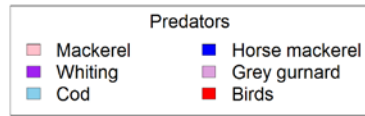


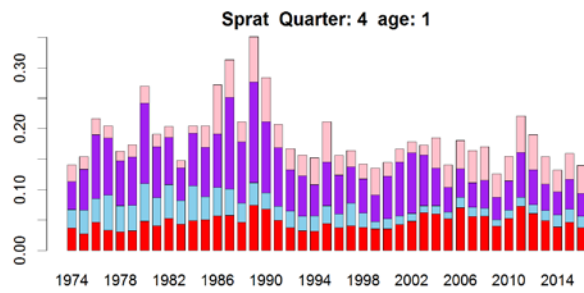
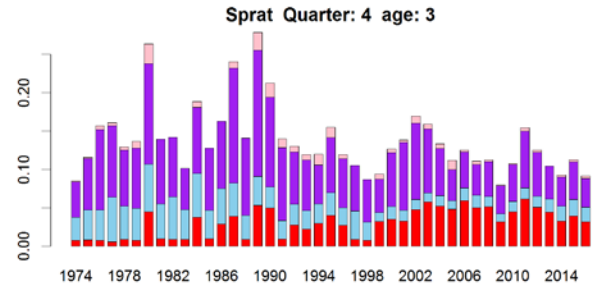
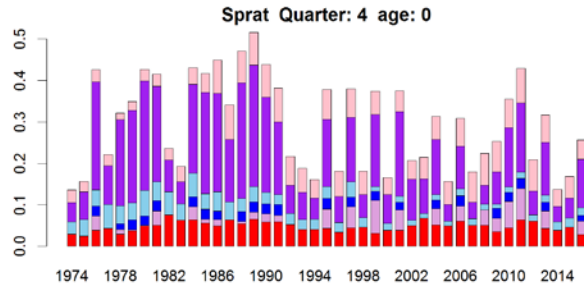
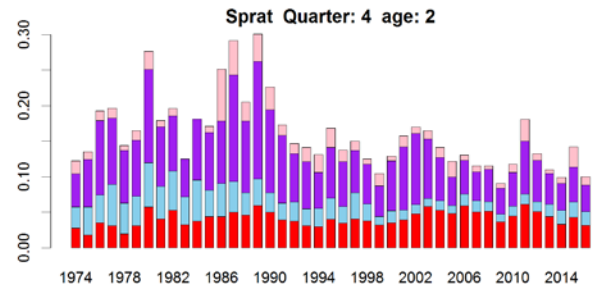
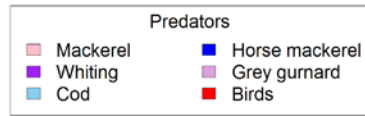


Predation mortality for Sprat









Annex 5: ToR C: Consider methods to assess the skill of multispecies models intended for operational advice

Predicting the diet of Baltic Sea cod using stomach data and an age-length base model

- A multispecies model for cod, herring and sprat in the Baltic Sea for 1974–2013 was implemented in the Gadget
- Changes in prey length preference and species composition investigated with respect to ontogenetic groups.
- Ontogenetic shifts in predicted species composition fitted quite well to observations.
- Temporal shifts fitted better to observed composition at the beginning of time-series.
- Length preference for sprat fitted better towards the end of time-series
- Length preference for herring haven't fitted that well.

One of the main reasons to use multispecies models in stock assessment is to improve our understanding and estimation of the impact of predators on the stocks of preys which are simultaneously exploited by the fisheries. Whenever available values of the natural mortalities used into single species stock assessments are derived from multispecies models. However realistic estimation of natural mortality relies on model's ability to predict the predator diet well. Gadget (Begley, 2004) is a candidate modelling framework for this for few reasons:

- First of all, stomach data, which are used to inform the multispecies models, often have gaps in sequence of sampling years. Gadget is able to handle data, which have temporal gaps.
- Predator-prey interactions in aquatic systems depend to a large extent on sizes of both predator and prey (Scharf, 2000). Size of prey that predator is able to eat is determined by predator gape size, which is often correlated with predator size. Additionally, foraging success of predator and escape success of prey depend on their sizes since larger individuals are faster (Webb, 1976, Folkvord and Hunter, 1986). Gadget models are age-length structured and thus suitable to model length-dependent interactions.

A multispecies model for cod, herring and sprat in the Baltic Sea for 1974–2013 was implemented in Gadget. We investigated two aspects of cod diet: prey length preference and species composition. To improve the prediction we informed the model with prey length and species composition from cod stomach data as input. Ontogenetic shifts in predicted species composition (Figure 1) fitted quite well to observations. The model is able to reproduce the decrease in the proportion of *Saduria entomon* and *Mysis* spp. in the diet as cod grows (seen on Figure 1 as position of points moving from left to right through the vertical blocks indicating ontogenetic cod groups). The model also captures the general patterns observed for the two clupeids. In the specific, sprat increases earlier

in the diet becoming the most relevant prey for the cod size group 25–35 cm, but its contribution to the diet decreases for very large cod. Herring enters later in the diet of cod but its importance increases throughout the ontogeny of the predator. Fitting to the observed composition was usually better at the beginning of the time-series as visible from figure 1 where the distances between observations (blue) and predictions (red) are in general smaller in 1974–1988 than in 2007–2013. The prediction was also better for medium size cod (35–60 cm), for which more stomachs were sampled. Predicted length composition (Figure 2) fitted better towards the end of time-series for sprat, possibly because the model was informed with length distribution of sprat from survey only from the mid-1990s. On the contrary, the length composition of herring in the stomachs is poorly represented and additional work is required on the parameters controlling the length selection of herring by cod.

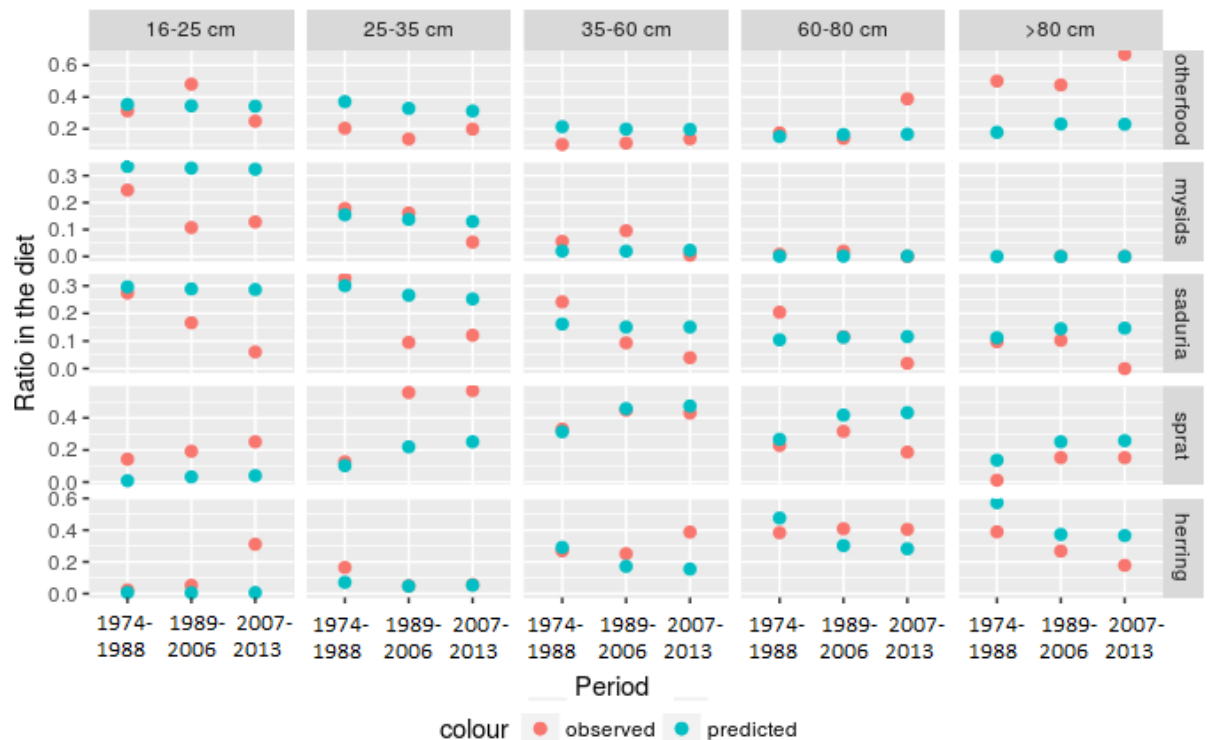


Figure 1. Comparison of observed (red) and predicted (blue) species composition of main prey groups (indicated on right) in the diet of cod of different size groups (indicated in the top).

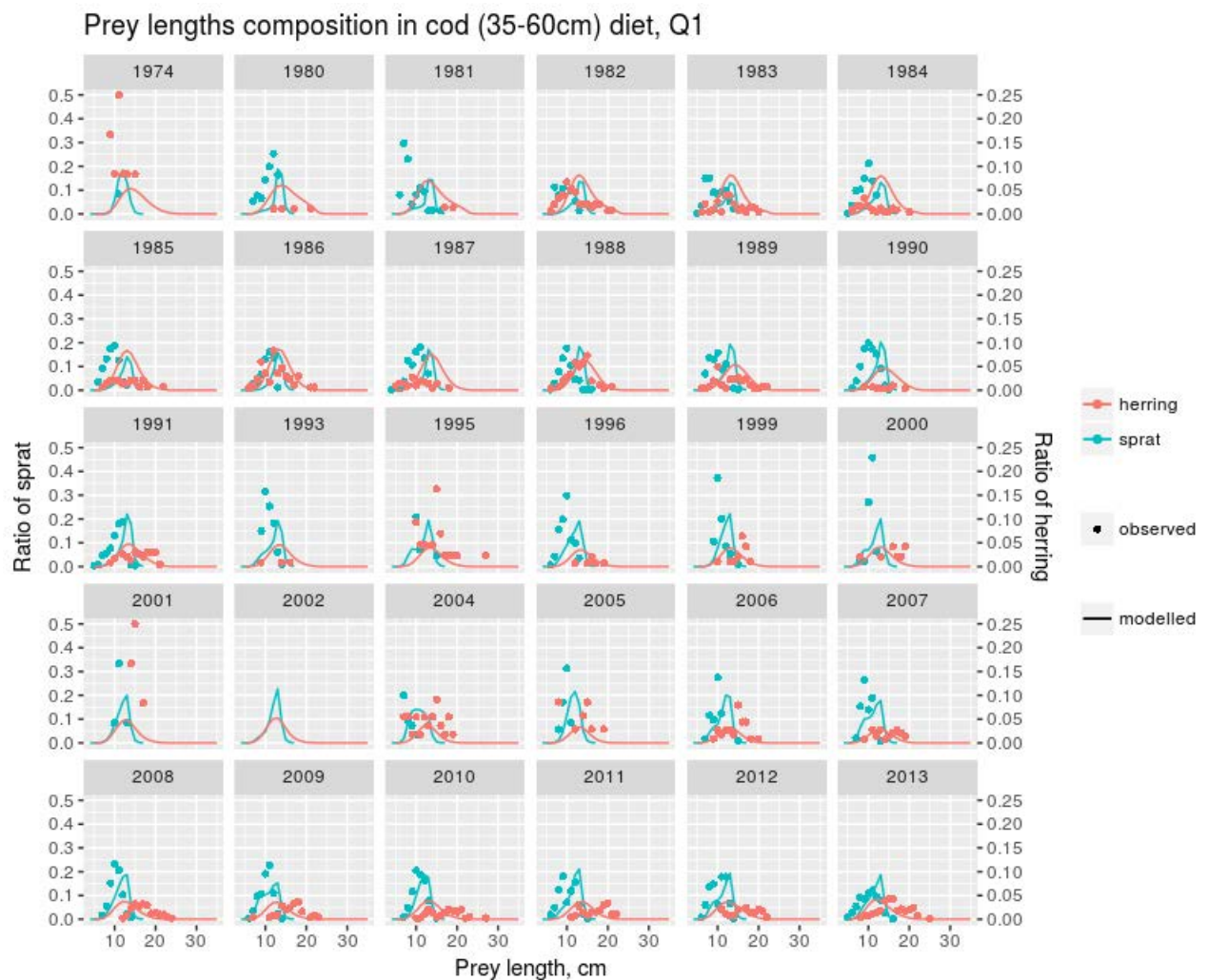


Figure 2. Comparison of observed (points) and predicted (line) length distribution of herring (red) and sprat (blue) in the diet of cod of 35–60 cm.

The multispecies cod-herring-sprat Gadget model for the Baltic is able to represent the overall cod diet composition and size preference. However, the current model is still unable to predict the marked changes in the role of benthic and pelagic components in the diet of cod which have been observed during the last four decades. In addition, further tuning of the predator-prey size selection parameters is expected to improve the fitting of the herring and sprat size compositions in the stomachs.

References:

- Begley, J. 2004. Gadget user manual. Marine Research Institute, Reykjavik.
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Can EwE mimic the Atlantis ecosystem?

- The Atlantis model for Icelandic waters was used as an operating model to test the performance of EwE.
- Balancing and fitting routines were written to make the modelling process more automatic and less subjective.
- The EwE model was able to replicate the Atlantis ecosystem.
- The forecasting ability of the model was however not reliable.

Ecosystem model has been constructed for Icelandic waters using the Atlantis modelling framework and used as an operating model to test the performance of Ecopath with Ecosim (EwE). A methodology was developed to extract data from Atlantis and take into the R-version Rpath (<https://github.com/sluccey/RpathDev>) for EwE. Balancing and fitting routines were written to make the modelling process more automatic and less subjective. A data rich scenario was carried out where the best possible knowledge of the ecosystem was assumed, e.g. the true biomass for the vertebrate groups was assumed known. This scenario was carried out to see whether EwE could replicate the Atlantis ecosystem. Also, the forecast ability of the model was evaluated. This methodology was used for two version of the Atlantis model, version 1 and 2. Version 2 is an updated version of version 1 where respiration has been activated and recruitment includes annual scales for cod, haddock, saithe, herring and capelin but recruitment scales for cod and haddock were only included in version 1. The forecasting was carried out by assuming status quo, i.e. the harvest rate of the last year in the simulation (hindcast) of both Atlantis and EwE was assumed to continue for 20 years. In version 2 the recruitment scalars were repeated from the last 20 year of the simulation in the 20 years in the forecast for the Atlantis model.

The hindcast of EwE fitted reasonable well to the Atlantis ecosystem when both version 1 and 2 of the Atlantis model were used (Figure 5 and Figure 7). However the EwE model had easier time mimicking version 1 of the Atlantis ecosystem than version 2. The average correlation in the hindcast for nine harvested groups (cod, haddock, saithe, redfish, Greenland halibut, flatfish, herring, capelin and other codfish) in version 1 was 0.9 but 0.64 for version 2. Version 1 was also easier to forecast where the correlation with EwE was 0.35 but was -0.25 for version 2. For version 1 the correlation in the hindcast gave some evidence on how well the EwE model would be able to forecast (Figure 6). However, this was not the case for version 2 where recruitment was scaled to resemble the past 20 years. In this case correlation in the hindcast did not give much evidence on how the correlation would be in the forecast (Figure 8).

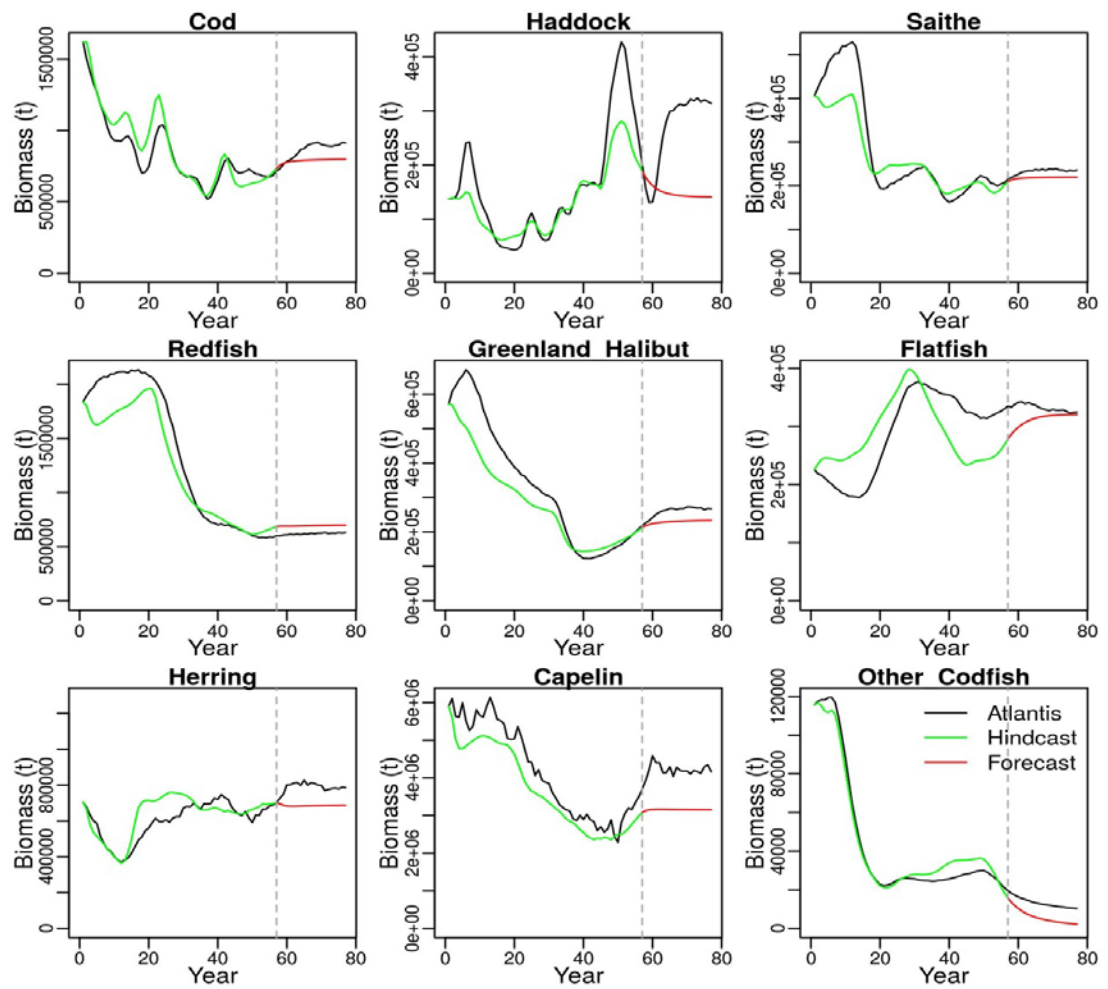


Figure 39. Simulated biomass from EwE compared to the Atlantis model for harvested groups from version 1 of Atlantis.

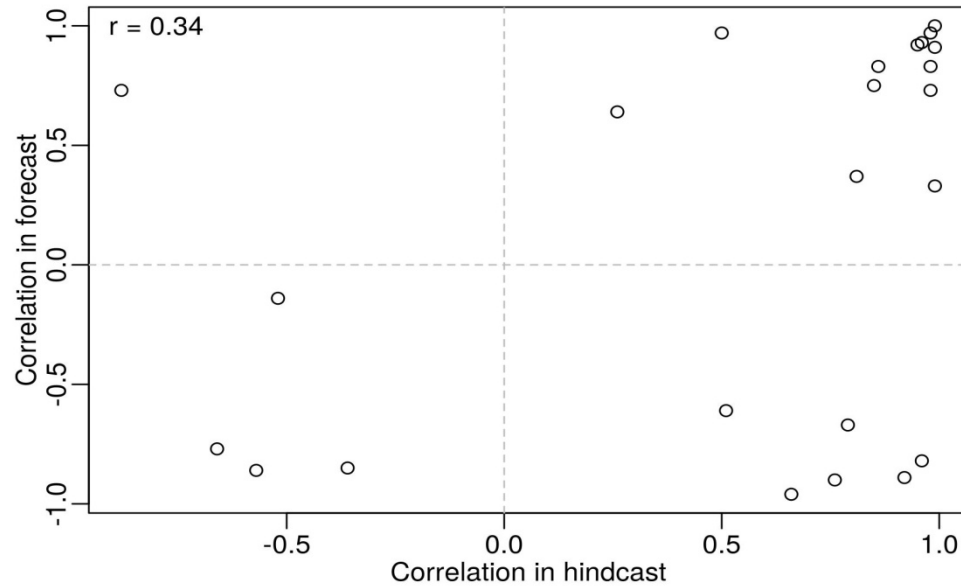


Figure 40. Correlation in hindcast and forecast between biomass of EwE and Atlantis for the vertebrate groups in version 1.

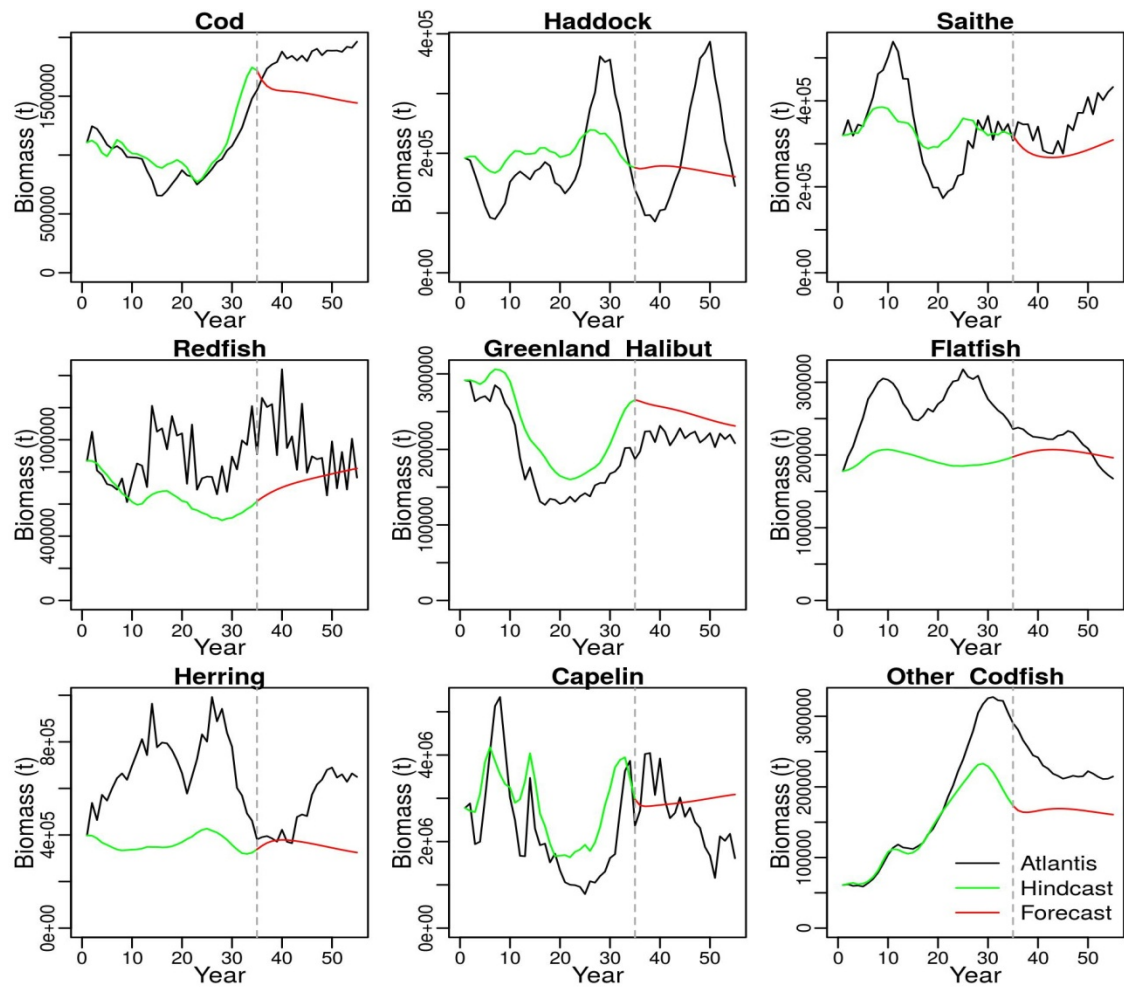


Figure 41. Simulated biomass from EwE compared to the Atlantis model for harvested groups from version 2 of Atlantis.

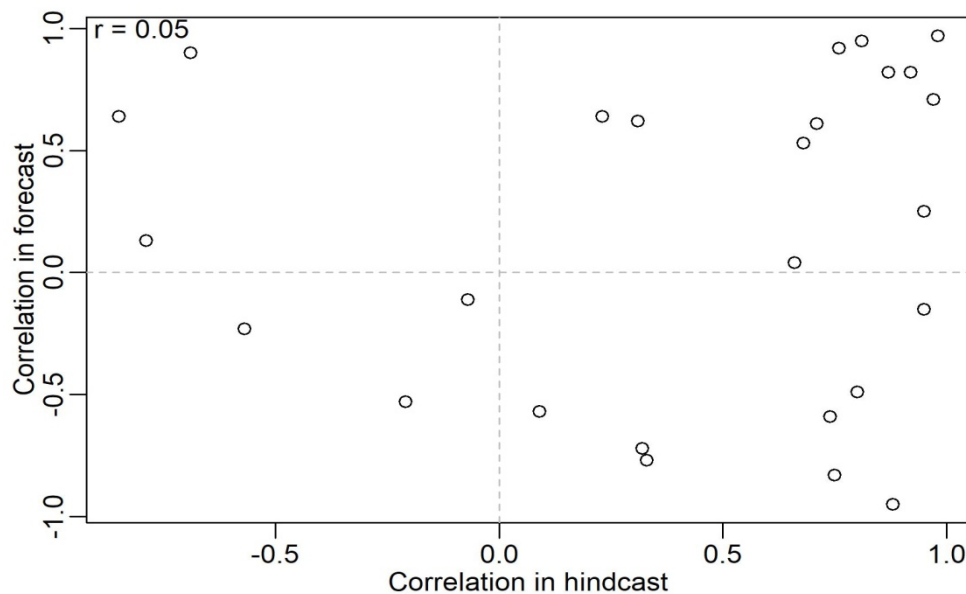


Figure 42. Correlation in hindcast and forecast between biomass of EwE and Atlantis for the vertebrate groups in version 2.

Influences of model structure on estimates of community dynamics and multispecies biomass production

We examine how estimation model structures influence estimates of species-pair interactions within a complex of ten commercially important species on Georges Bank, USA, as well as model predictive ability. We estimate species-pair interactions in a Bayesian state-space multispecies production model fit to annual catches, abundance indices from bottom-trawl surveys, and diet composition data. To assess model structure uncertainty, we evaluate how model fits vary from models that 1) include species interactions in the biomass production model, 2) include diet composition data as an index of interactions, 3) include diet composition data in the likelihood function during estimation, and 4) define priors on interaction terms according to ecological theory. Model fits are compared using evaluations of posterior predictive loss and cross-validation error in the final five years of the survey indices. Contact: Robert Wildermuth (rwildermuth@umassd.edu)

New state-space multispecies age-structured stock assessment model for Georges Bank finfish community

For summary:

Four simulations models of 1000 iterations were run to test their performance (bias in estimated parameters and derived outputs) against the new state-space multispecies stock assessment model created for Georges Bank cod and haddock. The importance of estimating process errors in the model was also investigated (state-space vs. statistical catch at age (SCAA) models).

- The state-space multispecies model (same configuration than the operating model) has a tendency of underestimating fishing mortality at age and overes-

timating predation mortality at age and this is accentuated in the multispecies SCAA model.

- As expected, the SCAA models perform less well than the state-space models.
- Overall, the bias in recruitment and SSB is below 20% but the state-space single species model performs better than the state-space multispecies model which is unexpected. This may come from a problem in the simulated diet data since the estimated diet parameters are the only ones showing a large bias. This may also come from the low interactions between the fish species in the models. This will be investigated further in the next months.

Main text:

Vanessa Trijoulet presented a general state-space multispecies age-structured stock assessment model developed in Template Model Builder (TMB). It accounts for species interactions and stochasticity in population processes and associated observations. This model was applied to two finfish stocks on Georges Bank, fitting to survey indices of abundance, catch at age (from both fishery and surveys), and diet composition data.

A simulation study was performed to test the performance of the model before extending it to more fish species (10 EBFM finfish species of interest on Georges Bank). The state-space multispecies model is used as an operating model to simulate 1000 data sets of observed data with observation and process errors using the TMB simulation tool. Four simulation models (SM) were fitted to the simulated data to investigate the performance of the multispecies model (SM1) compared to a single species model (SM2) but also the performance of the state-space models compared to statistical catch at age (SCAA) models by ignoring the process errors in the models. The simulation models were as follows:

- SM1: multispecies state-space model (same configuration than the operating model)
- SM2: single species state-space model (predation is turned off)
- SM3: multispecies SCAA (recruitment and survival are deterministic)
- SM4: single species SCAA

The performance of the simulation models is evaluated by calculating median relative differences (and 95% confidence interval) between estimated and true values for the model parameters and the derived outputs (SSB, recruitment, mortality).

- The SM1 has a tendency of underestimating fishing mortality (F) at age and overestimating predation (P) mortality at age and this is accentuated in the SM3 model (Figure 1).

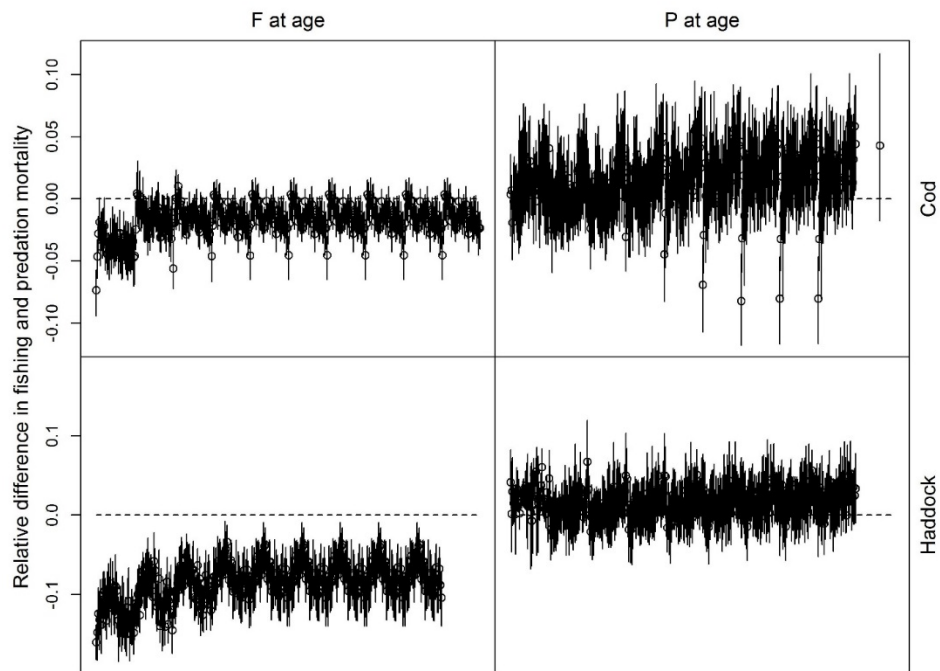


Figure 43. Median relative difference in fishing mortality (F) at age and seal predation mortality (P) at age with 95% confidence interval around the median in the SM1 model.

- As expected, the SCAA models (SM3 and 4) perform less well than the state-space models (SM1 and 2, Figure 2).

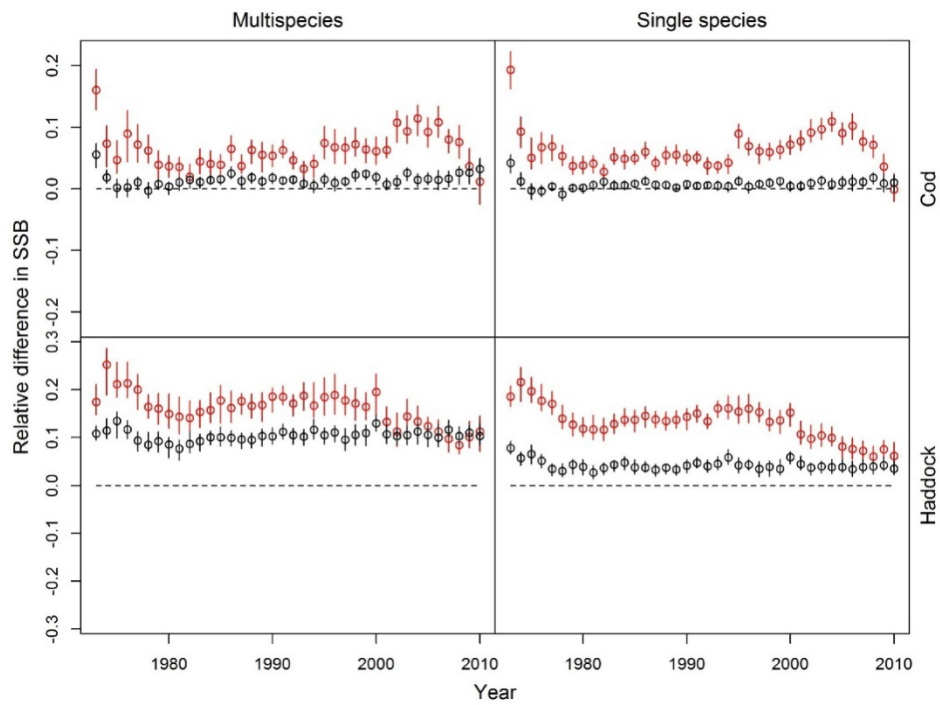


Figure 44. Median relative difference in SSB and 95% confidence interval around the median in the state-space models (black) and SCAA models (red).

- Overall, the bias in recruitment and SSB is below 20% but the SM2 performs better than the SM1 model which is unexpected (Figure 3). This may come from a problem in the simulated diet data since the estimated diet parameters are the only ones showing a large bias. This may also come from the low interactions between the fish species in the models. This will be investigated further in the next months.

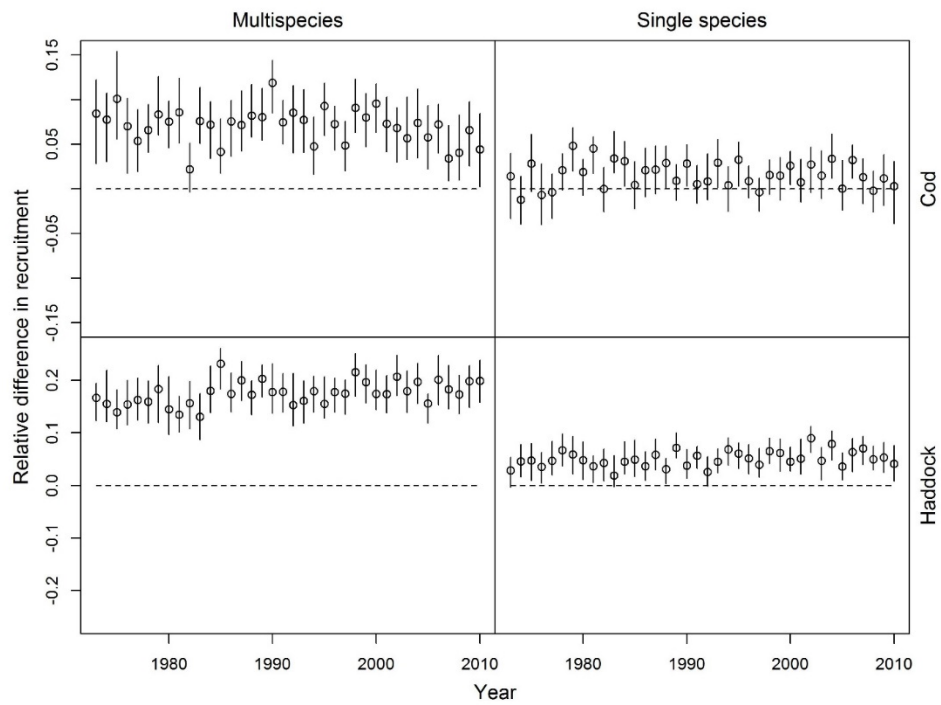


Figure 45. Median relative difference in recruitment and 95% confidence interval around the median in the SM1 and SM2 models.

Retrospective patterns in SMS hindcasts and forecast

To test the skill of SMS to provide robust estimates of natural mortalities retrospective patterns were analysed. Also the stability of forecasts was tested. Following conclusions could be drawn:

- The SMS keyrunhindcasts from 2015 showed no serious retrospective patterns for cod, whiting, haddock, sprat and Northern sandeel. SMS can provide consistent estimates of natural mortalities over the years for these stocks. For Norway pout and Southern sandeel retrospective patterns occurred.
- Retrospective bias in forecasts occurred for both single species and multi species versions of SMS. Forecasts with more than 4 years provided often considerably different results compared to short term forecasts.
- SMS multi species forecasts tended to be overpessimistic with regard to stock dynamics at low abundances. The Holling type II functional feeding response may be causing this.

Retrospective patterns in multi species models are in most cases not analysed despite it is a standard diagnostic for single species assessments. Therefore, the SMS keyrun from 2015 was taken as example to see whether robust estimates of natural mortalities can be provided by SMS. In addition, changes in forecasts in dependence of the starting year were analysed.

From the 2015 keyrun input data were reduced in one year steps and the model was fitted to the shortened time-series with the same settings as used in the keyrun. The last assessment year was reduced by this routine until 2004 was reached. The only change compared to the keyrun was that the 2013 stomach data for hake and grey gurnard had to be dated back to 2000 to be able to use them in runs with final assessment years before 2013.

The results show that the SMS multi species hindcasts provide robust estimates of SSB, F , recruitment and natural mortalities for cod, whiting, haddock, herring, sprat and Northern sandeel. For these stocks currently no larger retrospective patterns were observed and especially natural mortality estimates did not change much by adding more data points (see Figure 1 for cod as example). Norway pout and southern sandeel had the worst retro patterns and for these stocks a rescaling of natural mortalities occurred by adding more data points (see Figure 2 for Norway pout as example).

From the different hindcasts the stocks were forecasted until 2015. Relative trends in F_{bar} over time between the last assessment year and 2015 were used from single species assessments. The absolute level of F_{bar} was taken from the last assessment year in SMS hindcasts to avoid inconsistencies between the level of abundance and F . F values for Southern Sandeel and Northern Sandeel were directly taken from SMS (F_{2014} and $F_{2015} = F_{2013}$) because single species assessments use different stock definitions.

Three types of forecasts were conducted:

1. Multi species: Trends in F_{bar} known; Stock recruitment relationships (SRRs) fitted from SMS hindcast results

2. Single Species: Trends in F_{bar} known; SRRs fitted from SMS hindcast results
3. Multi species: Trends in F_{bar} known + trends in recruitment taken from single species assessments

From the forecast results (see Figure 3 for option 1 as example) it can be concluded that both single species and multi species forecast showed retrospective bias in forecasts when starting from different years. Especially forecasts over more than 4 years provided different results compared to shorter forecasts.

Short lived species were problematic when trying to forecast stocks without known recruitment trends. For these stocks the recruitment determines how much can be fished based on an escapement strategy. In the moment F and recruitment are no longer in line with each other unrealistic forecasts are the consequence. However, also some longer lived species tended to die out in medium-term multi species forecasts while the single species forecasts turned out to be too optimistic. Medium term forecasts in MS mode often led to unrealistic behavior at low abundances and the recovery potential was underestimated. This may have something to do with the Holling type II functional feeding response leading to unrealistic high predation mortalities with decreasing prey abundance.

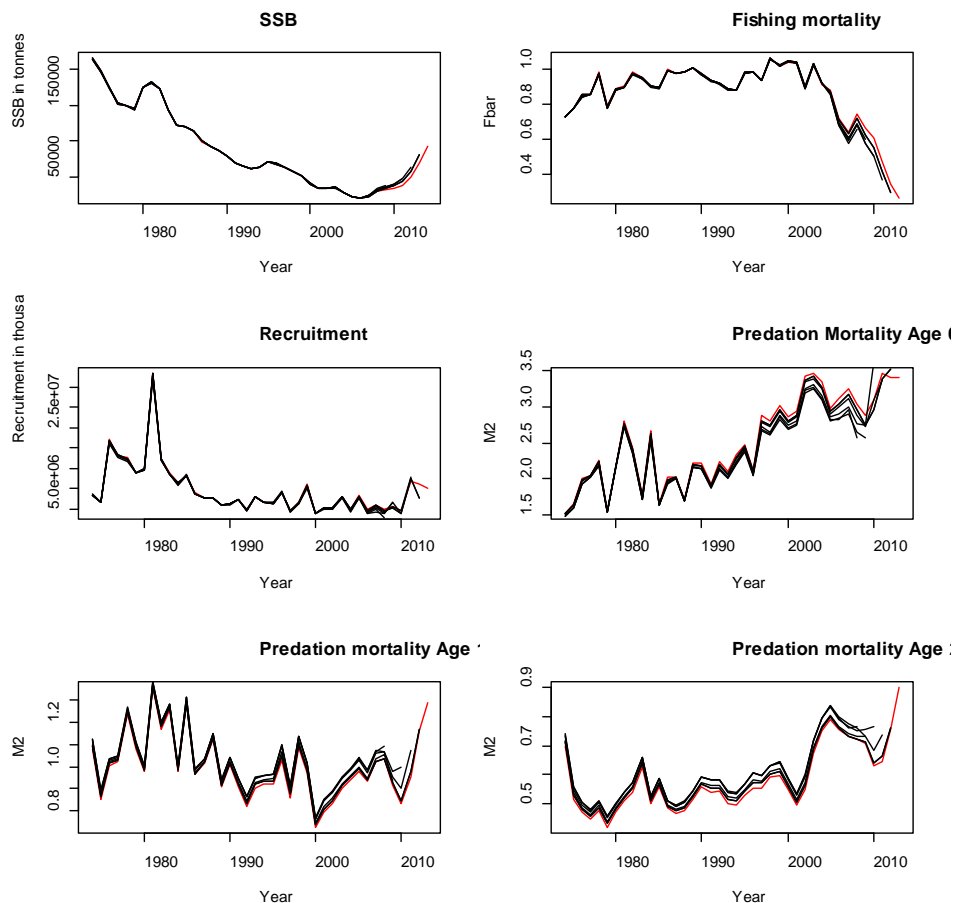


Figure 1. Retrospective patterns in the SMS keyrun from 2015 for cod.

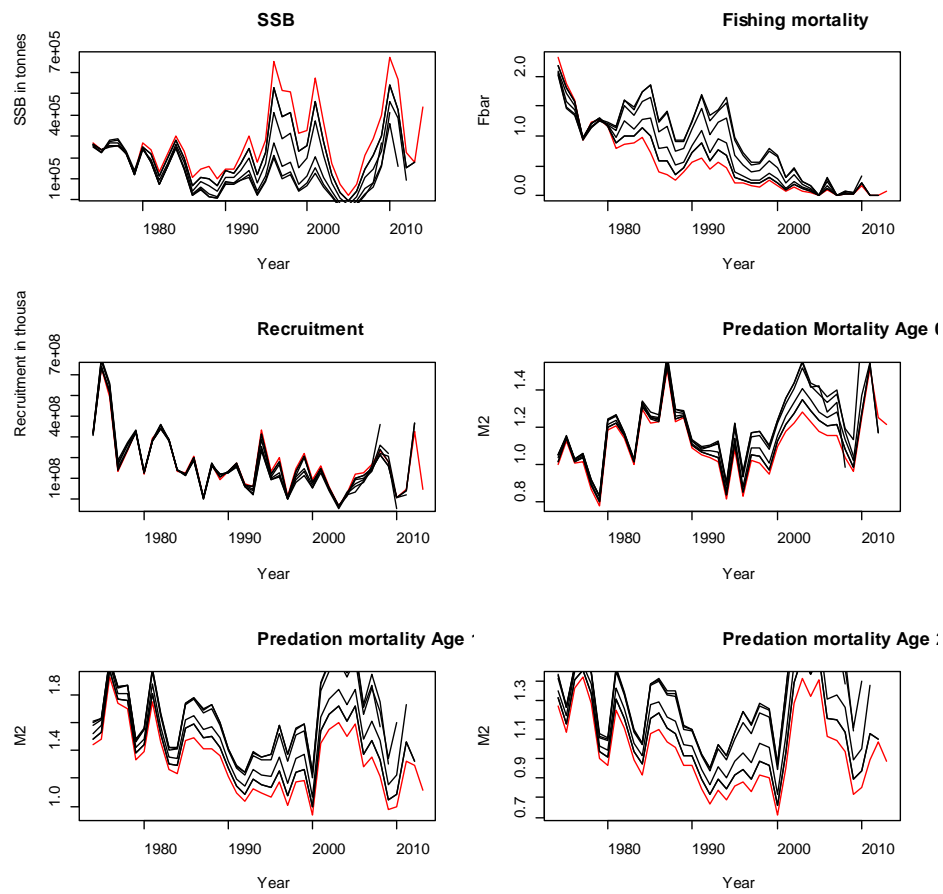


Figure 2. Retrospective patterns in the SMS keyrun from 2015 for N pout.

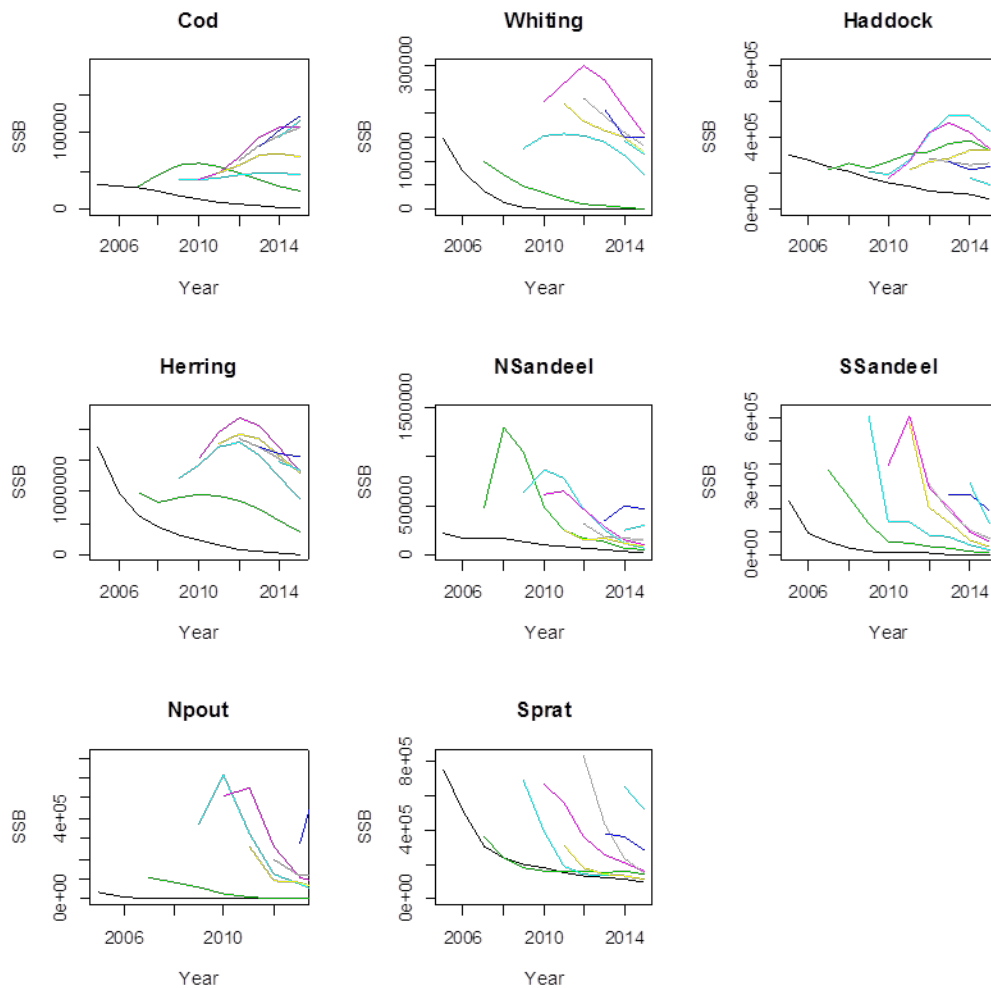


Figure 3. Multi species forecasts until 2015 from different final assessment years with known trends in F from single species assessments and SRRs fitted to the SMS assessment results.

Global sensitivity analysis of a multi-species model

- Used global sensitivity analysis to quantify how uncertainties in the inputs contribute to uncertainties in the outputs.
- The aim was to use variance based sensitivity analysis. However, due to the large number of inputs this is computationally infeasible.
- Used derivative based sensitivity analysis to decide which inputs had a negligible effect on the sensitivity of the output. This enables the variance based sensitivity analysis to be more tractable.

A sensitivity analysis of a multispecies size spectrum model of the North Sea was presented (Blanchard *et al.* 2014). The method was based on variance-based sensitivity (Saltelli *et al.* 2008), which measures the sensitivity of an input as the expected reduction in the uncertainty of the output if we were to learn the exact value of the input. It can be computationally infeasible to do to find the sensitivity indices for each of the inputs. De-

derivative based sensitivity analysis (Sobol' and Kucherenko, 2009) was used to screen the inputs to find ones that had very little effect on the uncertainty of the model. This will make the sensitivity analysis computationally feasible. This method will help when fitting complex ecosystem models and could advise empiricists and modellers about what data is needed to be collected to reduce the uncertainty of the model outputs.

For more details email Hayley Bannister: hjbannister1@sheffield.ac.uk

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.

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Estimating hake natural mortality based on multispecies model results and longevity (Santiago Cerviño and Camilo Saavedra).

- Cetaceans are the main predators of hake in the Iberian peninsula
- Multispecies models provide a way to estimate preys M-at-age as the sum of two components (M1 and M2). M2 is based on predator diet but M2 can be difficult to estimate if there is not a minimum likelihood at realistic M1 values.
- The use of life history parameters (longevity) was explored as a tool to select and adequate M1.

Multispecies models can help single species models providing M-at-age figures. Natural mortality in a key parameter in assessment model usually set as a constant at time and age (or length). However it is known that M varies on age (or length). Changes in M at age (or size) are dependent on life history processes like growth (small fish has more potential predators) or maturity process that triggers senescence. The objectives of the current work are to present a method that combines two different approaches to estimate a thorough variable M-at-age to be used in the single species models. To this aim we have used different sources of information. A multispecies model that provides a combination of M1 and M2 mortality coming from both, predation and other causes of natural mortality and (2) the known relationship between longevity (t_{max}) and M (Hewitt and Hoenig, 2005), which is further extended to explain how it relates with variable M-at-age and how can it be used to select the best M1 in an multispecies model.

Longevity (t_{max}), similar than life expectancy, refers to long-lived members of a population, and it is lower when mortality increases. In absence of fishing, M is the only source of mortality. Exponential models for M [$N_t = N_0 * \exp(-M*t)$] do not help to calculate t_{max} properly since population abundance decreases until $t = \infty$. To interpret the relations between M and t_{max} with exponential models we need to define the level of population reduction (N_t/N_0) at $t=t_{max}$. Hewitt and Hoening (2005) found that t_{max} happens when population reduction ($N_{t_{max}}/N_0$) is about 1.5%. Then M can be calculated with the following equation:

$$M = -\ln(0.015) / t_{\max}$$

In this way, given a known t_{\max} , a constant M-at-age can be estimated. However notice that M is the mean M in the life span from $t=0$ to t_{\max} , and t_{\max} is independent of the age allocations of M.

$$M = \sum_{t=0}^{t_{\max}} M_t / t_{\max}$$

In this way M can be allocated differently at different ages if additional information about M variable at age is available. The equations do not allow allocating different M-at-age but whether other method is available longevity (or t_{\max}) can be used to check whether the estimates are acceptable.

Taken in consideration that hake longevity is around 10 years, this means that the mean M for ages 0 to 10 would be 0.40. The multispecies model provides a different share of M-at-age since cetaceans feed on small hake (mainly smaller than 30 cm, i.e. ages 0 to 3). The most adequate M1 would be this that provides a total accumulated M-at age ($M1+M2$) for ages 0 to 10 equal than the M assumed. The final selected M-at-age (corresponding with $M1=0.2$) was implemented in the single species model and the fit quality compared between constant and variable M-at-age. The model likelihood shows that the hake model with variable M improves the current likelihood assessment model in a 12%.

The analysis performed shows that longevity can help to estimate variable M-at-age from multi-species models helping to set an adequate M1 level. However, to have a final estimate for Souther hake the impact of some sources of uncertainty should be analysed. Among these we highlight the following 3: (1) To what extend is our guess for hake longevity (t_{\max}) representing age at which the population is reduced to its 1.5%? ; (2) cetacean abundance is quite uncertain affecting directly to estimated M-at-age and its variability was not yet considered; (3) M at age 0 should be higher than age 1, since cetaceans do not prey this age group this mortality it is only considered in the M1 constant component which do not allow high Ms at age 0. Cannibalism could explain a higher mortality for this group although the way to consider it is not straightforward. The aim of this exercise was to provide a M-at-age for hake single species assessment. Next hake ICES benchmark will be developed in 2019.

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Hewitt, D. A. and Hoenig, J. M. 2005. Comparison of two approaches for estimating natural mortality based on longevity. *Fishery Bulletin*, 103(2), pp. 433–437.

Hindcast Evaluation of a North Sea Multispecies Model

- Robert Thorpe gave a presentation on a “hindcast evaluation of a North Sea multispecies model” outlining a test of a new model that is being developed to see how useful it might be in terms of a) forecasting biomass trajectories, and b) predicting whether limit reference points might be breached.
- The model was tuned to stock assessments for the period 1990–2010 and then used to make a “forecast” for the period 2010–2015 using the same level of

fishing as in the assessments for that period, with the assessed outcomes assumed to represent “truth”. Model outcomes were compared with simply persisting the 2010 assessment, to judge whether the model was adding useful skill when compared against the best available information at the time.

- We found that the raw model output was less useful than persisting the last available assessment, but once corrected for biases that would have been known at the start of the forecast period, it outperforms persistence, and is able to forecast the recovery of the cod stock.
- The raw model output was more skilful than persistence at predicting whether limit reference points had been breached, but had similar skill following adjustment. The method presented here is potentially useful for evaluating the utility of multispecies models.

A new model is being developed for the North Sea, which has 37 stocks (the 21 existing ones + 9 nephrops FUs, bass, turbot, megrim, hake, ling, dogfish, and thornback ray), along with 5 fleets. We have conducted a hindcast using a 200-member ensemble of the model with different parameter settings to see how useful it is, and the methodology is shown schematically in Figure 1:

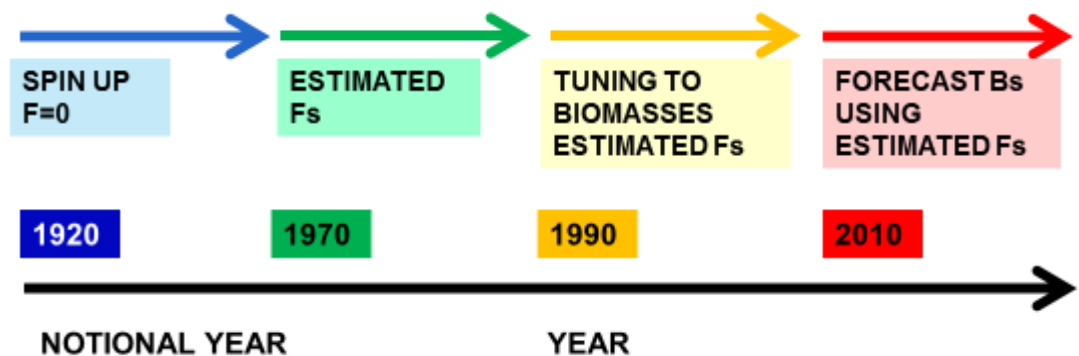


Figure 1. Schematic of hindcast experiment, showing how the multispecies model is tuned against data and then used in assessing outcomes of different fleet combination.

The ensemble members were selected based upon their ability to preserve stocks when unfished and to simulate estimates of stock status for the period 1990–2010. Each stock contributes to the model score. The stocks are grouped into three types depending upon the data available. The stocks that have a full assessment are tuned to the assessed biomass for each year. The stocks with landings information are tuned to the landing estimates, and the stocks with neither are tuned to lie within a broad range of mean survey estimated abundance. The overall score of the model is a weighted average of the individual stocks, with most weight being placed upon the stocks that perform most poorly (i.e. the model has a strong incentive to improve on its handling of the worst performing stocks).

The ensemble hindcast for the assessed stocks is shown in Figure 2. Biomasses are too low for sandeel and plaice, and too high for cod, haddock, and whiting. The variability of some stocks is also underestimated, but that is to be expected given that a deterministic stock-recruitment function was used in the model.

- BLACK == assessment
- Coloured == model ensemble
- SAN, PLE too low.
- COD, HAD, WHG too high.
- Deterministic recruitment

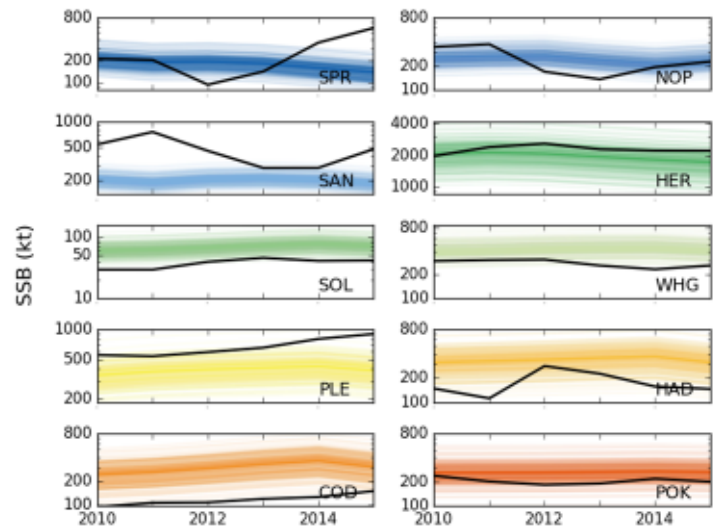
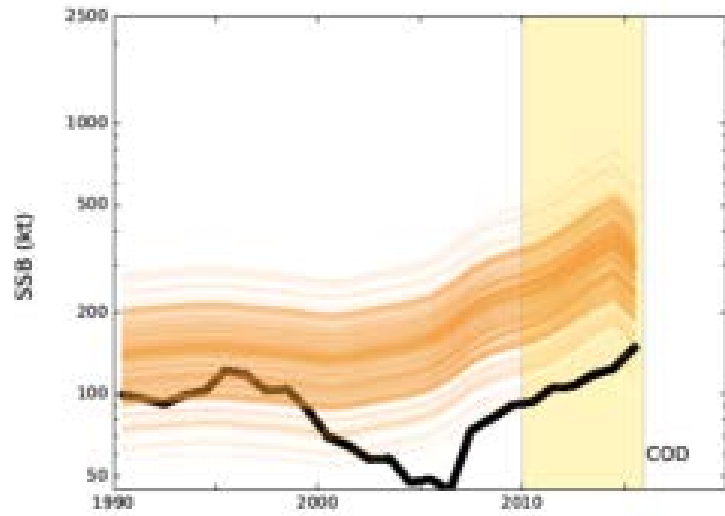


Figure 2. Model ensemble estimates of biomass for the period 2010–2015 for the 10 assessed stocks.

Cod is a particularly interesting case because the cod stock began to recover after around 2005. From the perspective of 1990–2010 (the tuning period), the recovery did not seem remarkable, but during 2010 to 2015 the estimated cod biomass moved above its peak in the tuning period. Figure 3 shows biomass outcomes for 1990–2015 for cod for a) the raw modelled output, and b) model output adjusted for bias. Once bias is taken account of, the model makes a good prediction of cod recovery, even though this was to levels not seen during the tuning process.

The ability of the model to predict whether the limit biomass was breached ($B < B_{lim}$) was also investigated, comparing the predictions of a breach with whether one occurred according to the assessment. 8 stocks had published Blims and 5 years (assessments for 2011–2015) were considered, making 40 instances in total. Outcomes were assessed on a 2x2 matrix, showing the correct forecasts, false alarms, and missed events (Figure 4).

RAW
MODEL



ADJ
MODEL

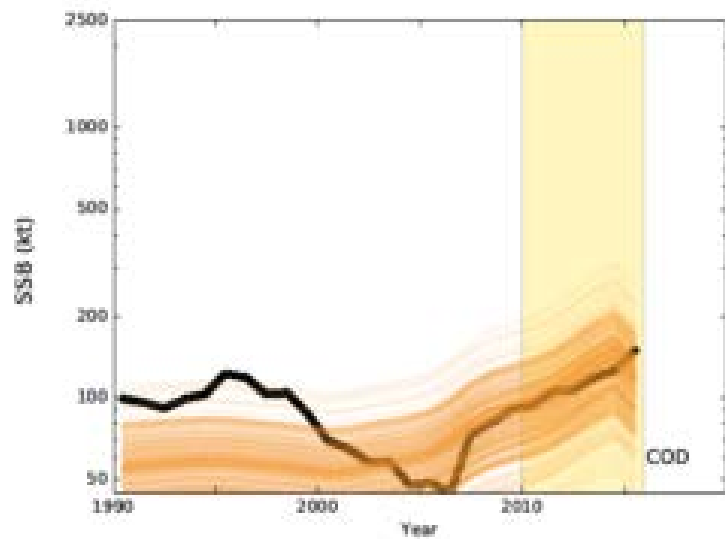


Figure 3. Ensemble biomass outcomes for cod for a) the raw model output, and b) the model as adjusted for known bias in 2010.

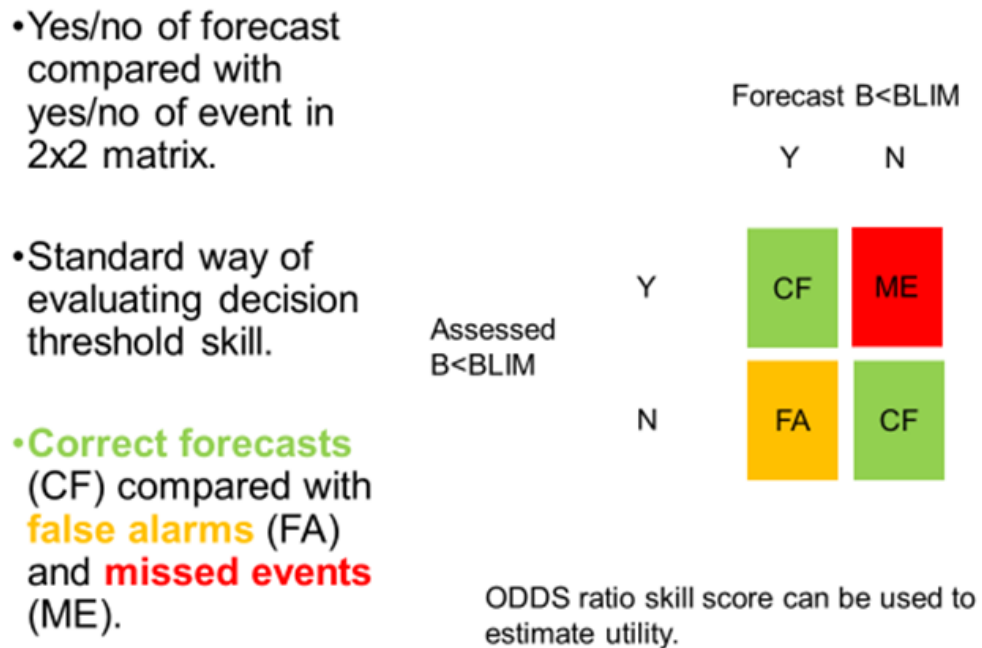


Figure 4. Schematic showing the way in which model skill at forecasting breaches of the limit biomass was assessed. The odds ratio skill score is defined and its properties are analysed in Stephenson *et al.* (2000).

We found that for this metric, the unadjusted model performed better (with one false alarm and 3 missed events), than persistence, which missed 6 events, but adjusting the model made things worse (with 6 missed events and one false alarm), suggesting that any benefits of the model on these timescales would depend on forecast methodology and might not be robust.

Overall, the methods presented here are potentially useful for assessing the skill of multi-species models as aids for management.

Reference:

Stephenson D.B. (2000) Use of the "Odds Ratio" for Diagnosing Forecast Skill, *Weather and Forecasting*, 15, 221–232

Annex 6: ToR D: Investigate the performance of multi-model ensemble in comparison to single model approach

A multi-model approach to understanding the role of Pacific sardine in the California Current food web

- Multi-model approach is useful, but is only a stepping stone toward true ensembles and more directly comparable models
- Structural assumptions influence predictions: taxonomic resolution (whether brown pelican modelled explicitly) and age structure and density dependence (Atlantis and MICE vs Ecopath/PREP)
- Monte Carlo approaches (MICE) give probability distributions, but we can also translate to make MICE comparable to Atlantis, Ecosim, etc.
- Atlantis identifies species MICE might consider: Dolphins, large flatfish (halibut) – both decline at moderate levels of sardine abundance; other birds

For the California Current on the West Coast of the US, Mexico, and Canada, we have developed a multi-model approach to explore how sardine *Sardinops sagax* abundance impacts the ecosystem and predators. In this region sardine and anchovy *Engraulis mordax* have recently declined to less than 10% of their recent peak abundances. As part of the Ocean Modelling Forum (<http://oceanmodellingforum.org/>), we have applied three ecosystem modelling approaches: Ecopath (Koehn *et al.* 2016), MICE (Punt *et al.* 2016), and Atlantis (Kaplan *et al.* 2017); we also used static Ecopath diets to predict impacts to predators using a statistical generalization of the dynamic Ecosim model (PREP, (Pikitch *et al.* 2012)). Note this was not a true ensemble approach, but nonetheless the collaboration, workshops with experts and stakeholders, and comparison of multiple models strengthened the effort and conclusions. Lessons learned for modelers included various methods to force the three models into a common currency for comparison. Results from both ecosystem models for which we have brown pelicans *Pelecanus occidentalis* modelled at the species level (MICE and Ecopath/PREP) emphasize the vulnerability of brown pelicans to low sardine abundance due to diets that are high in sardines and another species with high variability, anchovy. Two of our ecosystem models (MICE and Atlantis) suggest that California sea lions *Zalophus californianus* should exhibit relatively minor responses to sardine depletion, due to having broader diets and lower reliance on anchovy. On the other hand, Ecopath/PREP suggests that sardine declines will have a strong impact on California sea lions. This discrepancy reflects structural differences in the models: weaker responses in Atlantis and MICE are likely attributable to the explicit representation of density dependence and age-structure. The work has now been presented to a subcommittee of the local fishery management council; however there is no immediate management application since the sardine fishery remains closed due to low stock abundance. Contact: Isaac C Kaplan (Isaac.Kaplan@noaa.gov)

(full list of collaborators: Tessa B Francis, André E Punt, Laura E Koehn, Enrique Curchitser, Felipe Hurtado-Ferro, Kelli F Johnson, Salvador E. Lluch-Cota, William J Sydeman, Timothy E Essington, Nathan Taylor, Kirstin Holsman, Alec D MacCall, and Phillip S. Levin)

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Alaska Climate Change Integrated Modelling (ACLIM) model ensemble

- Five Climate-Enhanced (CE) models (stock, multispecies, ecosystem, fleet and human community) will be used together to evaluate potential responses to projected climate change in the eastern Bering Sea.
- Alternative management strategies will be evaluated under different climate projections

The Alaska Climate Change Integrated Modelling (ACLIM) project represents a comprehensive, collaborative effort to characterize and project climate-driven changes to the Bering Sea ecosystem, from physics to fishing communities, and to understand how different fisheries management approaches might help promote adaptation to climate-driven changes and long-term sustainability in fish and shellfish populations. To address this goal, ACLIM strives to evaluate fishery management strategies under different climate change scenarios in the Bering Sea. It connects research on scaling of climate models, climate-enhanced biological models, and socio-economic and harvest scenarios. ACLIM is a multi-year, interdisciplinary collaboration between 19 physical oceanographers, ecosystem modelers, socioeconomic researchers, and fishery management analysts from NOAA AFSC, NOAA PMEL, and the University of Washington. A major focus of the project is to quantify scenario, parameter, and structural uncertainty through a multi-model projection suite, which will aid in evaluating the performance of resource management strategies under different future scenarios. Projections of climate conditions are complete, and projections of catch for core species under baseline status-quo fishing conditions are underway for several fish and invertebrate species from the Eastern Bering Sea (EBS), for which changes in productivity have been linked to climate variability. Results will include projections of the future ecosystem state of the Bering Sea, probability of changes in catch under different management tools, and spatial and temporal schedules of expected change. A core component of the work is to evaluate alternative management strategies to adapt to changing conditions. ACLIM would like feedback from the council and stakeholders about potential strategies and metrics to evaluate during the next phase of the project. The purpose of the presentation to the council will be to present preliminary findings and initiate discussions on this front.

Primary target species (others included as available):

Walleye pollock (*Gadus chalcogrammus*)

Pacific cod (*Gadus macrocephalus*)

Arrowtooth flounder (*Atheresthes stomias*)

Northern rock sole (*Lepidopsetta polyxystra*)

Snow crab (*Chionoecetes opilio*)

Human fishing fleets and communities

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Comparisons of Models using the Jacobian Matrix

John Pope NRC(Europe) Ltd.

An interesting way of comparing multispecies models springs from the approximate fits of the Multispecies Schaefer models to the results from steady state solutions of more complex models. The Schaefer model describes steady state yield in terms of fishing mortality by N equations defined by:-

$$Y^1(i) = A(i)F'(i) + F'(i) * \sum_{j=1}^N (B(i, j) * F'(j)). \quad \text{Equation NS1}$$

Where $Y^1(i)$ is the steady state yield (landings) of species i, when fishing mortality rate on all species are set to status quo, i.e. when all $F'(i) = 1$ where $F'(i)$ Fishing mortality rate of species i written as the proportion of its status quo F. A_i and $B(i, j)$ are the constant terms of the quadratic equation and both i and $j=1:N$ where N is the number of species included in the ecosystem model.

This approximation is constructed from outputs from more complex models of yield such as SMS, (equivalent equations can also be used for discards (where known) and SSB (as a linear equation)) at the future steady state at status quo fishing, together with the equivalent long term steady states to be expected with a 10% increase in each individual species fishing mortality above status quo. This is the minimum information needed to solve parameters. Since if we know the $Y^1(i)$ and the $Y^{1.1(j)}(i)$ from the more complex model where $Y^{1.1(j)}(i)$ is the steady state yield when $F'(j)$ is set to 1.1 and all other $F'(i) = 1$ where $j \neq i$.

Then where $j \neq i$

$$(Y^{1.1(j)}(i) - Y^1(i))/0.1 = B(i, j) \quad \text{Equation NS 2}$$

Where $i=j$

$$(Y^{1.1(i)}(i) - Y^1(i)) = A(i) * .1 + B(i,i) * .21 \quad \text{Equation NS 3}$$

We may then substitute the $B(i, j)$ results of equations 2 back in to equation NS1 to get

$$Y^1(i) = A(i) + B(i, i) + \sum_{j \neq i} B(i, j) \quad \text{Equation NS 4}$$

and we may solve equations 3 and 4 for $A(i)$ and $B(i, i)$

Note that equation 4 indicates that the sum of $A(i) + \sum_{j \neq i} B(i, j) = Y^1(i)$

So that dividing all $A(i)$ and all $B(i, j)$ by $Y^1(i)$ standardizes them to a sum of 1. This is convenient for comparing the sizes of interactions of different models.

For comparative purposes it is also revealing to work with the Jacobian Matrix itself. When the $a(i)$ and $B(i, j)$ have been calculated $N \times N$ the Jacobian Matrix $J = [\partial Y^1(i) / \partial F'(j)]$

May be written as $J = B + \text{diag}(A) + \text{diag}(B)$

Where B is the $N \times N$ matrix of the $B(i, j)$. Where $\text{diag}(A)$ is an $N \times N$ matrix with the A_i on the main diagonal and $\text{diag}(B)$ is an $N \times N$ matrix with the $B(i, i)$ on the main diagonal and where all of diagonal terms are zero in both cases. That is

$$\partial Y^1(i) / \partial F'(j) = A_i + 2 * B(i, i) \text{ if } i = j$$

And

$$\partial Y^1(i) / \partial F'(j) = B(i, j) \text{ if } i \neq j$$

Figure 1 below shows the values of $A(i)$ and $B(i, j)$ for each species derived from the SMS run used to parameterize the GREEN model and an alternative parameterisation based upon a single realization of the Thorpe *et al.* (2015, 2016) model. This latter run is for illustrative purposes only since it only one run of an ensemble. Normally one would expect $A(i)$ to be positive and $B(i, i)$ to be negative. Positive $B(i, j)$ suggests that a species is a predator so that fishing it more should increase the i species. This is the case with the Green Model for pok on had, her, nop and whg but these interactions are not seen in the particular run shown of the Thorpe model.

Figure NS2 below shows radar plots for the $\partial Y^1(i) / \partial F'(j)$ of each species derived from the SMS run used to parameterize the GREEN model and an alternative parameterisation based upon a single realization of the Thorpe *et al.* (2015, 2016) model. This latter run is for illustrative purposes only since it only one run of an ensemble.

Note that if all $\partial Y^1(i) / \partial F'(j)$ were zero when $i \neq j$ then we would just have a series of single species models. This seems very nearly to be the case for plaice(PL), saithe(P), sole(S) for the Green Model and for the Thorpe run results. That the Green Model Results are more reactive than the specific Thorpe run results is apparent from both plots.

What should we make of this? First it would seem how interactions are estimated has a large impact upon their size. It is perhaps interesting that haddock, herring, Norway pout and whiting all have noticeable interactions with pok (saithe). The saithe was the species where the stomach content data collected in 1981 and in 1991 gave notably differ-

ent estimates of suitability for MSVPA. (Rice *et al.* 1991 and ICES 1992, ICES,1994) and thus may give rise to particularly large uncertainties.

It will be interesting to add other North Sea multispecies models to this figure!

Apart from the direct examination of the $A(i)$ and $B(i,j)$ it is also possible to use these results to predict various reference points for the approximation though note that these may well not hold for the underlying models if they predict F' beyond the likely validity of the quadratic approximation of yield.

An obvious though probably stupid reference point is overall MSY. Note that the summed yield of the quadratic system is

$$\text{Total Yield} = \sum_i (A(i) * F'(i) + \sum_j F'(i) * B(i,j) * F'(j))$$

Considering that at the maximum total yield $\frac{\partial(\text{total yield})}{\partial F'(j)} = 0.0$

Leads to the matrix equation $A + |B + B'| * F' = O'$

Where A is the $N*1$ column vector of the $A(i)$ s B is the $N*N$ matrix of $B(i,j)$ and B' its transpose, F' a $N*1$ column vector of fishing mortality relative to its status quo value and O' a $N*1$ column vector of zeros.

Thus Total MSY occurs at

$$F' = |B + B'|^{-1} * -A$$

It would be a simple matter to compare the F' s from different models.

This could be more sensibly considered as a maximum value solution is the A and B matrices were first multiplied by a diagonal matrix containing the relative values of each species(i) on the diagonal.

It would also be simple to compute the Nash equilibrium at

$$F' = |B + \text{diag}(B)|^{-1} * -A$$

Where $\text{diag}(B)$ is the diagonal matrix containing only the $B(i,i)$ terms.

The Multispecies Working Group of I think 1989 also considered multispecies $F_{0.1}$. I must look this up since I was smarter in 1989 than I am now!

More complex questions such as MEY could of course also be considered by putting the A's and B's into the T-ONS model and using the Solver routine in EXCEL to discover constrained maxima of economic or social factors. Typically such maximizations take about 2 minutes so are very practical to compute.

However arrived at, the values of comparable F's would be a useful way to compare the predictions of two or more models and could readily be presented as Graphs. I will try to work out a few for you.

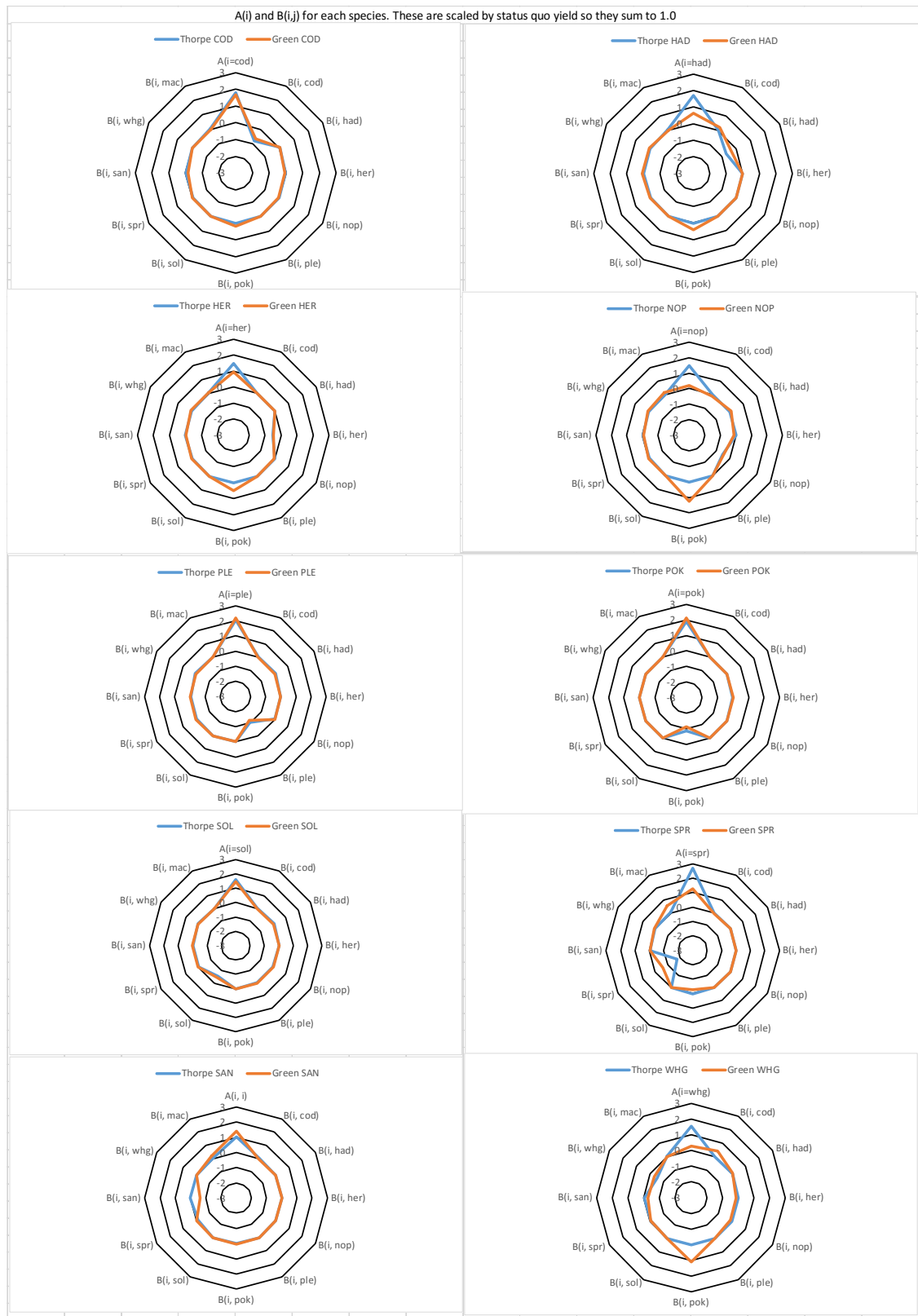


Figure 1. the A(i) and B(i,j) terms for each species i



Figure 2. Comparison of standardized interaction terms for each species derived from the Thorpe run and from the Green model. The i in the labels should be understood to be the species given in the legend.

A dynamic multi-model ensemble for marine ecosystem simulators

- A method for combining different ecosystem models with different outputs.
- The ensemble model exploits the strengths and discounts the weaknesses of each of the models.
- A proof of concept example was shown. It demonstrated that the ensemble model is able to predict, with quantifiable measures of uncertainty what would happen in the future under specific scenarios.

The ensemble model, developed as part of the Marine Ecosystem Research Programme, was presented which aims to combine outputs from different marine ecosystem models. 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 can be 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 grouped by their living habitat whereas in the LeMans model (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.

Using the ensemble model design in Figure 1, we are able to learn how each of the models is wrong relative to the simulator consensus known as the individual discrepancy, and then using noisy, possibly incomplete observations of the truth, we are able to learn how wrong the simulator consensus is from the truth, the shared discrepancy.

Two cases studies were presented to demonstrate the use of the ensemble model. The recovery times of indicators after a change in fishing pressure and what would happen if we were to stop fishing? These cases studies demonstrated the utility of the ensemble model and predicted what would happen in the future with robust measures of uncertainty (see Figure 2).

Using this framework, the ensemble model “exploits the strengths and discounts the weaknesses” (Chandler, 2013) of each of the models. We therefore advise that models should be very good at predicting a few aspects thing as opposed to being okay at predicting many things. We also advise that all available models be included in the ensemble, as there the ensemble model will discount the models if they are poor and take strengths from areas where they are strong.

With this in mind we are going to extend the methodology to include single species models as well as include models that are unable to run a specific scenario. For example, if we were interested in climate change, we would include the multispecies size spectrum model (Scott *et al.* 2014) despite it not having temperature built into it.

Current methods are able to predict what would happen under specific scenarios. Future work will extend the ensemble model to cope with a continuum of scenarios.

For more details see Spence *et al.* (2017).

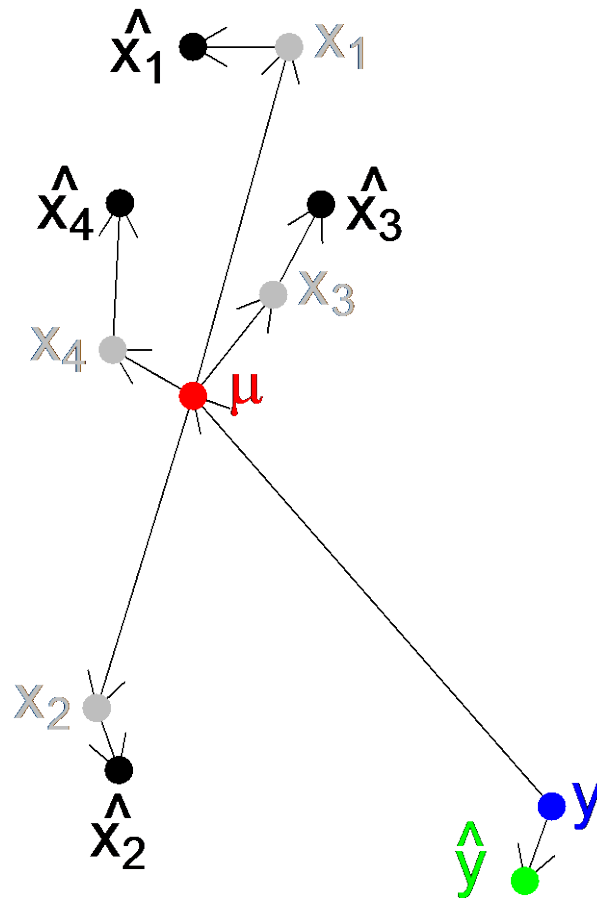


Figure 46 The idealized model outputs, or “best guess”, of each of the models, the value that the model would output if it were able to output all of the variables that we are interested in (e.g. what StrathE2E would predict if they were able to model cod) with no parameter uncertainty (i.e. fitted to an infinite amount of data) are the grey nodes and are centred on the simulator consensus, the red node. The actual model outputs are the black nodes. The difference between the model outputs and the simulator consensus is known as the individual discrepancy. The green nodes are noisy, possibly incomplete observations of the truth, the blue node. The difference between the truth and the simulator consensus is known as the shared discrepancy.

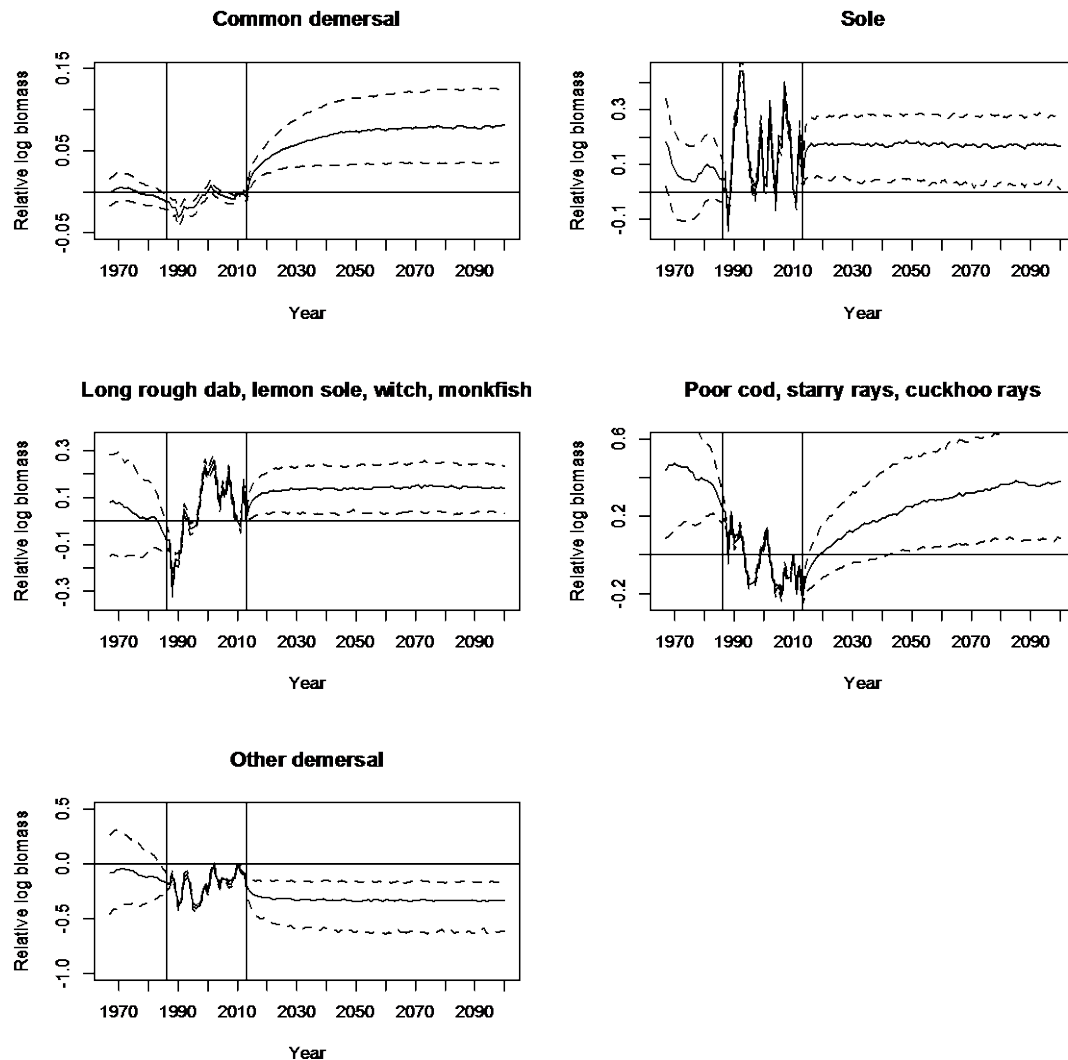


Figure 47 The results of the ensemble model. Using 5 ecosystem models we were able to predict what would have happened to demersal species if we had stopped fishing in 2013?

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Scott, F., Blanchard, J.L. and Andersen, K.H. 2014. mizer: An R package for multispecies, trait-based and community size spectrum ecological modelling. *Methods in Ecology and Evolution*, 5, 1121–1125.

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Annex 7: ToR E: Test performance and sensitivity of ecosystem indicators

No work has been presented under this ToR this year.

Annex 8: ToR F: Metanalysis of impact of top predators on fish stocks in ICES waters

Estimating abundances and spatial distributions of great and sooty shearwaters in the Bay of Biscay

Overall conclusions:

- Thanks to multidisciplinary surveys, the distribution and habitat use of species within the Birds and Habitats Directive in the Bay of Biscay can be determined.
- Relative abundance and temporal trends of marine top predators can be estimated with the information provided by these survey, supporting the detection of potential changes in their populations.
- High importance of considering oceanographic, prey and static variables to explain the abundance of marine top predators.
- Spatial predictions obtained matched with abundance patterns
- Both the survey used (JUVENA) and SAMM (Aerial Monitoring of Marine Megafauna) provide similar orders of magnitude despite methodological differences, areas and sampling times
- Despite inter-annual variability, some oceanographic phenomena (upwelling, river plumes) occur in similar places year after year, then high density areas could be limited in space.
- Overlap with human pressures, although additional data sources / programs are needed (bycatch, VMS).
- The results obtained are comparable to previous information found in the literature.

Big efforts have been done to identify the status of marine top predators in the Bay of Biscay since 2013, foccusing mainly on two seabird species (sooty and great shearwaters) and one marine mammal species (fin whale). Different species' habitats have been characterized and their relative abundance estimated. Marine habitats with high human pressure have also been identified.

An observer programme has been operationalized since 2013, when sightings from observers onboard multidisciplinary surveys operating in the study started to provide information about the relative abundance of different species. At the same time, abundances of different prey species (mainly anchovy and sardine) and other oceanographic (currents, salinity, teemperature) and static (bathymetry and distance to shelf) variables were measured. All this information was therefore used to develop an spatial abundance model that allow predicting abundance as a function of the previously mentioned variables. Inter-annual variability in areas of higher density was also analyzed. The Generalized Additive Models Information theoretic approach has been used for that (Louzao *et al.*, in review; García-Barón *et al.*, in review).

The line transect sampling methodology was used to estimate the relative abundance of these species from counts at sea and therefore compared with additional surveys (SAMM).

Reference:

Louzao, M., García-Barón, I., Rubio, A., Martínez, U., Vázquez, J.A., Murcia, J.L., Boyra, G.. Understanding pelagic seabird 3D environment from multidisciplinary oceanographic surveys. MEPS (in review)

Modifying longlining operational techniques to limit fish depredation by marine mammals

- Longline fisheries are the most impacted fisheries by mammal depredation.
- Recent technological approaches implemented proved to be less if not at all effective to limiting depredation impact to fisheries.
- Changes in operational techniques, specifically on fishing behaviours, have been tested taking into account the case study of longline fisheries in Crozet and Kerguelen Islands.
- Preliminary results indicate that depredation in these areas is mainly influenced by mammalian ecology and the conditions of fishing operations.
- A manuscript was recently submitted to Fisheries Research discussing the results obtained from this case study (see reference below).

Abstract:

Marine mammal depredation on fisheries (animals removing fish caught on fishing gear) is a worldwide issue involving socio-economic and ecological consequences. Longline fisheries are the most impacted fisheries by mammal depredation. While technological means have proved limited efficacy in reducing depredation, our study examined how the fishing practices influence both the proportion of depredated longline sets and the amount of fish removed by whales. We used an 8-year long fishing dataset from the Patagonian toothfish (*Dissostichus eleginoides*) longline fisheries operating in Crozet and Kerguelen Economic Exclusive Zones (EEZs) (South Indian Ocean) and GLMMs to investigate sperm whale (*Physeter macrocephalus*) depredation. Sperm whale depredation occurred on 60.5% of 5,260 sets in Crozet and 40.7% of 16,902 sets in Kerguelen, and resulted as the only depredating species in estimated toothfish losses of 266 tons and 1,281 tons, respectively, in the two areas. The probability of depredation decreased depending on the season, depth and following a fishing activity migration away from with occurring depredation. Aside from these, the soaking and hauling times were also observed as factors that may contribute to decrease depredation impact on fisheries. These observations can then be combined to implement strategies of avoidance in all fisheries facing similar depredation impact.

Keywords: Depredation, demersal longline, sperm whale, Patagonian toothfish, changing fishing practices.

Reference to paper:

A. Janc, G. Richard, C. Guinet, J.P.Y. Arnould, M.C. Villanueva, G. Duhamel, N. Gasco, P. Tixier. How do fishing practices influence sperm whale (*Physeter macrocephalus*) depredation on demersal longline fisheries? Submitted to Fisheries Research

Annex 9: ToR G: Explore the consequence of multispecies, mixed fisheries interactions and environmental factors in practical multispecies advice for fisheries management

The effects of density dependent clupeid growth on Nash equilibrium reference points in the Baltic Sea

- Fitting of the clupeid growth functions in the MSI-SOM model to data is improved with intra- and interdependent density dependence.
- The inclusion of density dependent clupeid growth in the MSI-SOM has minor effects on the Nash equilibrium reference points

The multispecies model in Norrström *et al.* (2016) does not include direct density dependent effects on growth between the Baltic Sea herring and sprat. There are however indications that growth in these clupeid species is density dependent (Casini *et al.* 2006, Möllmann *et al.* 2004). Therefore, the MSI-SOM model was updated to take density dependent growth in the clupeids into consideration. The intra- and inter species density dependence is based on the total stock size of each clupeid species and the evaluations of the growth models use Akaike information criteria. To reduce the number of evaluated models the evaluation was done in two steps, first the best fitting growth models without density dependence were found for each clupeid species. Thereafter density dependence was added as additive effects and, or, as effects interacting with the average weight at age of the evaluated species. The updated model with intra- and inter species density dependence perform better than the previous models, supporting the theory that density dependence in clupeid growth influences the system, where the effects of the best fitting growth models of herring and sprat respectively are:

$$G_H = S + T + N_s + N_H \cdot W + \frac{W}{S},$$

and

$$G_S = S + T + N_s + W + N_H \cdot W + \frac{W}{S}$$

where S is salinity, T is temperature, N_H is the herring stock size and N_s is the stock size of sprat.

The changes in the Nash equilibrium reference points due to density dependent clupeid growth are relatively small with the biomass at MSY (BMSY) for herring being the exception, increasing with 273 thousand tonnes (Table 1).

Table 1. Nash equilibrium reference points for herring and sprat according to Norrström *et al.* 2016, denoted no dd in the table. Updated values for the model with density dependent clupeid growth is denoted dd. Yield and BMSY is shown in thousand tonnes.

	FNE		BMSY		MSY	
	no dd	dd	no dd	dd	no dd	dd
Cod	0.47	0.45	211	295	76	102

Herring	0.3	0.27	460	733	115	167
Sprat	0.54	0.59	794	663	402	370

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.

Casini, M., Cardinale, M., Hjelm, J. 2006. Inter-annual variation in herring, *Clupea harengus*, and sprat, *Sprattus sprattus*, conditions in the central Baltic Sea: what gives the tune? OIKOS 112, pp. 638–650.

Möllmann, C., Kornilovs, G., Fetter, M., Köster, F.W. 2004. Feeding ecology of central Baltic Sea herring and sprat. Journal of Fish Biology. DOI:10.1111/j.0022-1112.2004.00566.x

Ecosystem FMSY Project

A project called “Ecosystem Based FMSY Values in Fisheries Management” is running to try and identify multi-species Fmsy estimates for ICES stocks with analytic assessments across the North Atlantic, and compare these with current single-species based estimates. This project spans a number of the ecoregions, the Baltic, North Sea, Norwegian and Barents Sea, Icelandic waters and NW Atlantic. The project does not examine the trade-offs inherent in multispecies management, rather it considers each stock in isolation and attempts to identify the estimates of Fmsy arising from multispecies models (without changing fishing patterns for other stocks). The project will use production models to give a simple route at finding these multispecies Fmsy estimates, and will also collate existing multispecies or ecosystem Fmsy estimates for the different stocks. The project will then examine similarities and differences between the multi-species and single-species estimates, and between the different multispecies models.

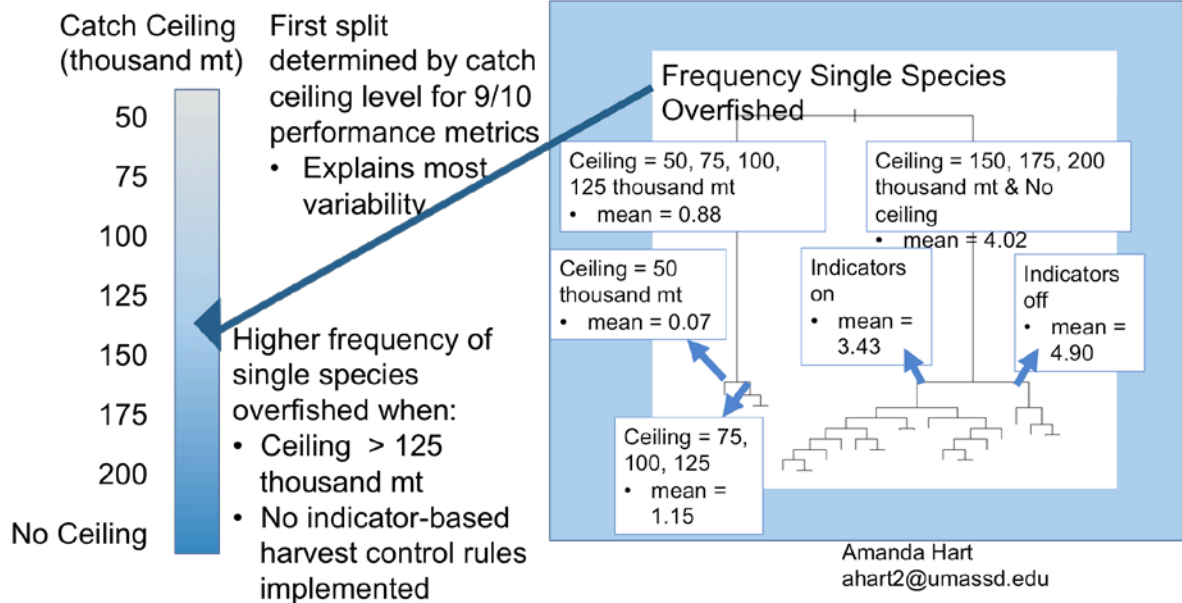
Evaluating an ecosystem-based fishery management procedure for Georges Bank using ceilings on system removals

- The ceiling level on total system removals explains most of the variability in performance metrics at the whole ecosystem, aggregate species groups, and single species level
- Implementation of indicator-based harvest control rules also explained a large portion of performance variability when ceilings were set to higher values.

We conducted a Management Strategy Evaluation (MSE) to test a proposed Ecosystem-Based Fishery Management (EBFM) strategy for Georges Bank in the Northeast U.S. We imposed a ceiling on total system removals in conjunction with indicator-based harvest control rules to evaluate combinations of management actions which may be effective for managing multiple species at once in an ecosystem context. Preliminary results suggest the level at which we set the ceiling on total system removals explains most of the variability in performance metrics at the whole ecosystem, aggregate species groups, and single species level. Generally, lower ceilings corresponded with lower frequencies of collapse and higher system diversity. However, the implementation of indicator-

based harvest control rules also explained a large portion of performance variability when ceilings were set to higher values. This indicates that these control rules are likely necessary to properly manage aggregate groups and single species when biomass removal from the system is high. Contact: Amanda Hart (ahart2@umassd.edu)

Ecosystem-Based Fishery Management Performance:



Ecosystem Based Fishery Management in New England, USA

- A proposed EBFM procedure is described, with components illustrated using model simulations

The main steps in the Ecosystem-based Management Procedure are:

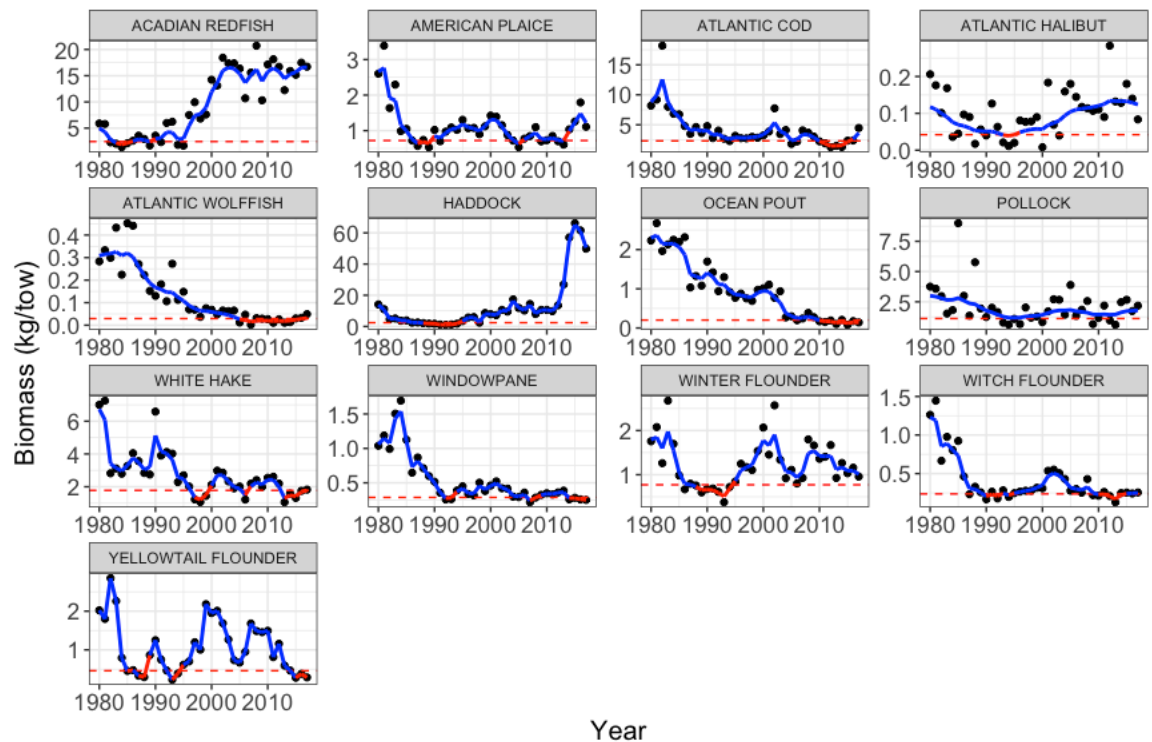
1. Specify spatial management units. EBFM is a place-based approach. We require a common spatial footprint for the species included in the management plan and in management-oriented models. By identifying Ecological Production Units (EPUs) we replace a very large number of stock areas currently used in management into a much smaller number of management units. EPUs are objectively defined based on physical and oceanographic features combined with measures reflecting the amount of energy at the base of the food web.

2. Establish specific management objectives and exploitation reference points directed at stock complexes rather than individual species. The approach adopted here defines overfishing at the species complex level. A species complex is defined as species that are caught together in specified fleet sectors and which share common life history attributes and roles in the food web. This is intended to simplify analyses (and management) by avoiding any attempt to quantify all pairwise interactions between species (e.g. predation, competition) and by recognizing the inherent difficulty in controlling fishing mortality rates for individual species in mixed-species fisheries.

In our simulations we considered three fleet sectors, demersal trawls, demersal fixed gear (longlines and seines), and pelagic trawls. We recognize three trophic groups (planktivores - herring and mackerel; benthivores - haddock, winter flounder, yellowtail flounder; and piscivores (spiny dogfish, winter skate, monkfish, silver hake, cod). As described below, we provide additional protection for species with vulnerable life histories related to reproductive dynamics. In this case, additional protection is provided for elasmobranchs. Here, we simulate the effects of exploitation rates ranging from 10–30% at the stock complex level. We examine the effects of exploitation rates at the species complex level on yield, revenue, population status (depletion), stability of catch and population size, and proportion of large fish in the catch and in the population. We define a limit exploitation reference point as the ratio of large to small phytoplankton production rates (a proxy for the proportion of new production in the system). This ratio has been increasing on Georges Bank and culminated in a peak of 0.28 in 2014. We set a maximum exploitation rate level at 0.3 and examined the performance characteristics of progressively reducing the exploitation rate to 0.15. This approach allows for applying different exploitation rates on different species complexes to find an optimum solution. For simplicity, here we will show results in which the same exploitation rate is applied for each species complexes. Once we choose a preferred exploitation rate, we can then calculate a corresponding catch at the species complex level by multiplying the exploitation rate by an estimate of the biomass.

3. Establish biomass thresholds (floors) below which the complex as a whole cannot fall (Option 1) or below which no species within the complex can fall (Option 2). In our simulation model, we estimate the biomass of species and species groups to estimate the catch. The model is designed to estimate biomass using simulated stock assessments based on production models, delayed-difference models with a simple age structure, and catchability corrected survey indices providing estimates of total catch.

In the simulations, we determine a stock complex to be depleted (Option 1) if the total biomass of the complex drops below 20% of the unexploited state. A more conservative option (2) determines whether any species in the complex drops below 20% of the unexploited state. [We also explore an alternative version in which depletion is defined as when species deemed to be vulnerable drop below 30% of the unexploited state; in our simulations, elasmobranchs are considered vulnerable species because of their life history characteristics]. For application to real-world populations, we would use a time-series (from surveys or stock assessment) that drops below a specified floor (e.g. below the 20th percentile of the time-series or below some designated 'safe' level). This is what is currently done in index-based single species assessments (which are now used for nearly half of the NEFMC-managed fish species).



4. Devise an Ecosystem-based Harvest Control Rule based on steps 2 and 3 designed to minimize the risk of overfishing for a range of exploitation rates at the stock complex level. In our simulations, we examine a range of exploitation rates from 0.15 to 0.30 in increments of 0.05. We then designate protective measures as described in Step 3 that include several major options including: (1) A threshold measure in which landings are prohibited for a species complex once the biomass falls below 20% of the unexploited state in the simulations and (2) strategies for phased implementation of protective measures in which (a) once either the functional group drops below 40% of the unexploited state, or (b) any species within a species complex drops below 40% of its unexploited state (option b). In an additional scenario we provide greater protection for vulnerable species in the phased implementation plan by starting reductions once the vulnerable populations drop below 50% of the unexploited state

5. Simulate the performance of a set of scenarios constructed under the EBMP using a suit of metrics including biomass, landings, revenue, probability of breaching a threshold biomass level, maintaining robust size structure of the populations (large fish index), and the stability of the landings. We find that low levels of exploitation rate (0.15–0.20) had the best overall performance in terms of biomass, yield, and risk of falling into the depleted status. Different species complexes have different risk tolerances with the planktivores being most sensitive, responding adversely to increases in exploitation rate above 15%. For the case where we apply an exploitation rate of 15% for each species complex, and examine the performance of the fixed threshold scenario, we estimate a catch total (cap) over all species complexes of 145 000mt and a biomass level of ~1940 000 mt. For the case of protection applied at the species complex level, we estimate a total catch of 155 000 mt and a biomass level of 2 250 000 mt. For the scenario in which

we invoke protections at the species level, the estimated overall catch is 150 000mt with a biomass of 2 430 000 mt.

Considerable caution is needed in comparing these results against real world catch and species status information. The simulation model is intended to be representative of the biological and ecological characteristics of the species included. Major modules of the simulation studies are estimated from real world data for these species. However, the simulation model is not simultaneously fit to all data sources to estimate outputs such as biomass and catch. Given these caveats, the simulation model returns catch and biomass results comparable to estimates of real world data. For example, the total catch (landings and discards) of the 9 NEFMC-managed species included in the simulation during the period 1977–2014 was just under 150 000 mt.

6. Identify and reconcile tradeoffs. The major tradeoffs involve catch, revenue, and species-complex or species status. We can gain a useful perspective on tradeoffs among a broader suite of indicators for the fixed exploitation rate threshold simulations and the graduated (ramp) analyses for the species complex and individual species protection scenarios, the best performance for the different metrics is indicated when the standardized value on each ‘spoke’ of the plot is at the outer margins. Conversely, poor performance for different metrics is indicated by values near the center of the diagram. The best overall performance for maximizing landings, revenue and managing risk was for an exploitation rate of 0.15 for the ramp-down procedure and determination of overfished status determined at the species level. This higher overall performance comes at the cost of a slight loss in yield, revenue, and stability of landings relative to the scenario in which we invoked protections at the species complex level with the benefit of reduced risk of population biomass falling below the overfished status level. Contact: Mike Fogarty (Michael.Fogarty@noaa.gov)

