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Executive summary

This is the fifth report of the pan-regional Working Group on Multispecies Assessment Methods (WGSAM). The group met at NOAA in Woods Hole, USA and reviewed ongoing multispecies and ecosystem modelling activities in each ICES ecoregion (including the North Sea, Baltic Sea, Bay of Biscay, Mediterranean, Iceland, Barents/Norwegian Seas, and eastern USA).

The participants provided an updated inventory, to supplement the information collated in 2007–2010 (ToR 'a' and 'b'). New information was presented for Iceland, Barents Sea, North Sea, Celtic Sea, Bay of Biscay, Baltic Sea, Northeast US, and the Pacific Northwest. The group reviewed a new key run for the North Sea SMS which included one new prey species (sprat) and two new predators (grey seals and harbour porpoise). In addition, a summary dataset of natural mortalities, stock numbers and biomasses of the modelled species was compiled and is available for download with the report. The key run-concept was further developed to the extent where the first Ecopath with Ecosim key run for the North Sea was reviewed by the group. Summary data from the run are also available for download with the report.

WGSAM works continually towards significant improvements in model functionality. This year, the work continued on the development of cross-model validation techniques and suggestions of how to test various multispecies models using a common, virtual dataset were discussed along with the necessary characteristics of such datasets (ToR 'd'). The group further reviewed the possibilities to include fleet dynamics in multispecies models (ToR 'e') and continued their work towards obtaining new stomach data from the ICES region (ToR 'c').

Simple relationships between natural mortality (M) and biomass of predators (B) were explored (ToR f). Several approaches were explored, with the conclusion that there are apt to be relationships between M and B, as conditioned upon several caveats. These may be useful inputs for single species assessments during interim years when MS key-runs are not updated.

The group reviewed estimates of consumption by top predators (including fish, seabirds, whales and seals) from a range of ecosystems and found that the consumption of marine mammals for some species exceeds the amount taken by other fish or the fishery whereas seabirds in most cases were responsible for less than 5% of total removals (ToR g). Estimates of M (and associated biological reference points) of some species derived from multispecies models are likely to be highly sensitive to the uncertainty in the population size estimates of top predators but leaving these predators from the models entirely is likely to lead to highly biased estimates of M. The group concluded that a rough estimate of the effect of top predators appears preferable to excluding this knowledge.

A proof of concept and worked example have been done to demonstrate the possibility of estimating diet compositions and predicting the trophic role of invasive and introduced species (ToR 'h'). The approach presented here is a useful protocol applicable for many other situations.

Following requests from HAWG, WGSAM analysed the consistency of the temporal patterns and level of M derived from different multispecies key runs from both 4M (a deterministic model no longer used by the group) and the two North Sea key runs provided with SMS (2008 and 2011). A comparison of the estimated natural mortalities shows that the temporal development in M derived from the three models show

similar patterns though the absolute level of M has increased over time as more predators have been added to the model.

1 Opening of the meeting

The **Working Group on Multispecies Assessment Methods** [WGSAM] met at NOAA in Woods hole, US from 10–14 October 2011. The list of participants and contact details are given in Annex 1. The two Co-Chairs, Jason Link (US) and Anna Rindorf (DTU-AQUA, Denmark) welcomed the participants and highlighted that like last year, the Working Group had a broad geographic scope, this year encompassing research in the Bay of Biscay, Barents Sea, Icelandic Sea, North Sea, Baltic Sea as well as the east and west coasts of North America. The Terms of Reference for the meeting (see Section 2) were discussed, and a plan of action was adopted with individuals providing presentations on particular issues and allocated separate tasks to begin work on all ToRs.

1.1 Acknowledgements

WGSAM would like to thank Jason Link and Jennifer Mindek (NOAA) for logistics during the meeting and Claire Welling of the ICES Secretariat for her continued support with the WGSAM SharePoint site.

2 Terms of reference

The **Working Group on Multispecies Assessment Methods** (WGSAM) co-chaired by Anna Rindorf*, Denmark and Jason Link*, US will meet in Woods hole, US from 10–14 October 2011 to:

a) Review further progress and report on key updates in multispecies and ecosystem modelling throughout the ICES region;

b) Report on the development of key-runs (standardized model runs updated with recent data, and agreed upon by WGSAM participants) of multi-species and eco-system models for different ICES regions (including the North Sea, Baltic Sea, Barents Sea, Bay of Biscay and others as appropriate);

c) Work towards implementing new stomach sampling programmes in the ICES area in the near future;

d) Explore how 'virtual multispecies datasets' (including survey, catch and stomach content data) for use in multiple multispecies models, especially for comparison and sensitivity testing, could be constructed;

e) Work towards inclusion of fleet dynamics in multispecies models;

f) Explore simple statistical relationships between M and B among predator and prey from output of multispecies models;

g) Improve quantification of the role of top predators (marine mammals, sea-birds, large pelagics) on forage fish in the ICES area ecosystems;

h) Explore the expected trophic role of invasive species using a simulation model package under anticipated conditions;

i) Address requests from other ICES Expert Groups as appropriate.

Of these, a and b are standing terms of reference, while c, d and e are 'multiyear projects'

3 ToR a) Review further progress in multispecies and ecosystem modelling throughout the ICES region

3.1 Ecoregion A: Greenland and Iceland Seas

Work is ongoing in Iceland on how to include marine mammals in Gadget. Both the modelling of their population dynamics and details on how to incorporate data on their diet and abundance are investigated. The Gadget model is currently used in a number of single species assessments and extended single species assessment of shrimp where natural mortality is related to the abundance of cod in the shrimp survey (as cod is a major predator on shrimp). Related to these single species models, work is ongoing to estimating "appropriate" weights on different likelihood components in Gadget, ultimately to obtain more refined confidence intervals by bootstraping the data. This work can potentially be extended to a multispecies Gadget model for Icelandic waters.

3.2 Ecoregion B: Barents Sea

3.2.1 Gadget models

The SYMBIOSES project to produce a linked series of models mentioned in the 2010 WGSAM report has received funding, and the project kickoff meeting was held on 3rd-4th October 2011. This model will couple the Gadget multispecies fish model to larval (LARMOD), plankton (SINMOD) and ecotoxicology models to estimate the potential impacts of oil-induced larval mortalities on the major fisheries in and around the Barents Sea. A first version of the tool is due by 2014.

3.2.2 Unquantifiable uncertainty in projecting stock response to climate change: Example from NEA cod modelled using the STOCOBAR model

Data from the years prior to 2005 suggests a positive relationship between recruitment of cod in the Barents Sea and the sea temperature at the Kola section during the year of spawning. However, analysis of subsequent data indicates that this relationship no longer holds. This change in the recruitment dynamics will clearly have an impact on our understanding of future stock dynamics and long-term yield. Such changes are likely in any prediction of response to future climate change that is based on historically derived (regression-based) recruitment relationships. This paper uses a multispecies “STOCOBAR” forward simulation model to evaluate the dynamics under a variety of climate scenarios and recruitment hypotheses, presenting the differences in modelled SSB and yields under temperature-dependent and temperature-independent recruitment situations. The divergence between the modelled populations and yields under the different recruitment hypotheses indicates the impossibility of predicting the future evolution of a stock with any degree of certainty, or even with any quantifiable degree of uncertainty. These results highlight the importance of having a management regime that is robust to unpredicted and unpredictable changes in stock dynamics, and the need for management strategy evaluations under a wide range of possible future scenarios.

3.2.3 ATLANTIS

Work is progressing to develop an ATLANTIS multispecies model for the Barents Sea. Planning work and initial parameterization has been completed, and the model

is almost ready for first full-scale test runs. This model covers the major benthic, planktonic, fish, seabird and marine mammal components of the Barents Sea ecosystem, as well has having a detailed three-dimensional spatial structure, and oceanographic inputs from the ROMS oceanography model.

3.2.4 DSF

A minimal model, The Dynamic Stochastic Food web (DSF) model, is under development in Tromsø. Marine ecosystem models are notoriously difficult to construct because of their complexity as well as the many unknowns concerning fundamental ecosystem processes. In the face of such difficulties, we adopt a simple approach based on a very limited set of constraints (mass balance, physiology and life-history traits) and stochasticity. Such a simple model is capable of mimicking a wide range of features observed in the Barents Sea. The Dynamic Stochastic Foodweb model (DSF) can serve as a reference against which other models can be tested and as an experimental device to test possible effects of environmental change on the Barents Sea ecosystem.

The model constraints include mass balance (i.e. the conservation of mass within the system), physiology (i.e. satiation: the maximum amount of food intake of a predator per year per unit biomass) and inertia (i.e. the maximum relative variation in biomass of a tropho-species per year). The first prototype of the model for the Barents Sea includes six tropho-species and the trophic interactions between them (Figure 2.2.1). Despite its extreme simplicity, the model can reproduce realistic time-series of fluctuations in biomass of individual trophospecies (Figure 2.2.2) as well as trophic relationships or diet composition (Figure 2.2.3). At the ecosystem level, the model also produces realistic fluctuations in trophic controls (top–down and bottom up oscillations), or apparent regime shifts. The DSF model shows that many of the properties that are observed in real ecosystems could simply result from a very minimal set of constraints. The model is under development to include additional features such as age-structured populations and multiple geographical units.

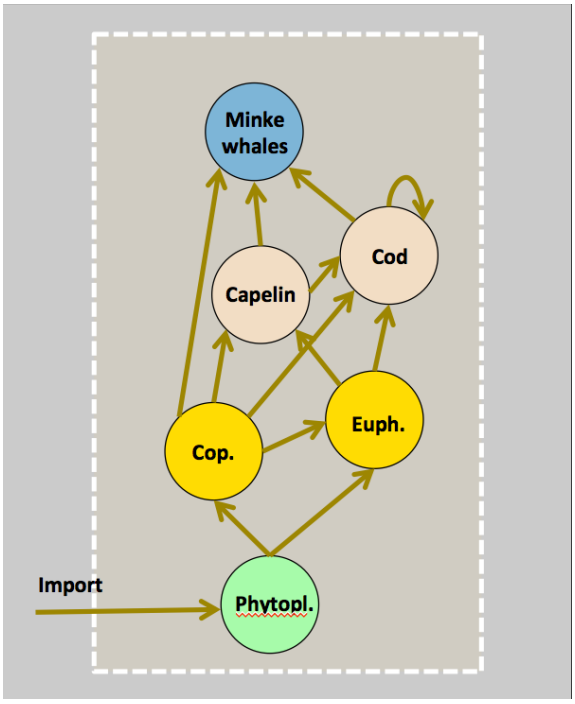


Figure 2.2.1.Tropho-species and trophic links of the DFS prototype for the Barents Sea.

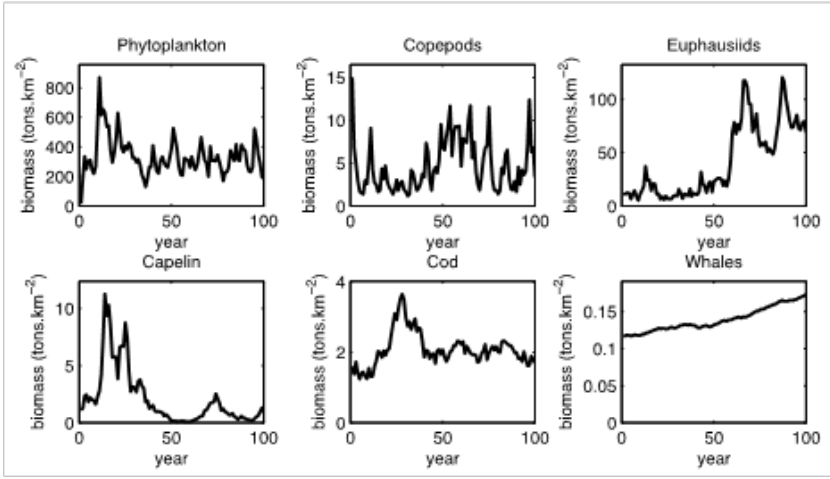


Figure 2.2.2. Temporal dynamics of the tropho-species included in the DSF.

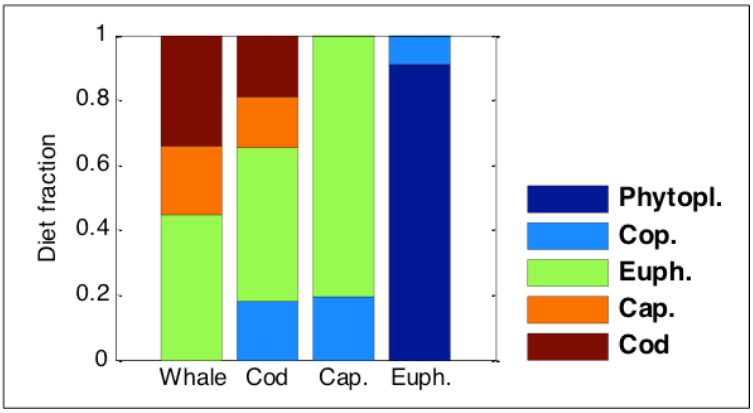


Figure 2.2.3. Modelled mean annual diet fraction of whales, cod, capelin and euphausiids.

3.3 Ecoregion C: Faroes

No updates were available in 2011.

3.4 Ecoregion D: Norwegian Sea

See Barents Sea for an update on ATLANTIS.

3.5 Ecoregion E: Celtic Seas

3.5.1 Ecopath in the Celtic Sea

Development of the Celtic Sea EwE model (by Cefas and University of Plymouth, UK) reported in WGSAM 2010, has focused on fitting model predictions to time series data and analyses of the effects of climate change and fisheries discard policy on seabird populations. The model description and its applications will be published in later 2011. Cefas will publish a technical report describing the model and its fit to time-series which will be made available in 2011/12.

3.6 Ecoregion F: North Sea

3.6.1 Ecopath with Ecosim

The North Sea EwE model (Mackinson and Daskalov, 2007; Mackinson *et al.*, 2009) has been applied to 3 published studies investigating the rebuilding of fish stocks (Worm *et al.*, 2010), the effects of subsidies on North Sea fisheries (Heymans *et al.*, 2011, see Figure 3.6.1) and the role of forage fish in marine ecosystems (Smith *et al.*, 2011, see Figure 3.6.2).

Development work since WGSAM 2010 has focused on calibration to time-series data 1991–2007, details of which are presented under ToR(b; Section 4) in this report, and coupling the foodweb to a physical-biogeochemical model (GOTM-ERSEM) so that future scenarios of climate change can be more adequately represented (see Beecham *et al*. submitted).

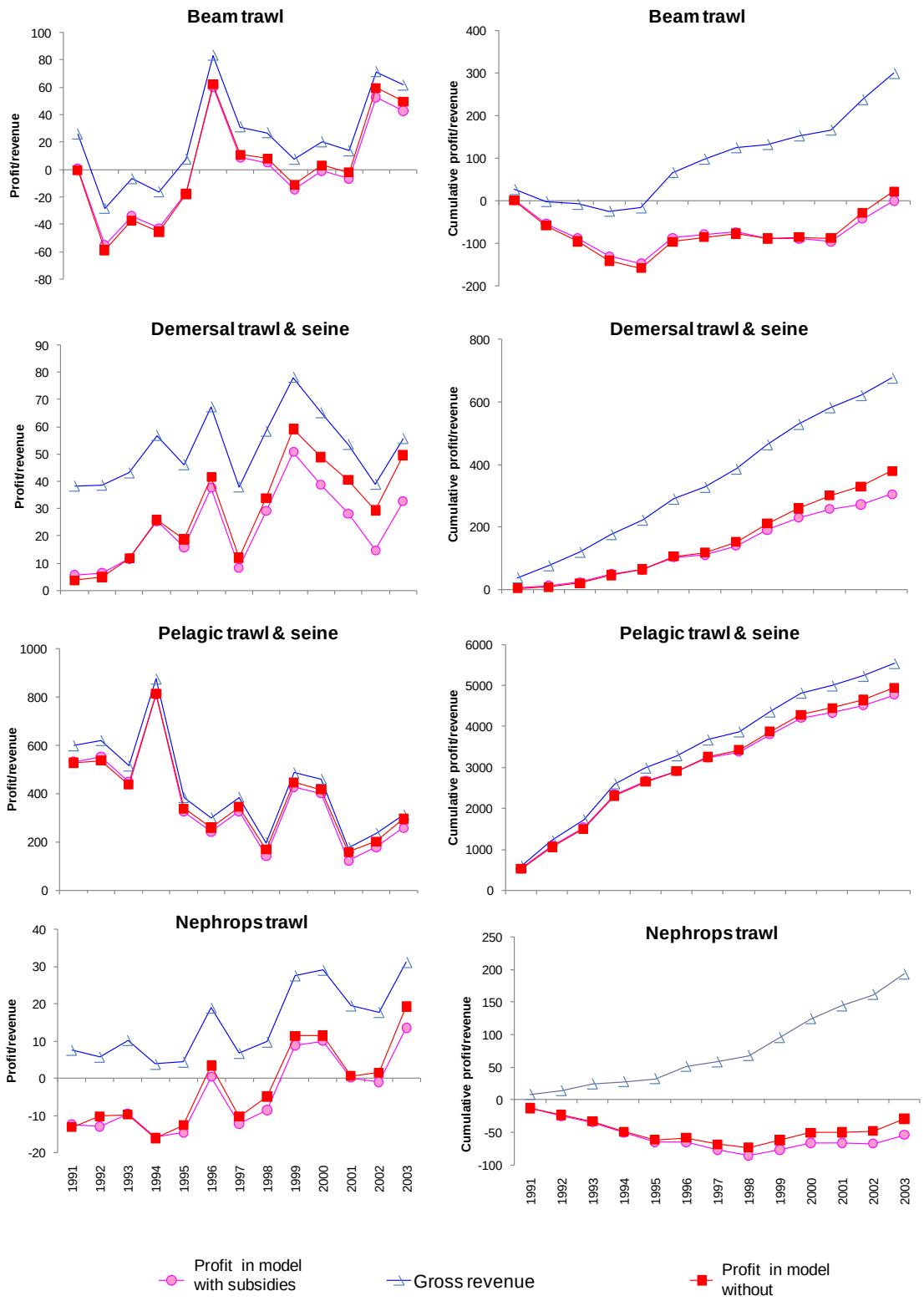


Figure 3.6.1. Profits, cumulative profit and revenue obtained with and without subsidies (in Euro million). Profits (pink) and gross revenue (blue) in the "with subsidies" model, pelagic trawl and seine fleet (2E and 2F) and the Nephrops trawlers (2G and 2H), with subsidies and profit when subsidies were removed from the model (red). All left hand figures show true values and right hand figures show cumulative values – all in Euro million (from Heymans *et al.*, 2011).

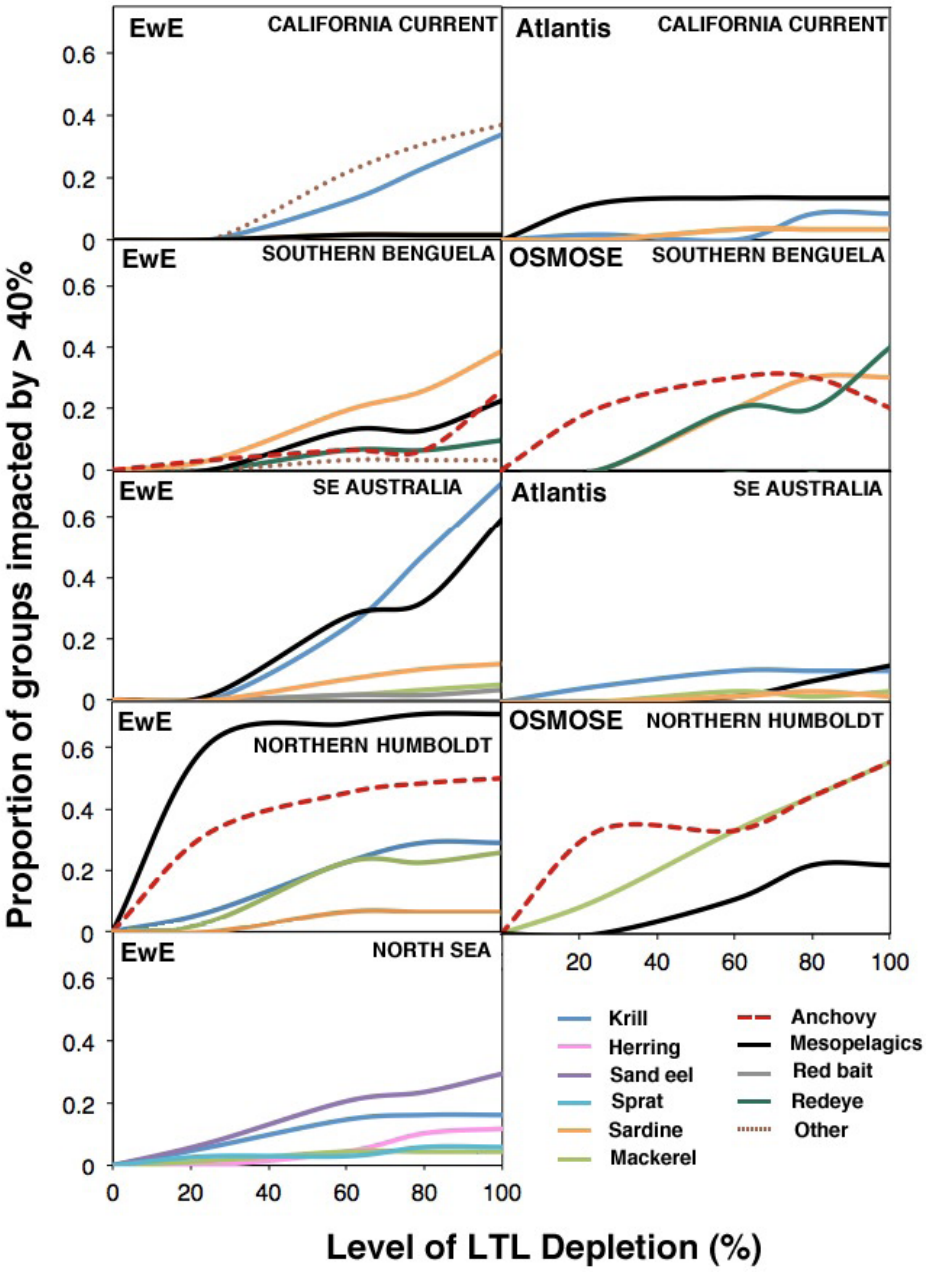


Figure 3.6.2. Impacts on other groups in the ecosystem at differing levels of depletion of forage fish caused by fishing. From Smith *et al.* (2011).

3.6.2 SMS and Estimation of consumption of top predators

Outputs of this modelling approach are treated in Sections 4.1 and 9.

3.6.3 Stochastic multispecies production model

A stochastic multispecies production (Schaeffer) model is being developed at DTU Aqua with the aim of deriving a simple model which can be used by external users with minimum training time and furthermore used to investigate the properties of multispecies production models when different observation datasets are used and differing assumptions are made regarding observation and process error. Preliminary results show very good fits to observations when parameters are estimated from biomass (estimated in multispecies SMS) and F (or catches) while attempts to estimate

parameters from survey indices and catch alone have highlighted the difficulty in estimating multispecies interaction parameters simultaneously with catchability parameters. Simulations indicate that fishing at a constant F will not ensure constant yield. Rather, yield will fluctuate over time due to the combined effects of process and observation error.

3.6.4 ATLANTIS model

Work on implementing Atlantis in the North Sea has just begun in the FP7 project VECTORS to analyse potential impacts of management measures on the ecosystem and economy. Specifically, the effects of installing wind parks in the North Sea will be investigated as well as various fishery closures and marine protected areas. Focus has until now has concerned the basic settings of the simulated areas (polygons) and basic parameterization will hopefully be finished in summer 2012.

3.7 Ecoregion G: South European Atlantic Shelf

3.7.1 Trophic data

The summer ecosystem survey annually conducted by AZTI-Tecnalia continued in 2011. Collected data are still being processed, but some preliminary results are available. As mentioned in the previous WGSAM report (ICES, 2010), one of the main tasks of this survey is to collect stomach content data and some general information related to the status of the ecosystem in Basque coastal waters. 3531 hake stomachs have been analysed since 2009 (individuals sized between 13–82 cm). Although most of the stomachs were empty (89%), it was found that blue whiting (*Micromesistius poutassou*) was almost 50% of the diet of hake, as found in similar studies carried out in the Cantabrian Sea (see Velasco, 2007). Other relevant species were horse mackerel (*Trachurus trachurus*, 17%), sardine (*Sardina pilchardus,* 11%) and anchovy (*Engraulis encrasicolus,* 10%), with anchovy being a very important prey species for the smaller individuals (juveniles) and horse mackerel for adults, bigger than 40cm. The cannibalism ratio was quite low compared to other studies executed in the Cantabrian Sea (Velasco, 2007) and the Bay of Biscay (Mahe *et al.*, 2007), being only 5% of the total diet and appearing only for adult individuals (> 40cm). In addition, several tuna stomachs have been analysed by AZTI-Tecnalia in the last years.

The annual autumn demersal survey was also conducted by the Instituto Español de Oceanografía in 2011 along the Cantabrian Sea and Galician waters. A total of 10073 stomachs of different demersal species were analysed this year (*M. merluccius, M. poutassou, L. boscii, C. conger*, etc.). The results of the analysis will be available to be used for multispecies modelling purposes in the Bay of Biscay and Iberian Peninsula.

3.7.2 A Coupled ECOROMS+APECOSM model

A project is underway to assess the capability of end-to-end models to capture the past evolution of the Bay of Biscay ecosystem and the potential impact of climate variability and critical drivers in future. In the case of Bay of Biscay, the main driver of the ecosystem is fishing.

APECOSM uses the lower-trophic-level results from the coupled physical-biogeochemical model ROMS-NPZD. The hindcast simulation of ROMS-NPZD covers the period 1998–2009, using NCEP re-analysis forcings and hydrological forcing from 20 rivers. This hindcast simulation has been validated using satellite images, climatologies and *in situ* data for temperature, salinity and chlorophyll-*a* biomass.

The model reproduces the seasonal evolution and vertical distribution of the temperature and salinity, as local and mesoscale processes. The simulated spring bloom of chlorophyll-*a* is about 1 month too early in the season, compared with satellite images, but the model captures the interannual variability (timing and intensity). The simulated slope current has been validated with ADCP data in 2008 and 2009.

The full "online" code of the ROMS-PISCES-APECOSM model is not yet available, and the version of APECOSM with open-boundaries usable for the Bay of Biscay is only available since September 2011. So, the APECOSM model is in implementation but no simulations have yet been performed.

3.8 Ecoregion H: Western Mediterranean Sea

No updates were available in 2011.

3.9 Ecoregion I: Adriatic–Ionian Seas

No updates were available in 2011.

3.10 Ecoregion J: Aegean–Levantine

No updates were available in 2011.

3.11 Ecoregion K: Oceanic Northeast Atlantic

No updates were available in 2011.

3.12 Ecoregion L: Baltic Sea

3.12.1 Biological ensemble modelling of climate impacts for the Eastern Baltic Sea

An ecosystem-based approach to fisheries management implies that management should explicitly account for interactions among species and other ecosystem processes. Thus, diversity and complexity of models used for predicting fish stock responses to management have increased. Yet, the structural uncertainty associated with alternative models is rarely accounted for. Here we present the biological ensemble modelling approach (BEMA, Gårdmark *et al.*, in prep.; ICES, 2010) to deal with such structural uncertainty.

Four single-species models, four multispecies models and one foodweb model were used to predict the response of Eastern Baltic cod (*Gadus morhua morhua*) to five alternative fisheries management scenarios and two climate change scenarios, assuming no climate change or a warmer and less saline future Baltic Sea. Although predictions differed qualitatively between the models, the BEMA provided a means to (i) present the full set of projected stock responses, (ii) assess whether these imply different conclusions on management, and (iii) draw general conclusions valid across all models used.

Table 3.12.1. Models used in the intersessional work on BEMA (for more details, also on the climatic scenarios, see ICES, 2009, Gårdmark *et al.*, in prep.).

1	Stochastic Cod model	Wikström *et al.* in prep.	Auto-regressive (AR(1)) model of total cod biomass.
2	MCMC cod long-term projection model	Aro, E. ICES, 2008b.	Modified ICES medium-term projection model, age-structured cod
3	Cod mini-model	Müller-Karulis, in prep.	Age-structured cod model, similar to medium-term prediction models for Baltic herring stocks
4	Dynamic cod-herring-sprat model	Heikinheimo, in prep.	Age-structured cod, sprat, herring model including cod predation, modified from MSVPA
5	SMS (stochastic multispecies model)	Neuenfeldt *et al.* in prep.; Lewy and Vinther (2004)	Age-structured cod, sprat, herring model including cod predation and cannibalism, with size-based diet parameterization
6	Stage-structured multispecies model	Van Leeuwen *et al.* (2008)	Size-structured cod and sprat, with 2 zooplankton and 1 zoobenthic resources, including cod predation and resource-dependent growth of cod and sprat
7	BALMAR	Lindegren *et al.* (2009)	Multivariate autoregressive (MAR(1)) model of total biomass of cod, sprat, herring, Pseudocalanus, including cod predation, negative effect of sprat on herring and cod
8	Baltic NEST EwE foodweb model	Tomczak *et al.*, submited.	Ecopath/Ecosim model of age-structured cod, sprat and herring, and total biomass of foodweb components on 7 trophic levels (incl. plankton groups, benthic groups and seals).

For the Eastern Baltic cod example, it was found that no recovery of the stock will occur if fishing returns to mean levels of 1996–2005, but that the stock will recover if the harvest follows the levels outlined in the management plan (even under climate change). The Biological Ensemble Modelling Approach (BEMA) has proven to be useful for collating and comparing possible future population developments, and for providing and communicating the range of projected outcomes. By identifying critical uncertainties, knowledge gaps and thereby structural causes of model ensemble variability, it is possible to focus the collection of field or experimental data and need for further model development, e.g. interactions, feedbacks and improved S-R models.

3.12.2 EwE models for Baltic Sea

To evaluate interactions between fisheries and the foodweb from 1974 to 2000, Harvey *et al*. (2003) created a foodweb model for the Baltic Sea proper, using EwE. Model parameters were derived mainly from multispecies virtual population analysis (MSVPA). Ecosim outputs closely reproduced MSVPA biomass estimates and catch data for sprat (*Sprattus sprattus*), herring (*Clupea harengus*), and cod (*Gadus morhua*), but only after making adjustments to cod recruitment, to vulnerability to predation of

specific species, and to foraging times. Cod was shown to exhibit top–down control on sprat biomass, but had little influence on herring. Fishing, the main source of mortality for cod and herring, and cod reproduction, as driven by oceanographic conditions as well as unexplained variability, were also key structuring forces. The model generated many hypotheses about relationships between key biota in the Baltic Seafoodweb and may ultimately provide a basis for estimating community responses to management actions.

For five Baltic coastal ecosystems (Puck Bay, Curonian Lagoon, Lithuanian Open Baltic coast, Gulf of Riga coast and Pärnu Bay) Ecopath models have been built to investigate trophic networks and carbon flows (Tomczak *et al*., 2009). Authors compared the models using 12 common functional groups. The studied systems ranged from the hypertrophic Curonian Lagoon to the mesotrophic Gulf of Riga coast. Interestingly, authors found that macrophytes were not consumed by grazers, but rather channelled into the detritus food chain. In all ecosystems fisheries had far reaching impacts on their target species and on the foodweb in general.

A number of additional EwE activities are underway in the Baltic area covering mainly small-scale ecosystems and coastal areas (e.g. Lithuanian coast – open Baltic and Curonian lagoon, CORPI; Gulf of Riga, LATFRI; Gdańsk basin, NMFRI). Furthermore, an Ecopath and Ecosim model for the Kattegat (Lindgren and Tomczak – work in progress) is in the construction phase. A preliminary version should be ready by the end of 2011.

The current BaltProWeb (Baltic Proper Food-Web model - NEST Ecopath with Ecosim model; Tomczak *at al., submitted to Ecological modelling*, Tomczak *et al.*, 2011) covers the area of the Central Baltic Sea (ICES SD 25–29 excluding Gulf of Riga) and contains 22 functional groups (Figure 3.12.1).

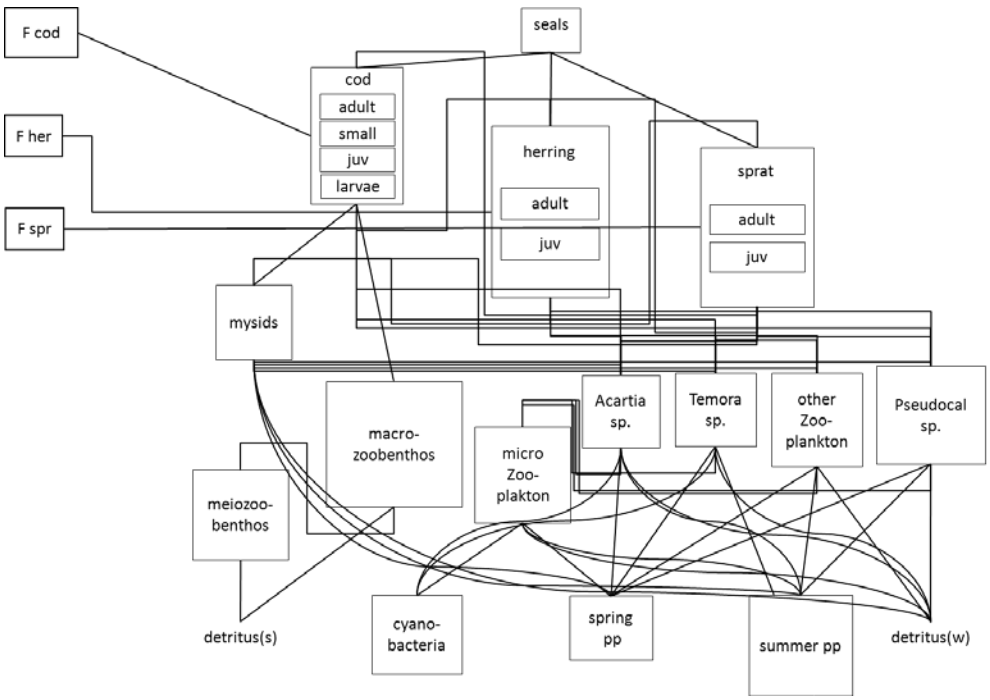


Figure 3.12.1. BaltProWeb model structure.

The model has been created based on extensive databases and literature. Cod, herring and sprat are split into multi-stanza groups to represent the main ontogenetic changes and shifts in diets (Stanza represent: Juvenile Cod – Age 1,2; Small Cod – Age 2,3, Adult Cod – Age 4+; Juv. Herring – Age 0–2, Adult Herring - Age 3+; Juvenile Sprat – Age 0,1, Adult Sprat – Age 2+). The mezo-zooplankton community is split into functional groups representing the three main species related groups and one group that combines other zooplankton components. Fisheries are represented by three fleets fishing on the main fish species; however current work is focused on a re-evaluation of this categorization. The mass-balanced model represents the state of the ecosystem in the middle of the 1970's and 1974 has been chosen as a baseline for the temporal Ecosim simulations. To fit and drive the Ecosim model, time-series of biomasses (fish, benthos, mysids, zooplankton, chl-a), catches (all fish species), fishing mortalities (all fish species) and environmental drivers (Temperature, Cod Reproductive Volume, Hypoxia) have been tested. Dynamics of the pelagic foodweb are described and the model fitted relatively well to the data; however the benthic part of foodweb needs to be improved.

The study "Challenges and unknowns in environmental change studies – a foodweb model sensitivity analysis" by Niiranen *et al.*, (*submitted to Ambio*) aims to highlight the importance of model sensitivity analysis in exploring uncertainty in model parameterization when working with foodweb models. Currently in the field of ecosystem modelling uncertainties and model sensitivity are seldom addressed. Authors highlight that especially when such models are used to project future environmental change the uncertainties can grow large and such analysis cannot be ignored. In this study authors have carried out a sensitivity analysis using a new Baltic Sea Ecopath with Ecosim (EwE) foodweb model (BaltProWeb). Results from the sensitivity analysis demonstrate that model uncertainties, if realized, can in extreme cases lead to different requirements from future ecosystem management than the original model results indicate.

A key-run for the Baltic Sea, based on BaltProWeb is under development but is not presented in the current WGSAM report; however a comparison with the prior Baltic SMS key run has been performed and is included here. Predation mortality (M2) and average annual biomasses (B_{bar}) were compared with the SMS key-run (ICES, 2009; Figure 3.12.2 and Figure 3.12.3). In contrast to SMS, predation mortality on adult cod, by seals, was included in BaltProWeb.

The M2 value estimates show significant differences between SMS and BaltProWeb (adult cod, small cod, juv. cod, juv. herring) as well as the level of M2 for all groups (Figure 3.12.4). Median M2 estimates from EwE were in most cases higher than SMS outputs, except for Adult Sprat (Figure 3.12.5). These differences could be partly explained by the inclusion in BaltProWeb of seals predation on clupeids, in addition to of cod predation.

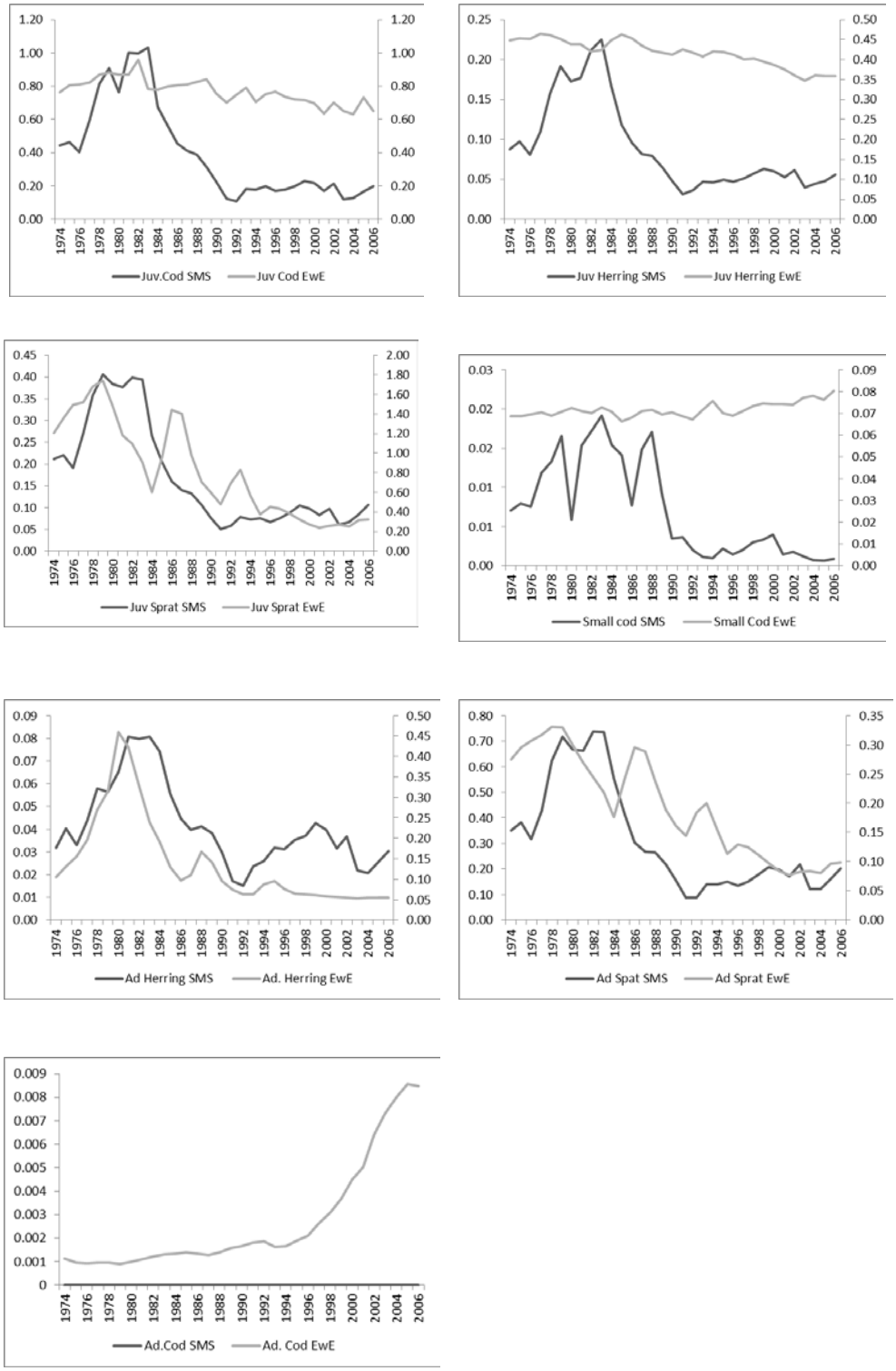


Figure. 3.12.2 Predation mortality coefficient (M2) over time from Baltic SMS (left y-axis) and BaltProWeb (right y-axis).

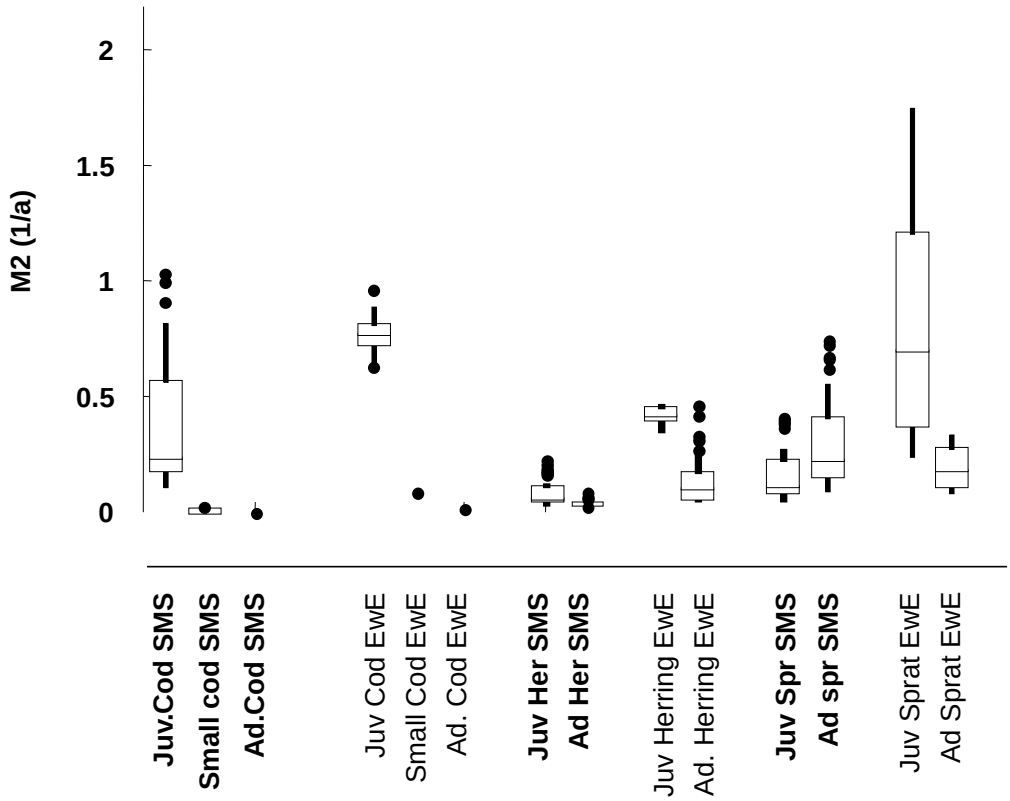
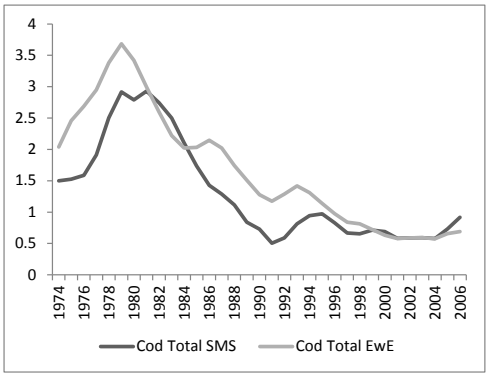
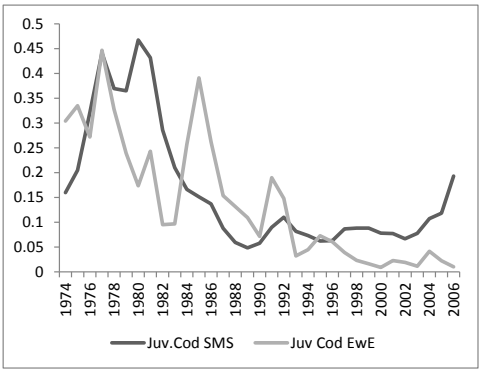
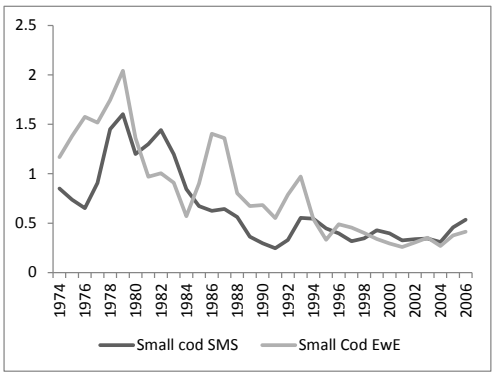


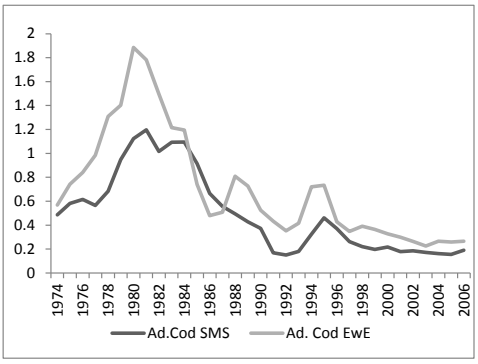
Figure 3.12.3. Predation mortality coefficient M2 comparison for time period 1974–2006. Line in the box indicate median, box - 25% and 75% quintiles, whiskers maximum and minimum values.

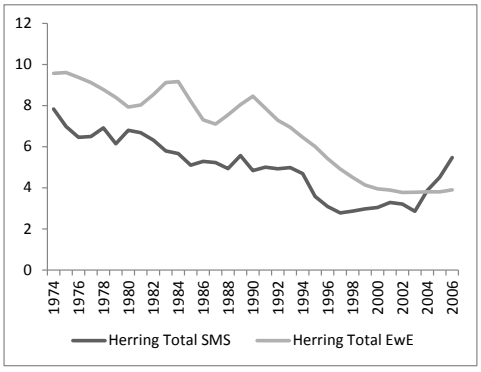
Average annual biomasses of the main fish groups show similar dynamic trends and levels; however there are some differences especially for clupeid groups. Biomasses of cod groups show similar trends as well as medians at similar levels for the time period 1974–2006. Comparison indicates the highest differences for both (juvenile and adults) sprat groups. As discussed in Tomczak *et al.* (submitted) the differences could be caused by several factors, including model limitation (constant v over time) or initial biomass conditions of biomass and production of sprat group.

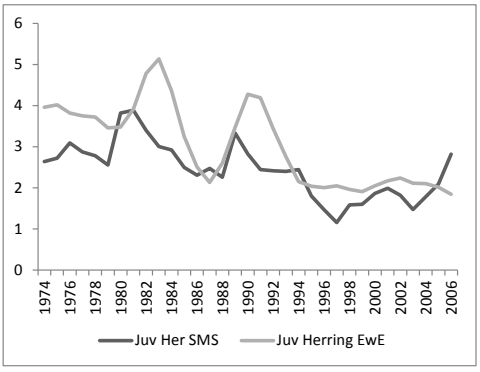

Cod Total SMS
Cod Total EwE

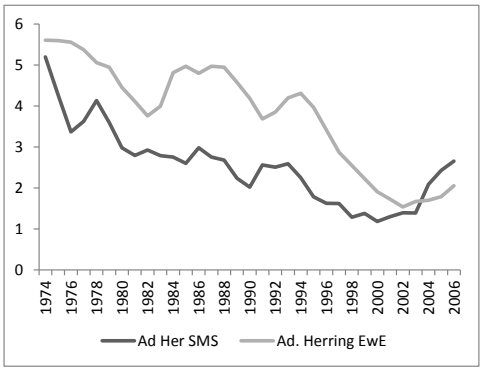

Juv.Cod SMS
Juv Cod EwE

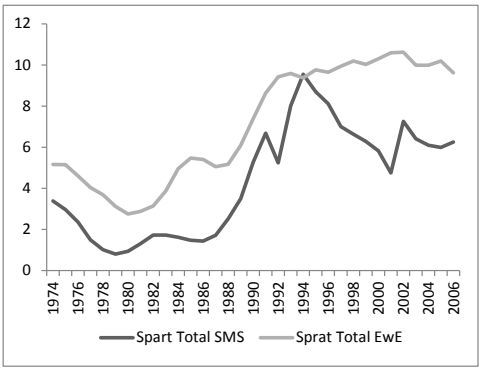

Small cod SMS
Small Cod EwE


Ad.Cod SMS
Ad. Cod EwE


Herring Total SMS
Herring Total EwE


Juv Her SMS
Juv Herring EwE


Ad Her SMS
Ad. Herring EwE


Spart Total SMS
Sprat Total EwE

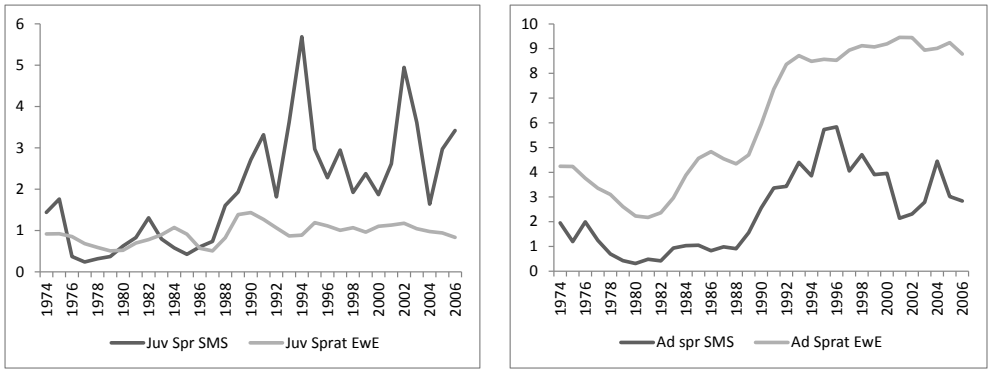


Figure 3.12.4. Comparison of average annual biomass (t/km²) dynamic over time of main fish groups from SMS and BaltProWeb-Ewe.

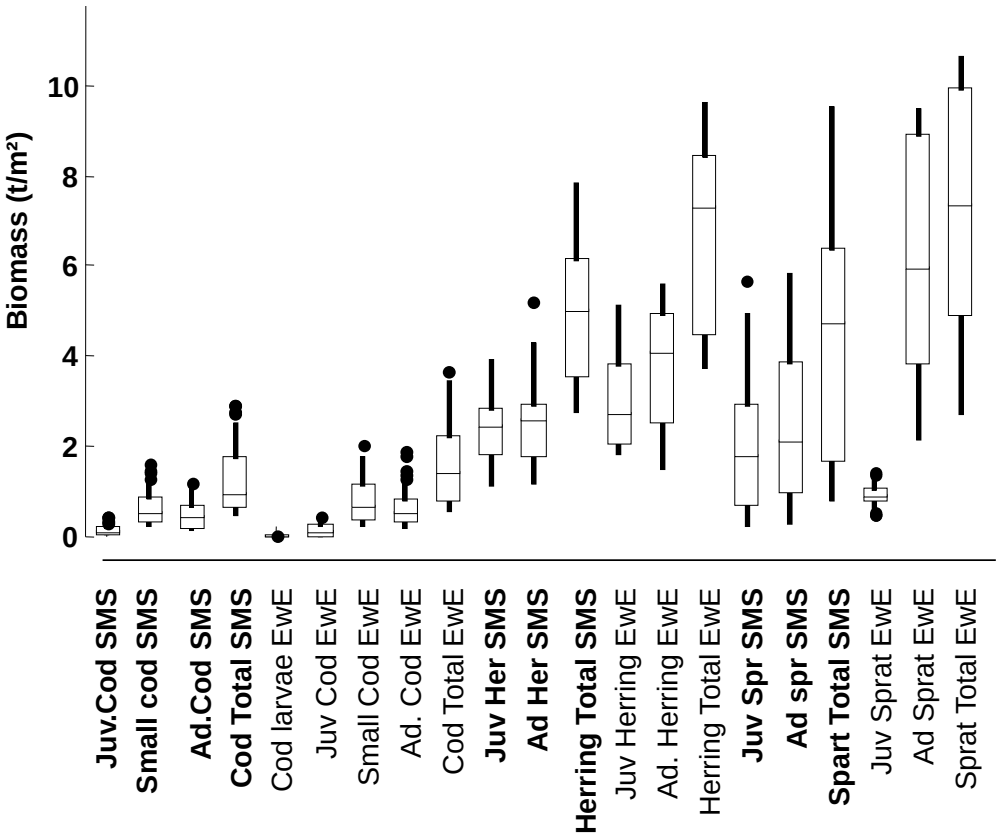


Figure 3.12.5. Median biomass comparison for time period 1974–2006. Line in the box indicate median, box - 25% and 75% quintiles, whiskers maximum and minimum values.

3.13 Ecoregion M: Black Sea

No updates were available in 2011.

3.14 Ecoregion: Canadian Northwest Atlantic

No updates were available in 2011.

3.15 Ecoregion: US Northwest Atlantic

3.15.1 Surplus Production Modelling

Two workshops (funded by the Comparative Analysis of Marine Ecosystems, CAMEO) were held in Woods Hole, MA with the goal of exploring the drivers that delineate the productivity of marine ecosystems, using surplus production models as a unifying analysis tool. A vast literature identifies three main processes that regulate the production dynamics of fisheries. These are collectively referred to as the triad of drivers, and are: biophysical, exploitative, and trophodynamic processes (Figure 3.15.1). There were 13 northern hemisphere ecosystems from Canada, the US, and Norway included in the analyses which used different levels of aggregations, and various drivers (including several ecological and environmental covariates) to make comparisons across ecosystems.

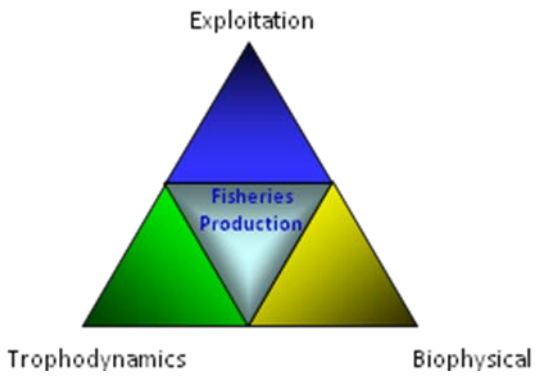


Figure 3.15.1. The triad of drivers that influence fisheries production in ecosystems.

Distinct, but complementary, information was provided in analyses that applied surplus production models at single-species (SS), multispecies (MS), aggregated species, and full-system levels across the 13 ecosystems.

The comparative modelling and empirical work resulted in the identification of key emergent trends and common patterns governing fishery productivity in Northern hemisphere temperate marine ecosystems. Holsman *et al.* (submitted) applied models at the single species level of organization, comparing both cod and herring production across ecosystems. Lucey *et al.* (submitted) applied models at intermediate levels of organization, including habitat-based groups (demersal and pelagic), size-based groups (small, medium, and large average adult size), and trophic functional groups (planktivores, zoopivores—shrimp-feeders, piscivores, and benthivores). Bundy *et al.* (submitted) applied models at the full system level. In nearly all cases, a production modelling approach provided better fits to the data than the null model across all levels of aggregation. A striking result of all three studies was the general similarity of estimated BRPs across ecosystems at each level of organization. In the two studies where they were included, environmental covariates specific to each ecosystem generally improved fits to the production models at both the single species and full ecosystem levels (Bundy *et al.* submitted, Holsman *et al.* submitted). It was confirmed

that full-system yield in any given ecosystem generally is less than the sum of SS yields – a result seen in a suite of other empirical and modelling studies.

3.15.2 Using Surplus Production Models as Operating Models in Management Strategy Evaluation

Two MSE examples have been developed in the North Pacific and Northeast United States Large Marine Ecosystems using a multispecies surplus production simulation model (MS-PROD) which incorporates predation and competitive interactions between species. The first usage of MS-PROD in an MSE context was to explore the effects of managing species by aggregate groups, specifically if certain aggregations are likely to result in stock collapses. The approach was to generate two "cartoon" ecosystems with MS-PROD based on the Gulf of Alaska (GoA) and Georges Bank (GB) ecosystems, each comprised of 10 representative species (Gaichas *et al.* submitted, Gamble and Link, submitted). These generated ecosystems were then considered to be 'the truth'. GoA was parameterized to emphasize predation while GB was parameterized to emphasize competition. Different levels of aggregation were then applied to each ecosystem: full system, taxonomic (groundfish, flatfish, small pelagic, elasmobranchs), habitat (demersal/pelagic), feeding guilds (benthivores, piscivores, planktivores), and size (small, medium, large), and no aggregation. Several biological reference points (BRPs; e.g. MSY and F_{MSY}) were estimated as production model parameters based on each level of aggregation, and then simulations were run using the estimated F_{MSY} BRPs to compare the equilibrium biomass and yield among the alternate fishing strategies. One of the main results from this study was that if stocks with very different productivities were aggregated, the less productive species were at risk of collapse under aggregate fishing strategies. However, if similarly productive species were aggregated, there was little loss of yield compared to a single species strategy and the equilibrium biomasses were also comparable. Additionally, it was shown that in most cases where stocks levels fell below 25% of unfished biomass, reducing F to the level where no stocks were below 25% of unfished biomass usually resulted in yield levels 80% or greater of aggregate MSY. The only exception was for cases where a species was much less productive than the others in the aggregate (e.g. Pacific ocean perch in GoA and mackerel in GB). This suggests that multi-objective, multi-species reference points can be created by combining a minimum biomass threshold with aggregate species yield (Figure 3.15.2), and that the yield trade-offs associated with managing for multiple objectives may not be as severe as previously thought.

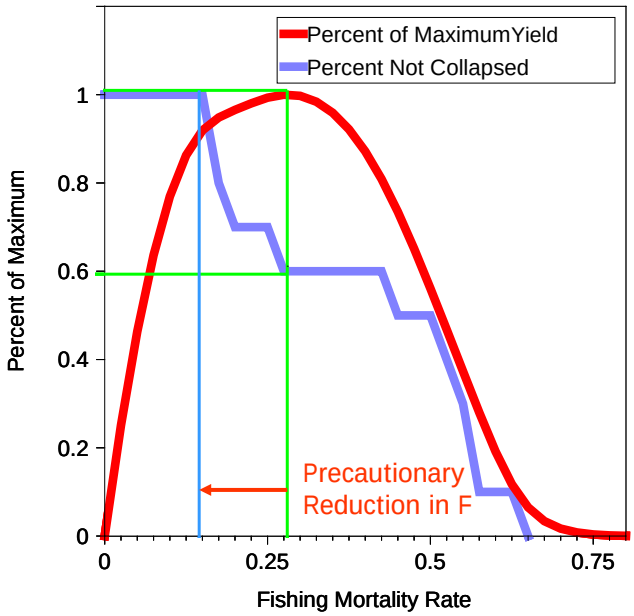


Figure 3.15.2. An example of how to create multi-objective reference points using a yield curve overlaid with the proportion of stocks not collapsed. In this case, 60% of stocks collapse at MSY, but by reducing F, it's possible to achieve ~90% of maximum yield while preventing any stock from collapsing.

The second usage of MS-PROD as an operating model in an MSE was the incorporation of simple climate effects (Gamble and Link, submitted). A simple modification of the NEUS MS-PROD model (Gamble and Link 2009) explored the effect of simulating climate effects by decreasing the intrinsic rate of growth for the stocks that make up the groundfish functional group by 10% based on other studies in the region which show that many groundfish stocks have been affected by climate change (Nye *et al.*, 2009). While very preliminary, the results indicate that even this simple modification can lead to complex and non-intuitive effects such as some groundfish stocks increasing in biomass compared to the non-climate scenario although they had their intrinsic rate of growth reduced by 10%. These results were primarily due to the trophodynamic interactions between species; specifically competitive release of other groundfish that were negatively affected by the climate scenario. Additionally, the removals of biomass (from predation, competition, harvest, and climate) can all be easily calculated to provide information on the relative importance of each driver on the system, aggregate groups, and individual stocks.

3.15.3 ATLANTIS NEUS Removal Scenarios

Large marine ecosystems are complicated, with a large amount of connectivity (via trophic interactions) between ecological components and their environment. Ecosystem-based fishery management (EBFM) is moving from asking *why* one would do EBFM towards asking *how* to do EBFM and recognition of the importance of these connections has increased as a result. Studies have shown that commercially important fish stocks may migrate out of regional areas and management units, but little research on the likely effects on the biotic community following such a shift has been done. Additionally it has been hypothesized that the lack of recovery of some commercially important species, despite significant management actions designed to allow them to do so, could be related to predation by or competition with other, less commercially desirable species. Again, little has been done to explore what the effects

of removals of these less desirable species might have on the species of interest, a community, or even the entire ecosystem.

Scenarios where individual species or groups are removed from a system, thus simulating either migration out of an area or targeted removals to achieve a management goal, were carried out in ATLANTIS NEUS (Link *et al.,* 2011) in order to elucidate some of the probable community or ecosystem responses following such removals. The species removed from the model were spiny dogfish, Atlantic cod, Atlantic herring, and seabirds. The final biomass of every functional group in the model was then compared to the base scenario to determine the effect of the removal. The results indicated that removals of species or groups from an ecosystem are unlikely to have simple effects due to the complexity of the interactions between species and their environment. Removals of spiny dogfish – often proposed to help the recovery of groundfish in the Northeast US large marine ecosystem– primarily resulted in predatory release on anadromous small pelagic fish and shrimp, with minor effects (0–10% change in biomass) on the other groundfish in the system. Removing Atlantic cod resulted in minor to moderate effects on the other groundfish in the system, primarily through release of competitive effects, and the behaviour of the fisheries in relation to the change in ecosystem structure. Other functional groups were less affected in this removal scenario. Removing Atlantic herring affected both upper and lower trophic levels, primarily causing: other pelagics to increase in biomass, shrimp to decrease in biomass, complicated effects in the phytoplankton and zooplankton groups, and a decrease in baleen whales. Finally, the removal of seabirds led to a less than 1% change in any group in the model. To generalize, the propensity for indirect effects and unanticipated consequences is high in these scenarios; and therefore the use of such ecosystem models to explore the full range of options is recommended to bound the scope of possible responses for the implementation of EBFM.

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4 ToR b) Report on the development of key-runs (standardized model runs updated with recent data, and agreed upon by WGSAM participants) of multispecies and eco-system models for different ICES regions (including the North Sea, Baltic Sea, Barents Sea, Bay of Biscay and others as appropriate)

4.1 North Sea SMS

4.1.1 Overview

The key run for the North Sea is produced with the SMS model. SMS (Lewy and Vinther, 2004) is a stock assessment model including biological interaction estimated from a parameterized 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, sandeel, Norway pout), predator only (saithe), no predator prey interactions (sole and plaice) and 'external predators' (8 seabirds, starry ray, grey gurnard, western mackerel, North Sea mackerel, North Sea horse-mackerel, western horse-mackerel, grey seals and harbour porpoise (Figure 4.1.1)). The population dynamics of all species except 'external predators' were estimated within the model.

Due to problems in the assessment of North Sea sprat (ICES, 2004, 2006b) and fitting problems for this stock inside SMS, it was decided to leave this prey species out in the 2008 key run. Since then, the acoustic survey series has become long enough to be of use as a cpue time-series and the internal consistency of the acoustic survey and the IBTS surveys has been confirmed by the HAWG to be reasonable (WD presented in HAWG 2011). As sprat is an important prey species in the model, it was therefore decided to include this species in the current key run.

Grey seals and harbour porpoise were not included in the 2008 key run. Since then, population numbers of seals have been updated as part of the FACTS project and diet of harbour porpoise has been made available in the DEFINEIT project. As reliable data were now available, these two species were included as additional external predators.

A further update concerned the use of the statistical distribution used to estimate the likelihood of the stomach content distribution. If the 2008 key run, the lognormal distribution was used whereas the Dirichlet distribution was chosen this year due to the fact that this distribution is theoretically more appropriate to modelling fractions (as diet composition).

The start year of the time-series used was reviewed by the group. It was considered that the data prior to 1974 were highly uncertain for some of the forage fish species, as catch-at-age data are not available for the sixties and the start of the seventies. However, including them had no discernible effect on the estimates provided for the period from 1974 onwards. It was decided to keep the data in the model but to give a word of caution that **the quality of these data are far from as good as the quality of the later time-series**.

4.1.2 Input data update

Catch-at-age (including discards), weight in the catch and in the stock were extracted from the 2011 assessment reports of WGNSSK, HAWG and WGWIDE. The assumptions used in official single species assessments were reproduced as much as possible. For cod the scaling factors for unallocated removals were taken into account by multiplying the input catch numbers-at-age with the respective factors estimated in the latest single species (SAM) assessment. In general, the survey time-series were identical with the ones used in single species assessments. Only for the 0-groups of some species and early years in the time-series additional surveys (e.g. ENGFS, SCOGFS) were utilized.

The working group reports unfortunately no longer provide information on the quarterly distribution of catches. To estimate these proportions, the average proportion of the catch taken in each quarter was calculated for the years 1972 to 2003 where quarterly catch data were available without discard and unallocated landings. However, in many cases data have been revised to include discard and unallocated landings and these revisions have not been accompanied by revised estimates of the proportion of the catch taken in each quarter. Using the average proportions from the years 1972–2003 corresponds to using the same discard percentage in all quarter. While this is probably not correct, the working group was unable to provide better estimates without new information on the quarterly distribution of catches, discards and unallocated landings.

4.1.2.1 Proportion of mackerel and horse mackerel stocks present in the North Sea

Historically, information on the proportion of the mackerel and horse mackerel stock which was inside the North Sea has been provided by the relevant working groups. However, in later years updated information has not been available and in 2007 it was decided not to continue the acoustic survey of mackerel due to large variation in the measurements. The quarterly proportions of the Western stocks in the North Sea were therefore assumed to be constant in the last 24 years. With the assumed proportions and the official assessment results the numbers of mackerel and horse mackerel in the North Sea could be estimated for each quarter and year.

4.1.2.2 Data on seabirds

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. For the current study, the ESAS database version 4.1 (as of September 2004) was used. The ESAS database 4.1 contains 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.

4.1.2.3 Data on grey seals

Seal diet data were derived from scats was sampled in 1985 and 2002 at haulout sites around the UK coast. An aggregated estimate of grey seal diet composition based on all these 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.

Population numbers are derived from seal counts at haulout sites in the North Sea and Orkney for the period 1984 to 2009. Populations prior to 1984 are estimated assuming exponential growth in the period up to 1990 (using 1984–1990 to estimate parameters). For 2010, the value in the last year is used as populations are assumed to be levelling off.

4.1.2.4 Data on harbour porpoise

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). Daily consumption was set to 2.4 kg (Sophie Smout, University of St Andrews, pers. Comm.). 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 4.1.1). 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 do as such have the full size range of fish sizes. Stomach data from each decade were assigned to years, 1985, 1995 and 2005 respectively.

Table 4.1.1. 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

No correction for differences in evacuation times between prey were applied. However, the effect of underrepresenting sandeel on the consumption of gadoids was investigated. If sandeel underrepresentation was as high as 20 times (only 1/20 of the otoliths eaten recorded), there was a large effect on consumption of sandeel but a much smaller effect on the consumption of other prey.

4.1.3 Modelling size preferences

The SMS model has two options for size preferences of predators: either prey is 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 (lognormal distribution). 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, Fs 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.

It was considered by the group 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 in the 2007 key run.

4.1.4 Modelling sprat

Sprat as prey has previously been considered as part of "other food", as it was not possible to estimate reliable stock size within the MSVPA, 4M and SMS. However, there have been large fluctuations in the sprat stock size and in its main predators, mackerel and whiting; such that it might be beneficial have sprat as a SMS species. The new stock estimate made by SMS is probably highly uncertain, but it seems better than a constant biomass as assumed for other food.

Input catch numbers and mean weights at age for sprat 1996–2010 were copied from HAWG (ICES CM 2011/ACOM:06). Data from the period before 1963–1995 were copied from the dataset used in the 2001 North Sea key run (SGEEMI, 2001). For this dataset, data for the period before the mid-seventies are considered very uncertain. Annual catch-at-age in the period 1 July to 30 June for 1967–1977 are given in ICES 1978/H:3, but no international data exist for the period 1963–1966 and these therefore

had to be guesstimated form the total annual landings and assumptions of age distribution.

Fisheries independent data were available from the IBTS Quarter 1 (1977–2011), IBTS quarter 3 (1991–2010) and the summer North Sea acoustic survey (2005–2010; (ICES CM 2011/ACOM:06).

Sprat single species assessment (no biological interaction) using the updated dataset showed a recruitment and SSB level for the period 1963–1970 in the range of what were estimated for the most recent 10 years. However, in the multispecies assessment, the recruitment and SSB at the beginning of the time-series were estimated to be in the same very high range as estimated in the period 1970–1980 with very large sprat catches and low herring stock size. The WG group considered that this high recruitment was unlikely given the large mackerel stock followed by the gadoid outburst (predation), the large herring stock in that period (competition) and the very limited sprat fishery. It was therefore decided to fix the recruitment of sprat in the period 1963–1969 to a value of 100 million, which is close to half of the long-term recruitment estimated from the multispecies assessment.

Key run summary sheet

Area	North Sea
Model name	SMS
Type of model	Age–length structured statistical estimation model
Run year	2011
Predatory species	Assessed species: Cod, haddock, saithe, whiting Species with given population size: North Sea mackerel, western mackerel, North Sea horse mackerel, western horse mackerel, grey gurnard, starry ray, fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin, razorbill, grey seal, harbour porpoise
Prey species	Cod, haddock, herring, Norway pout, sandeel, sprat, whiting
Time range	1963–2010. Model output for 1963–1973 is considered less reliable due to poor quality of catch data of mainly the forage fish before 1974.
Time-step	Quarterly
Area structure	North Sea
Stomach data	Fish species: 1981, 1985, 1986, 1987, 1991 Grey seals: 1985, 2002 Harbour porpoise: Decadal 1985, 1995, 2005
Purpose of key run	Making historic data on natural mortality available
Model changes since last key run	Inclusion of sprat, grey seal and harbour porpoise
Output available at	http://info.ices.dk/reports/SSGSUE/2011/WGSAM_SMS_summary.csv
Further details in	Report of the Working Group on Multispecies Assessment Methods 2011

4.1.5 Results

The population dynamics of all species except 'external predators' were estimated within the model. However recruitment of sprat, 1963–1969, was given as input. The key-run converged and the uncertainties of parameters and key output variables were obtained from the inverse Hessian matrix. A summary of results is provided in Figure 4.1.2.

The input and output from the model are available online as ASCII file downloads from http://info.ices.dk/reports/SSGSUE/2011/WGSAM_SMS_summary.csv

4.1.6 Comparison with the 2008 key run

Estimated stock sizes and fishing mortalities for the 2008 and the 2011 are pretty similar for most of the species except for cod and whiting (Figure 4.1.3). For these two species, the changed perception of the stocks is mainly due to the inclusion of the marine mammals in the 2011 key run. For whiting, the inclusion of harbour porpoise in the 2011 key run has increased the predation mortalities of age 2+ by almost one third, which leads to a higher recruitment, higher SSB and a lower F in the 2011 key run. For cod, the biomass eaten by harbour porpoise and grey seals comprises more than half of total biomass removed in the most recent years. Although cod up to 77 cm has been in the found stomachs of grey seals, the two marine mammals eat mainly smaller cod (age 0–2). As the SBB comprise mainly older cod, the difference in SSB from the 2008 and 2011 is rather small. For F (mean of F, age 2–4) the difference is notable and recruitment is increased substantial to compensate for the increased mortality.

4.1.7 Comparison with fixed M at age (single species) assessments

Previous work of the multispecies working groups have shown that SSB and fishing mortality are to a large degree insensitive to the addition of species interactions, especially for species with a limited predation mortality on the fished and mature part of the stock. Cod (Figure 4.1.4) is shown as an example of such species. For forage fish or fish with a relatively small body size, for example whiting (Figure 4.1.4), the level of both F and SSB are different when estimated in single (fixed M at age) or multispecies (variable M at age) assessment. The trends are however quite similar for the two approaches. For almost all the prey species the historical development in recruitment differs greatly between multispecies and single species assessments. This pattern is caused by the temporal variation in predation mortality and the assumed single species M (assumed known, total natural mortality) and multispecies M1 (additional natural mortality, assumed constant and known) applied.

4.1.8 Biomass eaten

The total prey biomass eaten by year is presented by predator in Figure 4.1.5 and by prey species in Figure 4.1.6. The amount eaten follows largely the biomass of the main fish predators, which was highest in the late sixties and early seventies (the "gadoid outburst"). At that time the main fish predators were cod, whiting, haddock and mackerel. In the most recent years the biomass eaten by cod has clearly decreased, while the proportion eaten by mackerel, horse mackerel and grey gurnards has increased.

The contributions from the individual prey species to total biomass eaten have changed considerably (Figure 4.1.7). Haddock contributed much more during the gadoid outburst than in most recent years. In the seventies with the low herring

stock, its role a prey species decreased while the proportion from sprat was high in that period. Although the contribution from sandeel is variable, this species contributes most to the biomass eaten over the full time-series.

4.1.9 Predation mortality (M2)

The overall picture of M2 at age (Figure 4.1.8) 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 in recent years. 0-group mortality for the other species seems to be without a clear trend for the period since 1970. Predation mortality of the relatively large prey such as age 2 cod, whiting and haddock is highly influenced by the declining abundance of large cod. Compared with the decline for age 2 haddock the decline in M2 for age 2 whiting and cod is smaller due to an increasing predation from grey seals and harbour porpoise.

M2 of the newly introduced sprat is highly dependent on the biomass of mackerel and horse mackerel present in the North Sea. Whiting and seabirds are also major predators on sprat.

Rather limited information is available on the stock size of the North Sea mackerel and horse mackerel, and the proportion in the North Sea of the Western stock of those two species is highly uncertain. This influences the quality of the predation mortality significantly, as predation mortality from mackerel and horse mackerel is estimated to increase in recent years. Stock sizes of grey gurnard and the derived M2 on mainly cod, whiting and haddock has also increased according to the SMS model. It is crucial to obtain more detailed information on the quantity of grey gurnard, mackerel and horse mackerel in the North Sea as well as of the diet composition of these three species.

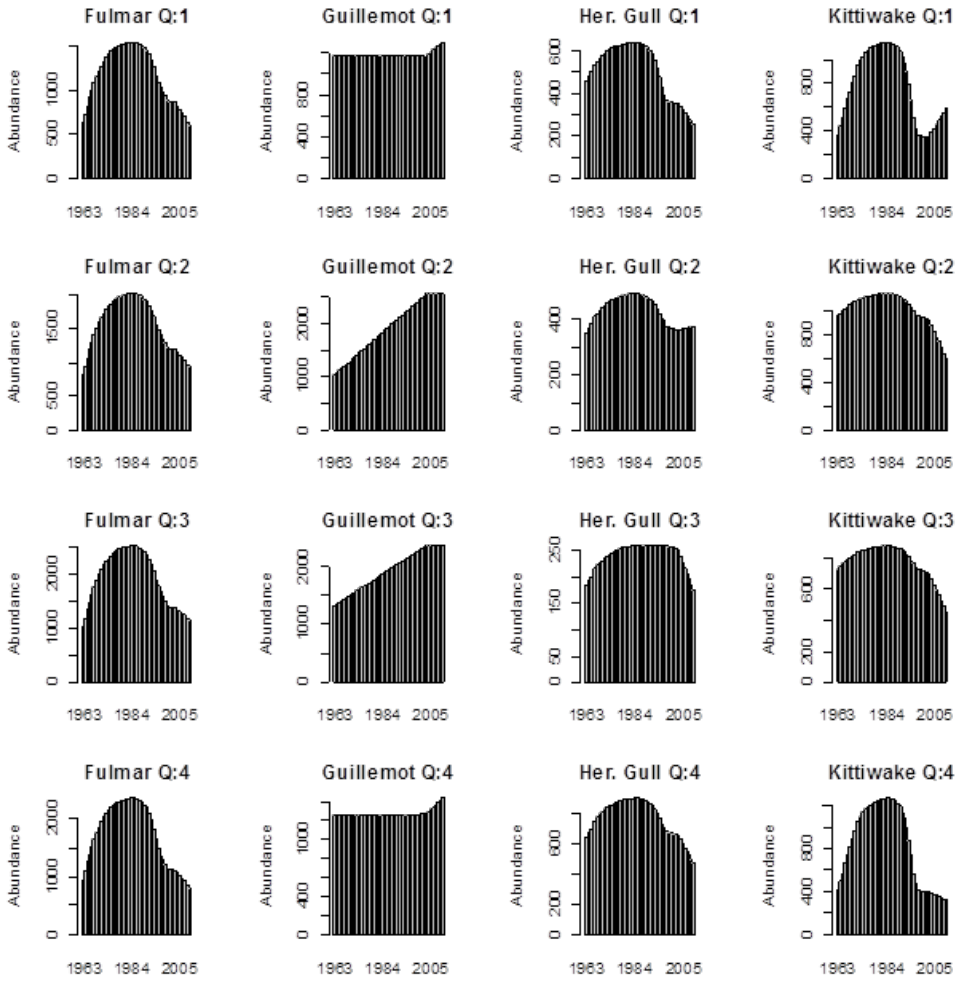


Figure 4.1.1. Estimates as used by SMS of the abundance of "external predators" present in the North Sea presented by year and quarter. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

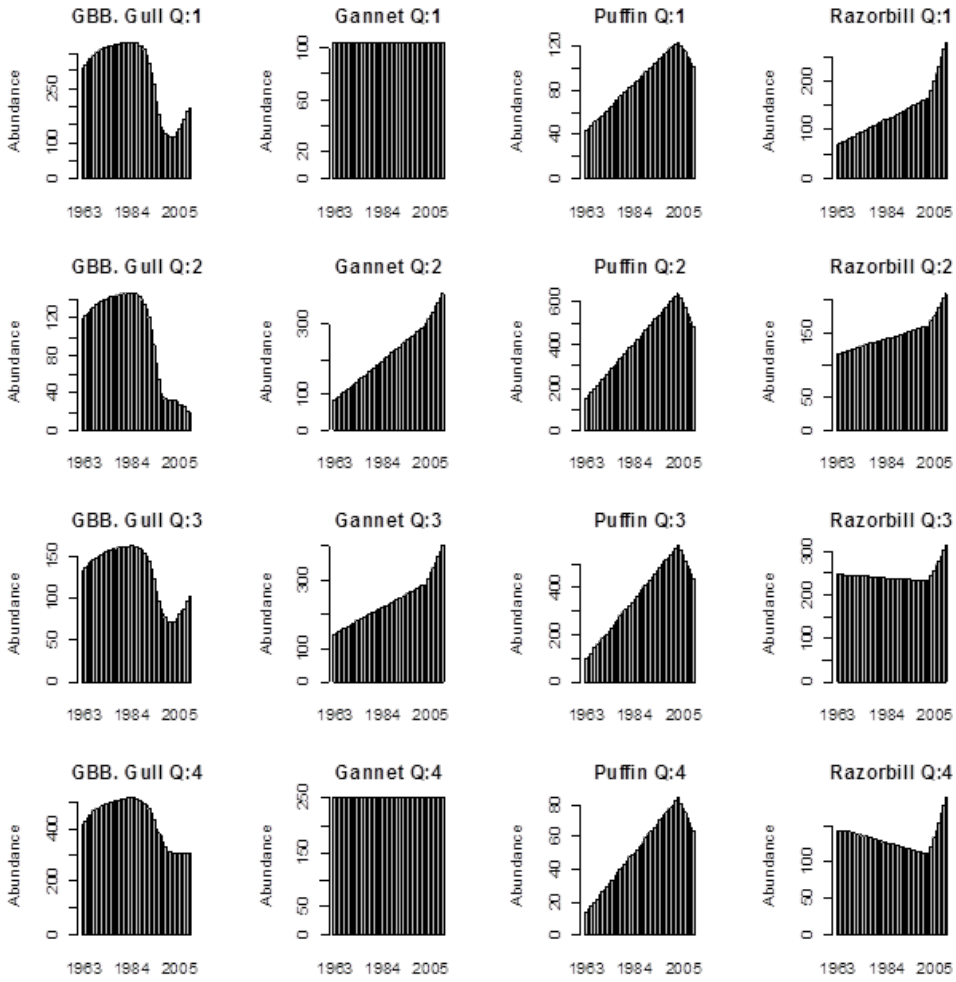


Figure 4.1.1. (Continued) Estimates as used by SMS of the abundance of "external predators" present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

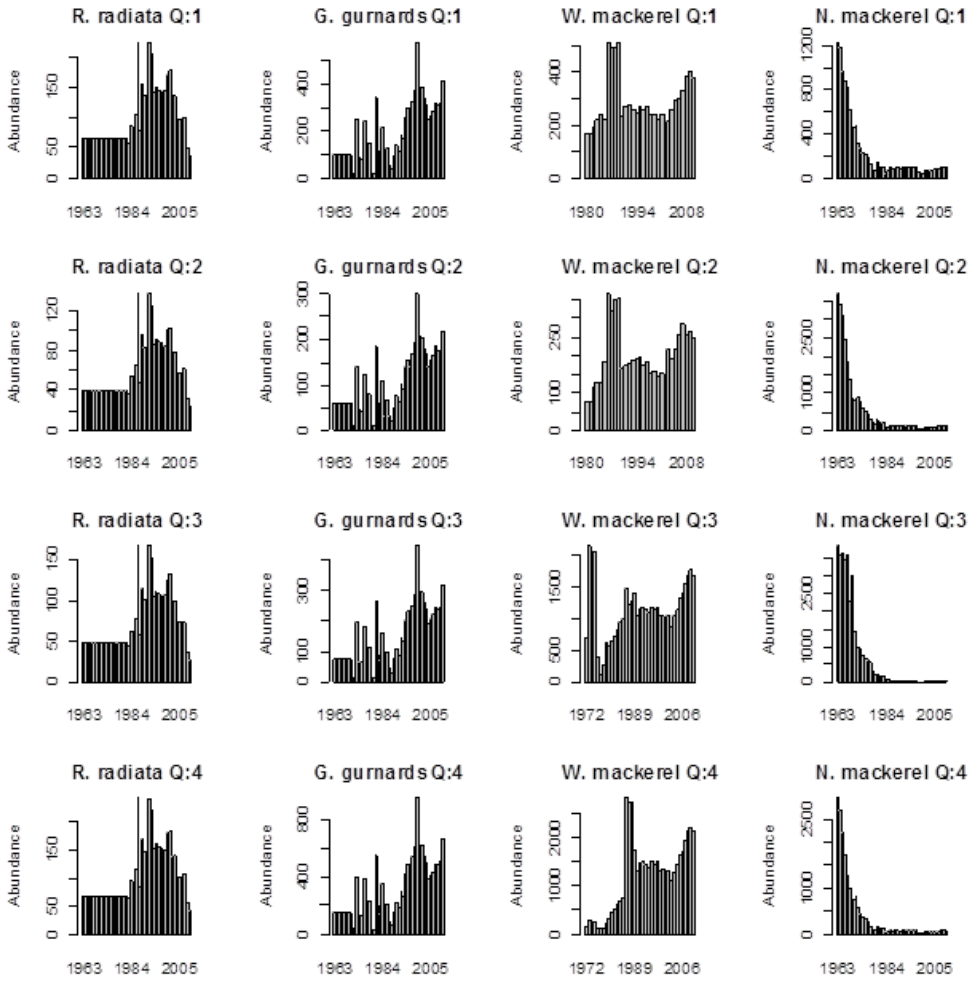


Figure 4.1.1. (Continued) Estimates as used by SMS of the abundance of "external predators" present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

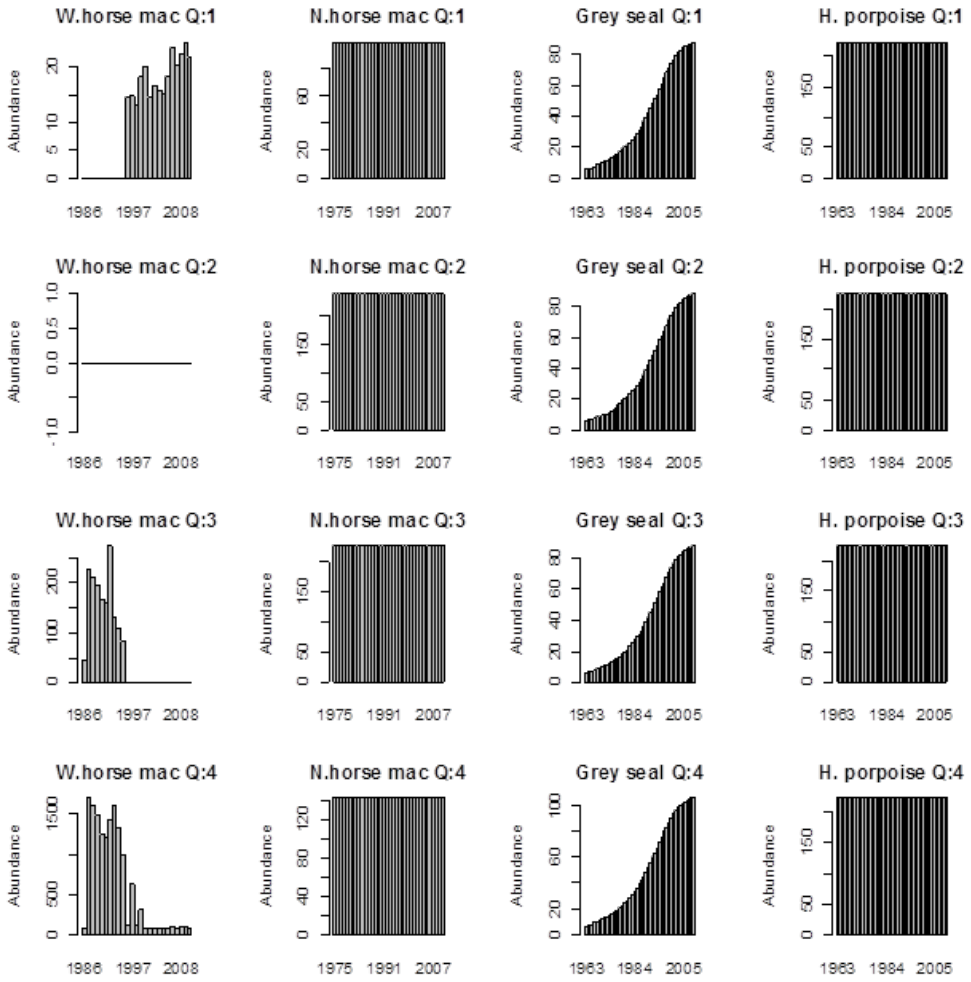


Figure 4.1.1. (Continued) Estimates as used by SMS of the abundance of "external predators" present in the North Sea. (Abundance of birds and marine mammals are given as numbers (1000), and as population biomass (1000 t) for fish species.

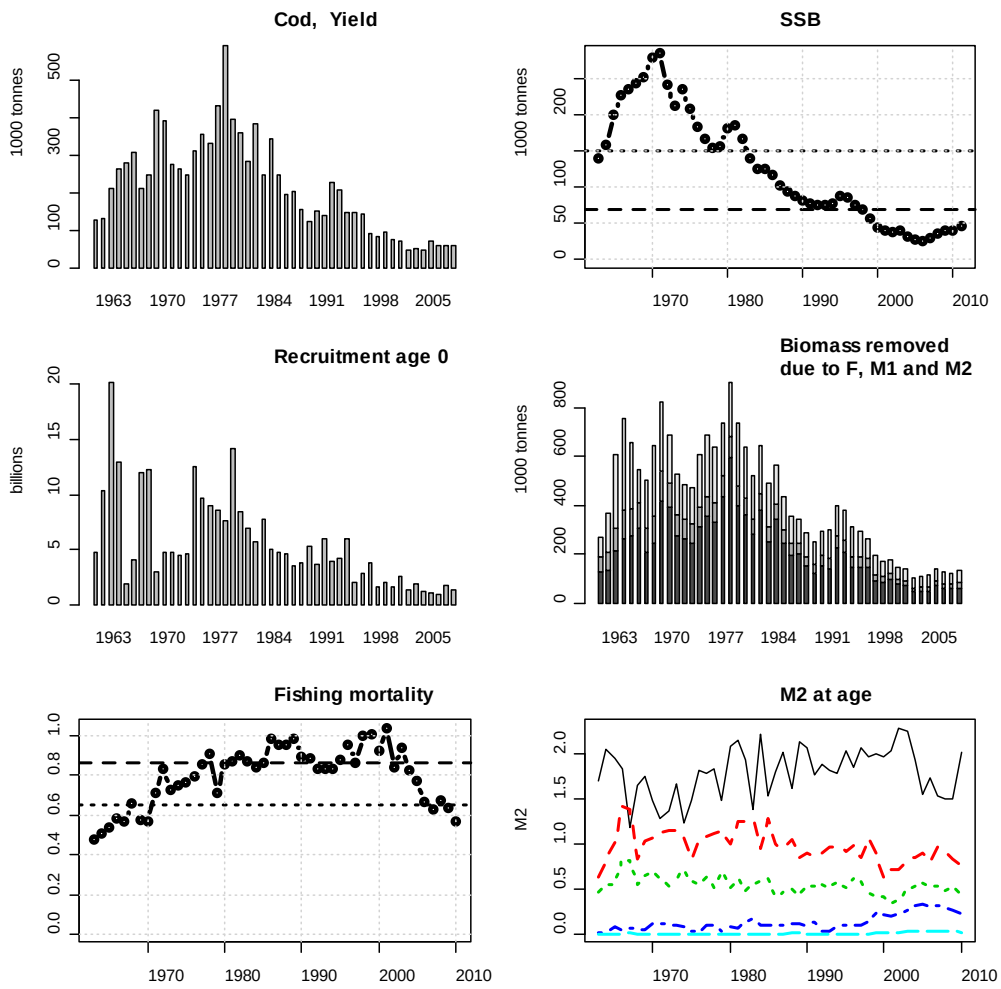


Figure 4.1.2. SMS output for cod. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

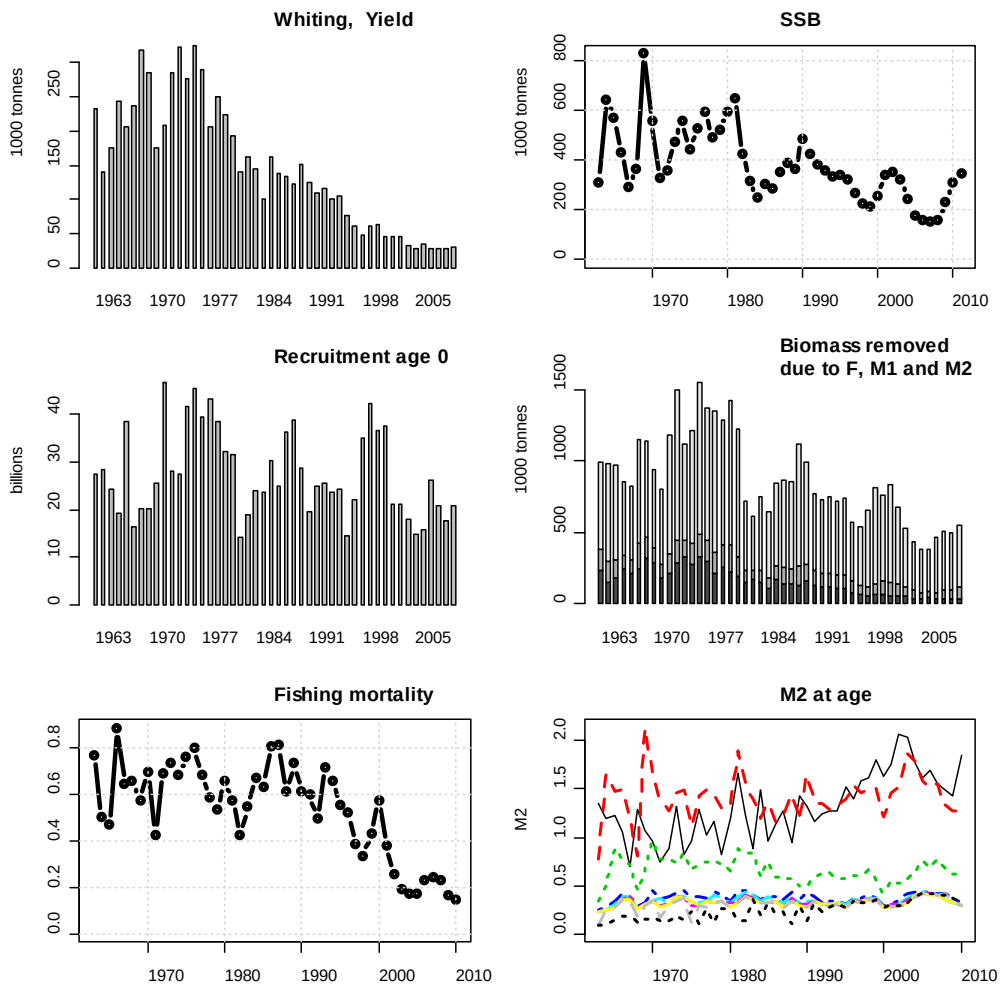


Figure 4.1.2 cont. SMS output for whiting. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

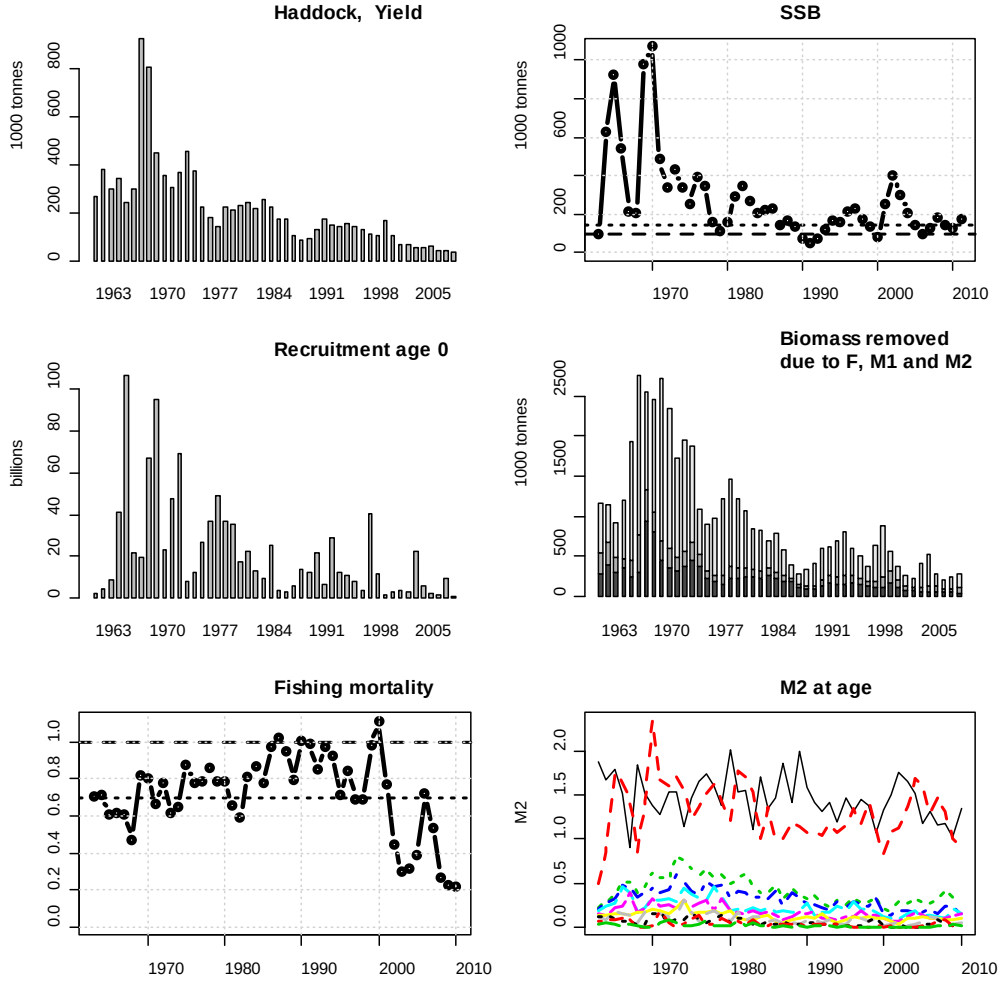


Figure 4.1.2 cont. SMS output for haddock. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

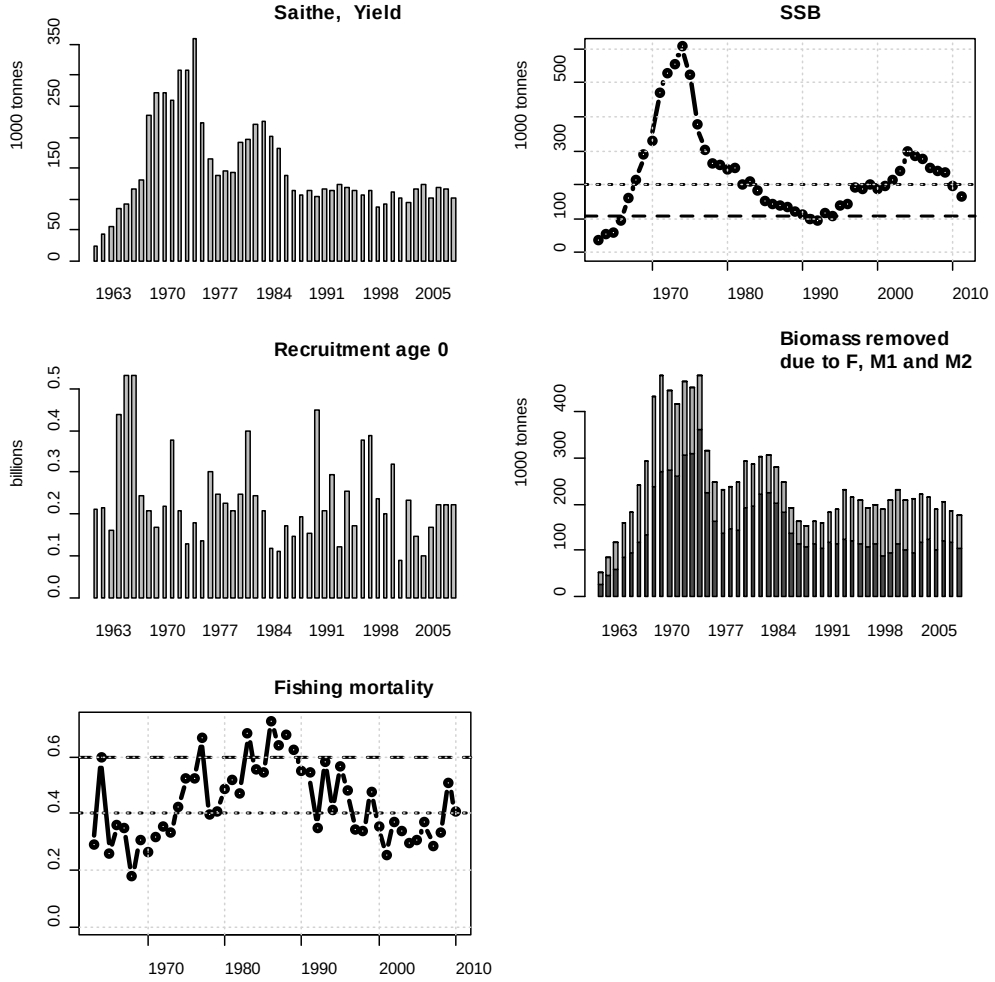


Figure 4.1.2 cont. SMS output for saithe. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

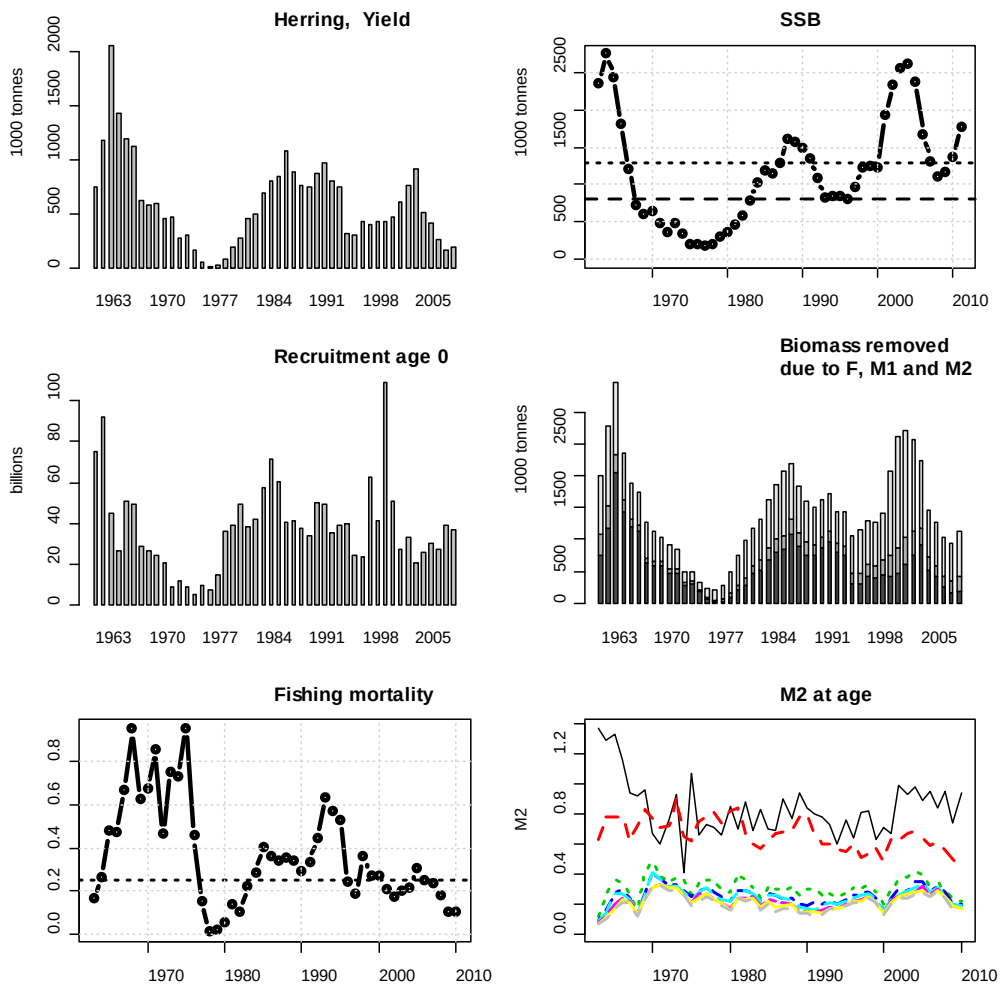


Figure 4.1.2 cont. SMS output for herring. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

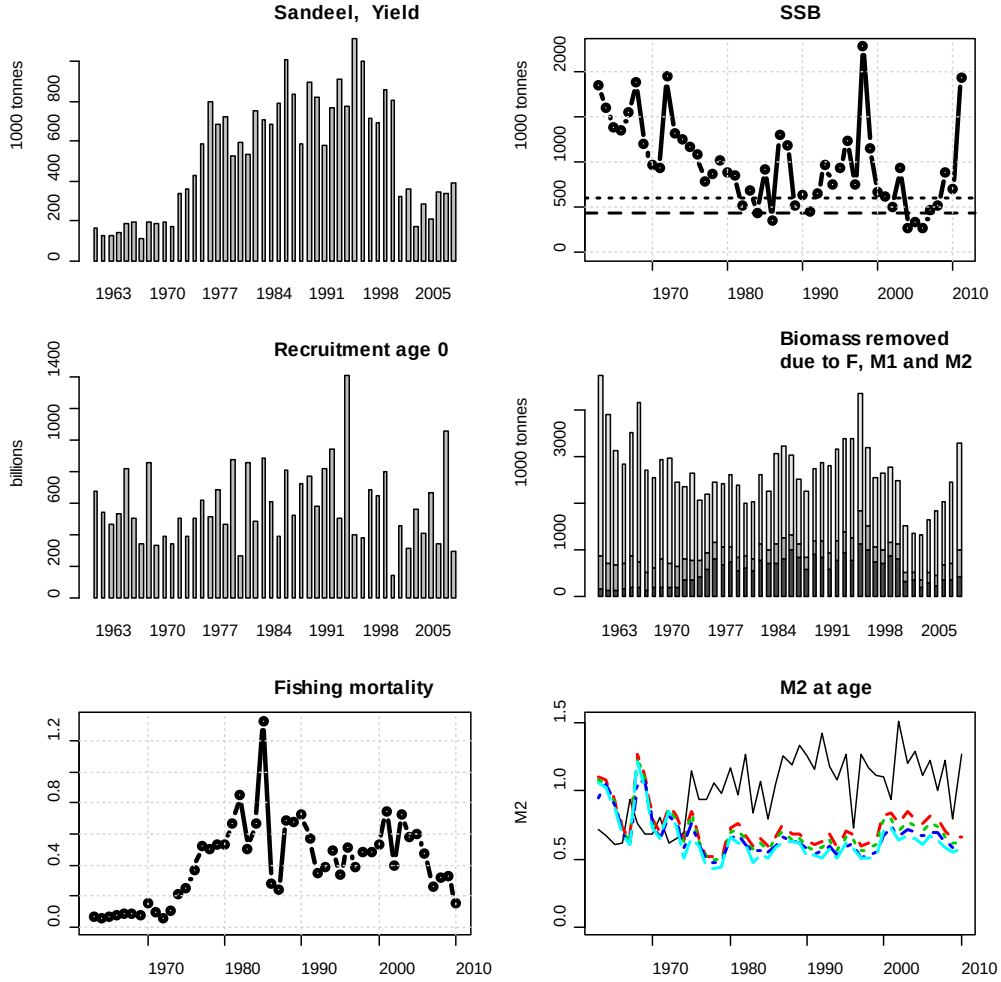


Figure 4.1.2 cont. SMS output for sandeel. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

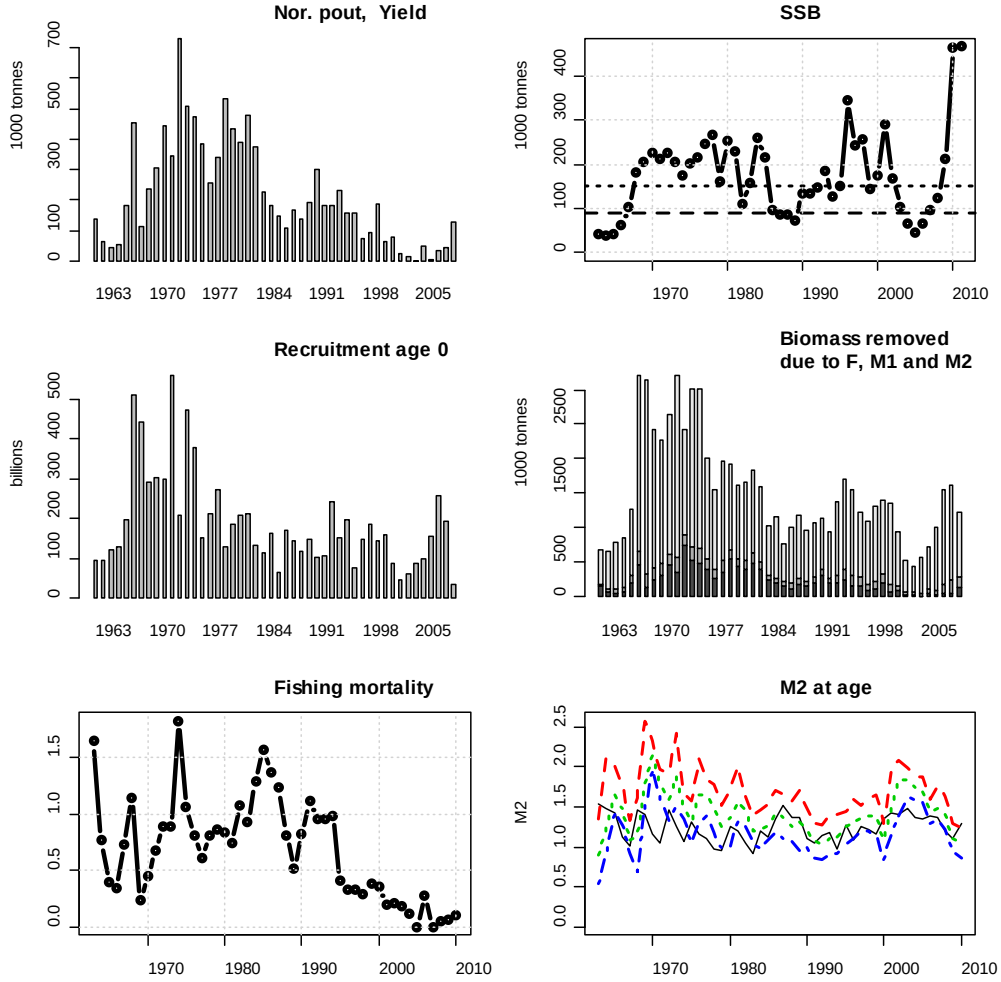


Figure 4.1.2 cont. SMS output for Norway pout. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

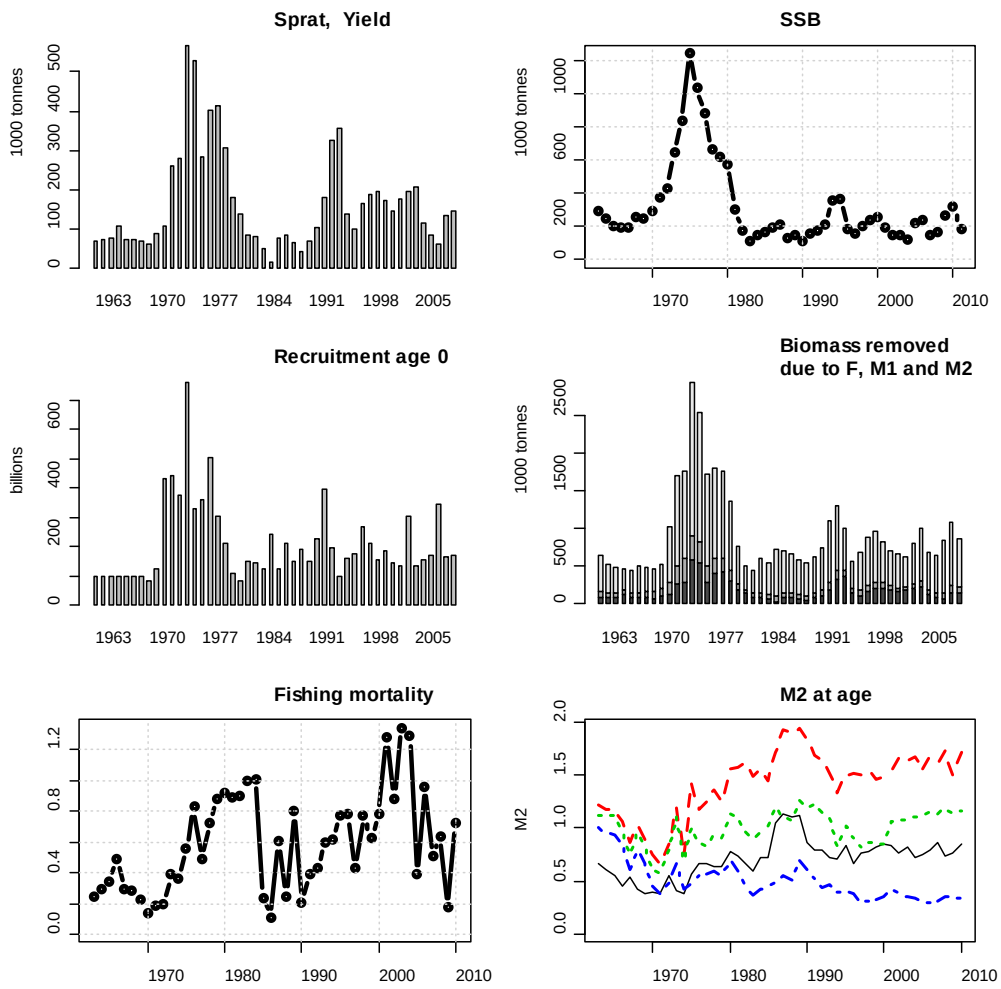


Figure 4.1.2 cont. SMS output for sprat. SOP (catch numbers * catch weight), Recruitment, F, SSB, Biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1). The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.

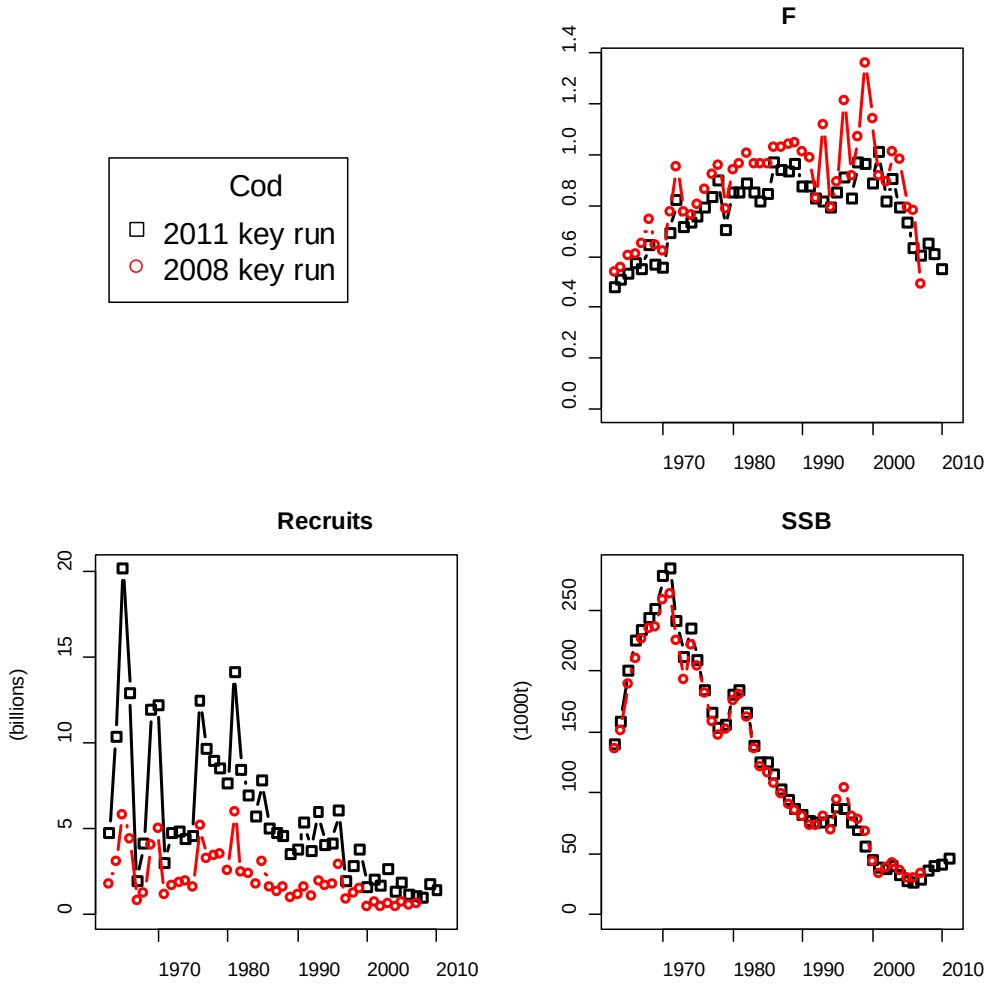
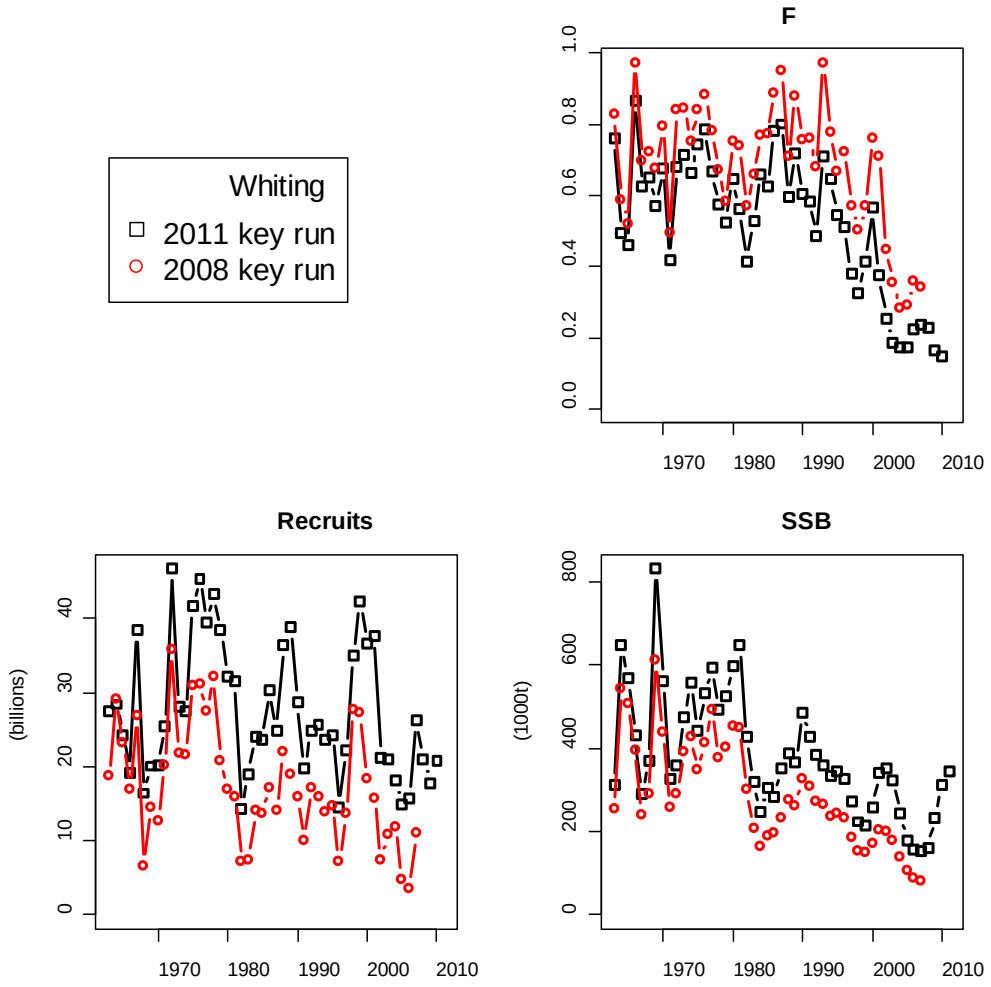


Figure 4.1.3. Comparison of estimates of recruits, fishing mortality and SSB of cod from the 2008 and 2011 key runs.



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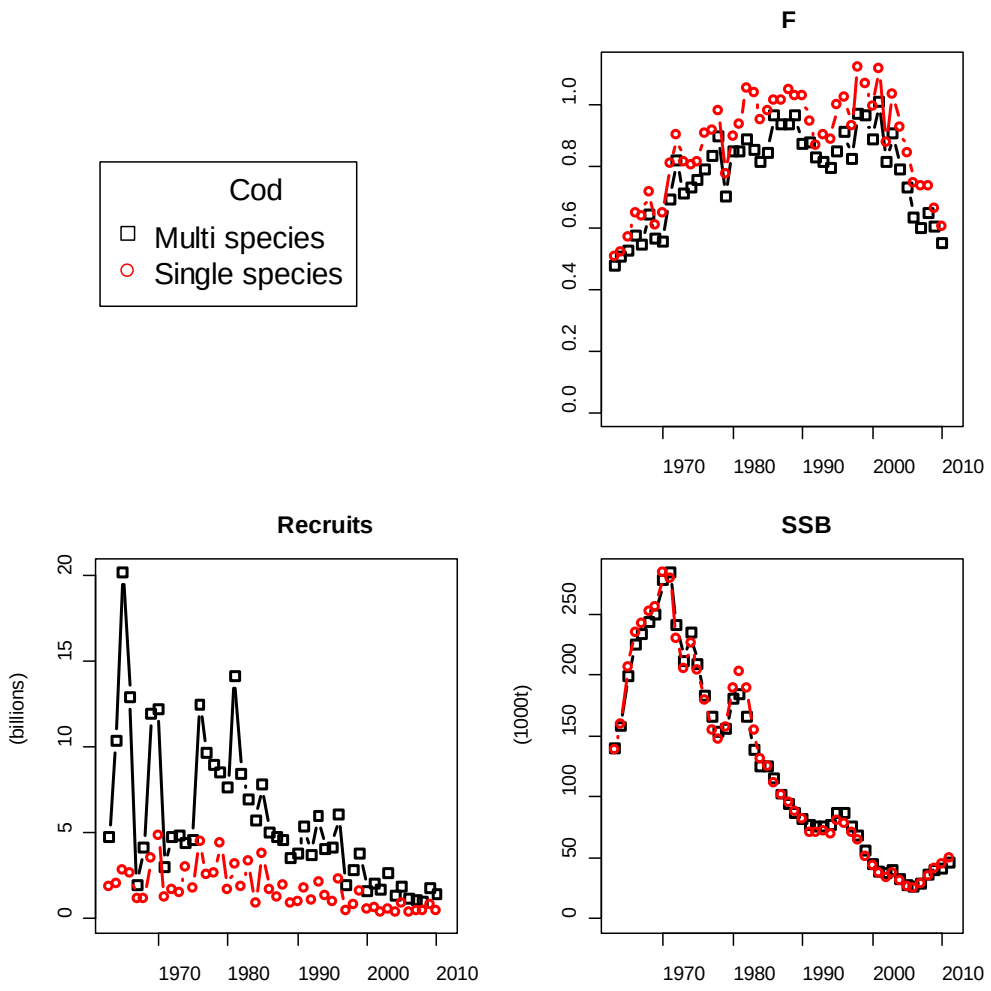


Figure 4.1.4. Comparison of estimates of recruits, fishing mortality and SSB of cod from single and multispecies SMS.

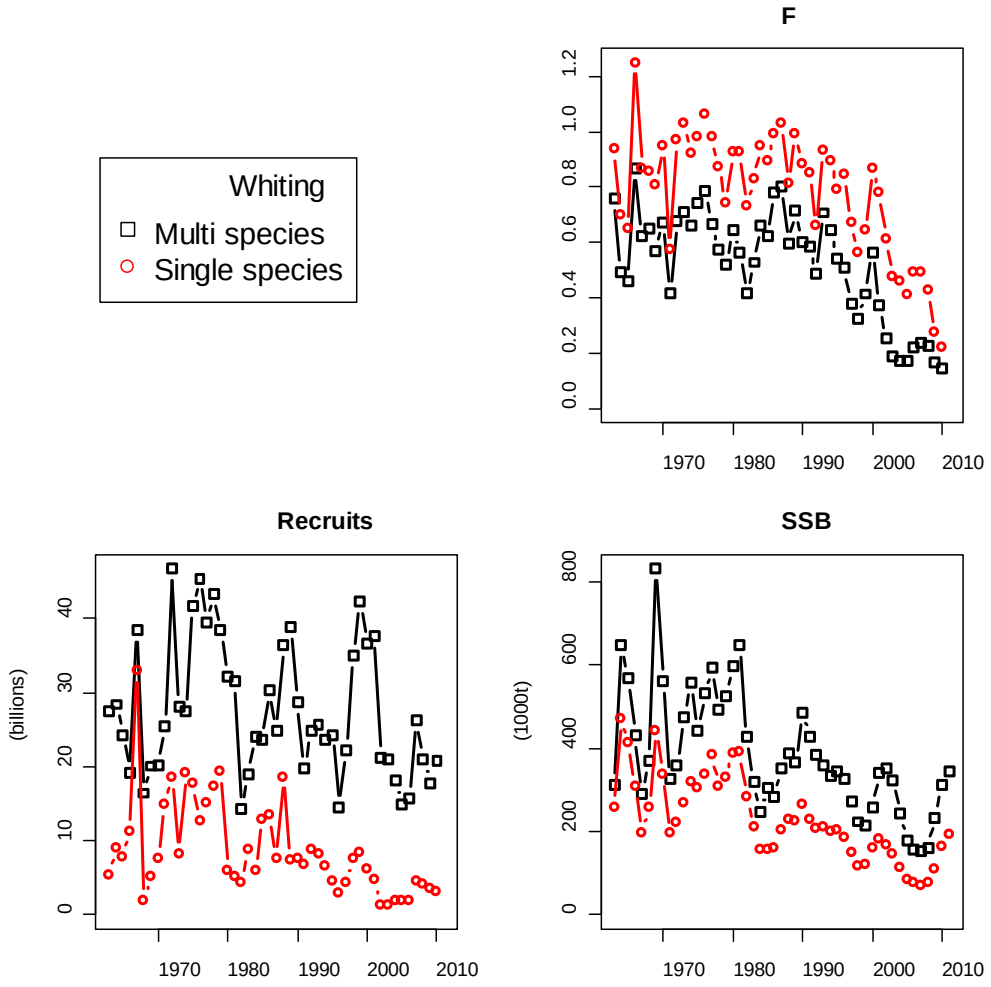


Figure 4.1.4. Comparison of estimates of recruits, fishing mortality and SSB of cod from single and multispecies SMS.

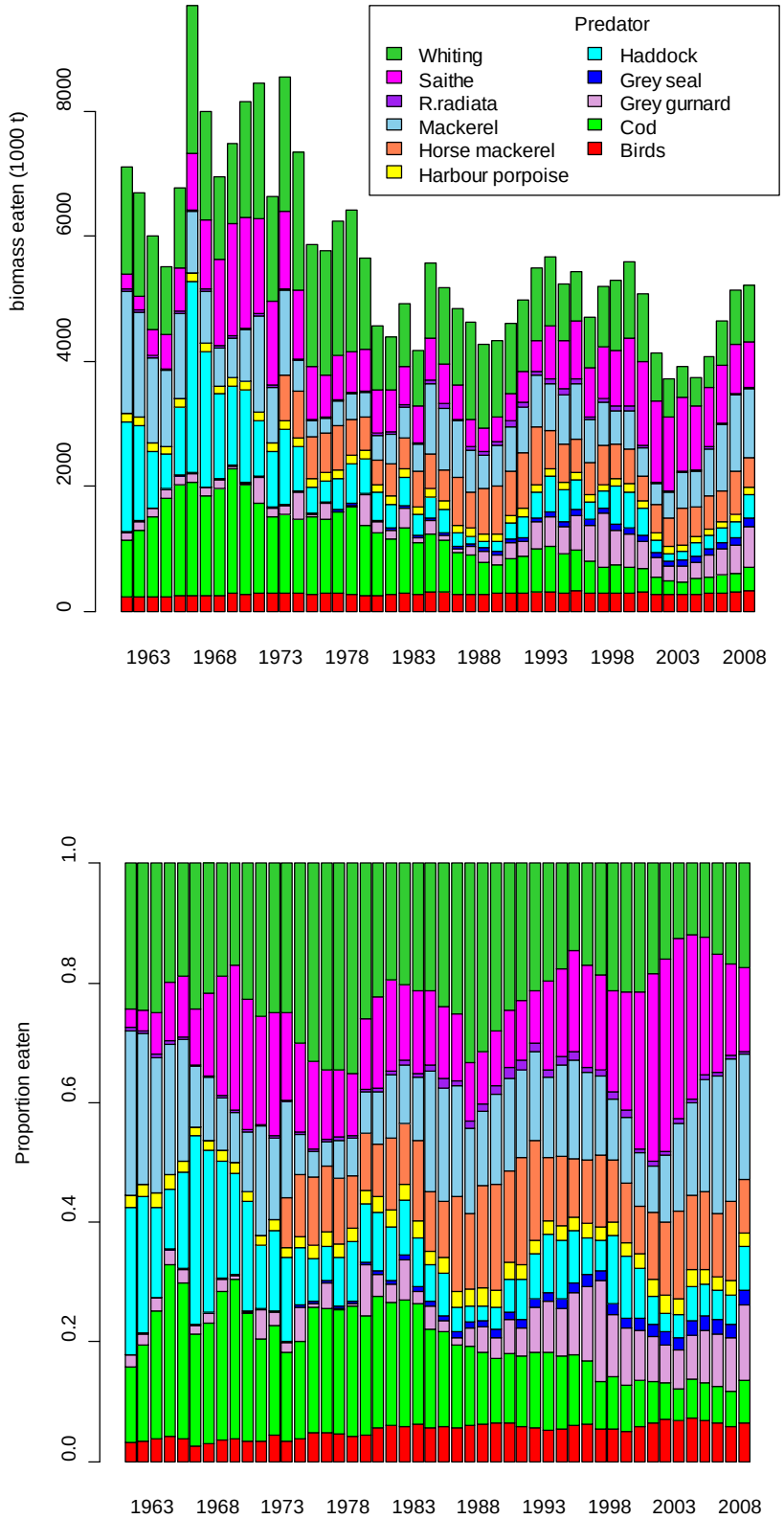


Figure 4.1.5. Prey biomass eaten by individual predators.

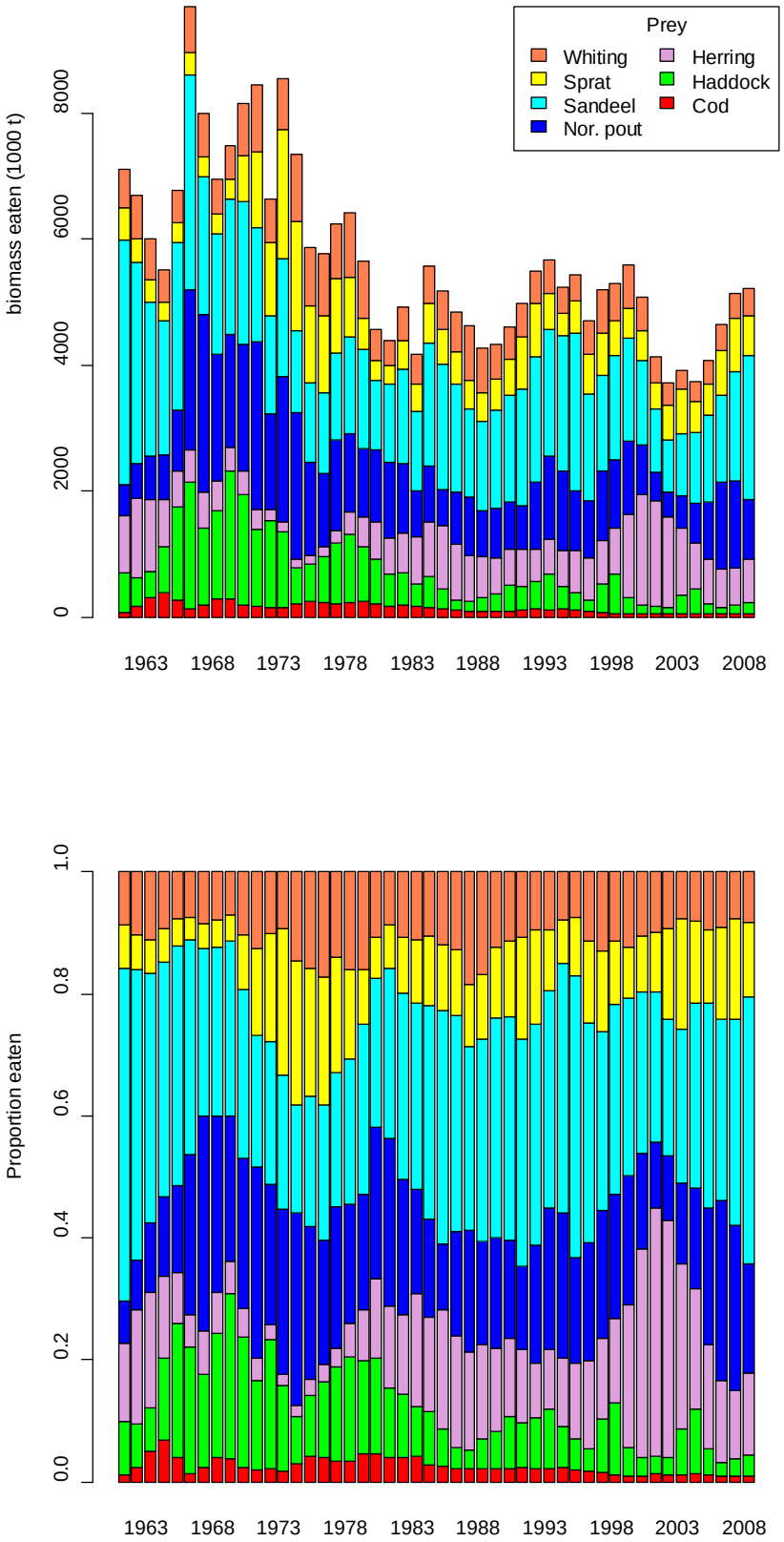


Figure 4.1.6. Prey biomass eaten divided on individual prey species.

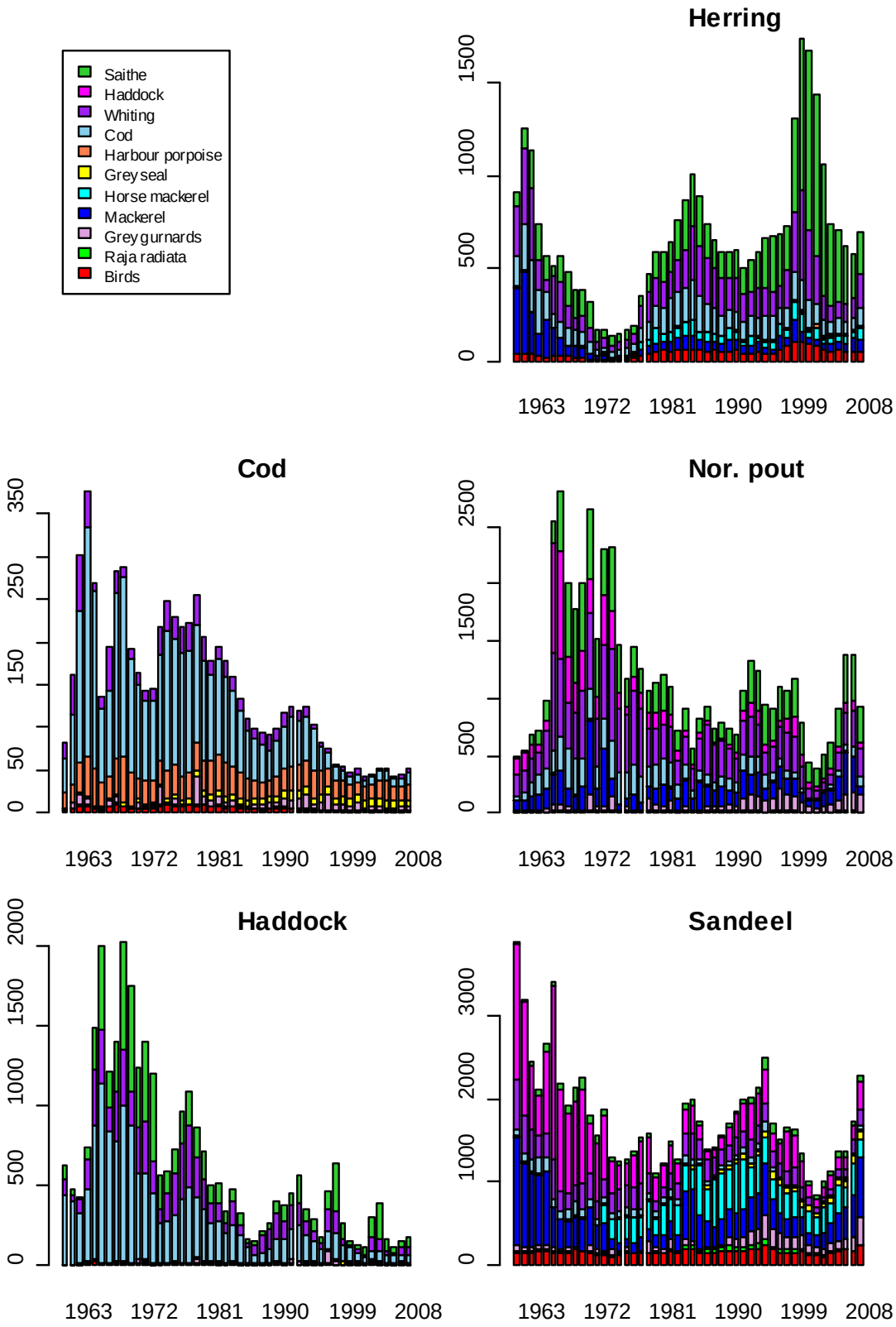


Figure 4.1.7. Biomass eaten (1000 t) for each prey species by individual predator species.

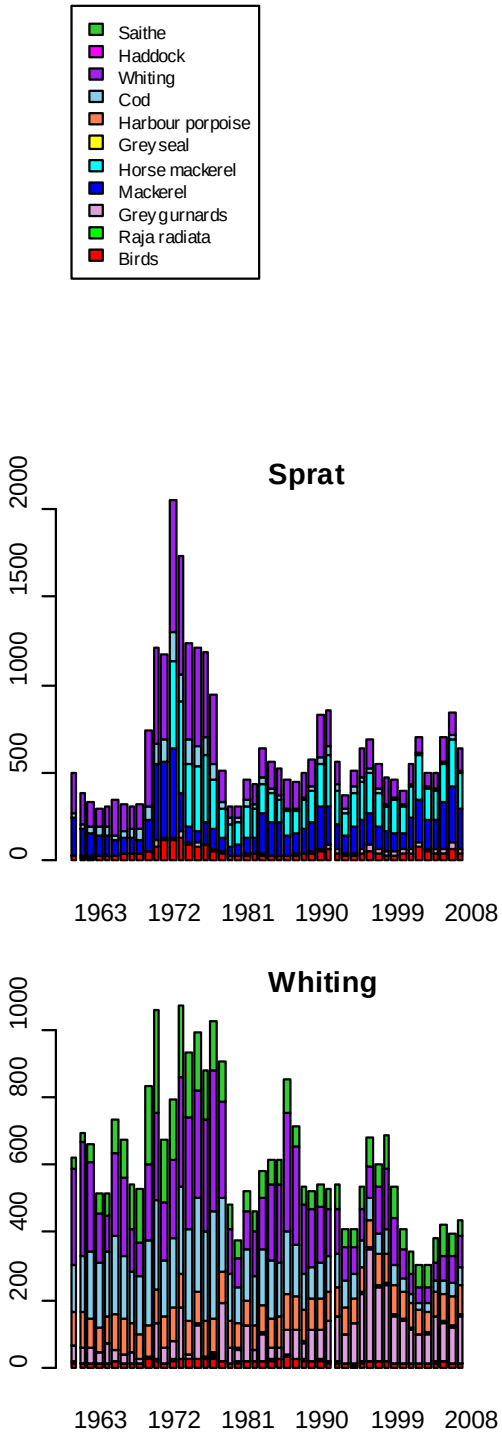


Figure 4.1.7. cont. Biomass eaten (1000 t) for each prey species by individual predator species.

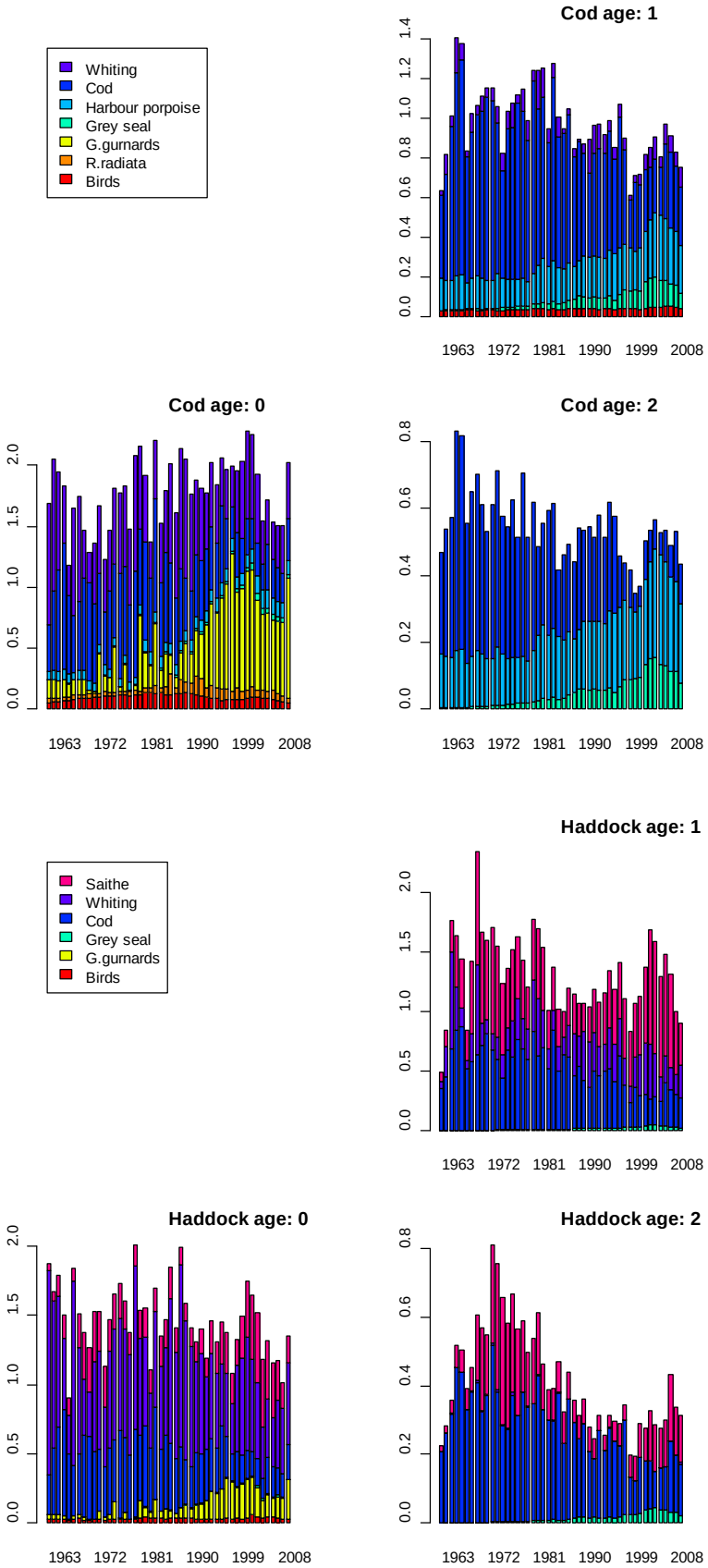


Figure 4.1.8. Predation mortality (M2) by prey species and age inflicted by predator species.

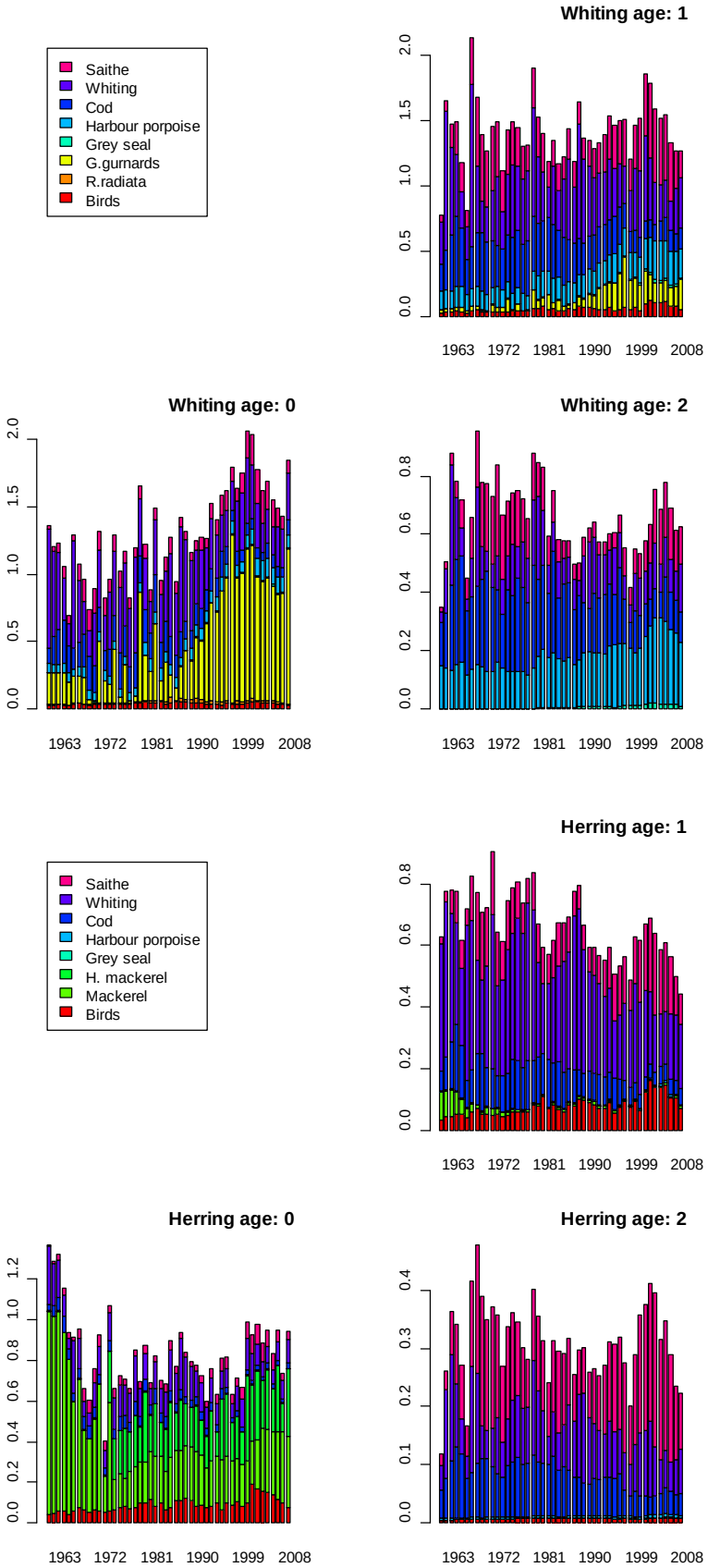


Figure 4.1.8. cont. Predation mortality (M2) by prey species and age inflicted by predator species.

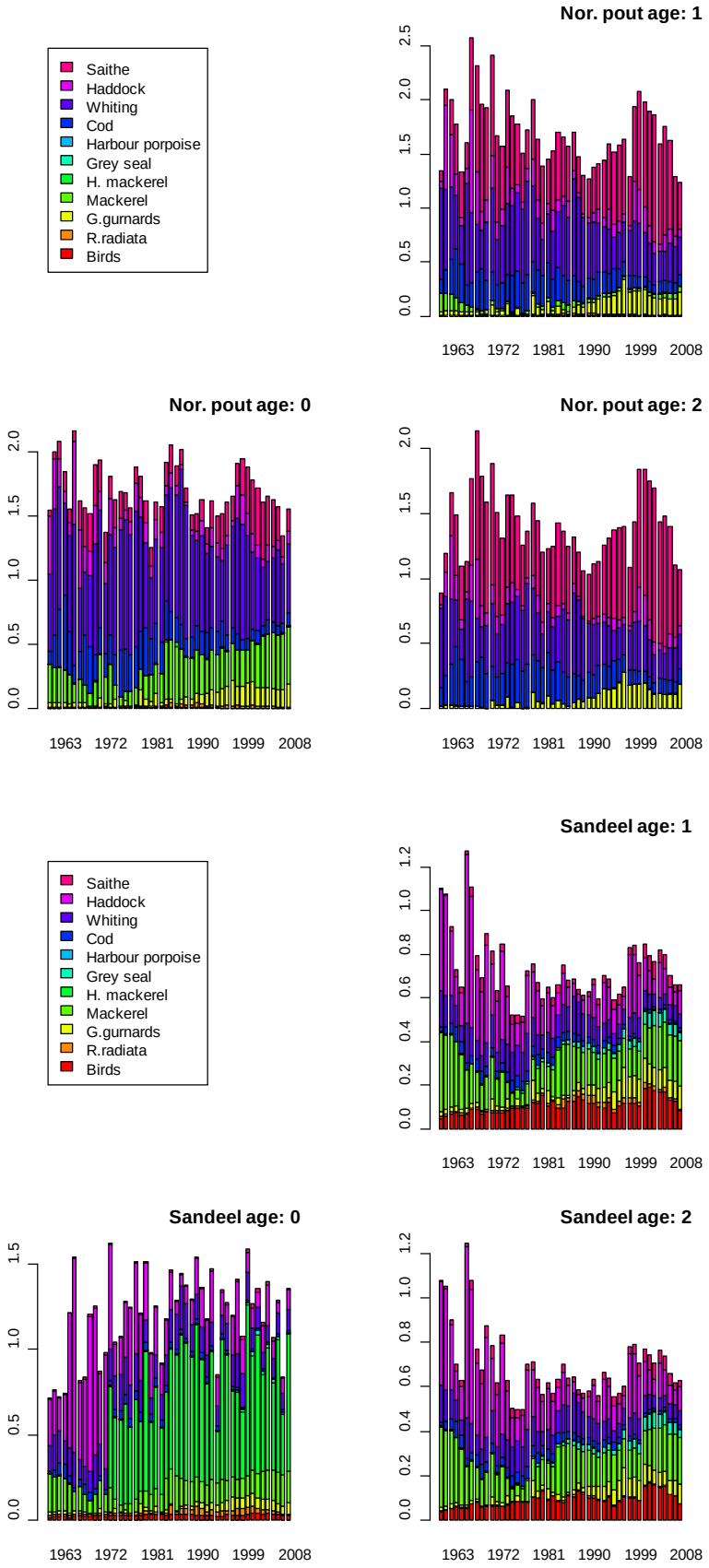


Figure 4.1.8. cont. Predation mortality (M2) by prey species and age inflicted by predator species.

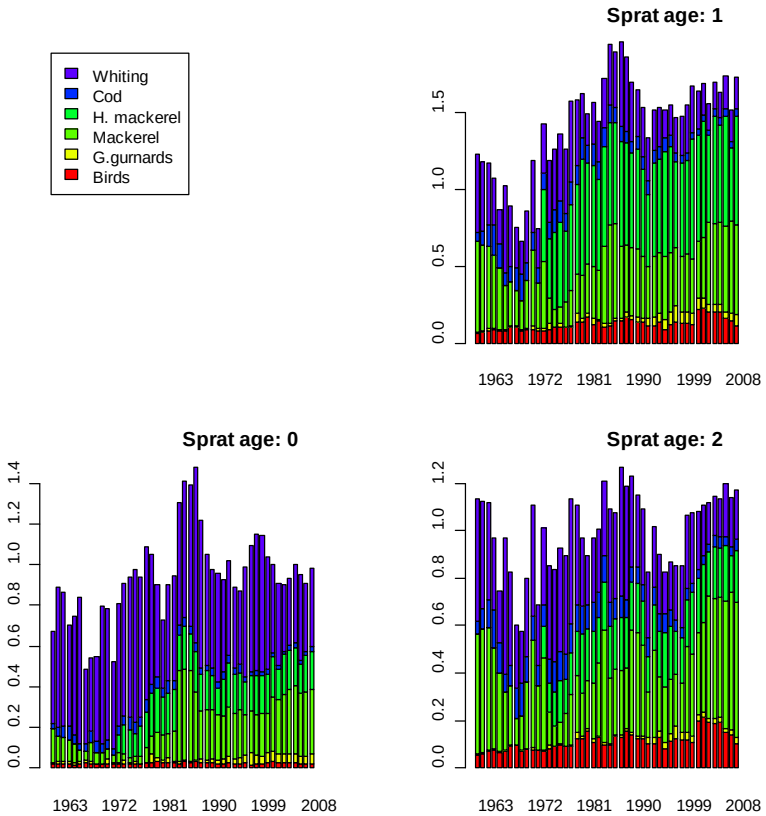


Figure 4.1.8. cont. Predation mortality (M2) by prey species and age inflicted by predator species.

4.2 North Sea –Ecopath with Ecosim

The North Sea EwE model published by Mackinson and Daskalov, (2007) has been updated and calibrated by fitting to time-series data from 1991–2007, and is presented here as the first Key Run. Full details are described in Annex 5. The calibration process included sensitivity analysis of parameters used in initializing and fitting model predictions to observations. Specifically, this involved the selection of initial vulnerability parameters, selection and weighting of time-series data used in fitting, and the parameterization of stock–recruitment dynamics. Additional details of the results and discussion of their sensitivity to model parameters and to the effects of fishing and environmental drivers are to be reported in a research manuscript (Mackinson, in prep). Sensitivity to the selection of data used in model fitting reveals that model performance is best when data on all functional groups in the ecosystem are included. When data on only selected important commercial species are included, the results conflict with empirical evidence. Results of analyses reported in Mackinson (in prep) indicate strong empirical evidence for water temperature as being an important driver of bottom–up changes in the North Sea ecosystem; a finding that is supported by evidence from the model key run.

ECOSYSTEM MODEL, 1991–2007

Key run summary sheet

Area	North Sea
Model name	Ecopath with Ecosim
Type of model	Foodweb compartment
Run year	2011
Species/ Groups	68 functional groups
Time range	1991–2007
Time-step	Monthly
Area structure	Model covers North Sea ICES Division IVa,b and c. No spatial aspects are modelled.
Stomach data	1991 year of the stomach and others
Purpose of key run	Describing changes in the North Sea ecosystem and providing model data
Model changes since last key run	First key run
Output available at	http://www.ices.dk/workinggroups/ViewWorkingGroup.aspx?ID=193
Further details in	Report of the Working Group on Multispecies Assessment Methods 2011, ANNEX 5

The term Ecopath 'Key-run' originated from earlier ICES multispecies working groups (ICES 2009, 2010) as a way to facilitate a common understanding among modellers using multispecies models. These have typically been for MS models and EwE models. In the present document and as noted in previous years reports (ICES 2009, 2010), we define what is meant by a 'key-run' and recommend a format for the reporting of results and model diagnostics in a way that facilitates comparisons among models, transparency and provides flexibility to tailor model outputs to specific questions. An EwE 'key-run' refers to a model whose time-series dynamics (Ecosim predictions) are fitted to time-series of observation data. It is sometimes referred to as a calibrated model. The procedure involves using historical data concerning fishing and environmentally driven changes in primary production (and possibly forcing of trophic interactions) to drive model predictions of changes in ecosystem components and catch from fisheries. A non-linear fitting routine is used to minimize the difference between observations and model predictions by estimating the interaction strength between predator–prey nodes in the foodweb (vulnerability parameters), and sometimes a time-series of primary production anomalies as well.

Essential components of the key-run are a balanced Ecopath model (a snapshot account of the foodweb), that conforms to a quality standard judged through publication/ and or established using quality criteria (see Link *et al.*, 2010), and the calibrated Ecosim predictions (Figure 1).

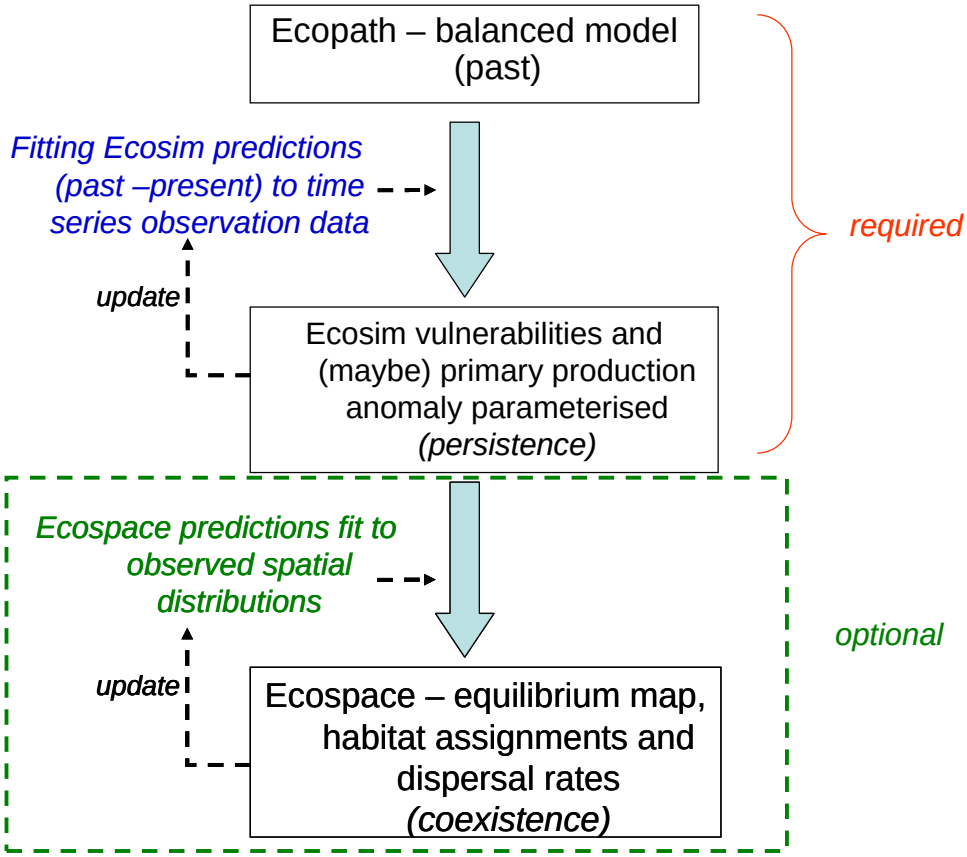


Figure 1. Schematic of an Ecopath with Ecosim key run.

Information components

There are 3 principal information components that comprise a standardized output for EwE runs:

1) Baseline Ecopath Model – should be published and/or quality established using clear criteria (e.g. see Link *et al.*, 2010)

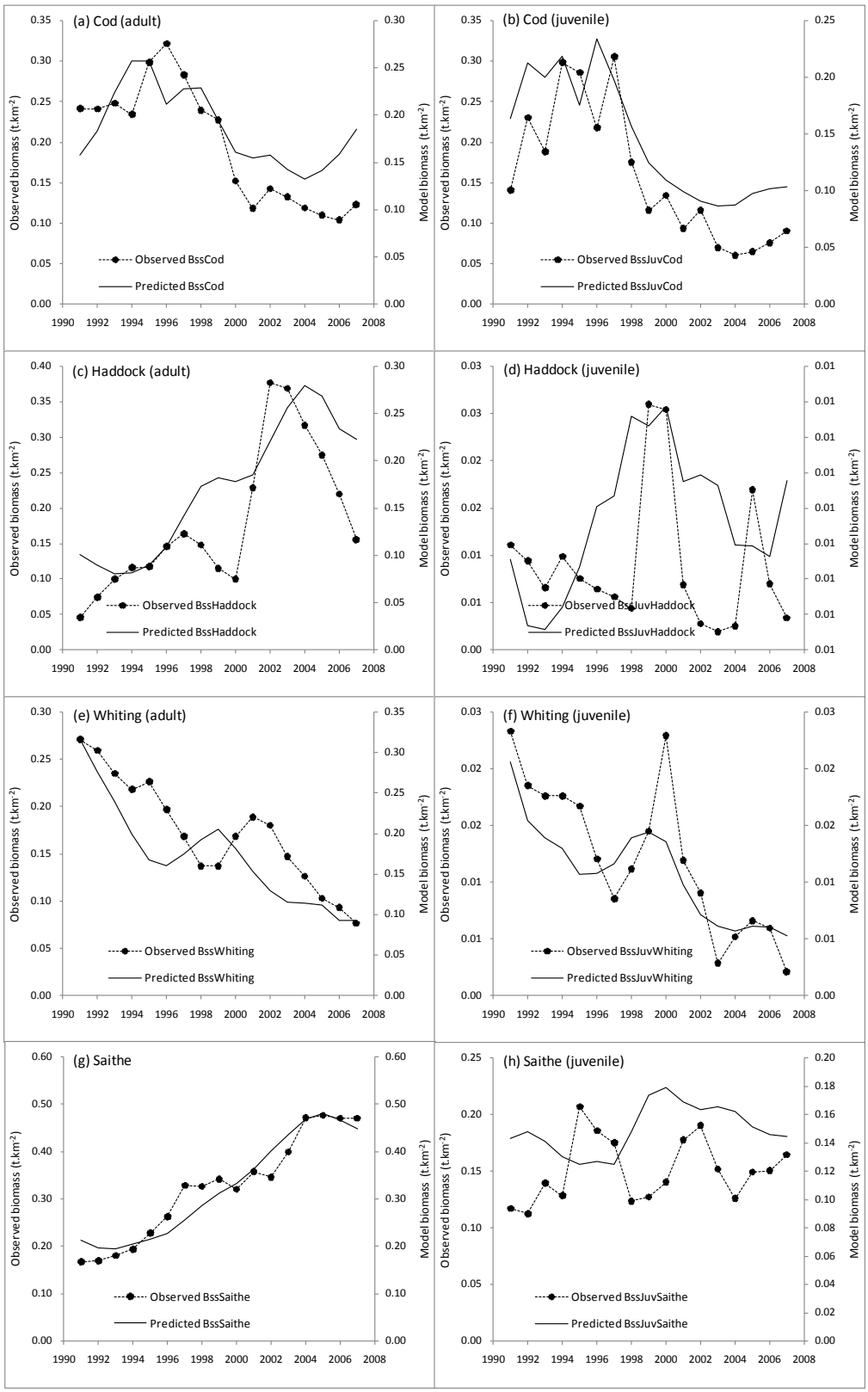
2) Observation Data used in forcing and fitting – the time-series data file should be made available, or at the very least the metadata so that users are able to understand clearly which data are driving the model and which are being used to constrain the model fit.

3) Fit of Ecosim prediction to observations – including the time-series trends and fitting diagnostics (see below for detail)

Standard format for EwE results based on the fit of Ecosim predictions to observations

Standardized outputs of key-runs should provide information on the functional groups, the fisheries, ecosystem indicators and diagnostics of the parameterization. The structure of this report is based upon the specification described in WGSAM 2009 (ICES, 2009, 2010).

The fit of model predictions to observed biomass data for selected key species of interest are shown in Figure 4.2.1. Changes in selected system and community indicators are shown in Figure 4.2.2. For further details, see Annex 5.


(a) Cod (adult)
Observed BssCod
Predicted BssCod
(b) Cod (juvenile)
Observed BssJuvCod
Predicted BssJuvCod
(c) Haddock (adult)
Observed BssHaddock
Predicted BssHaddock
(d) Haddock (juvenile)
Observed BssJuvHaddock
Predicted BssJuvHaddock
(e) Whiting (adult)
Observed BssWhiting
Predicted BssWhiting
(f) Whiting (juvenile)
Observed BssJuvWhiting
Predicted BssJuvWhiting
(g) Saithe
Observed BssSaithe
Predicted BssSaithe
(h) Saithe (juvenile)
Observed BssJuvSaithe
Predicted BssJuvSaithe
Observed biomass (t.km-2)
Model biomass (t.km-2)

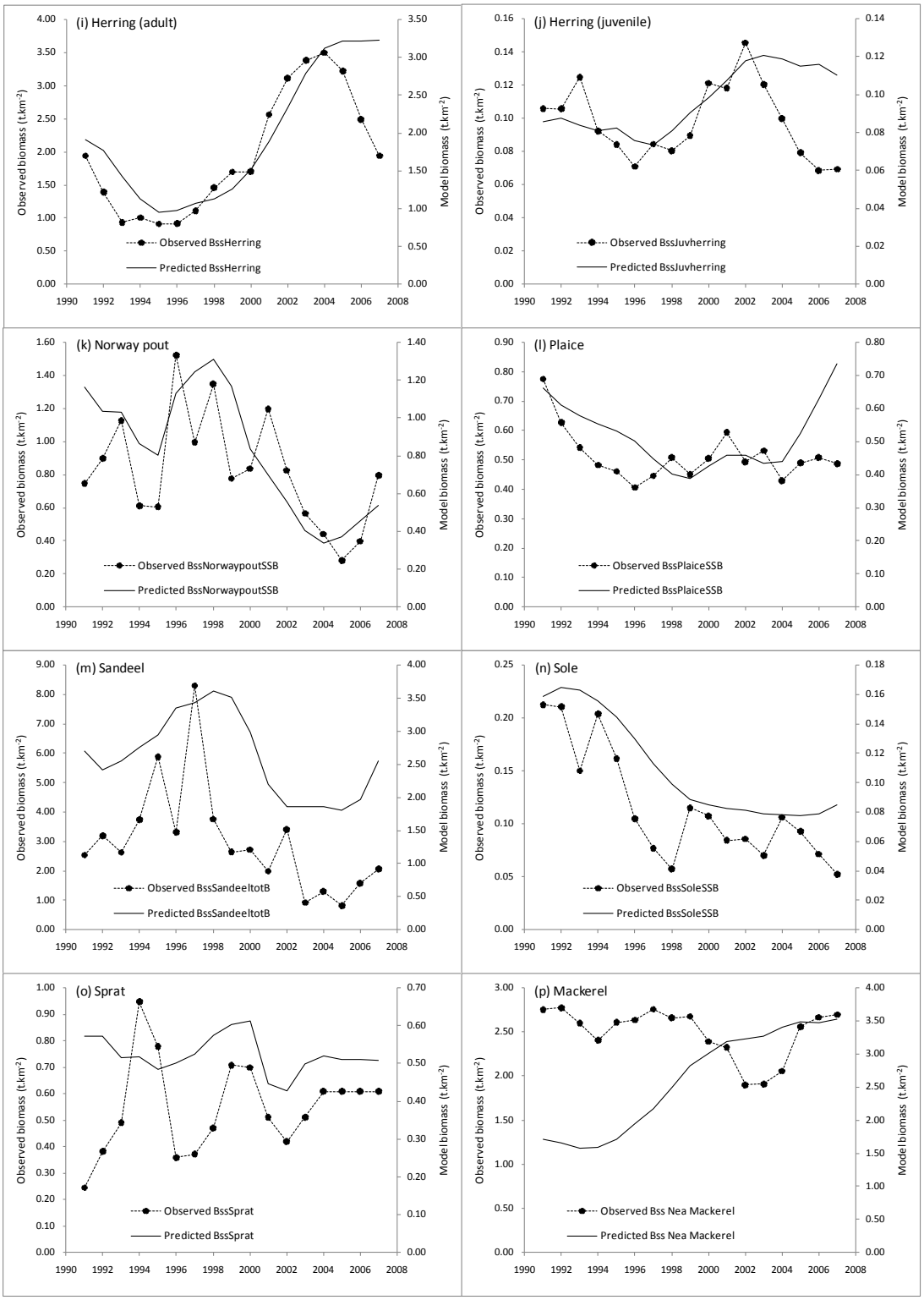
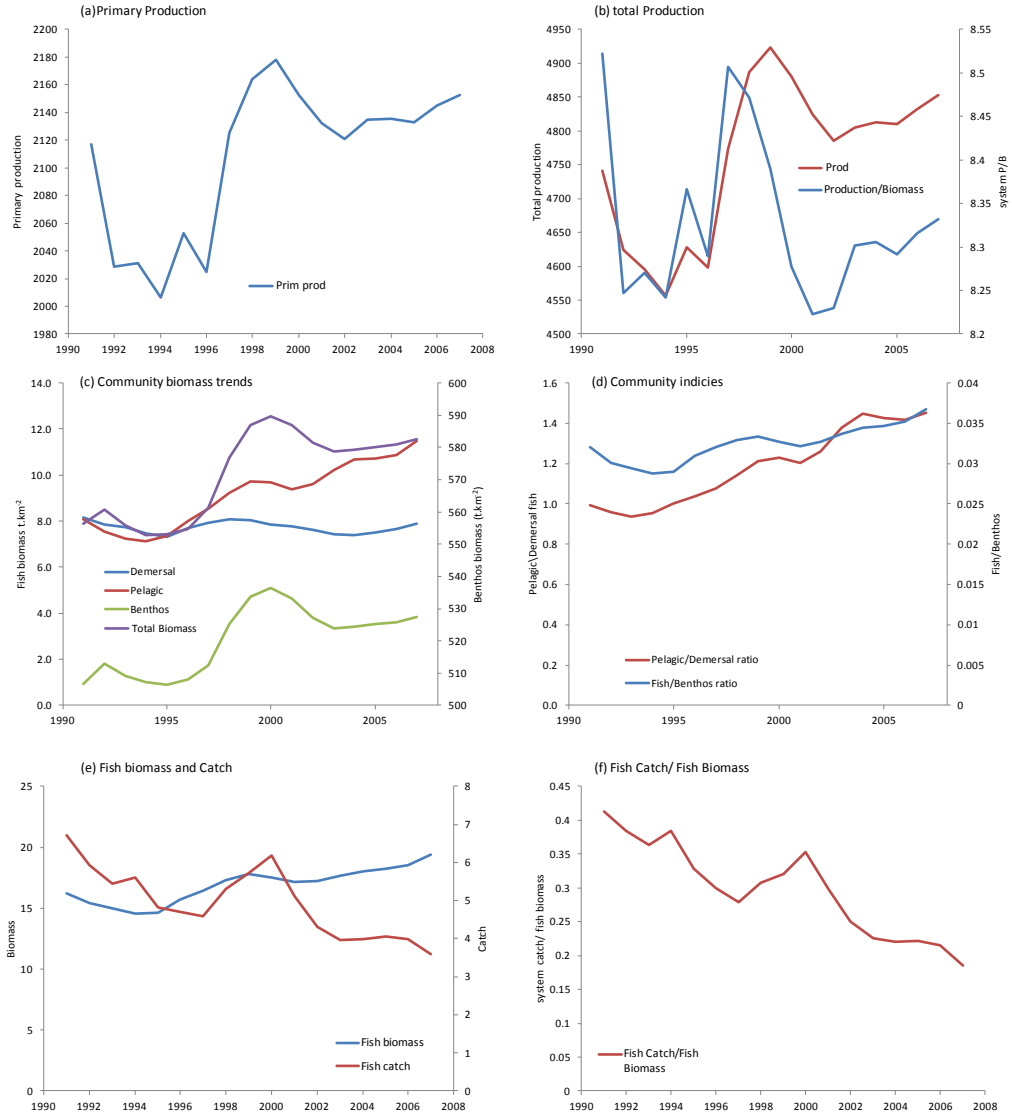


Figure 4.2.1. Relative biomass plots - observed and model predicted. 'Observed' data are derived from single species stock assessments.


(a)Primary Production
Primary production
Prim prod
(b) total Production
Total production
system P/B
Prod
Production/Biomass
(c) Community biomass trends
Fish biomass t.km-2
Benthos biomass (t.km-2)
Demersal
Pelagic
Benthos
Total Biomass
(d) Community indicies
Pelagic\Demersal fish
Fish/Benthos
Pelagic/Demersal ratio
Fish/Benthos ratio
(e) Fish biomass and Catch
Biomass
Catch
Fish biomass
Fish catch
(f) Fish Catch/ Fish Biomass
system catch/ fish biomass
Fish Catch/Fish Biomass

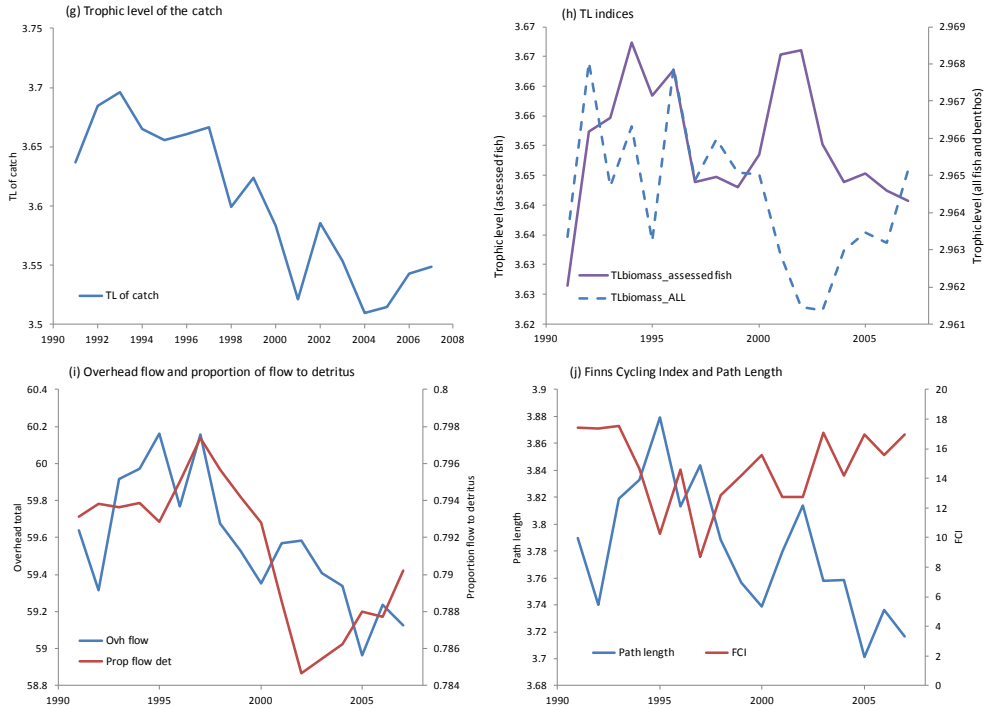


Figure 4.2.2. Ecosystem indicators derived from the model key run.

Mackinson, S. Empirical and model-based evidence explain long-term changes in the North Sea ecosystem linked to environmental change. In preparation.

5 ToR c) Work towards implementing new stomach sampling programmes in the ICES area in the near future

5.1 Continued requests for sampling

WGSAM has helped develop a letter detailing the rationale, value and utility for stomach sampling. The letter was sent to the European Commission in Nov., 2011. A copy of the letter follows. WGSAM recognizes that in all areas this need is important and supports all initiatives to initiate stomach sampling considerations not sampled systematically before, for example the Skagerrak, Kattegat, Celtic Sea, and similar areas.

To Maritime Affairs and Fisheries DG

Unit E/2 (Fisheries conservation and control Baltic and North Sea)

Att. Mrs Ilona JEPSENA

Dear Mrs Jepsena

In support of policies for sustainable management strategies of living marine resources, ICES and European scientific institutes are faced with growing demands for integrated ecosystem advice on the long-term impacts of fisheries and predict effects of climate change, acidification and species composition and dominance. Europe has a legal commitment to maintaining 'good environmental status' and to 'restore stocks to levels that can produce the Maximum Sustainable Yield (MSY)'. The restoration of stocks to levels capable of producing MSY is likely to result in major changes in the ecosystem, moving levels of fishing mortality resulting in MSY (F_{MSY}) away from the currently perceived values. The effect of increasing stocks on future values of F_{MSY} cannot be assessed using single-species models but instead requires models incorporating species interactions as the effects of fishing and environmental change are spread through the ecosystem by complex foodweb interactions. At the core of all these models is information on who-eats-who and how much.

The ICES Working Group on Multispecies Assessment Methods (WGSAM) includes scientists from throughout the ICES Ecoregions and has expertise on a range of ecosystem modelling approaches used in addressing research and management questions. It is the principal forum servicing requests for information on species interactions (e.g. what MSY means in a multispecies context, what is natural mortality on key stocks as mediated via predation, etc.). WGSAM is tasked, on a regular basis, to provide updated estimates of predation mortality for inclusion in single-species stock assessments (e.g. by WGBFAS addressing the Baltic and WGNSSK addressing the North Sea).

Based on WGSAM's review of the status of ecosystem models, the group considers that the lack of up-to-date information on 'who eats who' for several ICES regions (and how these foodweb dynamics have changed) makes it increasingly difficult to provide adequate advice in some regions. There is an obvious danger when models using patchy or grossly out-of-date information are used to make future predictions.

The last comprehensive investigation of species interactions in the North Sea and Baltic was conducted 20 years ago and is unlikely be representative of what is now a very different ecosystem; since the early 1990s, major changes in ecosystems have occurred, including shifts in benthic, plankton and fish communities. Providing up-

to-date estimates of natural mortality is becoming a challenging task because of the lack of recent information on the diet composition of predatory fish. No model can provide reliable predictions unless it is calibrated with up-to-date information. In the absence of recent data, the modelling work may continue but the accuracy and relevance on the predictions made will deteriorate accordingly. Furthermore, Member States are obligated under the Commission Decision "on criteria and methodological standards on good environmental status of marine waters" (2010/477/EU), to assess the extent to which GES of marine foodwebs is achieved (Descriptor 4). Identifying useful indicators of foodweb relationships and their development over time requires an effort to collect the data necessary for estimating the indicators. Changes in foodweb interactions are most directly assessed through changes in the abundance and diet of the important species or species groups.

Therefore, ICES and the ICES Working Groups on Multispecies Assessment Methods, Assessment of Demersal Stocks of the North Sea and Skagerrak and Assessment of Baltic Fisheries asks DG MARE to give due consideration to supporting the process of collecting food composition data on existing surveys under a future call for tenders.

Sincerely,

Poul Degnbol

ICES Head of Advisory Programme

Jason Link

Co-chair of ICES Working Group on Multispecies Assessment Methods

Anna Rindorf

Co-chair of ICES Working Group on Multispecies Assessment Methods

Michele Casini,

Chair of ICES Baltic Fisheries Assessment Working Group

Clara Ulrich

Co-chair of ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak

5.2 DAPSTOM

The DAPSTOM initiative was established following a visit by two Cefas scientists to the Alaska Fishery Science Centre (AFSC), Seattle in 2005. Researchers at AFSC had developed a website for use by stock-assessment scientists and policy-makers to help draft the now obligatory 'ecosystem' section of stock-assessment reports, and specifically to determine what a particular commercial species eats and what predators eat that species (as juveniles or adults). Cefas and its predecessor MAFF, had been collecting data on fish stomach contents for over a century, but it was recognized that much of these data were only available in paper form or alternately in hundreds of individual spreadsheet files on the hard-drives of scientists (where they were at risk of being lost forever). In 2006 the EU Network of Excellence programme 'Eur-Oceans' issued a call for 'data rescue' projects and DAPSTOM (Phase 1) was one of the first to be funded. The DAPSTOM initiative has established an Internet portal whereby external researchers can search and download csv files from the online database, with periodic updates to the website approximately every year.

In July 2011 Cefas scientists uploaded version 3.6 of the database (and its accompanying report) to the website. This database contains 177,443 records (from 157,351 stomachs) and provides dietary information for 144 different species of fish. As such, this represents one of the largest compilations of foodweb data anywhere in the world. The vast majority of the data are derived from the North Sea (66.5%), but DAPSTOM-3 has added substantial holdings from the Irish Sea (8.5%) and the area around Spitsbergen (6.5%), where the UK used to have major fishery interests – see Figure 5.2.1. 286 individual research cruises have been digitized and there is now coverage throughout the past 120 years (see Figure 5.2.2). The earliest data were collected in December 1893, the most recent in February 2010.

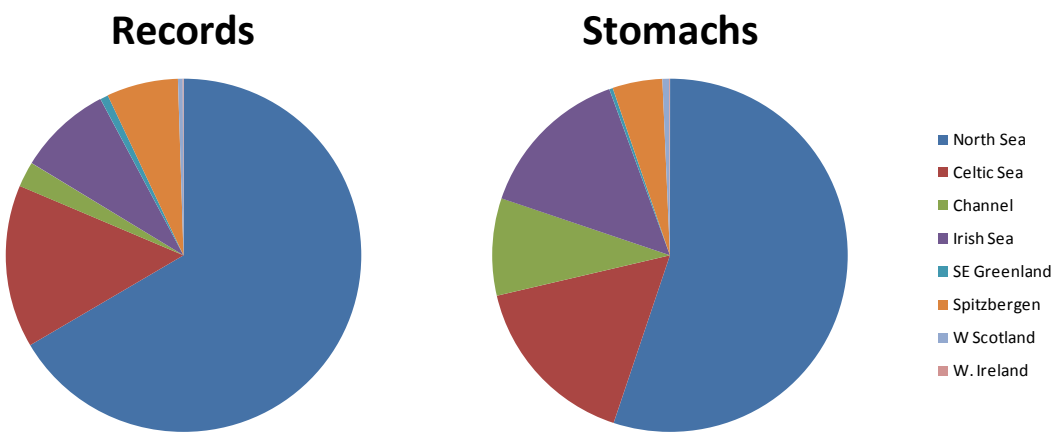


Figure 5.2.1. Proportion of all records and stomachs in the DAPSTOM database from each regional sea.

Figure 5.2.2. Temporal coverage of records contained in the DAPSTOM database by decade.

Major aspirations for the future will include: (1) to work with the ICES Data Centre to develop common database structures for holding historic and incoming fish stomach records, and perhaps to work towards a single database encompassing fish from all geographic regions; (2) further digitization of UK stomach datasets from the area around Spitsbergen, working with the Centre for Ecological and Evolutionary Synthesis (CEES) at the University of Oslo as well as Norwegian and Russian fishery scientists.

Phase 4 of DAPSTOM is currently being funded under the EU project 'EUROBASIN' and will result the addition of large quantities of blue whiting data from surveys throughout the ICES region. In addition efforts are underway to digitize further records from the 1880s, 1890s and early 1900s. Such data have already been used to highlight fundamental changes in the feeding preferences of North Sea fish over the past Century (see Pinnegar and Platts, 2011).

5.3 DINA – DIet and foodweb dyNAmics– exploration of old diet data with new analysis tools

The DINA initiative is a new project involving scientists from Centre of Ecological and Evolutionary Synthesis (CEES), Dept. of Biology, Univ. of Oslo, together with Institute of Marine Research (IMR), Bergen and Centre for Environment, Fisheries & Aquaculture Science (Cefas), Lowestoft. The project hopes to make use of fish survey data, fish stomach contents data and information on sea temperatures etc. collected by British scientists in the Barents Sea (and particularly around Bear Island) during the period 1936–1976. A proposal for this work was submitted to the Norwegian Research Council in August 2011 and will aim to 'better understand foodweb dynamics in the Barents Sea through analysis of spatial and temporal variation in diets and abundances of marine populations'. If the DINA initiative is funded, a technician will be employed in the UK to digitize the Barents Sea datasets, whereas statistical analysis will be carried out in Norway.

Initial analysis of the data spanning 1936–1950 has revealed significant differences in feeding preferences depending on season and locality, and that the diet of cod may have been somewhat different prior to the Second World War with many more records of cod eating herring and capelin (Figure 5.3.1). The dataset is of particular interest because it spans the 1930s and 1950s, periods when Barents Sea cod stocks and catches as well as sea temperatures were high and this is thought to be analogous to the present situation. It is important to deduce which prey species (e.g. capelin, krill, herring etc.) that might have been supporting this additional cod production. Much information on this issue has been and can be deduced from the extensive Russian diet dataset extending back to 1947 as well as being available for 1934–1938 (see e.g. Yaragina and Dolgov, 2011). However, for that dataset the raw data are not computerized and only aggregated data are available for analysis.

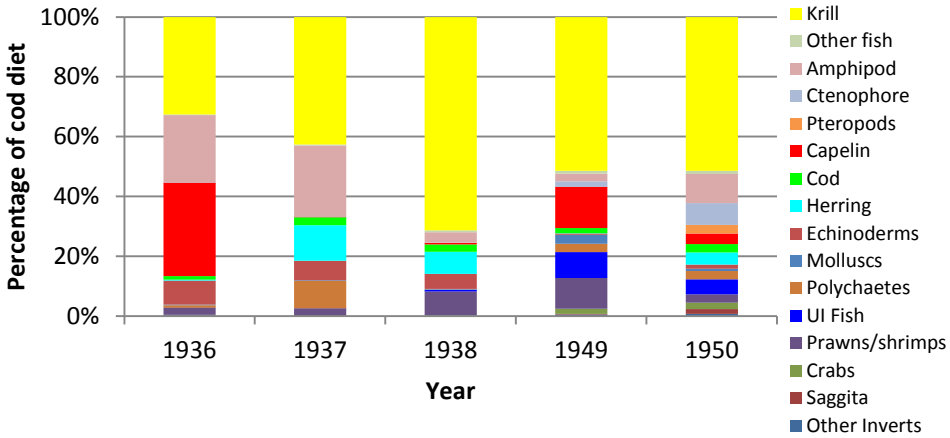


Figure 5.3.1. Proportion of diet represented by different prey types (number of prey items) in cod, for the area around Spitsbergen in 1936, 1937, 1938, 1949 and 1950 (number of records 67, 140, 329, 5004, 3136 respectively).

5.3.1 References

Pinnegar, J. K., and Platts, M. 2011. DAPSTOM - An Integrated Database & Portal for Fish Stomach Records. Version 3.6. Centre for Environment, Fisheries & Aquaculture Science, Lowestoft, UK. Phase 3, Final Report, July 2011, 35pp.

Yaragina, N. A., and Dolgov, A. V. 2011. Long-term variations in the importance of prey species for demersal fish in the Barents Sea under climate change. Poster, 15th Russian-Norwegian Symposium "Climate change effects on the Barents Sea marine living resources", Longyearbyen, September 2011 (Extended abstract will be made available in proceedings).

5.4 Icelandic stomach sampling in cooperation with fishers

Cod, haddock and saithe stomachs have been sampled by fishers Icelandic fishermen since 2001, 2008 and 2008, respectively. The sampling is coordinated by the branches of the MRI around Iceland and the data are analysed there. The fishers measure the length of the predator, take the stomach contents, put them into a plastic bag, register time and location and put them in a freezer. Location is often not registered but later obtained by MRI staff from the vessel's logbooks. The sampling is dispersed and only 2–4 samples are taken in each catch; the exact number depends on the catch as a species is only sampled if the catch is exceeds a fixed predetermined amount. The sampling involves a considerable amount of work, therefore the fishmen are paid approximately 400 Ikr (2.5 Euros) per sample.

The goal of the sampling program was to fill in spatio-temporal gaps in the extant stomach sampling programs (i.e. the surveys where the stomach contents, otoliths, etc. are collected). Fishing vessels are operating all year-round and use more types of gear than just bottom trawls. The groundfish surveys are conducted with a bottom trawl but large areas in Icelandic waters are untrawlable in groundfish surveys due to obstacles and low depth. Thus, the fishing vessels augment sampling in areas that would otherwise not be sampled.

The number of vessels participating vary from 7–12 in any given year. During the last two years MRI employees have started sampling from the landings of small handliners, which were the only vessels landing ungutted fish. MRI employees have also sailed aboard trawlers during the last two summers to assist with sampling, as considerably less is known about the diet of demersal fish in summer than winter and as such there has been interest specifically in knowing whether mackerel, which is now abundant in summer, is a major prey of large cod and saithe.

The constraint on the number of predators that can be sampled during each time period could limit the use of this kind of sampling in other areas where the species variety is higher. This problem could be addressed by making the guideline to sample the 3 species that seemed to be most abundant in each haul or use some other, similar criteria to also obtain samples from less common species.

The experience with the program demonstrates that it can indeed be done but involves significant communication with the fishers, with the results needing to be presented to them regularly. Experience has shown that the time spent analysing each sample is at least 20 minutes, including coffee breaks and other delays from work. But an additional benefit of this is that MRI personnel have time to communicate with the fishers, analysing the results. The average sample costs at least 2000–3000 Ikr.

Only one vessel has participated since the beginning. Usually sampling is discontinued when vessels are sold or key crew member leave the vessel.

In summary, the advantages of the sampling program are:

1) Continuous in time
2) Obtains samples from a variety of fishing gear.
3) Obtains samples from areas that cannot be trawled.

Disadvantages are

1) Only obtains samples from fish of catchable size

2) No control of where the samples are taken.

3) No control of the sampling. Is prey that is eaten in the trawl discarded and is the sampling random? These factors are not perfect in the surveys either, but merit consideration.

4) The sampling is not part of a survey that "in principle" gives spatial distribution of the predator. But the fishery does to some degree represent the spatial distribution of the predator, probably better if fishing effort is high and the predator is a large part of the total value.

5) The number of predators that can be sampled each time is probably limited.

6 ToR d) Explore how 'virtual multispecies datasets' (including survey, catch and stomach content data) for use in multiple multispecies models, especially for comparison and sensitivity testing, could be constructed

Multispecies models differ in their underlying structure and assumptions. To compare their performance and sensitivity to violations of these assumptions a common dataset is needed. Such a dataset would also create a test bed to accredit newly designed models. Additionally, systematic performance testing across models with increasing levels of complexity could be used to determine which details are necessary to adequately represent 'reality' for management decision-making, particularly for model skill assessment and management strategy evaluations.

The virtual dataset should initially focus on being relevant to a simple, minimum realistic model. Thus, the dataset should be one that adequately describes the dynamics of a system and includes the minimum data needed by the multispecies models (e.g. MSVPA, MSVPA-X, SMS, GADGET, SPMs) which will be compared. It is recognized that the virtual dataset could be expanded in future (e.g. to lower or higher trophic levels, additional species, more environmental covariates).

6.1 Structure of the Virtual System

It is suggested that to obtain an interesting multispecies system at least 2 forage fish, 2 predators (one of which is may be an apex predator), and at least one non-target species which has weak interactions with other species would be suitable (Figure 6.1.1).

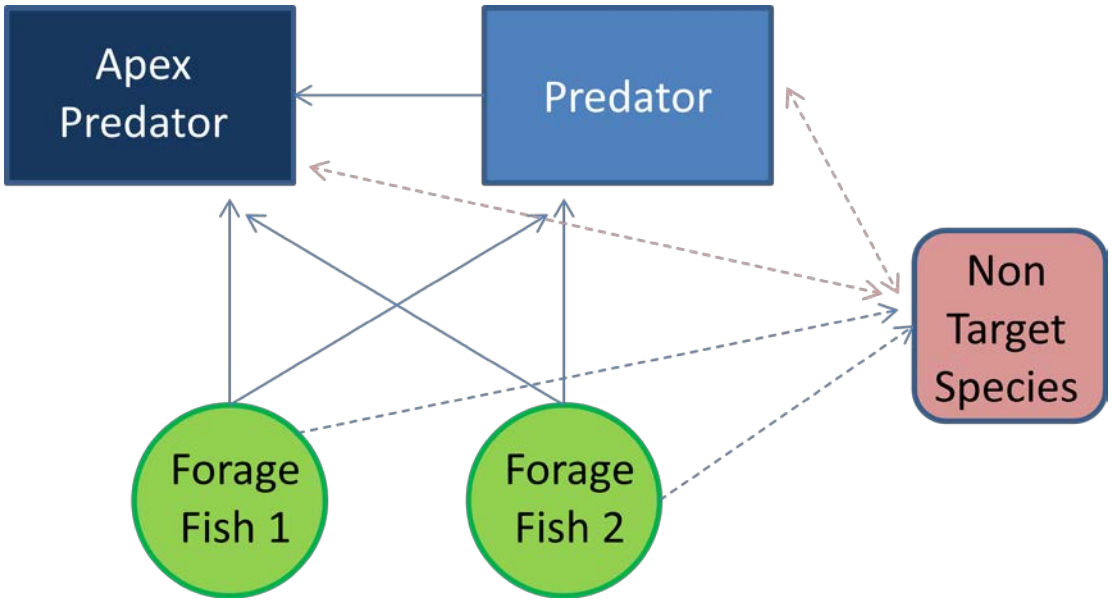


Figure 6.1.1. Schematic of the virtual ecosystem that would be the basis for the virtual dataset.

6.2 Input data needed

The full set of input data we propose for the virtual dataset will not be needed by all multispecies models, but it is important that the dataset can provide the necessary information for each model. The data should be at the finest temporal and spatial scale that is practicable. The data can then be aggregated to the level at which an individual model operates. The types of data and the considerations for such data for the virtual dataset are:

- All data, if applicable, generally needs to be age and length structured
- Survey indices of abundance/biomass
- Catches: total catches, preferable by landings and discards
- Diet composition (prey species and size/age of prey) and amount of food consumed
- Calorific value of each species
- "Other food" biomass estimates for different groups – size structured (e.g. small, medium, large)
    - benthic
    - pelagic
    - nekton
    - phytoplankton (likely not in the first cut)
- Length/age key and maturity ogive
- Length/weight key
- Ambient temperature – as an environmental covariate

6.3 Properties of Input Data

In addition to the data themselves, there are certain properties that need to be considered:

- **The data should be independent of the models used, internally consistent and biologically realistic**
- No significant outliers
- Spatial resolution: Initially 1 spatial unit, but the data should be able to be expanded to include more areas. A geo-referenced field could be added to the dataset to denote the spatial unit, or one could simply replicate the dataset for each spatial unit
- Temporal resolution: Should not necessarily be constrained by 'real world' sampling restrictions. It could be delivered with finer temporal resolution and then aggregated to the appropriate level for each model. Seasonal would be useful, but annual resolution is probably the minimum level required.
- Provision of the statistical distribution of each variable
- Data contrast desired: Different levels of variation in biomass/diets/etc both between species and across the time-series

Further, considerations of perfect data vs. errors are acknowledged as a non-trivial issue. To simulate real world usage of models, error structure should be included in the virtual data. The error is almost invariably not white noise, and often contains biases, skewed distributions, and auto- and cross-correlations. However, while defin-

ing the noise structure is difficult, actually converting the perfect data generated into data with a defined error structure is reasonably easy to implement by adding filters to the perfect data. Therefore, we consider it appropriate to focus initially on generating "perfect" accurate data, and leave designing suitable error generating filters to a later date.

6.4 How to create the Virtual Dataset

Creating the virtual dataset is not a trivial matter. Some of the data properties desired may contradict each other (specifically consistency and realism vs. independence from model assumptions). There are a number of possibilities to use in creating a virtual dataset. The consensus was to explore ATLANTIS, but to keep other options on the Table. First, the other options will be briefly described, and then the usage of ATLANTIS for this purpose will be more fully explored.

Excel Spreadsheet 'Model': This would have the inputs, equations, etc. to create the outputs by tweaking the values directly. It is a conceptually simple method to use, but errors in cell referencing are easy to make in complex excel models, and it would have to be a complex model to provide all the outputs desired.

Population Simulator: This is a data simulator designed to create populations with known parameters, processes and error structure. It was specifically made to provide virtual datasets for purposes of evaluating alternative estimation methods; however it does not include the possibility of environmental covariates, does not use diet data, and most importantly is single species in nature (NFT, 2011).

ATLANTIS: ATLANTIS is a bio-geophysical box model which can include processes ranging from sunlight to economics, although most ATLANTIS models are parameterized to emphasize the biological and exploitation dynamics of organism populations and the fleets that target (or at least catch) them (Fulton *et al.*, 2004a, b). Additionally, ATLANTIS has multiple functional forms to represent most of the included processes. Biological groups can represent individual species or multiple species with similar life-history characteristics. All vertebrates in the model are parameterized with 10 age classes, and invertebrates are either one biomass pool or split into adults and juveniles. There are numerous ATLANTIS models in various stages of development, but the two referenced as most likely to be useful in creating the virtual dataset described above are ATLANTIS NEUS which models the Northeast United States continental shelf large marine ecosystem (Link *et al.*, 2011), and an ATLANTIS model in the Baltic Sea. The Northeast United States model has the advantage of being a well developed model which is sufficiently advanced to produce the required outputs. In the contrast the Baltic Sea model is still in the early stages of development, but potentially offers a system with strong, well studied, environmental drivers.

One of the benefits of ATLANTIS is that it covers a wider range of ecosystem components than any of the multispecies models referenced above as being the models to be tested on the virtual dataset. Furthermore, with one exception described below, the model can output most of the required data either directly or with minimum post-processing (length-based data). Some of the outputs are biomass, catch (including landings and discards) per fleet, mortality per predator. It easily encompasses and exceeds the minimal spatial (any number of boxes) and temporal (any time-step – often half a day) resolutions required. The main issues discussed as being potential problems with ATLANTIS are:

- Only 10 age classes – for long lived species this would be a problem, but shorter lived species could be selected. For longer lived species some kind of interpolation would be required.
- Most current ATLANTIS models have 'smooth' outputs in their biomass, but there are parameters to allow for lognormal variation in recruitment which should permit less smooth outputs that are more representative of real world ecosystems. An initial test has shown that in principle, ATLANTIS can provide similar fluctuations in biomass using lognormal variation in recruitment, but the details remain to be worked out.
- The assumption of constant length-at-age – probably not a serious problem, but represents a potential limitation on the range of scenarios that could be explored.
- Stomach contents – this is the most serious problem identified in using ATLANTIS.

The ATLANTIS model can probably be used to create a diet matrix for the modelled predators. However the models to be tested take stomach data as inputs. Stomach data are typically zero inflated, noisy, and biased (by identification problems, differential digestion and evacuation rates). Creating artificial stomach content time-series from the diet matrices is an extremely non-trivial task. Although a lot of work has been put into the difficult task of trying to estimate diet compositions from stomach contents, attempting to go the other way is a new, and difficult, challenge. A transitional step could be to output the diet matrices and see if these can be used directly in the multispecies models (as fixed rather than optimized parameters perhaps). However this would represent a much easier task for the models than they actually face. The problem of stomach contents is not specific to ATLANTIS; rather this would be the key challenge for any attempt to create a virtual dataset for multispecies systems.

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7 ToR e) Work towards the inclusion of fleet dynamics in multi-species models

Modelling fleet interactions is important in multispecies models as fisheries are a major driver in most ecosystems and increased fishing effort will affect a number of species. However, the impacts depend, among other factors, on the gears used and the management systems in place.

In a system managed by TACs, an individual TAC is set for a number of species based on considerations of what might be an appropriate harvest rate for each species, but this does not necessarily lead to efficient fisheries due to variety of factors:

- Resiliency of different species to fishing varies, leading to different harvest rates.
- Availability of species to different gear varies.
- Assessment errors can lead to the realized harvest rate being considerably different from the intended one.
- The TAC portfolio of different fishing operations can make matters more complicated. Too large a portion of the TAC of some species might be owned by few companies (region dependent).

There is no link between these factors, so a species with low catchability could be short lived with even recruitment and thus tolerate high fishing pressure. Long lived species that can only sustain low fishing pressure could in contrast be easily caught and become a problem as bycatch. If some species abundance is below precautionary biomass levels, the advice will be no fishing making bycatch a major problem. The economic values of the species enter indirectly into the picture as fishers try to catch valuable species, even when they are difficult to get. From a fisher's point of view, value per effort unit is what counts.

Fishers have a number of tools available to catch their intended mixture of species:

1) Fishing location: This can range from relatively short distances that can be travelled in a few hours, or longer distances with locations that have to be selected before each fishing trip.
2) Time of day: This can affect the ease of catching target species (e.g. redfish during day, saithe during night).
3) Type of gear: Different gear catches different species even in the same area and time (e.g. longlines will get different catch composition from mobile gear).
4) Specific changes to the gear: e.g. bait if the vessel is a longliner, mesh size in gillnets and various minor changes to a trawl, even to another trawl.

Changing gear is a relatively expensive and difficult procedure for most fishing vessels, and therefore is often less of a possibility than the other methods fishers can use to influence their catch composition. There are exceptions though, for example longliners can change the type of bait they use relatively easily. It is likely that the fishing location is the most important tool that the captains have to change catch composition. How far vessels need to travel to get considerable change in catch composition varies, but locations with variable depth, bottom substrate and temperature give the best possibilities.

Over a period longer than one fishing trip, captains can plan ahead each year when they know what they are allowed to fish for each species and decide where to go for each trip. Larger companies have more flexibility, especially if they operate vessels using different types of gear. Such companies can also plan each year both based on expected seasonal changes in market prices for each species and the catch they are allowed to take for each species. Each of the captains receives orders about desired catch composition before each fishing trip. Plans for the year need to be adaptive as in all cases some species will be more difficult to catch than predicted, others easier.

From what is described above, it is clear that fishers can adapt to considerable variability in harvest ratios of different species in mixed fisheries, but as the harvest ratios become "less optimal" the fishery becomes less efficient, unreported mortality changes, and at some point the intended harvest ratios cannot be realized. But, although the total harvest ratios are reasonably balanced, different companies can realize unbalanced harvest rates due to their share of the TAC for different species.

As a specific example, in Iceland the species portfolio of each vessel often has some historical explanation as a result of their fishing areas over time, usually the areas close to their home harbour. Major changes in the spatial distribution of species can cause problems as the TAC might be distributed according to historical spatial distributions which no longer apply. In Icelandic waters this has happened with haddock, where fish of suitable size have been abundant in the north since around 2000. However, most of the TAC is owned by companies in the south, leading to a higher fishing mortality of haddock there, and further increasing the proportion inhabiting the north. In the long run, the fleets might adapt to the species composition in the region, but the species composition may change too rapidly compared to inertia in changing the fleet.

But how do we model these considerations in multispecies assessment models? Many of the issues are region specific and implementing all the details in fleet behaviour is difficult. We present four different methods that could be used to model fleet behaviour.

7.1 Gadget approach

The Gadget approach of multifleet, multiarea models is to treat the fleets as predators with suitability for each prey, and this has proved to be a useful approach (SMS or even biomass models should be able to do similar things within their limits of aggregation). The effort for each fleet, area, and time-step needs to be linked to the annual TAC for each species, compiled dynamically from measures like estimated biomass at the beginning of the year. The intended catch will most likely be impossible to achieve for all species under such a model.

7.2 Hypothetical management approach

An alternative method is to design the management system so that it will lead to sensible behaviour, with characteristics as proposed here.

For each prey and fleet there would be a length or age based suitability function. The length and age based function should preferably not be area based.

For each fleet there could be a penalty for a non-optimal ratio of target fishing mortality of species occurring together in the catch. Deviations from the optimal level will most likely show up in reduced efficiency of the fisheries and lead to discards when they become too large. Efficiency is reduced as fishers might have to avoid the best

fishing areas when they are trying to avoid some species where the TAC is easy to reach.

Reduced efficiency is something we need to consider as it can lead to increased trawling to get the required catch, something that is generally recognized as unsuitable for the ecosystem due to a variety of factors.

For work on mixed fisheries we could probably provide a matrix describing correlation between species. A correlation matrix would have to be based on analysis of logbooks or surveys and be used to mitigate or minimize potential bycatch and discards.

Additional complications arise from factors like relative price of each species as well as size composition in different areas. Effort control does not help much in solving the problem: some things are easier, others more difficult. Effort can in principle only be defined for fisheries with one type of gear in a relatively homogenous area. It can however be used in a relative sense by reducing number of effort units for each gear by x% if the intended harvest ratio for a species has been exceeded by x$. In multispecies fisheries it might be argued that the reduction will be according to the species where the intended harvest ratio is exceeded by largest amount.

The final point is that availability of species that occur in multispecies fisheries should be a factor in setting an appropriate harvest ratio, not solely resilience to harvesting based on single species approach.

7.3 ATLANTIS Approach

There are numerous parameters in an ATLANTIS model that can be used to simulate behaviour of the fisheries, from both the management and fishers side. Management can be based on TACs which can be static or dynamically change based on state variables within the model (e.g. biomass levels relative to starting levels). Management levers are not confined to TACs, and can also include marine protected areas (MPAs), changes to selectivities or catchabilities to simulate changes to the type of gear, and others.

In the Northeast US continental shelf ATLANTIS model (Link *et al.*, 2011), fleets are defined by gear type, target species, and bycatch on each non-target species in the model. Their behaviour is controlled through the following parameters:

- Initial effort allocation assigned to different boxes and depth layers
- Proportional effort change parameter – If one of the two thresholds described below is exceeded, this is the proportional change to effort applied to the fleet.
- A maximum cpue threshold – In ATLANTIS NEUS, if a fleet catches above this level, it increases effort by the proportional effort change parameter.
- A minimum cpue threshold – The same as the maximum cpue threshold, except if a fleet catches below this level, it decreases effort by the proportional effort change parameter. It is also possible to add the ability for fleets to do exploratory fishing which changes where and how deep they fish, if this threshold is exceeded.
- A maximum effort cap which prevents effort from rising above a limit.

This relatively simple effort model tends to lead to complex yet relatively reasonable results. It is important to note that the cpue is biomass based, and does not consider

economic value, and the model is probably best used in the context of exploring fleet behaviour in a relatively short time scale of about 5–10 years, as the lags between the fishery and target species can become too great – especially if the target species are reduced to very low levels and then recover.

7.4 FLBEIA Framework with multispecies component

A different approach has been developed in Azti, where a multispecies component has been added to an existing fleet dynamics model. It is the FLBEIA (FL Bio-Economic Impact Assessment) framework. This is a new R package (R Development Core Team, 2011) that has been built on top of FLR libraries (Kell *et al.*, 2007), aiming to provide a flexible and generic tool to conduct Bio-Economic Impact Assessments of harvest control rule based management strategies under a Management Strategy Evaluation (MSE) framework (Butterworth and Punt, 1999; De la Mare, 1998; Punt and Donovan, 2007). As such it is divided in two main blocks, the operating model (OM), and the management procedure model (MPM). In turn these two blocks are divided in three components. The OM is formed by the biological, the fleet and the covariates components and the MPM by the observation, the assessment and the advice components. The fleet component of the OM is where the fleet dynamics can be captured, whereas the covariates component is where the multispecies relationships can be included, by using a variety of mathematical functions. The stock component is obviously where all the relevant stocks of each system can be defined.

The model is multistock, multifleet and seasonal with uncertainty being introduced by means of Monte Carlo simulation. The algorithm has been coded in a modular way to facilitate the checking process and the flexibility of the model. The library provides functions that describe the dynamics of the different model components under certain assumptions and the user chooses which of the functions are used in each case specific model implementation. Furthermore, if in a specific case study or scenario, the functions provided within FLBEIA do not fulfil the requirements for some of the components, the user can code the functions that adequately describe the dynamics of those components and use the existing ones for the rest of the components. As the users can construct their own models by selecting existing submodels and constructing new ones, we can define this FLBEIA as a framework more than as a model. The package is still under development but most of its functionalities are already working. The package is being used in several case studies with very different peculiarities, Hake and associated mixed fisheries in Western Waters, Anchovy and associated sequential fisheries in Bay of Biscay, Northeast Atlantic Blue ling and associated Mixed Fisheries, Beaked Redfish and Sea bream artisanal fisheries. Except for the last two case studies the rest are multistock and multifleet-multimetier case studies. At the moment there are no functions to model trophic interactions but something is planned within the Anchovy case study. The main limitations of the model are that the stocks must be age structured or aggregated in biomass (length structure is not allowed) and that spatial dimensions are not considered explicitly. Spatial characteristics could be modelled by assigning stocks and/or fleets/métiers to specific areas.

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8 ToR f) Explore simple statistical relationships between M and B among predator and prey from output of multispecies models

Multi Species and ecosystem models are complex and their usage in standard assessments is avoided because of that. Nevertheless their output can be used to inform standard single species assessments. Estimates of historic natural mortality have already been used for the assessment of North Sea cod and whiting, for example. A further step could be to examine simple relationships between predation mortality (M2) and predator fields for the prediction of M2 values from single species assessment outputs.

In general, predation mortality estimates in multi species and ecosystem models depend on the following:

- Predator abundance
- Suitability (or something similar as vulnerabilities in EwE) as measure for preference and availability of prey
- Functional feeding response relating M2 to prey abundances (whole prey field)
- Assumptions on consumption rates

In the following three sections we tested methods for generating simple indices relating M2 of a prey to the biomass of its predators, and evaluated whether they are good enough to capture the most important trends in estimated predation mortalities despite ignoring variability caused by changes in the prey field (functional feeding response). Analyses were carried out for the North Sea and Baltic Sea (representing different degrees of foodweb complexity) using the same regression method. An alternative method is presented for Iceland in a much simpler predator prey system, cod and shrimp. Finally, a foodweb model was used to simulate a range of potential ecosystem and prey biomass levels and develop statistical relationships between walleye pollock predators and prey in the Gulf of Alaska.

8.1 North Sea and Baltic Sea

8.1.1 North Sea

The North Sea has a complex foodweb and the North Sea SMS model currently includes 26 predator and prey species. From the latest SMS key run, output predation mortalities for each prey and prey age combination as well as biomasses of the various predators were extracted. In addition, partial predation mortalities averaged over all model years for each predator, predator age and prey, prey age combination were utilized to give predators with a higher impact on the prey stock a higher weighting. With these standard output data two different linear models were fitted:

1. $$M2_{prey,preyage} = \sum_{p=1}^{n} BIO_{predator,predatorage}$$

2. $$M2_{prey,preyage} = \sum_{p=1}^{n} BIO_{predator,predatorage} * \overline{pM2}_{predator,predatorage,prey,preyage}$$

The partial mortality weighting factor $\overline{pM2}_{predator,predatorage,prey,preyage}$ is a measure for predator preferences but is also influenced by the general biomass level of a certain predator. Values of $\overline{pM2}_{predator,predatorage,prey,preyage}$ used are available for download at http://www.ices.dk/reports/SSGSUE/2011/WGSAM/pM2_for_relationship_between_M_and_B.csv. As just an average over all years is used, it only sets a constant weight for each predator. The dynamic between the years in the index is driven by the biomass of the predator only. Therefore, the annual predation mortalities and the weighting factor were regarded as independent enough to be used in the regressions. In lack of actual values, the weight of harbour porpoise and grey seals was set to 100 kg.

The second model led to better relationships for all predator–prey combinations (R^2 values were higher). In general, for 0 group prey the relationship between M2 and the index was weakest (Figure 8.1.1, Table 8.1.1). For 0-group cod and whiting the relationship was not significant. Other processes such as changes in the prey field (i.e. functional feeding response) were likely more important than the predator biomass in the sea. For older prey age groups all relationships were significant except for whiting age 3. However, R^2 values showed a wide range from 0.15 to 0.78 (Figure 8.1.1, Table 8.1.1). In the last 10 years hardly any larger systematic increases or decreases in the index occurred, and variability of estimated M2 was mainly driven by other modelled processes. This, however, can change in future if important predator stocks can be recovered.

8.1.2 Baltic Sea

The Baltic ecosystem is much simpler than the North Sea ecosystem. In the SMS model, cod is the only predator preying on cod, herring and sprat (+ Other Food). As for the North Sea, predation mortalities, predator biomasses and partial mortalities averaged over all model years were extracted from the latest key run and the same linear models were fitted.

The relationships between M2 and the index (including average partial mortality) are generally stronger in the Baltic than in the North Sea due to the simpler foodweb structure. Reasonable relationships were found for all prey species and age classes (Figure 8.1.2, Table 8.1.2). The R2 values of the linear regressions were all above or equal to 0.6 except for cod age 3 and all relationships were significant. In the last 10 years predation mortalities were nearly stable except in the last 2–3 years, where an increase in predation mortality in accordance with the recovery of Eastern Baltic cod could be observed (Figure 8.1.2, red points and lines). This shows that the need of a regular update of M2 values depends on the speed of recovery or collapse of predator stocks.

8.1.3 North Sea and Baltic Sea Conclusions

It turned out that there are indeed simple relationships between the estimated predation mortalities from complex multi species models and predator biomasses but not for all prey types. Especially the dynamic of 0-group predation mortalities in complex foodwebs as the North Sea was often driven by other processes such as the functional feeding response of predators. Additional weighting of predator biomasses with partial predation mortalities averaged over all years was beneficial. However, even if simple relationships exist for a number of interactions, this does not necessarily mean that predation mortalities depend mainly on predator biomass in reality. Important

processes such as changes in predator–prey overlap or inside the "Other Food" component are not taken into account in the SMS models used for the analysis in the Baltic and North Sea. Therefore, variability of predation mortalities coming from these additional processes is not visible in the model estimates of M2. Nevertheless, the simple linear regressions fitted in this analysis may help to inform single species assessment groups. A Table with the average partial mortalities will be provided on the working group site together with the equations in this report (Table 8.1.1 and 8.1.2). The Table with partial mortalities includes the weighting factors but also shows who eats whom and what predator species and predator age classes have to be taken into account for each prey when calculating the predation index.

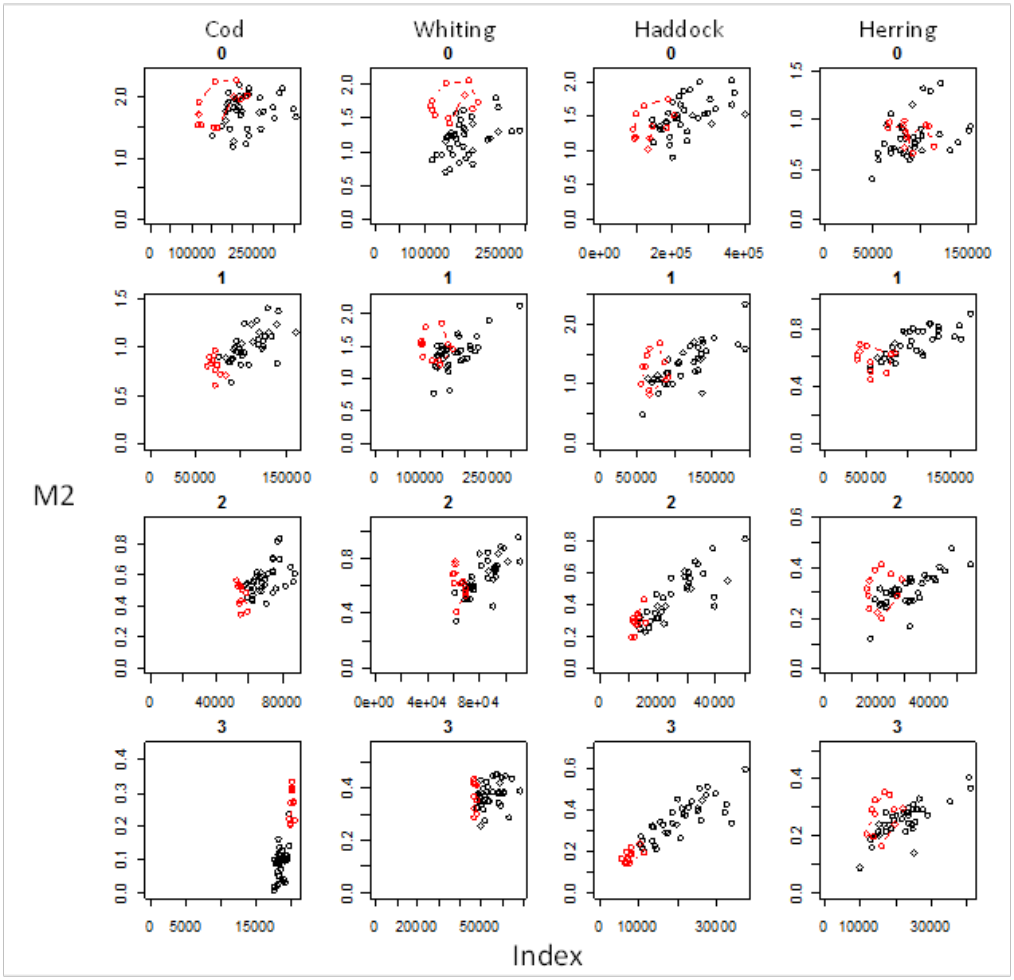

Figure 8.1.1. Relationships in the North Sea SMS key run 2011 between predation mortality M2 and an index taking into account predator abundance and average partial M2 over the model years for different prey species and prey age classes. Red points and lines indicate the last 10 years.

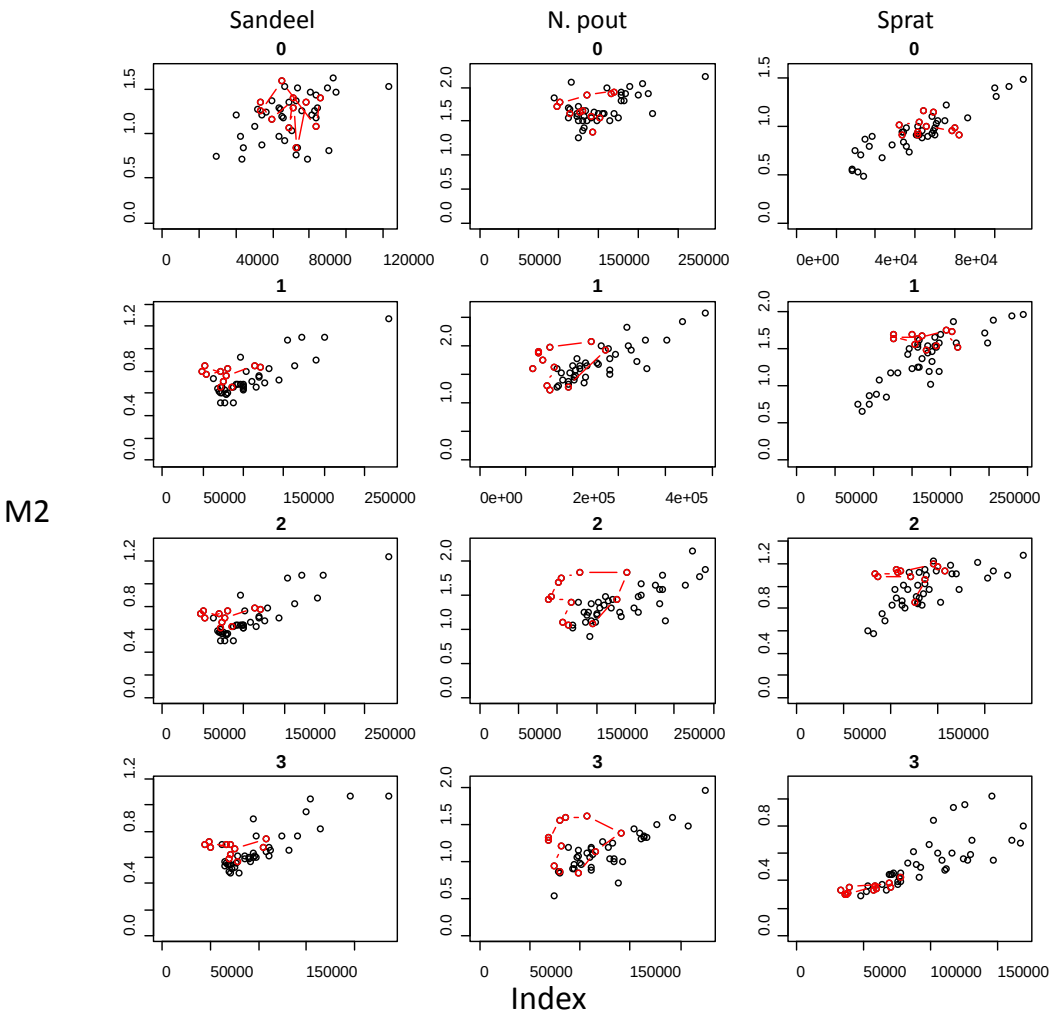

Figure 8.1.1 (continued). Relationships in the North Sea SMS key run 2011 between predation mortality M2 and an index taking into account predator abundance and average partial M2 over the model years for different prey species and prey age classes. Red points and lines indicate the last 10 years.

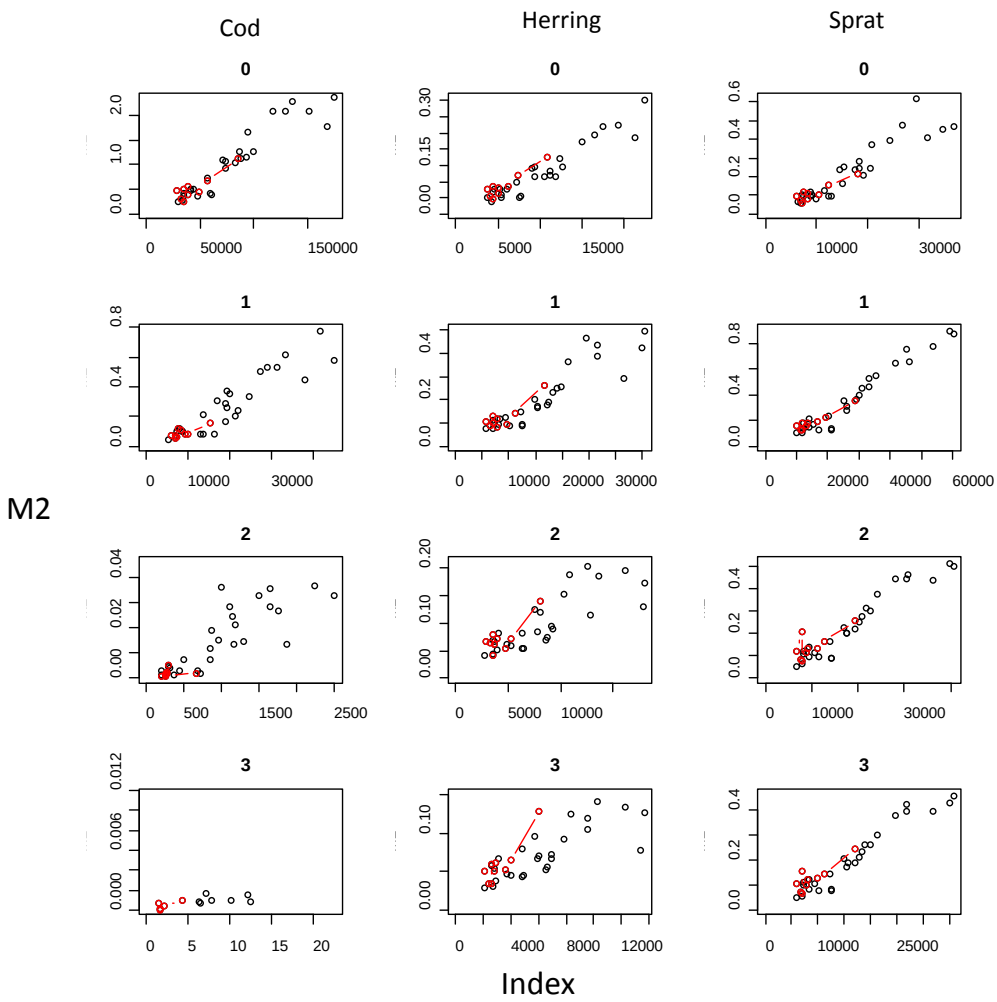

Figure 8.1.2. Relationships in the Baltic SMS key run 2010 between predation mortality M2 and an index taking into account predator abundance and average partial M2 over the model years for different prey species and prey age classes. Red points and lines indicate the last 10 years.

Table 8.1.1: Results for the North Sea of the linear regressions between M2 and an index taking into account predator abundance and partial mortalities averaged over all years. Bold values indicate significant relationships.

Species	Age	Type	Estimate	Pr(>|t|)	R^2_adjusted
Cod	0	Intercept	1.59E+00	0.00000	0.01
Cod	0	index	8.90E-07	0.22146	
Cod	1	Intercept	4.50E-01	**0.00000**	0.48
Cod	1	index	5.19E-06	**0.00000**	
Cod	2	Intercept	1.55E-01	**0.07303**	0.31
Cod	2	index	5.93E-06	**0.00002**	
Cod	3	Intercept	-1.43E+00	**0.00000**	0.64
Cod	3	index	8.29E-05	**0.00000**	
Whiting	0	Intercept	1.18E+00	0.00000	-0.01
Whiting	0	index	8.10E-07	0.49840	
Whiting	1	Intercept	1.02E+00	**0.00000**	0.15
Whiting	1	index	2.40E-06	**0.00345**	
Whiting	2	Intercept	2.12E-01	**0.01612**	0.37
Whiting	2	index	5.70E-06	**0.00000**	
Whiting	3	Intercept	2.90E-01	0.00010	0.01
Whiting	3	index	1.66E-06	0.19781	
Haddock	0	Intercept	1.09E+00	**0.00000**	0.27
Haddock	0	index	1.78E-06	**0.00010**	
Haddock	1	Intercept	6.83E-01	**0.00000**	0.42
Haddock	1	index	6.00E-06	**0.00000**	
Haddock	2	Intercept	1.27E-01	**0.00006**	0.72
Haddock	2	index	1.27E-05	**0.00000**	
Haddock	3	Intercept	1.03E-01	**0.00000**	0.78
Haddock	3	index	1.20E-05	**0.00000**	
Herring	0	Intercept	6.04E-01	**0.00000**	0.07
Herring	0	index	2.52E-06	**0.03397**	
Herring	1	Intercept	4.59E-01	**0.00000**	0.54
Herring	1	index	2.22E-06	**0.00000**	
Herring	2	Intercept	1.96E-01	**0.00000**	0.28
Herring	2	index	3.97E-06	**0.00007**	
Herring	3	Intercept	1.45E-01	**0.00000**	0.36
Herring	3	index	5.39E-06	**0.00000**	
Sandeel	0	Intercept	7.73E-01	**0.00000**	0.15
Sandeel	0	index	6.10E-06	**0.00367**	
Sandeel	1	Intercept	4.59E-01	**0.00000**	0.55
Sandeel	1	index	2.72E-06	**0.00000**	
Sandeel	2	Intercept	4.11E-01	**0.00000**	0.60
Sandeel	2	index	2.85E-06	**0.00000**	
Sandeel	3	Intercept	3.83E-01	**0.00000**	0.58
Sandeel	3	index	2.95E-06	**0.00000**	

Species	Age	Type	Estimate	Pr(>|t|)	R^2_adjusted
Norway pout	0	Intercept	1.28E+00	**0.00000**	0.22
Norway pout	0	index	2.75E-06	**0.00044**	
Norway pout	1	Intercept	1.10E+00	**0.00000**	0.44
Norway pout	1	index	2.52E-06	**0.00000**	
Norway pout	2	Intercept	9.58E-01	**0.00000**	0.24
Norway pout	2	index	2.60E-06	**0.00024**	
Norway pout	3	Intercept	6.77E-01	**0.00000**	0.30
Norway pout	3	index	4.10E-06	**0.00004**	
Sprat	0	Intercept	3.69E-01	**0.00000**	0.76
Sprat	0	index	9.21E-06	**0.00000**	
Sprat	1	Intercept	5.38E-01	**0.00002**	0.58
Sprat	1	index	5.28E-06	**0.00000**	
Sprat	2	Intercept	6.55E-01	**0.00000**	0.30
Sprat	2	index	2.56E-06	**0.00004**	
Sprat	3	Intercept	1.43E-01	**0.00175**	0.63
Sprat	3	index	4.04E-06	**0.00000**	

Table 8.1.2: Results for the Baltic of the linear regressions between M2 and an index taking into account predator abundance and partial mortalities averaged over all years. Equations in bold indicate significant relationships.

Prey	Age	Equation	R^2 adjusted
Cod	0	**-1.52E-1 + 1.51E-5* index**	0.9
Cod	1	**-5.47E-2 + 1.68E-5 *index**	0.86
Cod	2	**-1.86E-3 + 1.69E-5*index**	0.71
Cod	3	**2.99E-4 + 8.5E-5*index**	0.45
Herring	0	**-5.71E-3 + 1.39E-5*index**	0.89
Herring	1	**-7.32E-3 + 1.36E-5*index**	0.85
Herring	2	**2.65E-2 + 9.16E-6*index**	0.66
Herring	3	**2.77E-2 + 7.68E-6*index**	0.6
Sprat	0	**-3.74E-2 + 1.59E-5*index**	0.87
Sprat	1	**-7.13E-2 + 1.62E-5*index**	0.95
Sprat	2	**-1.52E-2 + 1.37E-5*index**	0.9
Sprat	3	**-1.26E-2 + 1.36E-5*index**	0.91

8.2 Iceland

A different way of testing if M is proportional to predator biomass was used for deep-water shrimp in Icelandic waters, where M was assumed to be proportional to some measure of biomass of cod, an important predator of shrimp. The measures tested were:

1) Total stock size of immature cod

2) Indices of cod from groundfish surveys in the area.

3) Indices of cod from a survey directed at shrimp, conducted in July.

The 3rd set gave much better fit to the data, something that was relatively obvious in this simple system. There are large deviations in some earlier years though, and in recent years the model incorrectly predicts reduction of the shrimp stock when is has been stable.

The system shown here is much simpler than those shown in the North Sea and Baltic and the effects of the predator more obvious.

Deep-water shrimp was among the most valuable species in Icelandic waters in the 1990's, with the landings reaching 60 000 tonnes from 1994–1997 (Figure 8.2.1). The fishing area is large, around 70 000 km^2 (Figure 8.2.2), in mostly cold, relatively deep waters off the north coast.

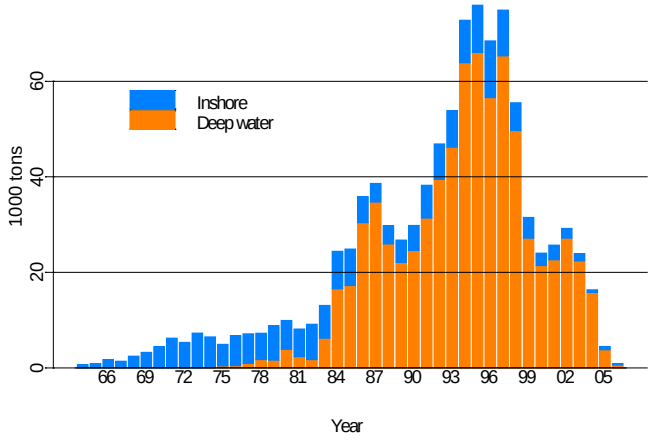

Figure 8.2.1. Landings of shrimp from Icelandic waters.

Stomach samples have indicated that cod eats more shrimp than the fleet catches, therefore possible reduction in shrimp fisheries was one of the options when an "optimal" HCR for Icelandic cod was evaluated in 1994. The proposal was that the yield from the shrimp stock would be reduced by 50% (with large confidence intervals) if the cod stock increased according to the premises of the HCR. Also, estimated consumption by cod was always included in all assessment models for the shrimp; most often, the predation on shrimp was linked to the size of the immature part of the cod stock. All models indicated that the predation was much less than direct calculation from stomach samples indicated.

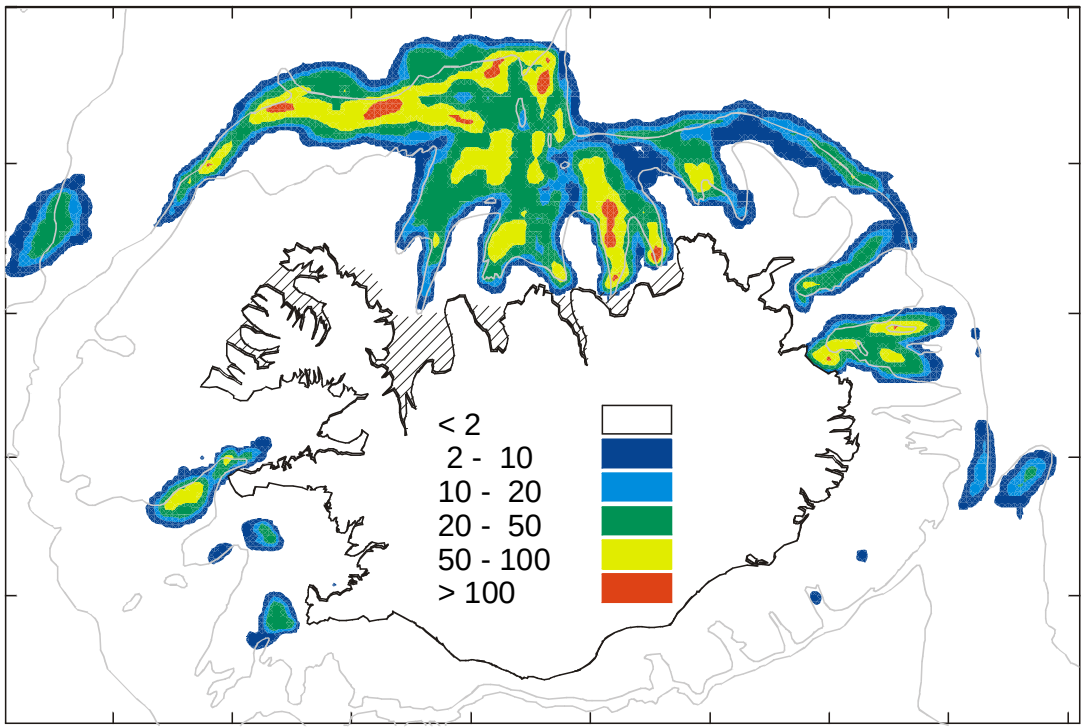

Figure 8.2.2. Fishing areas for shrimp in Icelandic waters 1987–2006. The figure shows tonnes per square nautical mile.

In 1997, the shrimp stock became severely reduced (Figure 8.2.3) as demonstrated by both cpue and, importantly, landings. Predation by cod is the most likely cause of this reduction, but abundance indices from the surveys in the areas did not indicate any major change in the cod stock (Figure 8.2.4). Abundance indices of cod from the shrimp survey did, on the other hand, indicate a major increase in the cod caught in the survey and the indices have been at high level since then (Figure 8.2.4). The discrepancy between indices from the groundfish surveys in the area and the shrimp survey may arise from seasonal differences in survey timing, and the shrimp survey is conducted in deeper colder water, sometimes on a bottom where applying the groundfish survey trawls is not possible. Why the cod are going into the deep waters is not clear, bottom temperature from the shrimp survey does not indicate any warming.

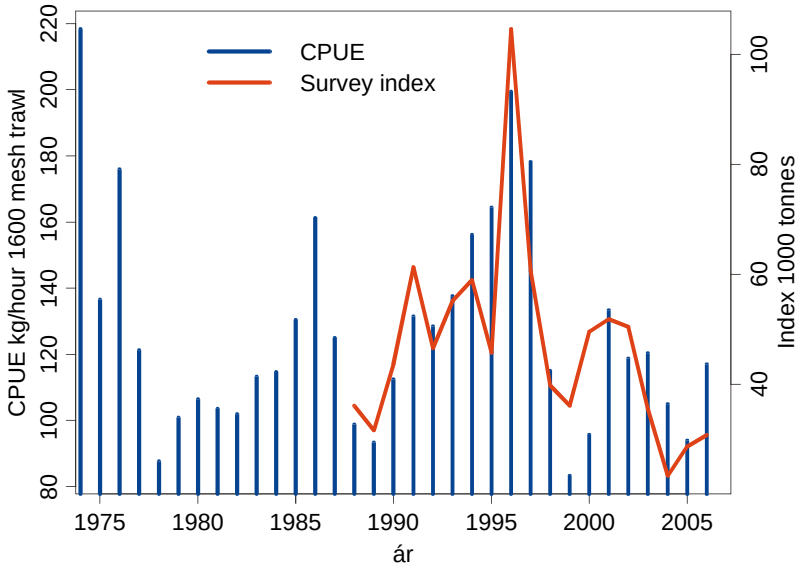

Figure 8.2.3. Abundance of shrimp in the July shrimp survey and cpue from the commercial shrimp fishery.

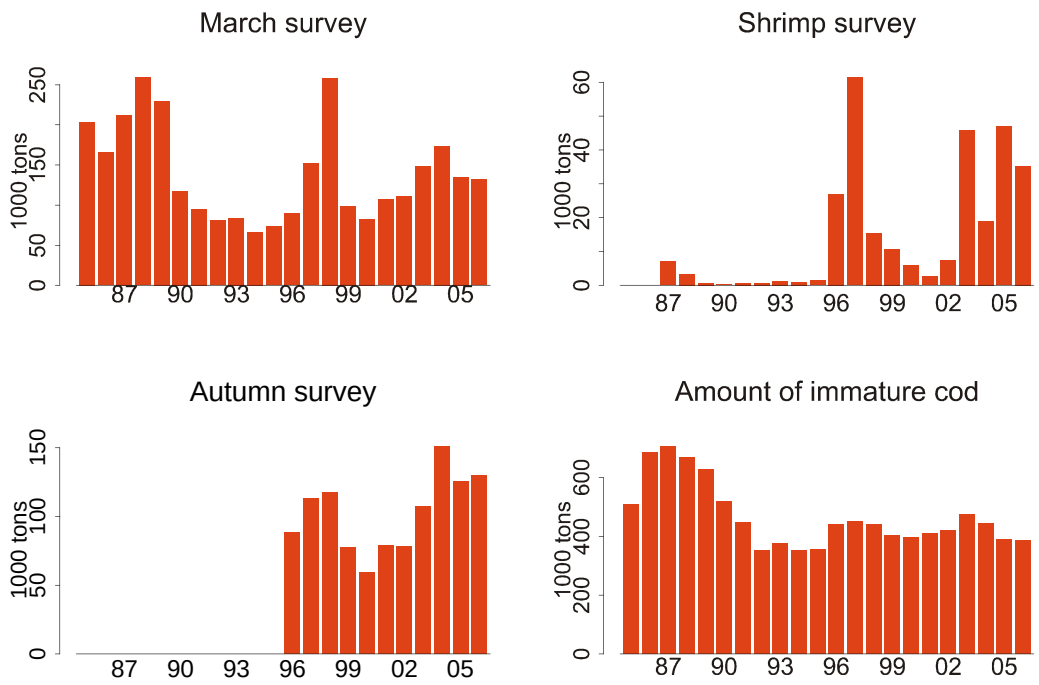

Figure 8.2.4. Different measures of cod abundance. The survey abundances are for the shrimp region and neighbourhood.

An assessment based on a 2 region Gadget model had real problems following this trend, so the model was simplified by assuming that the predation mortality on shrimp was proportional to an abundance index of cod in the area. Different abundance indices and combinations of indices were tested, but the indices from the shrimp survey gave the best fit to the data. However, they did not fit well to observations in 1987–1988 when the abundance of immature cod was high and the shrimp stock was at a low level.

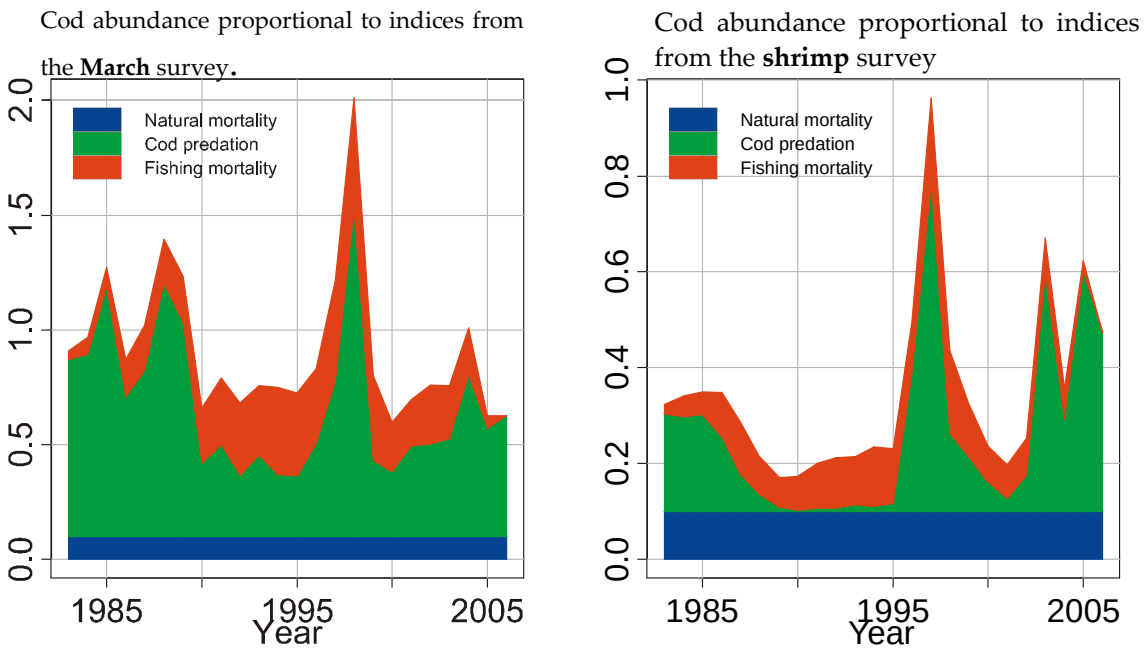

Figure 8.2.5. Mortality estimates from the Gadget model based on two different measures of cod abundance.

For a few years, the model where the predation of cod is linked to abundance of cod in the shrimp survey was used in assessment and prediction. In prediction, the assumption was that the cod abundance would be similar to last year or the mean of the last 3 years, but some stochasticity was included. Recruitment to the stock was estimated from the abundance of small shrimp in the shrimp survey, but they have been at low levels in recent years.

Given the combination of low recruitment (Figure 8.2.6) and high cod biomass (mortality), the model predicted continued depletion of the stock (Figure 8.2.7) assuming the cod abundance in the shrimp survey represents predation mortality on shrimp. The reality has been that with relatively limited catch, the shrimp stock has increased somewhat since 2007, mostly the largest shrimp, an indication of reduced mortality. Possible explanations for the discrepancy are

- Cod is preying less on large shrimp than the model anticipates.
- Interaction between cod predation and fishing effort, where fishing makes the shrimp more available for cod by preventing schooling.
- The shrimp survey takes place in July and does not describe the abundance of cod in other months. Using a mixture of all the surveys (one for each season) did not improve the fit.
- The survey indices of small shrimp are not proportional to number in stock. This is a known problem in many fish stocks and can be treated by a power curve. Usually the physics behind the relationship is not known.

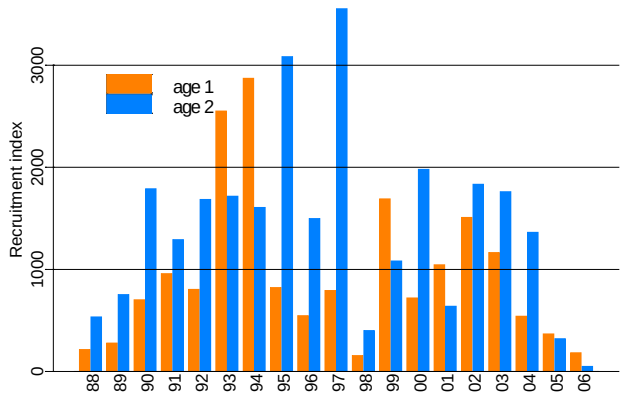

Figure 8.2.6. Recruitment indices for shrimp in the shrimp survey. The data are based on result from small fine meshed bag fixed to the trawl.

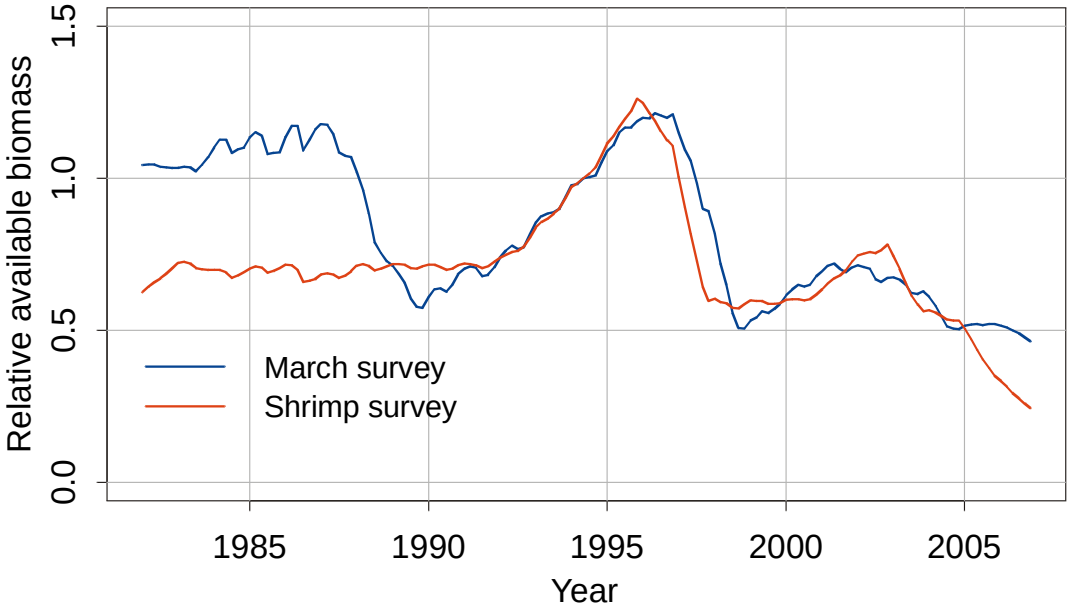

Figure 8.2.7. Development of the available biomass from the models where cod abundance is proportional to the March survey and the shrimp survey, same as in Figure 8.2.5.

8.3 Gulf of Alaska, North Pacific

Gulf of Alaska Walleye pollock (*Theragra chalcogramma*) are subject to an economically important commercial fishery and are also important prey for many groundfish and other predators in the ecosystem. A mass balance foodweb model (Aydin *et al.*, 2007) provided information on fishing and predation mortality sources for pollock (Gaichas *et al.*, 2010), and was used to develop simple statistical relationships between pollock M and B for key predators (Gaichas *et al.*, 2011). Previous work had shown that pollock M was likely to change over time in response to changing predator biomass (Hollowed *et al.*, 2000). Mortality estimates from the foodweb model identified arrowtooth flounder (*Atheresthes stomias*) as a key predator of juvenile (<20 cm) pollock, responsible for nearly half of total M. A combination of three groundfish predators—Pacific cod (*Gadus macrocephalus*), Pacific halibut (*Hippoglossus stenolepis*), and arrowtooth flounder—accounted for about 70% of adult pollock M in the base foodweb model. Using the EcoSense methods described in Aydin *et al.* (2007), parameter uncertainty was incorporated by generating over 9000 alternative feasible Gulf of Alaska foodwebs with a range of primary production regimes, pollock biomass, and predator biomass levels. Results from these Gulf of Alaska ecosystem model runs were used to estimate functional relationships between pollock mortality and predator biomass which considered up to three different levels of pollock biomass and ecosystem-wide primary productivity.

Results showed fairly strong relationships between the biomass of the three key predators and total pollock mortality; the relationship was especially strong between juvenile pollock mortality and arrowtooth flounder biomass (Figure 1). In general, these distilled results suggest that pollock mortality increases with predator biomass more quickly and to a higher level when pollock biomass is relatively low (red lines), and that individual predators affect pollock mortality to a lesser extent when Pollock biomass is high (green line). However, at low and intermediate pollock biomass (red and blue lines in Figure 8.3.1), ecosystem productivity can influence the relationship as well. For example, the thin blue line demonstrates that under conditions of low primary productivity, pollock mortality may increase quickly with predator biomass

although pollock biomass is at an intermediate level. Under conditions of higher ecosystem primary productivity, these distilled model results suggest that pollock mortality increases less quickly with increasing predator biomass (thick blue line).

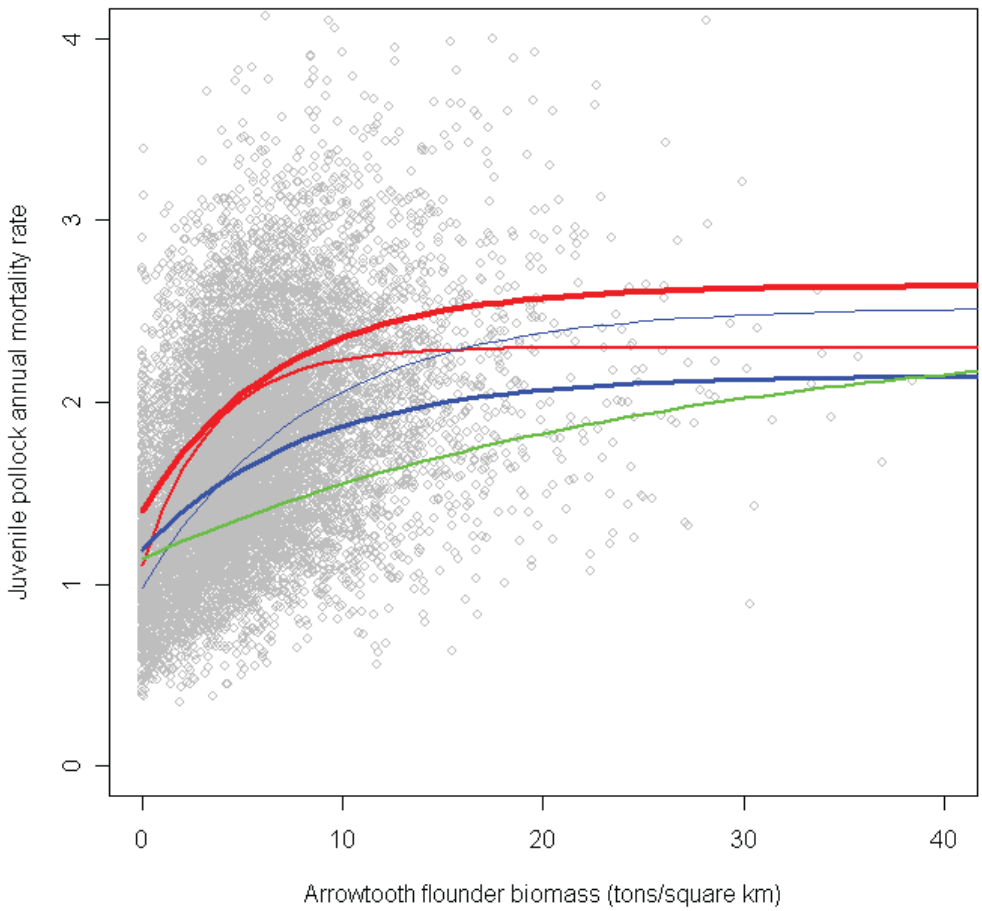

Figure 8.3.1. Relationship between arrowtooth flounder biomass and juvenile pollock mortality from 9,845 simulations with the Gulf of Alaska ecosystem model (Aydin *et al.*, 2007). Each open circle is the output of an individual model run; these model runs were conducted with parameters varying according to our uncertainty in the underlying information, so the whole spread of points represents a wide range of pollock and ecosystem conditions as well as our uncertainty in the underlying parameters. Each line represents the best fit relationship between flounder biomass and pollock mortality under different levels of pollock biomass and overall ecosystem production. Red lines indicate low pollock biomass (below the stock assessment reference point indicating 25% of unfished biomass), blue lines indicate intermediate pollock biomass, and the green line indicates high pollock biomass (above the stock assessment reference point indicating unfished biomass). Thickness of lines represents overall ecosystem production. Thinner red and blue lines indicate primary productivity at or lower than estimated current levels, and thicker lines indicate higher primary productivity than estimated current levels. Primary productivity had no impact on the relationship when pollock biomass was high (single green line).

8.3.1 References

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9 ToR g) Improve the quantification of the role of top predators (marine mammals, seabirds, large pelagic) on forage fish in the ICES area ecosystems)

9.1 Methods used to estimate consumption by top predators

Parameterizing predator consumption involves three steps; calculating food requirements (how much a single predator eats), diet composition (what type of prey they prefer), and predator population size (how many are doing the eating). There are a number of different methods available for estimating each of these, depending on the available data and the type of predator involved. The details will be different in each area and species, but some general principles are common to all areas. All involve different sources of uncertainties.

Food requirement is typically estimated using direct data on the diet, or by using bioenergetic modelling. Direct data may involve stomach samples, but can also come from scats (hauled out pinnipeds) or direct observation (nesting seabirds). All of these data sources have difficulties. Stomach samples give direct data on what was eaten, but this is complicated by uncertainties around digestion and evacuation rates, and differential speeds of digestion of different preys. Direct observation of seabirds bringing fish home to the nest gives information on how many the chicks are fed, but not on adult consumption throughout the rest of the year. Scats from hauled-out mammals combines difficulties with estimating digestion with the restriction that samples are typically only available for part of the year. A different approach is to use bioenergetic modelling, attempting to estimate the energetic requirements of the predator, and then converting that into the amount of food required to supply that energy. Data from captive animals may also be used, although it is often questionable how relevant this is to the actual consumption of a wild animal. A final resort where little information is available is to borrow strength and assume that the consumption will be similar to "similar" species.

Diet composition is typically estimated from the same sources of data, with similar difficulties, but compounded by the need to know the availability of potential prey. Stomach contents give direct information about what was found in the stomach at time of capture, but it is non-trivial to convert this to average consumption (due to difficulties with differential digestion and evacuation mentioned above). Shore-based analysis (scats or direct observations) give information about what the food composition was during the times the predator was close to the shore, but not on the rest of the year. Isotope analysis can analyse the chemical composition of the tissues of the predator, and give indications of what sources of food were used to grow those tissues, but the results can only give broad indicators rather than precise answers. Calculating the feeding functions (the desirability of different prey, and how this changes with changing abundance of different prey types) requires knowledge of the amounts of different prey available to the predator in the feeding areas. Total estimates of stock abundance are likely to be of variable quality (with commercial species typically better known), and knowledge of the local abundance in the feeding areas is generally lower than for the overall stock. Again, borrowing strength from similar species may be required where data is lacking.

A key difficulty in estimating *top predator populations* is that they are often not the subject of commercial hunting, and thus abundance data can be sparse and research

resources limited. For animals that breed on land (seabirds and pinnipeds), direct observations during the breeding season can give good data on population size, but data on whale and dolphin populations is less readily available. Sighting surveys can be used to give population estimates, and these are a relatively low cost addition to existing fisheries surveys, but the data can be rather noisy. A related difficulty is that the predators are often highly mobile, and it can be difficult to estimate what fraction of the population occupies a given area at a given time. It is often the case that uncertainties around population abundance can be the largest sources of uncertainty in overall consumption estimates.

9.1.1 Estimating population size

Abundance estimates of top predators have been shown to be one of the most influential input parameters for estimating population-level consumption by these predators (Col 2011). It is therefore critical to obtain the most accurate abundance estimates possible for marine mammals, seabirds and large pelagics. Where possible, dedicated aerial and shipboard cetacean surveys should be conducted using standardized transects with dive-time and boat affinity or aversion corrections, as well as circle-back methods for aerial surveys. Seabird abundance estimates can also be obtained on shipboard cetacean surveys. For cetaceans with identifiable characteristics (e.g. fluke patterns on humpback whales or barnacle patterns on right whale heads), a photo identification database can be used to estimate minimum population abundances for the area of interest. Commercial whale-watching tour boats can be ideal platforms for obtaining photos for these databases. Pinniped abundance estimates can be obtained through dedicated rookery-area aerial surveys during pupping season, with expansion factors used to convert from pup counts to entire populations.

9.1.2 Estimating daily food requirements

Bioenergetic modelling is the most commonly used method to estimate the energy consumption by sea mammals. To use this method, information about animals' field metabolic rate, energy storage and food digestibility is needed. Additionally, one needs to have knowledge of the population demography (length/age structure, sex ratio, age/length of maturity and pregnancy rate) of the population because the growth of juvenile animals and the cost of being pregnant must be accounted for. The energy consumption by a sea mammal population can be expressed as:

$$EC = \sum_{s=1}^{S} \sum_{t=1}^{T} \frac{(GF_{s,t} + FMR_{s,t} + P_{s,t})}{E_{fu}} \cdot N_{s,t}$$

where EC is the annual energy consumption, summarized across T equally spaced time-steps and S age/length groups, E_{fu} is faecal and urinary efficiency, GF is a growth factor, the additional energy required by juvenile animals, FMR is the basal metabolic rate and P is the energy stored and N is number of individuals.

In cases where some of the above parameters are not available, daily per capita consumption can be estimated using the Kleiber (1975) equation:

$$Y = \alpha M^b$$

Where Y is ingestion, M is body weight of a predator, and α and b are consumption parameters that can be obtained from published literature. In order to quantify the uncertainty of these consumption parameters, pairs of α and b parameters can be randomly sampled from a pool of published values for similar predator species.

9.1.3 Estimating diet composition

Diet compositions for marine mammals can be obtained from stomach samples of by-catch and stranded animals. Although there are biases associated with these opportunistic samples, these sources still provide valuable information on prey species, and should be considered the best available data in regions where directed sampling is not allowed. Conversion factors should be used to estimate biomass of prey species from otolith and cephalopod beak lengths when prey is well digested, taking into account erosion of the hard parts. Scat samples from haul-out sites can be similarly used to estimate diet composition by weight for pinnipeds. When primary data cannot be collected, diet compositions from published literature can be used to estimate average diet compositions for each marine mammal species in the area. It is preferable to use published sources of diet compositions from the region of interest, but if this is unavailable, data can be borrowed from other areas, substituting similar taxonomic prey groups that are found in the area of interest. However, care should be taken to only include diet compositions that are representative of the area of interest, especially with regard to depth, since diet compositions can differ greatly between nearshore and offshore areas.

9.2 Ecoregion B: Barents Sea (and Norwegian Sea)

Work is in progress to compile an Atlantis model for the Barents and Norwegian Seas, although this model is not yet operational. However the work involved in preparing the model involves collecting the best available knowledge of the populations, consumption and diet composition of the major top predators, and this information is summarized below (Table 9.2.1). Seabirds have been grouped into Arctic and Boreal groups, while the marine mammals have been treated as individual species. Total removals by top predators and the fishery are compared in Figure 9.2.1.

Table 9.2.1. Population size, consumption rate and diet composition of marine mammals and seabirds in the Barents Sea and Norwegian Sea.

Species	Mean body size	Population size	Consumption rate (% bodyweight Daily)	Diet composition
Polar bear	275kg (adult), 200kg (whole population)	1900 – 3600 (Aars *et al.*, 2009)	20% (based on stomach size)	50% adult bearded seal, 5% pup bearded seal, 20% adult ringed seal, 10% ringed seal pup, 15% harp seal (from stomach samples near Svalbard, but known to be large geographical variability). Also recorded eating walrus, narwhal, carrion, seabirds. (Derocher *et al.*, 2002)
Killer whale	5000kg	2000 in Barents Sea, 5000 in Norwegian Sea	3% of body weight (from bioenergetics, Williams *et al.*, 2004)	herring; mackerel; salmon; seals; young humpback, minke and fine whales (Mauritzen, pers. com.)

Species	Mean body size	Population size	Consumption rate (% bodyweight Daily)	Diet composition
Sperm whale – few data	34,322kg (Dommasnes *et al.*, 2001)	Population: 5,200 (Nøttestad and Olsen, 2004), only males in region (Mauritzen, pers. com.)	3%*bodyweight (from killer whale data)	
Humpback whale	32,500kg	500 Barents Sea, 500 Norwegian Sea (Nøttestad and Olsen, 2004)	3% of body weight (from cons. of 0.5–1 tonnes per day, American Cetacean Society)	krill, capelin, herring, polar cod, blue whiting, mackerel, haddock, saithe (Mauritzen, pers. com.)
Minke whale	5,000kg	c. 60,000 (Skaug *et al.*, 2004) to 80,000 in Barents Sea (Nøttestad and Olsen, 2004), c. 30,000 in Norwegian Sea (Nøttestad and Olsen, 2004).	3% of bodymass/day (total prey consumption c. 1.8 million tonnes)	haddock, saithe, NSS herring, NEA cod, capelin, krill, zooplankton (kovacs09 and haug02). Juv. Herring 38.4%, capelin 6.7%, cod 12.1%, other gadoids 11.5%, krill 27.7%, sandeel 2.8% (Folkow *et al.*, 2000), remainder unidentified other fish.
Fin whale	45,000kg	1700 in Barents Sea, 3000 in Norwegian sea	3% of bodyweight (assumed)	polar cod, capelin, shrimp, krill, zooplankton, mackerel, herring, blue whiting, haddock, saithe (more fish in north, more zooplankton in south; Kovacs *et al.*, 2009)
Bearded seal	275kg for fully grown adults, 35–40kg for pups, 230kg for whole population	4,000 – 100,000, very poorly known	4.9% day (assumed from ringed seals)	35% polar cod, 16% L. Medius, 8% cod, 6% cottidae, H. Platessoides 3%, crustacean 18%, gastropods 11%, bivalves 4% (Hjelset *et al.*, 1999)
Harp seal	80kg	from c. 2 million individuals (Nøttestad and Olsen, 2004), 1.36–1.63 million individuals (Øigard *et al.*, 2011) and 2.22 million individuals (Nilssen *et al.*, 2010) in Barents Sea, 437,700 in Norwegian Sea (Nøttestad and Olsen, 2004).	4.9% of body weight (assumed from ringed seals)	rustaceans, capelin, polar cod, herring, cod, other fish, spatially variable within the Barents Sea (Nilssen *et al.* in prep, no data on Norwegian sea diet). Remainder is invertebrates and unidentified fish.

Species	Mean body size	Population size	Consumption rate (% bodyweight Daily)	Diet composition
Hooded seal	270kg	13,100 (Nøttestad and Olsen, 2004).	4.9% of body weight (from ringed seals)	small pelagic, Greenland halibut, redfish, polar cod, cephalopods, krill (Kovacs *et al.*, 2009)
Ringed seal	50 kg	100,000 Barents Sea	4.9% of body weight (Ryg and Øritsland, 1991)	72% polar cod (mostly age 1), 2% gadoid larvae, 1.74% gadoid (mostly cod), rest predominantly crustaceans (Labansen *et al.*, 2007)
Boreal seabirds	663 g	6.1 million individuals	20% of body weight	herring, small pelagics (sandeel), 0-group gadoids, invertebrates 13% fatty fish, 61% lean fish, 26% invertebrates (Barrett *et al.*, 2006)
Arctic seabirds	640 g	20 million individuals	24.8% of body weight	45% fatty fish, 46% invertebrates (Barrett *et al.*, 2006)

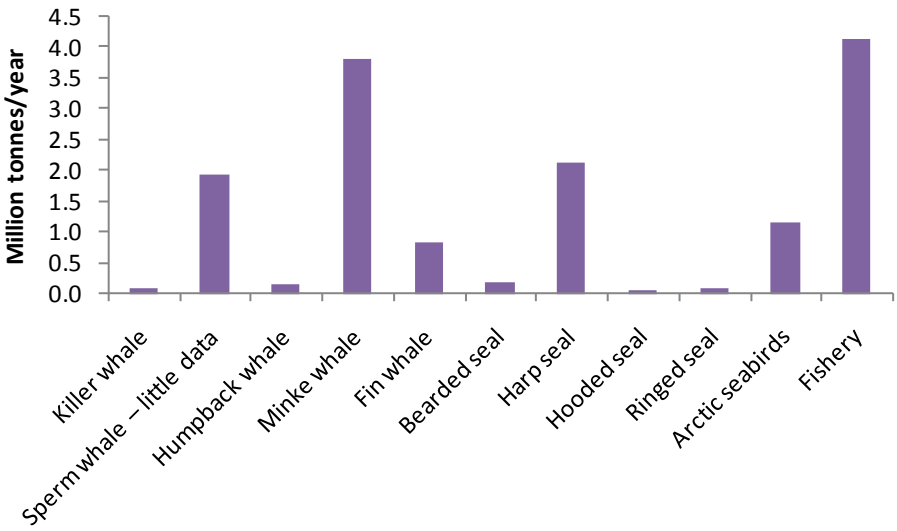

Figure 9.2.1. Total consumption by marine mammals, seabirds and the fishery in the Barents Sea and Norwegian Sea

9.2.1 References

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9.3 Ecoregion F: North Sea

The diet of the major predatory fish, seabirds, grey seals and harbour porpoise are modelled using the SMS model as described in Section 4.1. In addition to these predators, the diet of minke whale and harbour seal were also estimated.

9.3.1 Minke whale

A small resident population of minke whales are present in the North Sea year-round, but numbers increase during summer with the arrival of whales migrating into the North Sea from elsewhere. Seasonal changes in abundance were modelled using step functions, see Figure 9.3.1 (Nils Oien pers com).

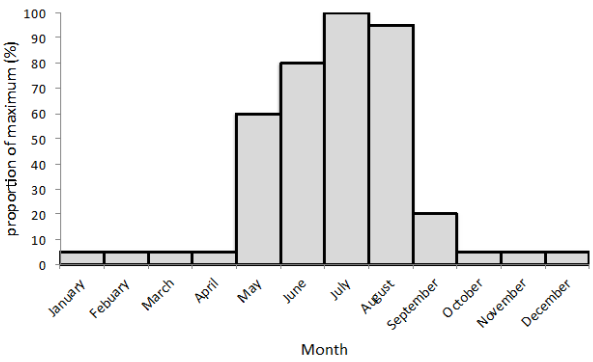

Figure 9.3.1. Seasonal variation in the abundance of minke whales in the North Sea. The size of the population through the year is represented as a percentage of the maximum population size, which is observed in July.

Annual estimates of peak abundance are available for some years, and these vary, though no general trend is evident (Skaug *et al.*, 2004). In order to estimate consumption for all years of the study, we set abundance at the mean value of all available estimates. We then calculated consumption rates based on the maximum and minimum observed population sizes. Diet composition of whales was estimated from a limited sample of minke stomach contents from 37 animals collected between 2000–2004 (Windsland *et al.*, 2007; Figure 9.3.2), and an 'average' diet was assumed for all North Sea whales. Stomachs were sampled in summer, but winter diet is likely to contain fewer sandeels (because sandeels are not available to pelagic feeders in winter). Therefore, we assumed a winter diet with identical prey composition, but without sandeel. An intake rate of 80kg per day for each whale was assumed.

9.3.2 Harbour seal

Focused, localized studies of harbour seal diet based on scat data are reported in the literature (Hall *et al.*, 1998, Brown and Pierce 1998, Sharples *et al.*, 2009, Tollit *et al.*, 1998, Thompson *et al.*, 1991, Tollit and Thompson, 1996; Figure 9.3.2). Though the pattern of sampling is not consistent over time and space, it is nonetheless clear that diets vary between sites, seasons, and years. This variation is probably related to local prey availability, but unfortunately we do not currently have sufficient data to attempt to fit a model of this relationship. Therefore, a fixed diet was assumed, and this was based on the mean of observations from diet studies carried out from 1993 onwards. The relative contributions of samples from different geographical areas were weighted according to the number of seals observed there during aerial surveys during this period. Daily consumption rates of 4kg per animal were assumed.

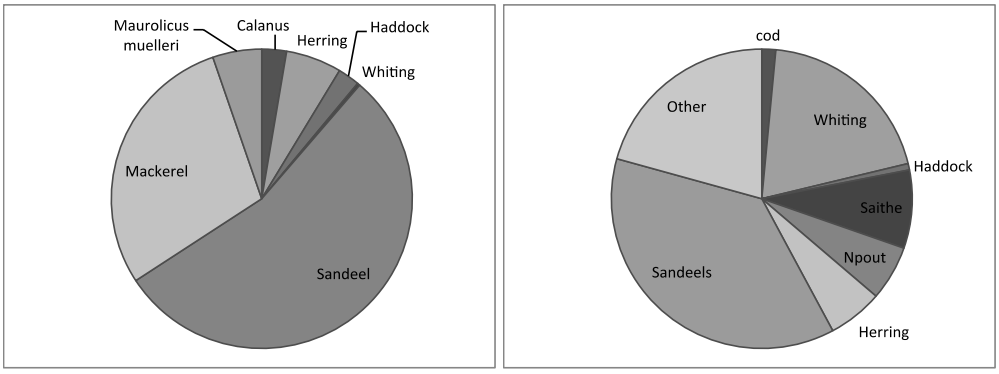

Figure 9.3.2. Diet composition of minke whale (right) and harbour seal (left).

Population estimates of harbour seals were based on counts carried out at UK haulout sites used by seals during the moult, corrected to take into account of the number of seals likely to be at sea at the time of the count (SCOS 2010). Survey results were available for all the important North Sea UK sites from 1993 onwards, but sites were not all surveyed in all years. As yet there is no population model for North Sea Harbour seals so, where some sites had not been surveyed in some years, the 'missing counts' were estimated using linear interpolation. Totals for the North Sea were then calculated based on the sum over all sites.

9.3.3 Distribution of removals on predators and fishery for different prey species

Predatory fish are by far the most important source of fish removals in the North Sea for the majority of species. However, when comparing the fishery to removals by marine mammals, seabirds and fish predators, the fishery comes out as most important in mass removals only for cod and herring (Figures 9.3.3 and 9.3.4). Whales (harbour porpoise) contribute significantly only to removals of cod and whiting. Seabirds are important for the three pelagic species, though their share of total removals never exceeds 10%. Seals are significant predators on cod and haddock.

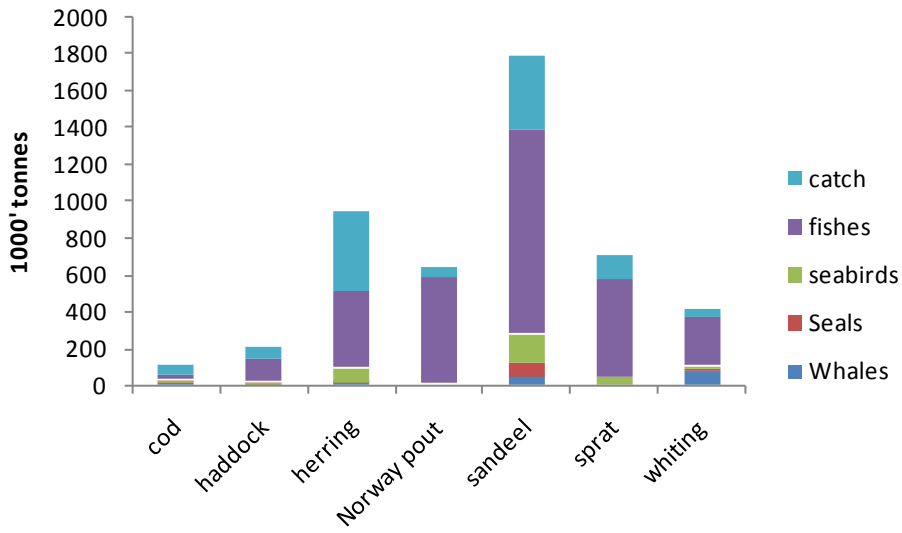

Figure 9.3.3. Yearly removals of different prey species by different sources on average over the years 2000 to 2010.

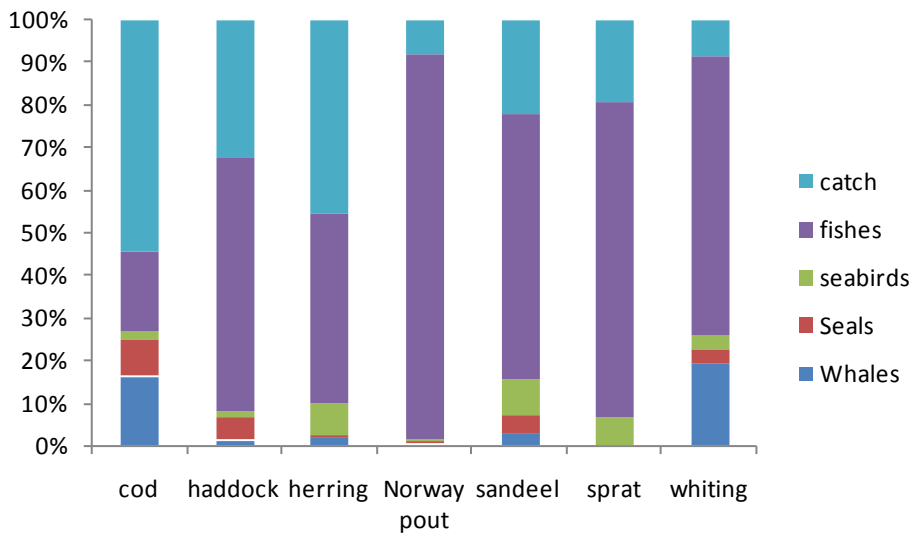

Figure 9.3.4. Relative contribution to removals of average fish consumption and fisheries catch in the years 2000 to 2010.

9.4 Ecoregion G: South European Atlantic Shelf

The diet of albacore and bluefin tuna in the Bay of Biscay and surrounding waters has been assessed in AZTI-Tecnalia through stomach content analysis. 1354 albacore and 579 bluefin tunas were sampled between 2004 and 2010. Stomach sampling is also going on in 2011.

One of the most important features of albacore feeding in the Bay of Biscay and surrounding waters is the geographic variability of the type of trophic resources and in the associated foraging behaviour displayed by albacore. The main pattern in this variability is the difference between shelf break areas and more oceanic areas. Albacore diet in shelf break areas appears to have a globally larger proportion of fish (saury, blue whiting, anchovy) than in oceanic areas in which small crustaceans (krill, amphipods) represent a higher part of this diet (Goñi *et al.*, 2011a). Average daily consumption of anchovy by albacore is around 10 individuals by day and predator, but reached a maximal value of 103 individuals in 2010 after anchovy recovery (Goñi *et al.*, 2011b)

In terms of feeding strategy, in shelf break zones albacore appear to feed in the epipelagic layer during both daytime and night, whereas in oceanic zones they appear to feed in the epipelagic layer by night and to dive into mesopelagic and/or bathypelagic layers to feed by day (Goñi *et al.*, in prep).

Bluefin tuna distribution in the Bay of Biscay is more reduced than albacore distribution, and focused on the Southeastern corner of the Bay, which is the core area of anchovy distribution. Although opportunist, bluefin tunas appear to have a more selective diet than albacore, with a preference for anchovy that – when present – can constitute 80% to more than 99% of the mass of bluefin stomach contents. Anchovy predation rates by bluefin tuna are estimated to be around twice the predation rates by albacore, with an order of magnitude of 20 individuals by day and predator.

9.5 Ecoregion: US Northwest Atlantic

Recently there have been a number of questions raised regarding the economic and ecological impacts of marine mammals through their consumption of fish and interactions with commercial fish stocks. Laurel *et al.* looked at estimates of fish predation by marine mammals on the Northeast US (NEUS) continental shelf, and compared daily marine mammal consumption estimates from the NEUS to daily consumption in other regions. For the NEUS, they compared annual population-level marine mammal consumption to commercial fisheries catch and fish predation to provide context for the orders of magnitudes of these estimates. Their study provides initial quantitative bounds on marine mammal consumption of six prey groups including clupeids, scombrids, sand lance, hakes, large gadids (cod, haddock, pollock), and flatfish. Species-specific consumption was estimated for humpback whales, fin whales, minke whales, pilot whales, bottlenose dolphin, Atlantic white-sided dolphin, common dolphin, harbour porpoise, grey seals, and harbour seals, using ranges of daily individual consumption and diet compositions compiled from literature values. Consumption was expanded to annual population levels based on abundance estimates and annual residence of each species in the area. Bounds on consumption estimates of each marine mammal species were determined using Monte Carlo simulations, and total marine mammal consumption was then summed for each prey group.

Results indicate that daily per capita consumption was similar in estimates from the NEUS and in previous studies off of Europe for two marine mammal species of interest, grey seal and harbour porpoise. When compared to mean individual weights, grey seal daily individual consumption was about 3% of body weight (90% CI: 1.6–4.8%, Figure 9.5.1), and harbour porpoise daily individual consumption was about 7% of body weight (90% CI: 3–11%) for the NEUS estimates (Figure 9.5.2).

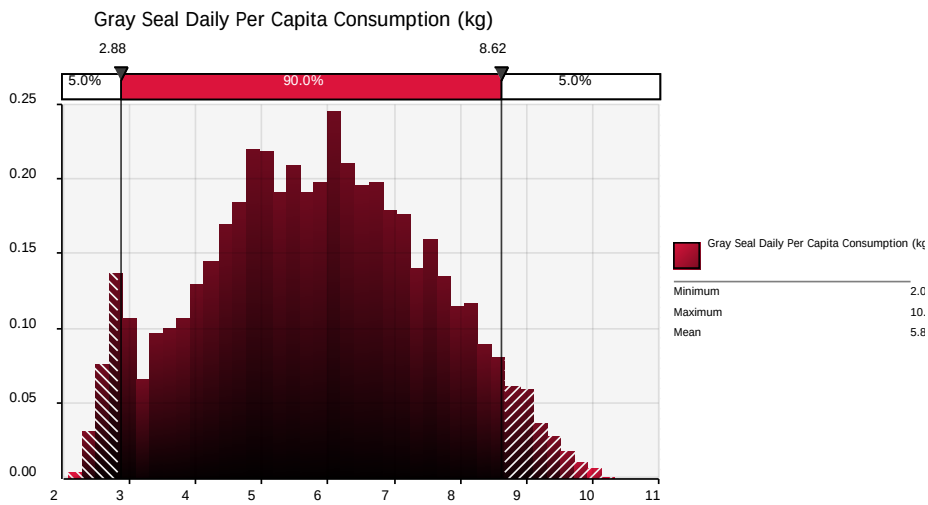

Figure 9.5.1. Ranges of estimated grey seal daily per capita consumption, incorporating uncertainty from marine mammal individual weights and random sampling of daily consumption parameters from published literature.

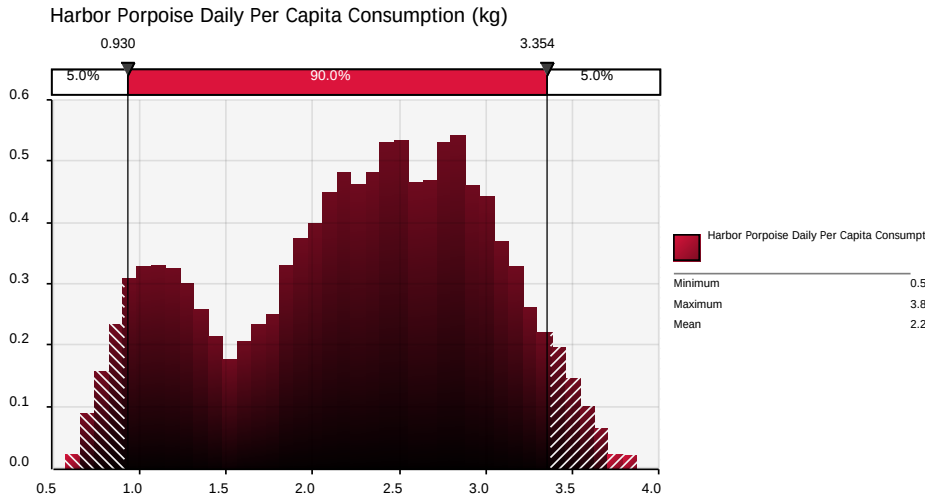

Figure 9.5.2. Ranges of estimated harbour porpoise daily per capita consumption, incorporating uncertainty from marine mammal individual weights and random sampling of daily consumption parameters from published literature.

Estimates of total marine mammal consumption off the NEUS (Figure 9.5.3) are often similar in magnitude to commercial fishery landings (Figure 9.5.4) for small pelagic and groundfish prey groups, although previous studies have indicated that targeted sizes may differ. Marine mammals are potential top predators off the Northeast US, and as we move toward ecosystem-based fisheries management, consumption by marine mammals should be included as a natural removal from prey populations when assessing fish stocks and in whole-system ecosystem models.

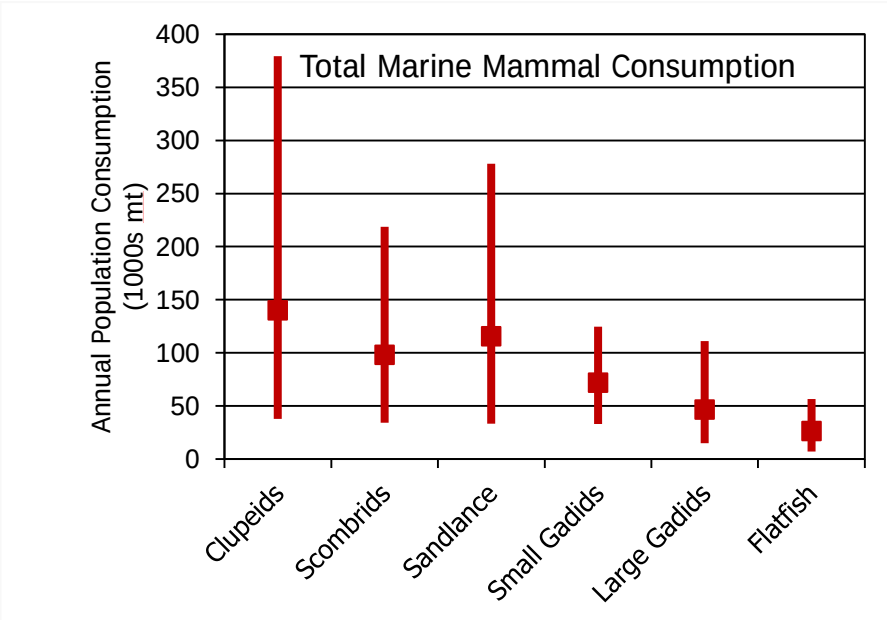

Figure 9.5.3. Annual population-level marine mammal consumption for six prey groups off of the NEUS.

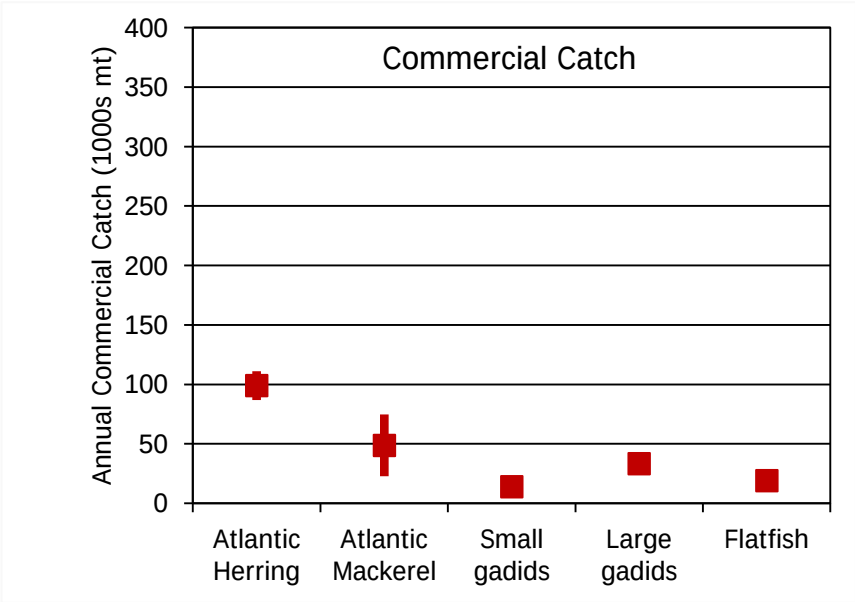

Figure 9.5.4. Annual commercial fisheries catch for five targeted fish groups off of the NEUS.

9.6 North Pacific ecosystems

The role of top predators were quantified in three north Pacific ecosystems using mass balance foodweb models (Aydin *et al.*, 2007). Biomass, consumption rates, and diet composition (not shown) of marine mammal, seabird, and fish predators were compiled for the Aleutian Islands (AI), eastern Bering Sea (EBS) and Gulf of Alaska (GOA) marine ecosystems during a comparative modelling project (Table 9.6.1, Figures 9.6.1, 9.6.2; for details see Aydin *et al.*, 2007, Tables B2, B5, B6, and B8). Detailed comparisons of consumption by predators compared with fisheries removals were presented for two prey eco-types: Pacific cod (*Gadus macrocephalus*) representing the large gadoids, and walleye pollock (*Theragra chalcogramma*) representing the medium gadoids. (Similar information is available for large and small forage fish from the same models, but was not compiled during the meeting.)

In these three ecosystems, Pacific cod's role is better characterized as a top predator than as prey. Consumption of cod by marine mammals, birds, and other predators is estimated to be an order of magnitude lower than commercial catch (Figure 9.6.3). In contrast, walleye pollock were consumed by predators in the EBS and GOA at rates equalling or exceeding commercial fisheries, while in the AI the early 1990s pollock fishery catch exceeded estimated consumption of pollock by predators (Figure 9.6.4). More recent AI pollock catch is much lower (the directed fishery is effectively closed) and well below estimated predator consumption.

Table 9.6.1. Biomass (B) in tons (t) and annual consumption rate per unit biomass (QB) for predators in the Aleutian Islands (AI), eastern Bering Sea (EBS), and Gulf of Alaska (GOA), from Aydin e*t al.* (2007).

Group	AI B (t)	AI QB (yr−1)	EBS B (t)	EBS QB (yr−1)	GOA B (t)	GOA QB (yr−1)
Toothed Whales						
Transient killer whales	22	11.2	67	11.2	40	11.2
Sperm and Beaked whales	8,339	6.61	8,753	6.61	13,159	6.61
Resident killer whales	223	11.2	669	11.2	397	11.2
Porpoises	3,745	30	1,792	30	4,399	30
Belugas			6,068	30		
Baleen Whales						
Grey whales			16,177	8.87	16,177	8.87
Humpback whales	8,149	7.58	5,991	7.58	52,059	7.58
Fin whales	2,502	6.52	225,195	6.52	77,659	6.52
Sei whales	361	8.79	3,137	8.79	1,848	8.79
Right whales	202	8	1,760	8	1,037	8
Minke whales	5,555	7.78	11,904	7.78	689	7.78
Bowhead whales			3,552	8.68		
Pinnipeds and otters						
Sea Otters	391	73	372	73	1,007	73
Walrus Bd Seals			56,578	15.4		
N. Fur Seal_Juv			435	82.2	86	84.1
N. Fur Seal			16,157	39.0	2,639	39.0
Steller Sea Lion_Juv	54	55.3	13	55.3	81	56.3
Steller Sea Lion	2,964	24.1	726	24.1	4,650	24.1
Resident seals	206	17.4	6,660	17.4	961	17.4
Wintering seals			15,000	19.2		
Seabirds						
Shearwater	103	73	198	73	67	73
Murres	76	72	4,033	72	1,410	72
Kittiwakes	28	110	328	110	260	110
Auklets	459	110	869	110	87	110
Puffins	234	73	235	73	1,892	73
Fulmars	278	73	258	73	240	73
Storm Petrels	201	144	1	144	67	144
Cormorants	61	73	74	73	102	73
Gulls	33	73	52	73	166	73
Albatross Jaeger	5	75	50	75	67	75
Adult groundfish						
W. Pollock	324,133	4.44	9,154,704	3.17	1,620,500	3.78
P. Cod	166,964	2.28	1,220,697	2.28	405,680	2.19
Arrowtooth	30,239	2.61	467,655	1.16	1,682,887	1.44
P. Halibut	40,110	1.1	109,887	1.1	443,529	1.1

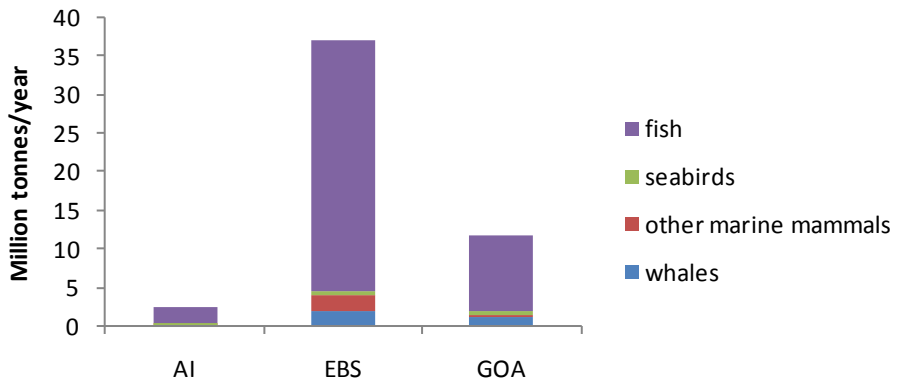

Figure 9.6.1. Comparison of estimated predator total consumption in the Aleutian Islands (AI), eastern Bering Sea (EBS), and Gulf of Alaska (GOA).

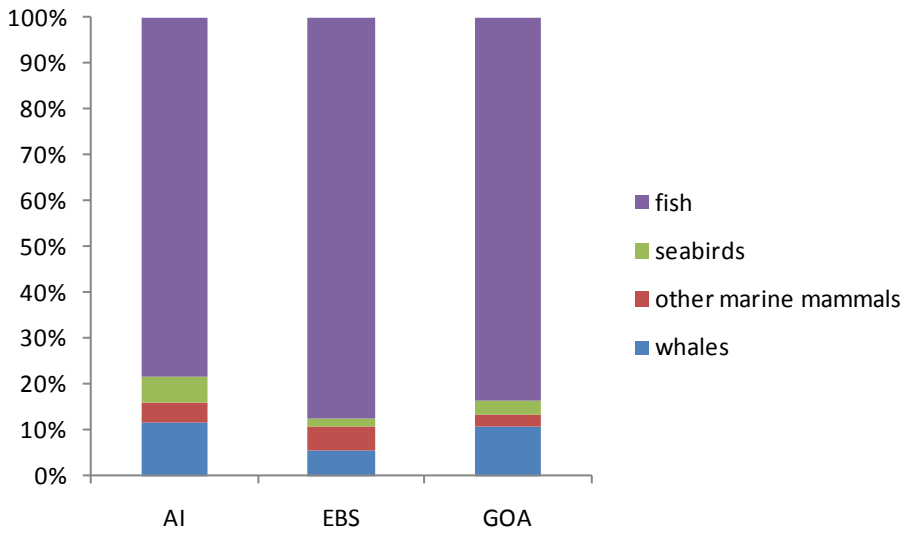

Figure 9.6.2. Comparison of relative distribution of estimated predator total consumption in the Aleutian Islands (AI), eastern Bering Sea (EBS), and Gulf of Alaska (GOA).

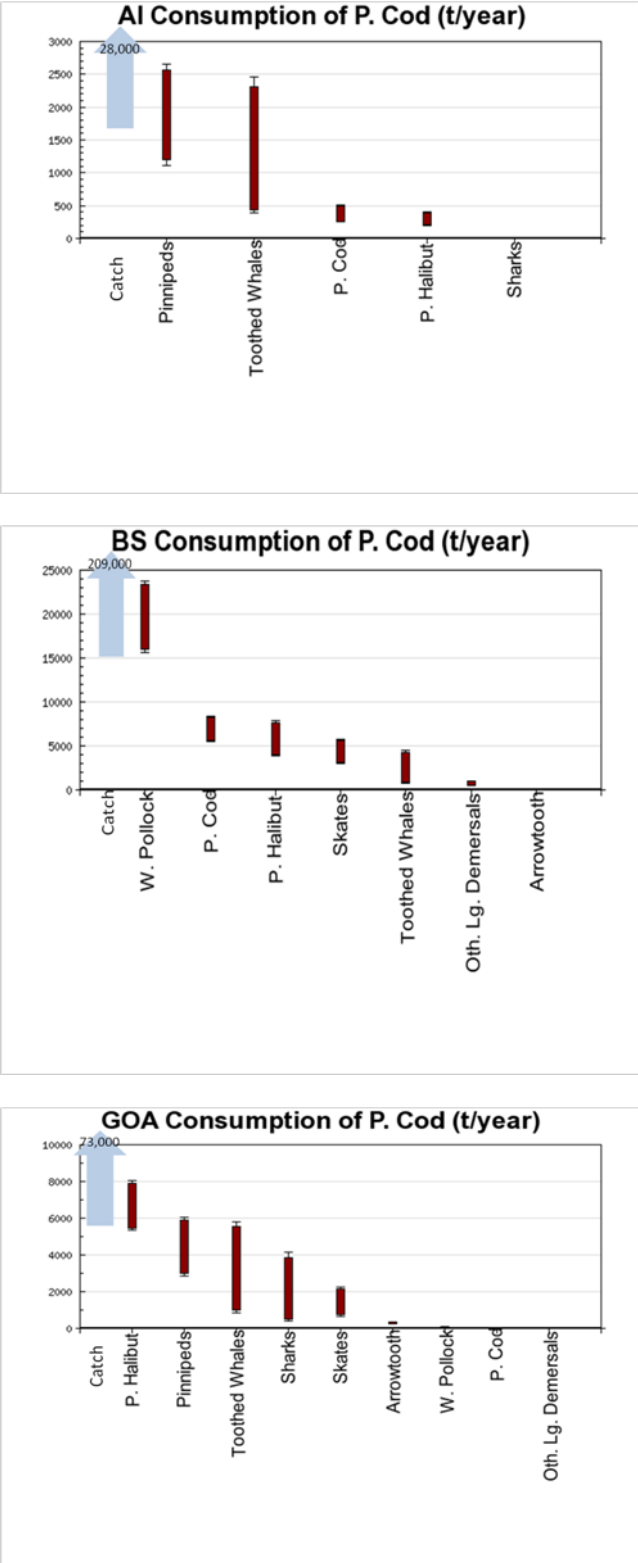

Figure 9.6.3. Comparison of estimated predator consumption of cod with cod catch in the Aleutian Islands (AI), eastern Bering Sea (EBS), and Gulf of Alaska (GOA).

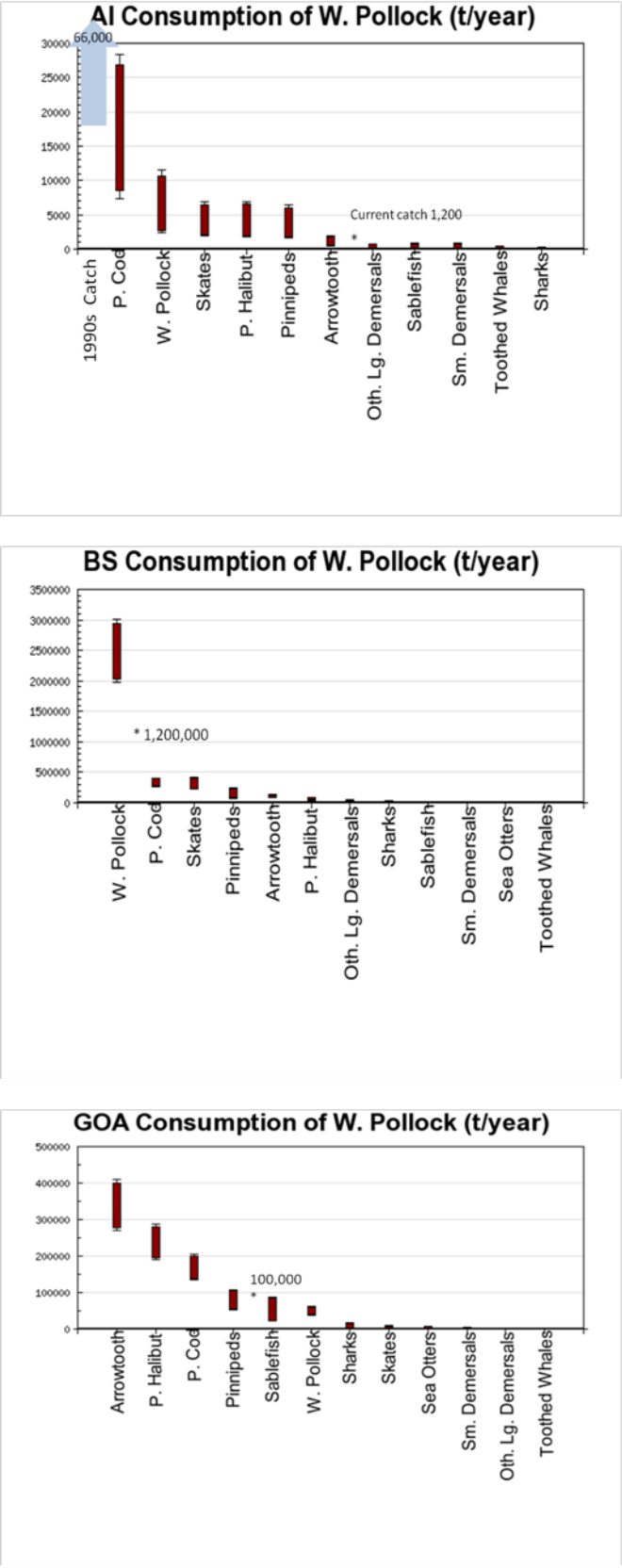

Figure 9.6.4. Comparison of estimated predator consumption of pollock with pollock catch (*) in the Aleutian Islands (AI), eastern Bering Sea (EBS), and Gulf of Alaska (GOA).

9.7 Conclusions

In general, total top predator consumption of fish greatly exceeds the commercial catch of most species. The fish predators remain the most important top predators, but on certain prey species, the consumption by marine mammals can exceed both that of fish and the total catch in the fishery. Seabirds remain of minor importance in terms of fish removals, typically removing less than 5–10% of the total removals of fish.

By far the greatest source of uncertainty is the estimates of top predator population size. In comparison, consumption estimates are usually known with greater accuracy. Estimates of M (and associated biological reference points) of some species derived from multispecies models are likely to be highly sensitive to these factors. However, leaving these predators from the models entirely is likely to lead to highly biased estimates of M and a rough estimate of the effect of top predators appears preferable to excluding this knowledge.

9.8 References

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10 ToR h) Explore the expected trophic role of invasive species using a simulation model package under anticipated conditions

WGSAM included this Term of Reference in order to contribute directly towards the stated "High priority research topic" raised in the 2009–2013 ICES Science Plan, namely *"Introduced and invasive species, their impacts on ecosystems and interactions with climate change processes"*. In 2010 WGSAM investigated the various types of models that are available to examine the impacts that non native species might have on foodwebs once established and also reviewed some of the more well-known introductions within the ICES area, including species that have offered new opportunities in terms of fisheries catches as well as those that have posed a significant threat. In this section we have revisited some of the more promising modelling approaches (e.g. that proposed by Link 2004), and have applied these to a case-study in the Mediterranean, including possible projections of consequences for native fish populations and artisanal fisheries.

As the primary working group within the ICES system responsible for multispecies modelling and prediction, WGSAM needs to consider the following key questions with regard to incoming and invasive species:

1) Can we predict what an incoming species will eat – on the basis of its diet elsewhere (or other methods)?
2) If a new species becomes established can we predict what predators will predate upon it (is this a more difficult question)?
3) If nothing eats a new species will it expand and out-compete native species (for food and habitat)?
4) How will the introduction of the new species impact fisheries catches and revenues?

10.1 Case study – Lessepsian Species in the Mediterranean

The Mediterranean Sea is a region that has been particularly characterized by changes in fish communities as a consequence of non-native species introductions, both from the North Atlantic but also invasions through the Suez Canal. The opening of the Suez Canal in 1869 allowed entry into the eastern Mediterranean of Indo-Pacific and Red Sea biota, where these so-called Lessepsian migrants now dominate the community structure (50–90% of fish biomass). For many decades, this migration was limited, partly due to extremely high salinity within the Suez Canal where dry salt valleys had existed previously, but the process has accelerated in recent years associated with a warming trend of the seawater and a significant drop in the canal's salinity to 'normal' marine levels. Recent records of lessepsian fish species in the Sicily Channel and in the southern Tyrrhenian Sea (Castriota and Andaloro, 2005) show that these alien species are spreading throughout the Mediterranean. In some cases (e.g. the rabbit fish *Siganus luridus*), species took advantage of vacant ecological niches (i.e. very few 'native' herbivorous fish species), however in other cases (e.g. *Upeneus moluccensis*), introductions have resulted in the complete exclusion of native Mediterranean species (Perlmutter, 1956). More than 63 fish species have now penetrated into the Mediterranean by way of the Suez Canal.

Fistularia commersonii, the blue cornetfish, is a reef-associated species that is widely distributed in the Indo-Pacific. The first individual recorded in the Mediterranean Sea

was captured in January 2000 by a fisher along the coast of Israel at a depth of 35 m (Golani, 2000), and the species has subsequently spread towards the eastern Mediterranean basin (Karachle *et al.*, 2004). It reached the coast of Tunisia in autumn 2002 (Ben Souissi *et al.*, 2004), the southern shores of Italy in December 2002 (Azzurro *et al.*, 2004), the North Tyrrhenian Sea in October 2004 (Micarelli *et al.*, 2006), the coast of Sardinia in October 2005 (Pais *et al.*, 2007); Malta and Gozo in 2006 (Sciberras and Schembri, 2007), the south coasts of Spain in November–December 2007 (Granada and northernmost Palamos; Sanchez- Tocino *et al.*, 2007), the coast of Algeria in winter 2008 (Hemida and Capape 2009). The first observation of *F. commersonii* along the French Mediterranean coast was made in November 2007 in the waters of Porquerolles Island (Var, France; Bodillis *et al.*, 2011). In August 2008, two individuals were captured in the northeastern part of Corsica (Miniconi, 2010). With such a widespread range of occurrence over a short period of time, *F. commersonii* is the fastest and farthest lessepsian fish migrant ever recorded (Golani *et al.*, 2007). Its spreading is characterized by very short time intervals between new records, followed by a rapidly increasing abundance in the invaded areas (Joksimovic′ *et al.*, 2008).

F. commersonii is a midwater coastal predator that primarily targets small fish (pelagic and reef-associated), but also occasionally crustaceans and cephalopods (Takeuchi *et al.*, 2002; Kalogirou *et al.*, 2007). In the Mediterranean, stomach contents data only exist for the Aegean Sea, around the Greek island of Rhodes (Corsini *et al.*, 2002; Kalogirou *et al.*, 2007) as well as the coast of Lebanon in the far eastern part of the Mediterranean (Bariche *et al.*, 2009). This data is characterized by a preference for *Spicara smaris, Boops boops* and *Mullus* spp., but a variety of small benthic fish especially gobiids, blennies and newly hatched fish constituted an important component when characterized in terms of number.

F. commersonii has now been recorded all around Corsica and this includes the Bay of Calvi (NW Corsica) in October 2010. The Bay of Calvi has been the subject of extensive survey work over the past 30 years, and this includes assessments of the littoral fish community (e.g. Spyker and van den Berghe, 1995), but also attempts to model the foodweb, using the Ecopath with Ecosim (EwE) modelling framework (Pinnegar, 2000; Pinnegar and Polunin, 2004). Given that the EwE model described by Pinnegar and Polunin (2004) attempted to replicate the ecosystem during the period only a few years before the arrival *F. commersonii*, this offers an opportunity to try to predict or anticipate the likely impact of *F. commersonii* on the wider foodweb of the region (which is substantially different from that around Rhodes). In particular, this would to try to assess the impact an expanding *F. commersonii* population might have on native fish populations and on commercial fisheries.

10.2 The Rank Proportion Algorithm (RPA) model applied to *Fistularia commersonii*

Link (2004) developed a rank proportion algorithm (RPA) model that predicts prey preference from first principles of predation that, when coupled with ambient prey concentrations, can predict prey utilization (i.e. diet composition). In 2010 WGSAM discussed the possibility that this approach might be used to provide diet composition data for newly arrived, or even yet to arrive, non-native marine species and thereby to investigate the role that the new species might play in marine foodwebs in future. In this section we have applied the RPA model to *Fistularia commersonii*, using a Table of potential prey biomasses (in t/km^2) in the Bay of Calvi from Pinnegar (2000) as well as information on potential prey sizes ($L\infty$) and expert knowledge regarding prey behaviour.

The limited literature available concerning feeding habits of *Fistularia commersonii* (Takeuchi *et al.*, 2002; Kalogirou *et al.*, 2007; Bariche *et al.*, 2009) has indicated that this species feeds on a wide variety of different fish prey types, including both benthic and pelagic organisms. Therefore, of the 39 fish species included in the analysis of Pinnegar (2000), 22 were taken forward for inclusion in the portfolio of prey items for the RPA. The stomach contents data provided by Kalogirou *et al.* (2007) for a site elsewhere in the Mediterranean (Rhodes) included several species that are not present in the Bay of Calvi in any great numbers. Also, this dataset was primarily concerned with *F. commersonii* individuals that were caught over *Possidonia oceanica* seagrass beds where *Spicara smaris* and *Boops boops* are considerably more abundant. This is not the case in the rocky littoral areas of the Bay of Calvi (that are dominated by the planktivore *Chromis chromis*, and various rocky reef-associated wrasses and sparids, most notably *Coris julis* and *Symphodus* spp.).

The portfolio of species selected for the RPA was based on taxonomic similarities with those listed by Kalogirou *et al.* (2007), Bariche *et al.* (2009) or Takeuchi *et al.* (2002) as well as the size of potential prey items. Large piscivorous fish and scorpaenids were excluded since there was no evidence that such species are targeted anywhere by *F. commersonii.* Given that this case study largely concerns a hypothetical invasion in future the spatio-temporal overlap (O_{ij}) between the fish predator j and each prey species i was set to unity and the RPA executed assuming complete overlap of predator and prey.

The next step in the RPA is to rank (R) each prey item i with respect to each factor m of the predation process, 1 being the highest rank and N (i.e. the total number of prey) the lowest, using an average value for the ranks of ties. Here it is important to determine whether there are any special prey attributes that would alter the ranking of any prey item for any component of the predation process. Prey attributes such as size (especially relative to other prey), cryptic behaviour, swimming speed and motion, site fidelity and so forth, are vital in determining the importance of each step in predation. Given these considerations, ranks for detection, reaction, capture, and ingestion were estimated for every feasible prey I (see below). The model also has an ''icing factor'' whereby one can rank a particular prey item higher based on some special or localized knowledge, in this case we used the data provided by Kalogirou *et al.* (2007) to 'up-weight' certain prey items slightly, although in reality this had limited impact on the model outputs. Each factor m is weighted the same in the RPA across all the possible factors (M; here equal to 5). This step of ranking each prey i for each factor m is arguably the most important part of the RPA and relies heavily on prior empirical observations.

In the present case we used the following criteria to rank each prey type according to their detection, reaction, capture, and ingestion qualities (see Table 10.2.1). This categorization is based upon Holling's (1959) "components of predation", further elaborated upon by O'Brien (1979).

Detection – this was based on the assumption that *F. commersonii* is predominantly a midwater predator and hence planktivorous species were ranked highest followed by more demersal species such as sparids and wrasses, followed by benthic and cryptic fish such as *Mullus surmuletus*, blennies and gobies, and finally crustaceans were ranked lowest – assuming they are the most cryptic prey type. The damselfish *Chromis chromis* was ranked highest, given that it is the most abundant fish species in the ecosystem, it is frequently the most obvious species in midwater (when it is feed-

ing on plankton during the day) and it falls within the required size range for the predator (see below).

Reaction – this was the most challenging attribute to try to enumerate and in the end species were ranked on the basis of their obviousness to the predator and their relative evasive ability. *Chromis chromis* were ranked highest as being the most obvious prey-item available (large 'clouds' of individuals picking copepods in the water column), followed by the other pelagic species *Boops boops* and *Atherina presbyter.* The large (and very obvious) sparid species *Diplodus annularis, D. vulgaris, D.sargus* and *Oblada melanura* were given an intermediate score, followed by the wrasses, blennies/gobies and small crustaceans respectively, on the basis of their cryptic habits and ability to evade predators.

Capture – this attribute was ranked on the basis of the different swimming abilities of the various potential prey items. A useful indicator that was examined was the 'aspect ratio' of the different fish species (Pauly, 1989). Reef fish such as wrasses typically have relatively rounded caudal fins, deeper bodies and consequently low aspect ratios that are indicative slower swimming speeds compared to faster swimming planktonic species. Therefore wrasses were considered the easiest prey for *F. commersonii* to capture and were ranked lowest, followed by the 3 *Diplodus* species, *Oblada melanura* and *Chromis chromis,* with slightly forked caudal fins and higher aspect ratios. The fast-swimming planktivores/omnivores *Boops boops* and *Atherina presbyter* were ranked next followed by the cryptic blennies, gobies and crustacean species that were judged most difficult to capture (despite possessing low aspect ratios).

Ingestion - this attribute was ranked on the basis of prey size. Kalogirou *et al.* (2007) has demonstrated that *F. commersonii* typically selects prey animals that are <10cm length, therefore crustaceans were ranked highest, and smaller fish species were ranked higher than larger fish. Prey were ranked on the basis of their f $L\infty$ from Pinnegar (2000). Of the fish species, blennies, gobies and *C. chromis* were ranked highest. Most of the wrasses received an intermediate score, whereas the sparids (*Diplodus sargus, D. Vulgaris* and *Oblada melanura*) were given lowest scores, since *F. commersonii* would only be able to consume such prey when they are small juveniles.

Icing factor - The model also has an ''icing factor'' whereby one can rank a particular prey item higher based on some special or localized knowledge (e.g. the fish have learned to feed on a particular type of prey, have a historical preference for a certain type of prey in a particular ecosystem, etc.). Like relative abundance or overlap, the icing factor is optional and can be set to unity for all prey if not used. In the present analysis we included an ''icing factor'' based on the observed diet composition in Rhodes (Kalogirou *et al.*, 2007).

After ranking, an adjusted (or inverse) rank R' for each factor should be calculated as follows:

$$R'_{ijm} = |(n+1) - R_{ijm}|$$

to account for the fact that 1 is the highest possible rank. Next, one should sum all the adjusted ranks for each factor m across all N prey items and then divide the adjusted rank by that sum to obtain a rank proportion (P) for each prey i:

$$P_{ijm} = \frac{R'_{ijm}}{\sum_{i=1}^{N} R'_{ijm}}$$

These rank proportions indicate the relative importance of each prey *i* for each factor *m*. Then the rank proportions should be multiplied across all factors (*M*) for each prey *i* to estimate the ultimate rank proportion (*Sij*):

$$S_{ij} = \prod_{m=1}^{M} P_{ijm}$$

This product can be considered an estimate of a preference proxy of predator *j* for each prey *i*. The value of *Si j* can be multiplied by the spatial overlap (*Oij*)and the relative abundance (*Ai*) (when the latter two are available) of each prey *i* to provide a proxy for prey utilization (i.e. diet composition, *Dij*):

$$D_{ij} = S_{ij} \cdot O_{ij} \cdot A_i$$

Table 10.2.1 provides the ranked scores for each attribute. These have been used to estimate the corresponding diet composition for *F. commersonii* in the Bay of Calvi (Table 10.2.2, Figure 10.2.1).

Table 10.2.1. Assigned rank of each prey type according to their detection, reaction, capture, and ingestion qualities. The assigned rank for the 'icing' factor reflects the diet composition of *F. commersonii* in Rhodes (based on Kalogirou *et al.*, 2007).

Potential Prey Item	Biomass (t/km2)	A (%)	Overlap	Detection	Reaction	Capture	Ingestion	Icing
Mullus surmuletus	2.233	3.1	1	15	11.5	10.5	20	2
Symphodus tinca	6.7	9.3	1	10.5	11.5	4	17	15
Symphodus mediterraneus	0.194	0.3	1	10.5	11.5	4	13	15
Diplodus annularis	1.16	1.6	1	5.5	5.5	10.5	15	12
Tripterygion sp.	0.066	0.1	1	18	19	19	6	6
Gobius sp.	0.83	1.2	1	18	19	19	7	6
Diplodus sargus	2.4	3.3	1	5.5	5.5	10.5	22	18.5
Diplodus vulgaris	0.5	0.7	1	5.5	5.5	10.5	21	18.5
Coris julis	7.3	10.2	1	10.5	11.5	4	16	10
Symphodus roissali	0.12	0.2	1	10.5	11.5	4	9	15
Aidablennius sphynx/ Parablennius zvonmiri	0.052	0.1	1	18	19	19	4	6
Symphodus melanocercus	0.022	0.0	1	14	11.5	4	11	15
Lipophrys sp.	0.024	0.0	1	18	19	19	3	6
Parablennius rouxi / P. incognitus	0.18	0.3	1	18	19	19	5	6
Chromis chromis	10	13.9	1	1	1	10.5	12	9
Boops boops	2.91	4.1	1	2.5	2.5	14.5	18	1
Symphodus ocellatus	0.132	0.2	1	10.5	11.5	4	8	15
Atherina presbyter	0.57	0.8	1	2.5	2.5	14.5	10	3
Oblada melanura	1.336	1.9	1	5.5	5.5	10.5	19	11
Thalassoma pavo	0.453	0.6	1	10.5	11.5	4	14	20
Decapods	24.63	34.3	1	21	19	19	2	21.5
'Other' Crustacea	10.002	13.9	1	22	19	19	1	21.5
SUM	**71.814**	**100.0**	**22**	**253.0**	**253.0**	**253.0**	**253.0**	**253.0**

Table 10.2.2. Estimated diet composition D^{ij} of *F. commersonii* in the Bay of Calvi, Corsica.

Potential Prey Item	S_{ij}	Overlap	A (%)	D_{ij}	D_{ij} %
Mullus surmuletus	6.99E-08	1	3.1	2.173E-07	1.10
Symphodus tinca	1.26E-07	1	9.3	1.18E-06	5.95
Symphodus mediterraneus	2.11E-07	1	0.3	5.694E-08	0.29
Diplodus annularis	3.25E-07	1	1.6	5.249E-07	2.65
Tripterygion sp.	2.23E-08	1	0.1	2.05E-09	0.01
Gobius sp.	2.1E-08	1	1.2	2.426E-08	0.12
Diplodus sargus	1.66E-08	1	3.3	5.554E-08	0.28
Diplodus vulgaris	3.32E-08	1	0.7	2.314E-08	0.12
Coris julis	2.4E-07	1	10.2	2.437E-06	12.30
Symphodus roissali	2.95E-07	1	0.2	4.931E-08	0.25
Aidablennius sphynx/ Parablennius zvonmiri	2.49E-08	1	0.1	1.805E-09	0.01
Symphodus melanocercus	1.82E-07	1	0.0	5.579E-09	0.03
Lipophrys sp.	2.62E-08	1	0.0	8.769E-10	0.00
Parablennius rouxi / P. incognitus	2.36E-08	1	0.3	5.919E-09	0.03
Chromis chromis	8.99E-07	1	13.9	1.252E-05	63.14
Boops boops	3.79E-07	1	4.1	1.536E-06	7.75
Symphodus ocellatus	3.16E-07	1	0.2	5.812E-08	0.29
Atherina presbyter	8.96E-07	1	0.8	7.112E-07	3.59
Oblada melanura	1.77E-07	1	1.9	3.298E-07	1.66
Thalassoma pavo	7.11E-08	1	0.6	4.488E-08	0.23
Decapods	9.72E-10	1	34.3	3.335E-08	0.17
'Other' Crustacea	5.09E-10	1	13.9	7.094E-09	0.04

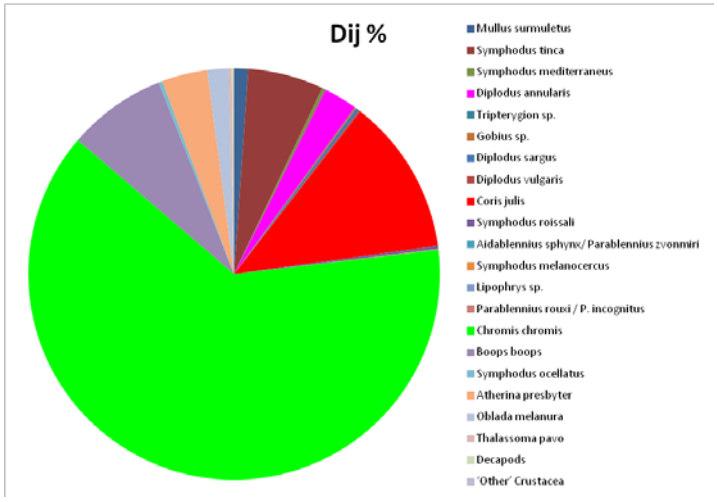

Figure 10.2.1. The suggested diet composition for *F. commersonii* in the Bay of Calvi.

The results of this analysis suggest that the diet of *F. commersonii*, if they were to establish themselves in the rocky littoral areas of the Bay of Calvi, would largely focus on *Chromis chromis* and *Coris julis* but also *Boops boops* and *Atherina presbyter*. These latter two species are important prey items elsewhere in the Mediterranean, however their overall contribution was observed to be much greater elsewhere (Kalogirou *et*

al., 2007; Bariche *et al.*, 2009). Part of the reason for this discrepancy is that the existing stomach content studies that have been conducted so far have tended to sample *F. commersonii* over *Posidonia oceanica* seagrass beds, where *Boops boops* and *Atherina presbyter* (as well as *Spicara smaris*) are significantly more abundant. In the Bay of Calvi *Posidonia oceanica* seagrass beds also exist (53.1 ha of rocky sublittoral habitat, 179 ha of *Posidonia oceanica* seagrass meadow and 14 ha of sandy substrate), but fish abundance surveys and foodweb modelling has not been conducted for this habitat, and the *F. commersonii* that were observed by Bodillis *et al.* (2011) in the Bay of Calvi in October 2010 were caught by fishers within the rocky littoral zone.

10.3 Bay of Calvi – Ecopath Model

An Ecopath model was constructed to represent the rocky sublittoral community of the Bay of Calvi, Corsica by Pinnegar (2000). The Ecopath model was developed using an alpha-version of the Ecopath 4, but has subsequently been re-balanced in response to improved diagnostics within Ecopath 6 ('tweaked' the unassimilated consumption of 'suspension-feeders' from 0.6 to 0.2 so that the respiration was no longer negative). The model is composed of 27 compartments, including seabirds, 11 groups of fish, 12 groups of invertebrates, 2 primary producers, bacteria and detritus (see Figure 10.3.1), as well as a single commercial fishing fleet.

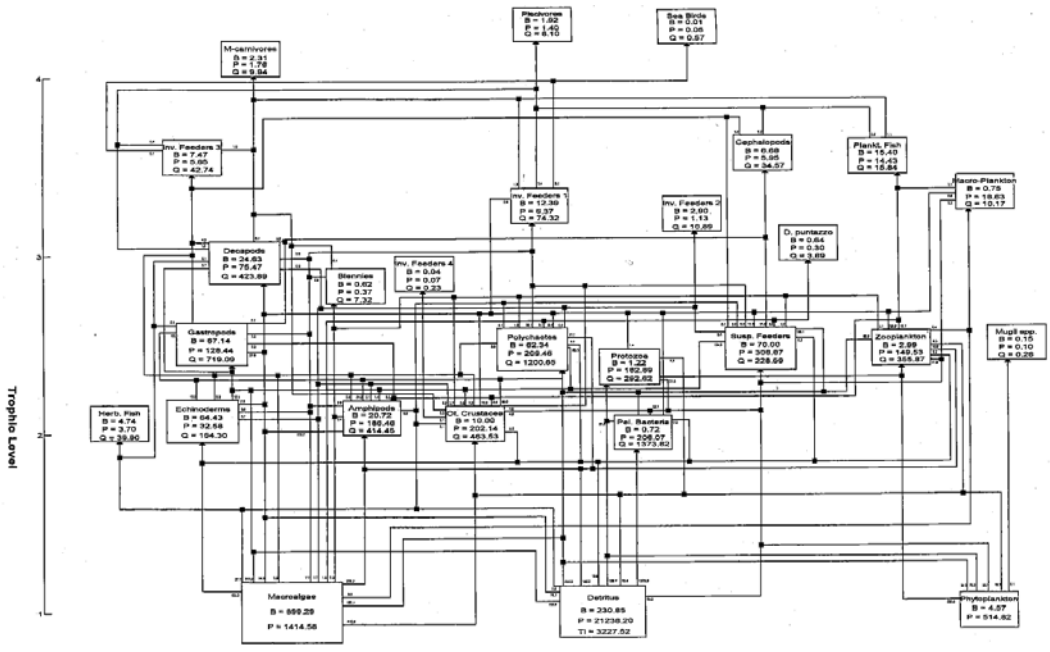

Figure 10.3.1. Flow diagram for the Ecopath model developed by Pinnegar (2000) for the rocky sublittoral community of the Bay of Calvi, Corsica.

A new functional group has been added to the EwE model to represent *F. commersonii*. This has been assigned a very low biomass (0.005 t/km^2), with consumption rate and natural mortality (P/B ratio) based on allometric equations (Pauly *et al.*, 1990; Pauly 1980) [Q/B = 5.86, P/B = 0.63]. Note that we have used the standard length in these calculations rather than the total length, due to the long ribbon like nature of the caudal fin in *F. commersonii.* Diet composition has been assigned on the basis of the RPA analyses described above. Some predation on *F. commersonii* has been added on the basis that seabirds (in this model the osprey *Pandion haliaetus* and the shag

Phalacrocorax aristotelis) typically select mid and surface water fish species (assumed that *F. commersonii* represents 0.03% of their diet) and also a small amount of predation by piscivorous fish (0.03% of their diet) since species such as groupers, snappers and sharks are known to take *Fistularia* spp. where they exist elsewhere in the world (e.g. Randall 1967). EwE was left to calculate ecotrophic efficiency and this revealed a need to reduce predation mortality on *F. commersonii* from the initially inserted values.

Once balanced, the new Ecopath foodweb was used to conduct Ecosim, dynamic simulations whereby the biomass of *F. commersonii* was steadily 'forced' upward over time to investigate the indirect consequences for the wider foodweb, but also the impact on commercial fisheries in the Bay of Calvi. A number of different 'forcing' scenarios were tested, first a 4% year-on-year increase in *F. commersonii was* tested but also a 6% and 8% increase for the full 100 years (scenarios S4, S6 and S8 - resulting in an *F. commersonii* biomass in year 100 of 0.243, 1.600 and 10.184 t/km^2 respectively). In addition, scenarios were tested using different 'vulnerability' settings for the linkages between *F. commersonii* and their prey. Vulnerability settings of 10 (top–down control), 2 (mixed, top–down and bottom–up control) and 1 (bottom–up control) were tested for each of the three biomass scenarios (see Table 10.3.1), all other vulnerability settings (between other predators and their prey) were left at the default setting of 2.

10.4 Results

Interestingly, the estimated trophic level for *F. commersonii* was the highest of any species in the ecosystem (*TL* = 4.511 compared to 4.269 in seabirds and 4.279 in piscivorous fish). The "S8 scenario" where *F. commersonii* was increased by 8% per year resulted in a marked decline in the biomass of planktivorous fish (including *Chromis chromis*) and invertebrate feeders N° 1 (including *Mullus surmuletus* and *Symphodus tinca*), with the decline particularly marked when vulnerability of prey was set at 10 (i.e. top–down control). Those functional groups that declined the most (see Table 10.3.1) were the species that were preyed upon most by *F. commersonii.* By contrast, seabirds and piscivorous fish (both of which were simulated to predate on *F. commersonii*) were suggested to benefit from an increase in *F. commersonii* biomass, with a doubling of the predator biomasses in the "S8" scenario (i.e. a 'bottom up' effect). Similar trends were witnessed under the "S6" and "S4" scenarios, although the magnitudes of change were much smaller. There is some evidence of indirect 'trophic cascades' in the S8 scenario with omnivorous blennies decreasing slightly in response to the increase in piscivorous fish (which predate heavily on blennies as well as *F. commersonii* in this model). Slight increases in polychaetes, decapods and gastropods may be a response to the decline in abundance of invertebrate feeders N° 1 and blennies.

Overall fisheries catches were projected to increase under the "S8" scenario and under the "S6" scenario when vulnerability of prey was set to 2 or 1 (but not under top–down control). This reflects the anticipated 'bottom up' increase in piscivorous fish, that are a main target of the commercial fishery. It is important to note that the simulations did not include *F. commersonii* as a target of the fishery itself, although over the past few years this species has become a regular feature of fish markets in Sicily and many of the individual fish reported by Bodillis *et al.* (2011) from the island of Corsica, were in fact derived from fisheries catches. It is possible that *F. commersonii* might become an important commercial species in future, thereby further supplementing the total catches of Mediterranean fishers and their revenues.

10.5 Conclusions

In this section, we have clearly demonstrated through a worked case-study, noting that it is possible to make an assessment of the indirect foodweb consequences that might occur as a result of the arrival of a hither-to not previously observed non-native species. We have shown that the likely diet composition of such a species can be estimated as well as the possible implications for commercial fisheries catches and hence fishery revenues. The various integrated approaches and protocols demonstrated here could have great utility for risk assessment of non-native fish introductions; a further case-study is now underway using these methods for the non-native round goby *Neogobius melanostomus* that has been introduced to the Gulf of Gdańsk in Poland and is now spreading rapidly.

Table 10.3.1. Outputs from Ecosim model runs where the biomass of *F. commersonii* was forced upwards throughout the 100 year period, by 8% (S8), 6%(S6) and 4%(S4) per year. Vulnerability settings of 10, 2 and 1 were tested for each of the three biomass scenarios. Figures indicate the biomass in year 100 divided by the biomass in year 1.

	S8	S8	S8	S6	S6	S6	S4	S4	S4
	V=10	V=2	V=1	V=10	V=2	V=1	V=10	V=2	V=1
Seabirds	2.021	2.057	2.062	1.101	1.126	1.129	0.996	1.014	1.017
Piscivorus fish	2.149	2.178	2.182	1.141	1.165	1.169	1.002	1.021	1.024
Macrocarnivorous fish	0.844	0.866	0.869	0.947	0.968	0.971	0.976	0.992	0.995
Invertebrate feeders 1	0.747	0.744	0.743	0.948	0.941	0.940	0.997	0.991	0.990
Invertebrate feeders 2	0.945	0.946	0.946	0.989	0.989	0.989	0.998	0.999	0.999
Invertebrate feeders 3	0.953	0.958	0.958	0.987	0.991	0.992	0.995	0.998	0.999
Sessile invertebrate feeders	0.963	0.963	0.963	0.993	0.992	0.992	0.999	0.999	0.999
Invertebrate feeders 4	0.984	0.986	0.987	0.994	0.997	0.997	0.997	0.999	1.000
Omnivorous blennies	0.985	0.976	0.975	1.005	0.996	0.995	1.007	1.001	0.999
Planktivorous fish	0.728	0.854	0.872	0.821	0.952	0.971	0.872	0.976	0.995
Fistularia commersonii	2037	2037	2037	320	320	320	48.6	48.6	48.6
Zooplankton	1.001	0.995	0.994	1.005	0.999	0.999	1.004	1.000	1.000
Echinoderms	1.020	1.019	1.019	1.004	1.004	1.004	1.001	1.001	1.001
'Other' crustacean	0.982	0.982	0.982	0.996	0.996	0.996	0.999	0.999	0.999
Amphipods	0.994	0.996	0.996	0.997	0.999	0.999	0.998	1.000	1.000
Decapods	1.021	1.018	1.018	1.006	1.004	1.004	1.003	1.001	1.001
Polychaetes	1.035	1.036	1.036	1.005	1.007	1.007	0.999	1.001	1.001
Gastropods	1.002	1.003	1.003	1.000	1.000	1.000	1.000	1.000	1.000
Total Fisheries Catch	1.202	1.241	1.247	0.984	1.022	1.027	0.968	0.998	1.003

10.5.1 References

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11 ToR i) Address requests from other ICES Expert Groups as appropriate

11.1 HAWG to WGSAM

In preparation to the benchmark of North Sea Autumn Spawning herring (NSAS) in 2012, the Herring Assessment Working Group (HAWG) has requested WGSAM to look into the estimates and possible trends and variability of natural mortality (predation mortality) of North Sea herring.

Natural mortality-at-age is used as input to most stock assessment models as well as to the North Sea autumns spawner assessment where a fixed natural mortality-at-age is assumed since the start of the time-series in 1960. Although some single species assessment models are technically capable of estimating natural mortality (Lee, 2011), the estimated values are seldom used as they are regarded unrealistic. Multispecies (stock assessment) models such as SMS (Vinther and Lewy, 2004) and EwE (Christensen, 1992) might however provide more reliable estimates and trends in natural / predation mortality. In these models, two types of natural mortality can be addressed: mortality by natural death (M1) and mortality by predation (M2). In contrast to single species models estimating total natural mortality (M), these models incorporate predator–prey overlap and diet composition observations which could be used to estimate the predation mortality more reliable from data.

11.1.1 Methods

The results from the North Sea EwE 2010 key-run (spanning 1991–2007), the 4M 2003 key-run (spanning 1963–2003) the SMS 2007 key-run (spanning 1963–2007) and SMS 2010 key-run (spanning 1963–2010) have been used here to study the trends in natural mortality-at-age or stage over time. Comparisons between the 4M 2003, SMS 2007 and SMS 2010 key-run were also executed to address variation in natural mortality estimates over time.

Both the EwE as the 2007 and 2010 SMS run provide, besides total mortality for each prey species, also the mortality by predator (partial M2). Based on the average partial M2 by predator over the time-series, the contribution of each predator to the total M2 can be calculated. The trends in total M of North Sea herring is a result of the contribution of each of the predators to the predation mortality, and hence it is important to understand which predators account for most of the M2. Biomass trends for the most important predators is also given to indicate if changes in predator biomass could explain changes in natural mortality estimates. A predator is regarded an important contributor to M2 of herring if it contributes, on average over 5% of the total M2. Linear regression, corrected for autocorrelation, is applied to the time-series to test for significant trends in M2 over time. The applied linear regression is however not always an appropriate statistical tool to test for changes over a longer period. For this reason, lowest smoothers have been added to the time-series to indicate changing trends within the time-series. P values are added to the trend lines to indicate significance.

Although the 4M and SMS models cannot be compared one-to-one as the species composition in these models differs, viewing trends and absolute values side by side provides valuable information on the variability of model estimates of M2. If for ex-

ample trends change considerably between the three runs, the credibility of the estimated trends in M2 diminishes.

11.1.2 Results

11.1.2.1 Trends in natural mortality estimates

Figure 11.1.1 shows the trends in M estimates for North Sea herring over the full time-series from 1963 to 2007 (left) / 2010 (right). Given the full time-series, a significant trend can only be observed for the 1-year olds. All other trends are not significantly different ($P < 0.025$) from a horizontal line fitted through the time-series. P values are only given for fitted linear trends. However, the smoothed red lines indicate in all age groups > 1 an increase in natural mortality in the early period from approximately 1963 to 1978, most likely linked to the gadoid outburst in the late 1960's as herring is primarily eaten by cod and saithe. From approximately 1979 onwards, natural mortality decreased while the size of the gadoid population was reduced. From approximately 1991 onwards, close to the period where a regime shift in the North Sea is assumed, an increase in natural mortality can be observed again. The time-series has been split up into these three periods to study the changes in natural mortality in more detail, with a main focus on the most recent period.

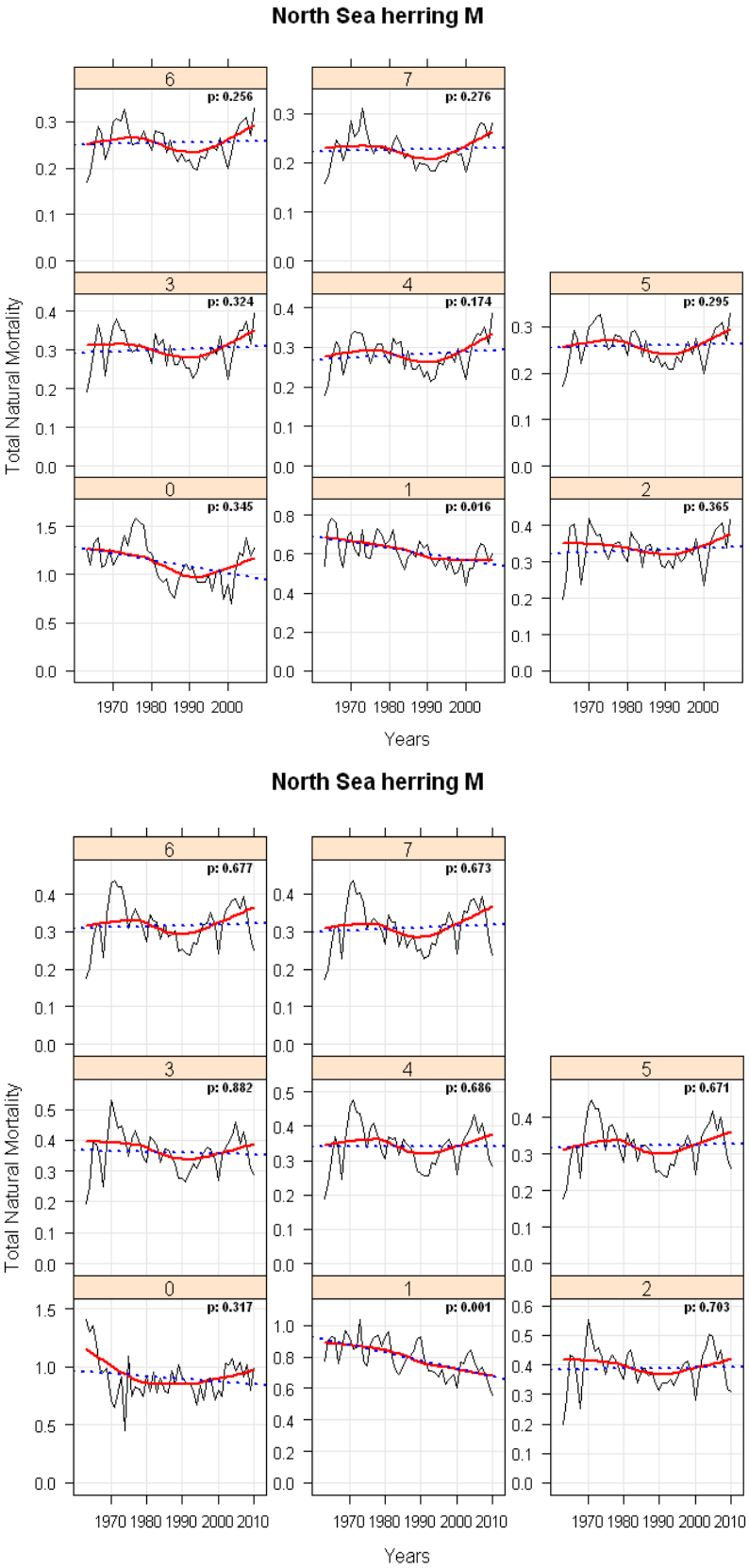
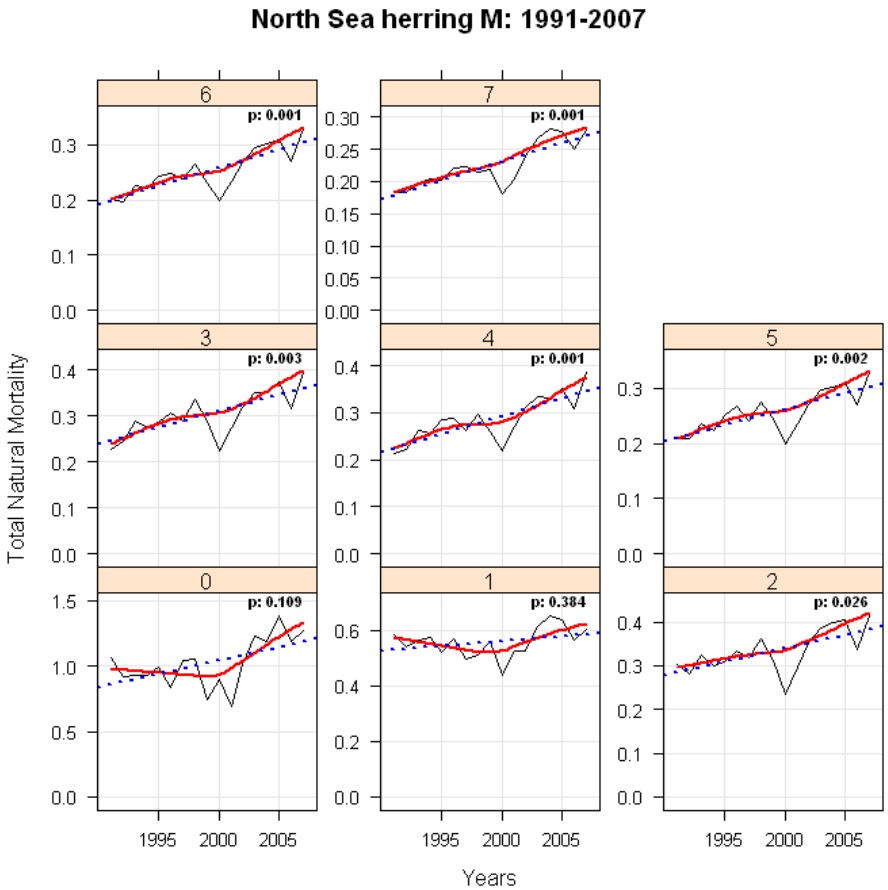
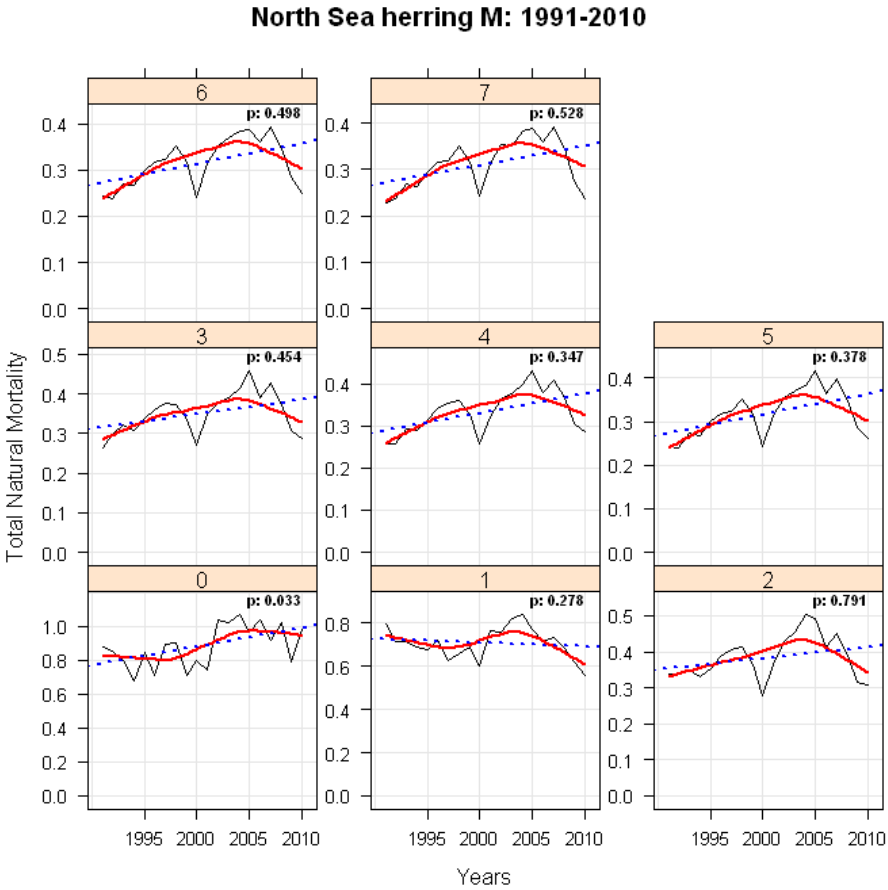


Figure 11.1.1. Total mortality estimates of herring (black solid line) derived from the SMS key-runs. Added lowest smoothers are given in red while fitted linear model is given in blue dashed lines. P values are provided in upper right corner. Panels represent the age groups. Top: 2007 SMS key-run. Bottom: 2010 SMS key-run.

Figure 11.1.2 shows the natural mortality estimates by the 2007, 2010 SMS key-runs and 2010 EwE run over the years 1991–2007/2010/2007. Again, lowest smoothers are added to the time-series while linear regression is applied to the shortened time-series. The 2007 SMS results do not differ considerably from the 2010 SMS results, while trends in ages 2–7 are in close agreement with the EwE run too. All series show that from 1991 onwards natural mortality increased. However, in the more recent years (2008–2010) natural mortality seems to have decreased. Increasing trends in natural mortality (aged 2 and older) are all significant ($P<0.025$) in the 2007 SMS key-run while, due to the recent drop in natural mortality, linear trends are not significant in the 2010 SMS key-run. The trend in adult M2 from the EwE run is significant too.

Natural mortality for ages 0 and 1 are difficult to estimate and highly variable. The estimates as obtained from the SMS or EwE runs might not provide accurate information on these age or stage groups, and as trends are not significant in all key-runs, they are no longer considered in this analyses.


North Sea herring M: 1991-2007
6
p: 0.001
7
p: 0.001
3
p: 0.003
4
p: 0.001
5
p: 0.002
0
p: 0.109
1
p: 0.384
2
p: 0.026
Total Natural Mortality
Years


North Sea herring M: 1991-2010
6
p: 0.498
7
p: 0.528
3
p: 0.454
4
p: 0.347
5
p: 0.378
0
p: 0.033
1
p: 0.278
2
p: 0.791
Total Natural Mortality
Years

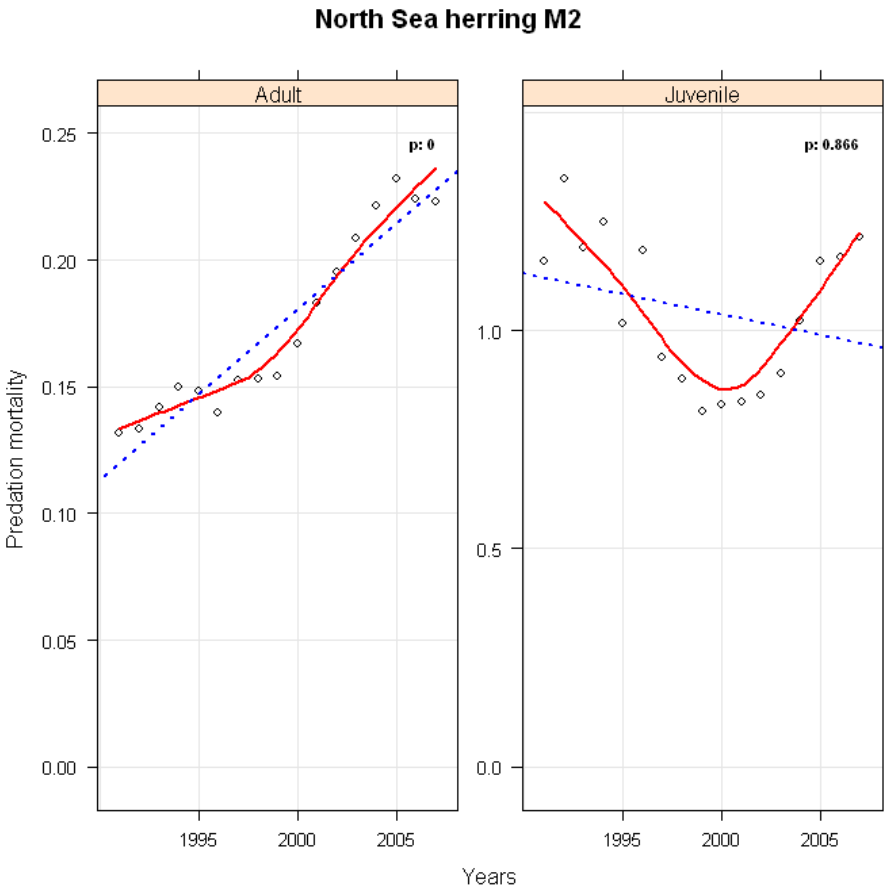


Figure 11.1.2. Total mortality estimates of herring (black solid line) derived from the SMS key-runs and EwE run. Added lowest smoothers are given in red while fitted linear model is given in blue dashed lines. P values are provided in upper right corner. Panels represent the age groups or stage groups. Top: 2007 SMS key-run for the period 1991–2007. Middle: 2010 SMS key-run for the period 1991–2010. Bottom: 2010 EwE key-run for the period 1991–2010.

Different predators have herring in their diet. Figure 11.1.3 shows the partial M2's of the most important predators by age and their average contribution to the M2 of North Sea herring. Young age groups are primarily eaten by cod, saithe and whiting, where whiting mainly predates on age 2 herring and to a lesser degree on age 4 herring. The contribution of saithe and cod alone makes up for nearly 90% of M2 from age 4 onwards. Partial M2's for cod have decreased within this period while M2's have gone up for saithe, except for the past 3 years where M2's have decreased rapidly. A trend or shift in partial M2 can be a result of a change in predator biomass or a diet shift. Hence, figure 11.1.4 shows the biomass development of the most important predators on herring.

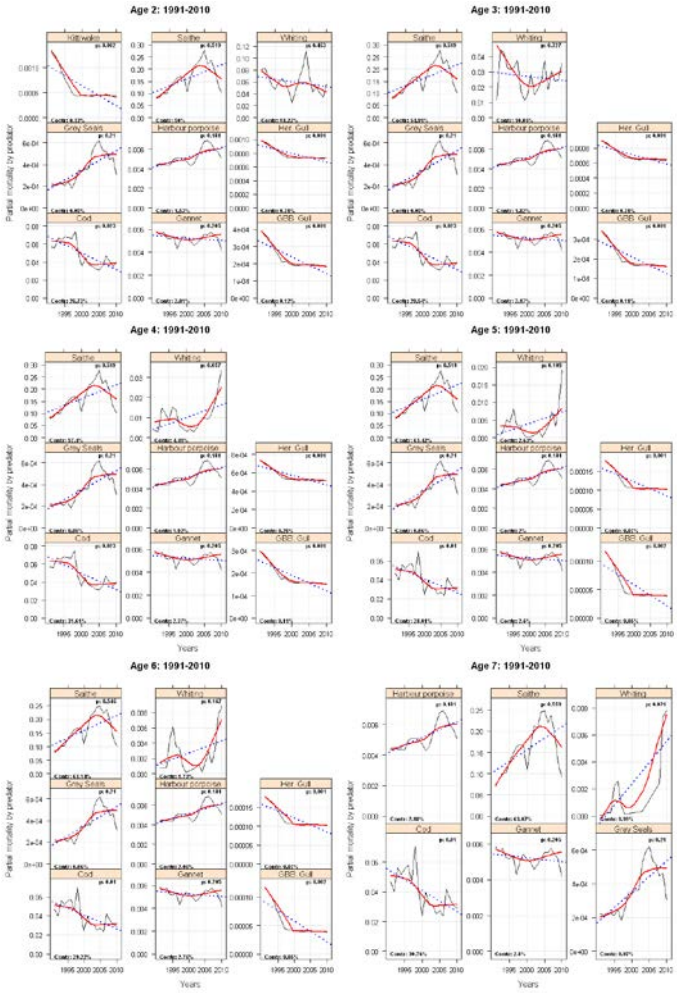


Figure 11.1.3. Partial predation mortality estimates by predator (black solid line) derived from the SMS 2010 key-run. Added lowest smoothers are given in red while fitted linear model is given in blue dashed lines. P values are provided in upper right corner. Contribution to the total M2 by predator and age is given in the bottom left corner. Panels represent the predator

Cod shows a nearly continuous decline in biomass while saithe has increased considerably over the years up to 2005 but has declined severely in the most recent years. These trends in cod and saithe biomass are in agreement with the trends observed in the single species assessments. Herring mortality (ages 2 and older) has, according to the SMS key-runs, increased over the period 1991–2007 but seems to have decreased in the more recent years. This trend is in close agreement with the development of the saithe stock, while the decline in the cod stock decline seems to have been compensated by the saithe increase over the years.

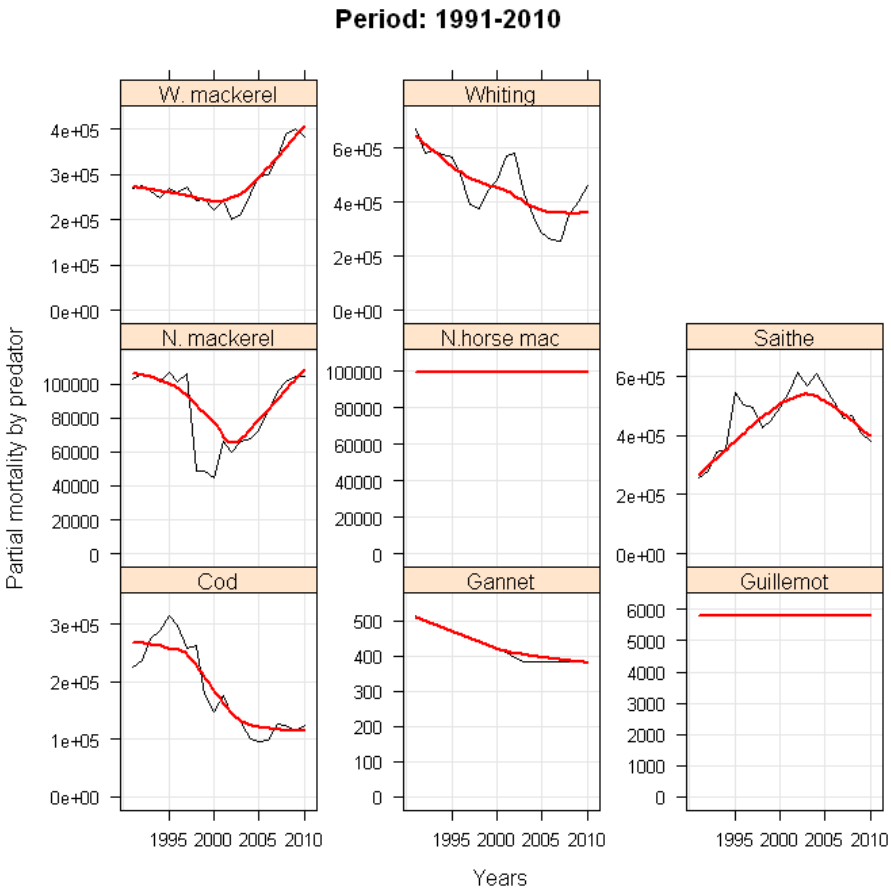


Figure 11.1.4. Predator biomass development (black solid line) derived from the SMS 2010 key-run. Added lowest smoothers are given in red. Panels represent the predator.

11.1.3 Variability of natural mortality estimates

Although trend analyses provide valuable information on changes in predation mortality of North Sea herring, applying a retrospective view on these estimates will show how consistent these trends are. I.e. if trends change considerable in between SMS key-runs, one might want to question the credibility of the trends. For that reason, the 4M 2003 key-run, the ancestor of the SMS model, is compared with the 2007 and 2010 SMS run. As mentioned before, these models cannot be compared one-to-one as not only the diet information has been updated in retrospect, but also the number of predators included in the model has increased.

Figure 11.1.5 shows the trends in total natural mortality for the 3 different runs. Particularly the 2007 and 2010 runs are in close agreement. Absolute differences in total mortality of North Sea herring occur for all ages, being less distinct over ages 0–3. The comparison of the 2007 and 2010 SMS runs shows that natural mortality is higher in the 2010 run. This is at least partly explained by the introduction of other predators in the 2010 run. However, as only the addition of seals and harbour porpoise, which have a relative low contribution to the M2, this might not explain the full difference. Within the model optimization, SSB, recruitment, F and M have strong interactions while trying to fit to the data. This can imply an increase in estimated natural mortality can result in an increase in recruitment estimates.

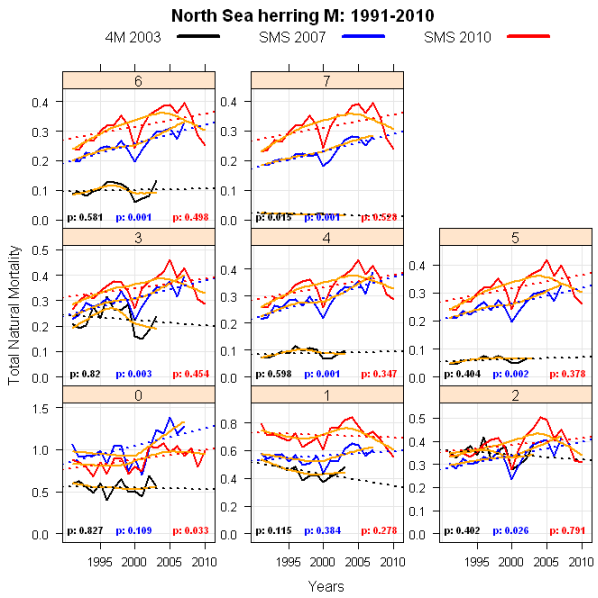


Figure 11.1.5. Total mortality estimates (black solid line) derived from the SMS and 4M key-runs. Added lowest smoothers are given in orange while fitted linear model are given in dashed lines. P values are provided in on the bottom of the panels. Panels represent the age groups or stage groups. The 4M model is given in black solid lines, the 2007 SMS run in blue and the 2010 SMS run in red.

The recruitment estimates, as shown in Figure 11.1.6, confirm this. The recruitment estimates in the 2010 SMS run are considerably higher than in the 2007 model. Hence, the population dynamics as estimated in the 2010 model estimate a bigger population due to the increased recruitment required when natural mortality is increased.

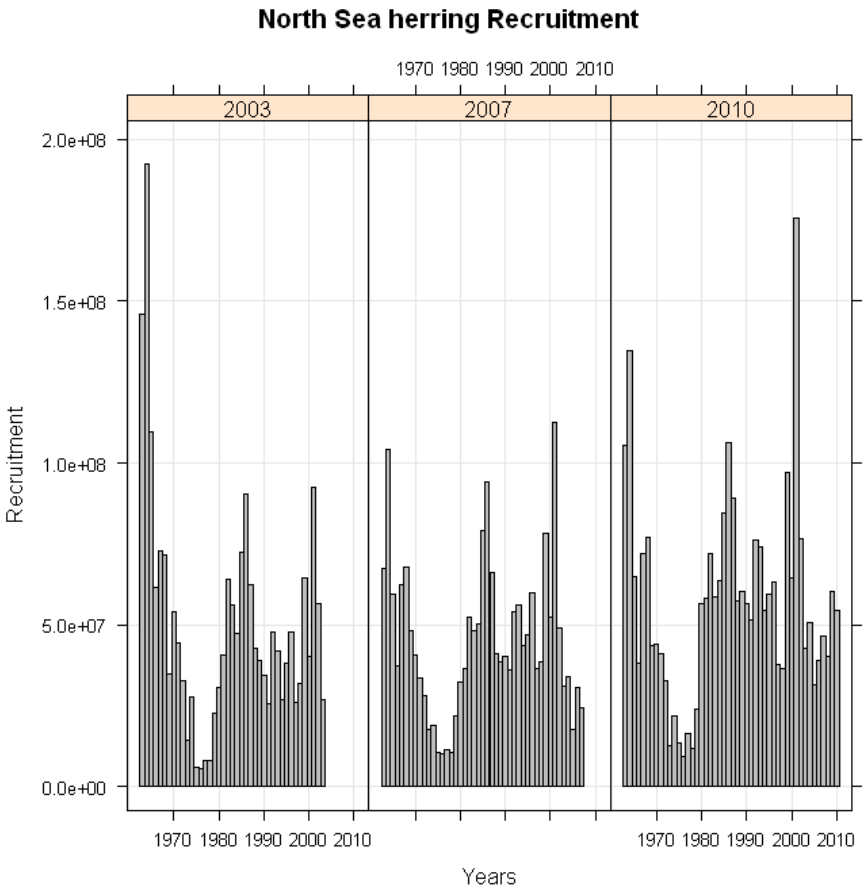


Figure 11.1.6. Recruitment estimates of the model key-runs. Bars represent recruitment estimates by year while panels represent the three different models.

11.1.4 Conclusions

- From the trend analyses it can be concluded that natural mortality-at-age (age > 2) has increased over the period 1963–1978, decreased over 1979–1990 and increased again in the period 1991–2007. The SMS runs and EwE model are in broad agreement on the trends observed in the most recent period and show significant increases in natural mortality.
- Absolute natural mortality estimates are highly variable given the assumptions of the model used. EwE estimated considerable lower natural mortality estimates than the 2007 and 2010 SMS runs, while the 4M model indicates even lower M values.
- Differences in estimated number of recruits by year between the 2007 and 2010 SMS run can be considered a driving factor behind the absolute differences in natural mortality estimates of these models.
- Variability of relative natural mortality estimates are low. The 2007 and 2010 SMS run are in close agreement for ages 2 and older. The EwE estimates are more smoothed than the SMS runs but encompasses similar relative M2 estimates.

11.1.5 Recommendations

- WGSAM considers that the observed trends in natural mortality reflect the actual changes in the predation mortality on North Sea herring given that stomach data and suitability data shows that both cod and saithe predate on North Sea herring. The biomass dynamics of these stocks, and therefore the increase or decrease in partial M2 agree with the changes in North Sea herring M2.
- WGSAM suggests not using raw M2 outputs as can be obtained from the SMS key-run, but suggests to use a smoothed time-series as shown in this document. A moving average of 5 years is currently applied to natural mortality estimates for cod and might be a suitable approach for North Sea herring too.
- WGSAM suggests performing an assessment running with changing natural mortality and let assessment fits decide whether fixed or changing natural mortalities are more appropriate.

11.2 WGBFAS to WGSAM

'WGSAM is recommended to supply WGBFAS at the beginning of 2012 with updated time-series of predation mortalities for Herring in SD 25–27, 28.2, 29 and 32, and Sprat in SD 22–32.'

WGSAM shall either provide spatial M2s after our key run update next year, else shall send a member to the WGBFAS meeting to work on this issue, as WGBFAS prefers.

11.3 WGNSSK to WGSAM

'As a new ToRs, the WGNSSK was asked to comment on the Strategic Initiatives on Marine Strategy and Marine Spatial Planning. The WG recommends increased collaboration between among others WGNSSK, WGMIXFISH, WGSAM, WGINOSE, WGIMM and WGECO to create synergies to best address the new ToRs and avoid duplicate work.

The WGNSSK has also reviewed the possibility of including spatially resolved data on a more routine and integrated basis. The WG was aware of many initiatives, but not of any regular mapping of e.g. landings distribution by métier at the scale of the whole North Sea International data. In the case that such regular mapping cannot be found, and in the case of the establishment of the DCF Regional DataBases would be delayed beyond 2012, the WG suggests that the WGMIXFISH could be an appropriate temporary group for providing such maps. The WGNSSK recommends that this issue is discussed during the WGMIXFISH data workshop on 30 August 2011.'

WGSAM can continue to coordinate with these other working groups, and shall endeavor to continue to establish future WGSAM ToRs that are germane to the work of WGMIXFISH, WGINOSE, WGIMM and WGECO. Further, WGSAM members shall plan to variously attend some of these other WG meetings.

WGSAM expresses some concern over the degree of spatial resolution of the data and model outputs noted. The primary concern is that multispecies and ecosystem models are not usually as finely resolved as has been noted, or the ones that may be are not yet operational. However, several of the ATLANTIS applications contain a finer degree of spatial detail and should be monitored for further considerations on this

issue. WGSAM shall coordinate with WGMIXFISH and send members to the next meeting of that WG.

11.4 WGBIFS to WGSAM

'The WGBIFS recommended to WGSAM that clear decisions are required related to the proposed procedure of stomach sampling and the analyses of the stomachs until August 2011 to start with the sampling during the BITS in November 2011.'

WGSAM have replied to WGBIFS (see letter below) requesting further details regarding specifically the issues WGBIFS needed clarification upon. Once we obtain their reply, WGSAM shall address those points as a high priority, even intersessionally if need be.

From: Anna Rindorf
Sent: 12 September 2011 16:34
To: 'olavi.kaljuste@slu.se'; Henrik Degel
Cc: 'jason.link@noaa.gov'; Stefan Neuenfeldt
Subject: FW: ICES EG Recommendations 2011 for WGSAM

Dear Olavi and Henrik

WGSAM has been tasked by WGBIFS to provide clear decisions regarding the stomach sampling procedures, preferably by August this year so sampling can start in November 2011. This request has only just reached us as WGSAM meets in the second week of October. However, we hope that there is still time to solve these issues if we act quickly. Would it be possible for you to specify exactly what clarifications you are looking for, and then we will get back to you as soon as possible?

Best wishes

Anna and Jason

WGSAM chairs

11.5 WGDIM to WGSAM

'It is recommended that the co-chairs of WGDIM and WGSAM and the ICES Data Centre get into contact intersessionally to discuss the feedback from WGSAM'.

WGSAM concurs that this is a good idea and shall meet with the appropriate personnel accordingly.

11.6 References

Lee, H.-H., Maunder, M. N., Piner, K. R., and Methot, R. D. 2011. Estimating natural mortality within a fisheries stock assessment model: An evaluation using simulation analysis based on twelve stock assessments. Fisheries Research, 109: 89–94.

Lewy, P., and Vinther, M. 2004. A stochastic age-length-structured multi-species model applied to North Sea stocks. 20. 33 pp.

Christensen, V., and Pauly, D. 1992. ECOPATH II — a software for balancing steady-state ecosystem models and calculating network characteristics. Ecological Modelling, 61: 169–185.

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Annex 2: Agenda

Monday 10-October

1000 Opening of the meeting
Adoption of ToR and Agenda
Volunteers to work on different ToRs and overview of presentations prepared for the meeting
1100 Coffee
ToR a), Presentations describing further progress in multispecies and ecosystem modelling
1300 Lunch
1400 Reconvene
ToR a), Presentations describing further progress in multispecies and ecosystem modelling
1600 Tea
Wrap up on ToR a)
Develop Following Day Workplan, Updates from earlier in day, Wrap up
1730 Adjourn

Tuesday 11-October

900 Initial presentations of ToR b): Key runs.
Key runs scheduled for 2011:
Ecopath North Sea and Baltic Sea, SMS North Sea, Gadget Bay of Biscay, Others?
Update runs: Baltic Sea, Others?
1030 Coffee
Discussion on ToR d), volunteers IDed to write up
1300 Lunch
1400 Reconvene
Discussion on ToR e), volunteers IDed to write up
1600 Tea
Continue discussion on ToR d), e), Subgroup reporting
Develop Following Day Workplan, Updates from earlier in day, Wrap up
1730 Adjourn

Wednesday 12-October

900 Revisit ToR b), Key run reports
Revisit, as need be, ToR d)-e)
1030 Coffee
ToR g) and h), discussion and workplan
1300 Lunch
1400 Reconvene
ToR i), discussion and workplan
1600 Tea
Discuss ToR c)
Subgroup reporting
Develop Following Day Workplan, Updates from earlier in day, Wrap up
1730 Adjourn

Thursday 13-October

900 Preliminary check on WG report elements
Revisit, as need be, ToR b)-e)
1030 Coffee
ToR g), h)
1300 Lunch
1400 Reconvene
ToR f), i) discussion and workplan.
1600 Tea
ToR f, g), h),i)
Subgroup reporting
Drafting session
Develop Following Day Workplan, Updates from earlier in day, Wrap up
1730 Adjourn

Friday 14-October

900 Revisit, as need be, ToR b)-e)
Revisit, as need be, ToR f)-i)
1030 Coffee
Subgroup reporting
Drafting session
Scope out next year meeting plan, schedule, ToR
1600 Adjourn

Annex 3: WGSAM terms of reference for the next meeting

The **Working Group on Multispecies Assessment Methods** (WGSAM) chaired by Anna Rindorf, Denmark and Jason Link, US, will meet in Venice, Italy, 22–26 October 2012 to:

a) Review **further progress** and report on key updates in multispecies and eco-system modelling throughout the ICES region;

b) Report on the development of **key-runs** (standardized model runs updated with recent data, and agreed upon by WGSAM participants) of multispecies and eco-system models for different ICES regions (including the Baltic Sea, and others as appropriate);

c) Work towards implementing new **stomach sampling** programmes in the ICES area in the near future;

d) Explore how '**virtual multispecies datasets**' (including survey, catch and stomach content data) for use in multiple multispecies models, especially for comparison and sensitivity testing, could be constructed;

e) Develop **foodweb and ecosystem indicators** relevant to the marine strategy framework directive (MSFD) from outputs of model key runs and input data;

f) Evaluate and explore **size spectra models** and compare to multispecies models in use in the ICES region;

g) Explore the **trophic role** of pelagic cephalopods (i.e. squid);

h) Explore the concept of Maximum Sustainable Yield (**MSY**) within a multi-species context and how it affects other biological reference points (BRP);

i) Address **requests** from other ICES Expert Groups as appropriate.

Of these, a, b and i are standing terms of reference, while c and d are 'multiyear projects'

Longer-term aspirations (possible ToRs for future years)

Review estimates of abundance and productivity at **lower trophic levels**, and work towards the inclusion of such information in multispecies models

Evaluate the major **sources of uncertainty** when making projections using multi-species and ecosystem models and explore possible best practices for addressing said uncertainties

Explore the **trophic role** of other non-assessed, but suspected important predator species (e.g. gurnard, starry ray)

Evaluate feasibility of including more **spatial structure** in models, and apply this e.g. to investigating the effects of marine protected areas

Provide advice on the '**infrastructure**' needed to support ecosystem/multispecies advice and modelling (data collection including process studies, modelling needs, database management, communication of results)

Explore how to utilize ecosystem/multispecies models in a management strategy evaluation (**MSE**) context

Work towards linking ecology and **economy** – valuation of goods and services from the ecosystem, exploring trade-off between MSY, MEY and conservation objectives

Work towards inclusion of **fleet dynamics** in multispecies models

Work towards multispecies harvest control rules (**HCR**)

Connection with the ICES science plan

The work outlined above fits well with the high priority research topics given in the ICES Science Plan for 2009–2013, and applies to all three thematic areas (Understanding ecosystem functioning, Understanding interactions of human activities with ecosystems, and Development of options for sustainable use of ecosystems).

Supporting information

Priority	Multispecies assessment modelling is essential to the development of viable long-term management strategies.
Scientific Justification and relation to action plan	The increased emphasis on ecosystem management (e.g. under the revised Common Fisheries Policy), and a move away from advising on single-stocks in isolation, necessitate consideration of interactions between key fish stocks and the ecosystems of which they are part. Historically the various ICES multispecies working and study groups have acted as a useful conduit, drawing together advice and quantitative outputs from many different assessment groups and combining these into an integrated product of direct use to managers and researchers. The past several meetings of WGSAM showed that there is much ongoing work within this field, and that there is a need for a pan-European (and more fully, northern hemisphere) forum for reviewing progress, and for learning about the 'best practice' of other research groups (ongoing ToR a). Multispecies models have often been used to provide updates of natural mortality M for inclusion in conventional single-species stock assessments (ToR b). Consequently it is considered useful to have occasional 'key-runs' for each region, whereby time-series are updated and model configurations are agreed and 'peer reviewed' by a number of regional experts. WGSAM will continue to work towards improved key-runs in the Barents Sea, Bay of Biscay, Baltic and North Sea, as well as working towards significant improvements in model functionality, for example the better characterization of the trophic role of understudied organisms, the development and application of new model types such as production models and size spectra models and the development of cross-model validation techniques (ToRs d, f and g). Stomach content data serve as the basis for a plethora of multispecies, extended single-species, and ecosystem models. Having a solid foundation of adequate stomach content data are a prerequisite for implementing the ecosystem-based approaches to fisheries. Stomach sampling has been annual in some areas, while in other areas (e.g. the North Sea) a large effort ('Year of the Stomach') has been made sporadically. At the 2012 WGSAM meeting the group will continue work towards implementing new stomach sampling programmes throughout the ICES area in the near future by reviewing protocols, pursuing new funding opportunities and gathering institutional support (ToR c). The ICES Science Plan for 2009–2013 highlights a top research priority for better understanding the impacts of fishing on marine ecosystems, in particulas on the biodiversity and helath of ecosystems. In 2012, WGSAM will increase their efforts in the development of foodweb and ecosystem indicators (ToR e), including both existing and new potential

	indicators in support of this top research priority. Other priority research areas that have been highlighted in the ICES Science Plan and which will be addressed by WGSAM at its 2012 meeting include: impacts of fishing on marine ecosystems (ToR b, e, f and g), and marine living resource management tools (ToRs a, b, d, e, f and h).
Justification of venue (in a non-ICES Member Country)	We propose to hold the next meeting of WGSAM in Venice, Italy in order to continuing widening the breadth of knowledge and experience in multispecies assessments. The University of Venice has a history of exploring ecosystem and multispecies models, particularly EwE as applied to Venice Lagoon and the Adriatic Sea. There is a keen interest there in our work, particularly to explore other approaches to ecosystem modelling and to mutually share best practices in such modelling. By meeting in Venice we will be able to meet with more of the staff involved with the modelling programs for those and associated ecosystems, and likely draw more of our colleagues from other Mediterranean facilities. The potential for regional capacity building would be an indirect benefit from holding the meeting at this venue, as would be the ability to compare and contrast different types of ecosystems, models and datasets on similar ecological and fisheries processes. Meetings beyond the traditional ICES locations typically have a beneficial effect upon attendance, although the dedication of WGSAM members generally means that attendance is likely to be good wherever we meet. The travel costs for attending a meeting in Venice will not be higher than for meeting in many ICES Member Countries, indeed they will be considerably lower than meeting in Copenhagen.
Resource Requirements	–
Participants	Approx 20. Expertise in ecosystem, modelling and fish stock assessment from across the whole ICES region.
Secretariat Facilities	None
Financial	No financial implications
Linkage to Advisory Committees	ACOM
Linkage to other Comities or groups	WGDIM, WGBIFS, IBTSWG, WGECO,WGMIXFISH, WGFE, most assessment Expert Groups, most EGs in the regional Seas Programme (e.g. WGINOSE, WGAIB, WGNARS, WGE2E
Linkages to other organizations	–

Annex 4: Recommendations

Recommendation	For follow up by:
1. BEWG: Produce a digitalized map of average benthos production and biomass by quarter and area for the North Sea (see explanation above)	BEWG
2. BEWG: Consider whether stomach data could provide information on the spatial and temporal changes in abundance of species or species groups difficult to sample with traditional gear types, and if the answer to this is affirmative, consider whether there would be interest in cooperating with WGSAM, IBTSWG and WGBIFS on planning and conducting future stomach sampling programmes	BEWG
3. WGBIFS: Provide further written information on questions raised concerning the stomach sampling manual	WGBIFS
4. IBTSWG: Provide further written information on questions raised concerning the stomach sampling manual	IBTSWG

Requests to other groups:

BEWG: Produce a digitalized map of average benthos production and biomass by quarter and area for the North Sea (Same as last year and the year before) or send a reply stating why this is not accomplished.

Explanation

Benthic food plays a large role in the diet of several North Sea predators. Among these are haddock and grey gurnard, two species which are important predators of sandeel (haddock), cod and whiting (grey gurnard). Unfortunately, the WGSAM does not have any information on the yearly variation in benthos production and biomass and is therefore forced to assume these as constant. However, future developments of the SMS will likely be able to include spatial differences in biomass and production of prey and the BEWG should be able to describe these to WGSAM. With these data, the model can take account of whether e.g. northern areas differ from southern in the amount of benthos present.

BEWG: Consider whether stomach data could provide information on the spatial and temporal changes in abundance of species or species groups difficult to sample with traditional gear types, and if the answer to this is affirmative, consider whether there would be interest in cooperating with WGSAM, IBTSWG and WGBIFS on planning and conducting future stomach sampling programmes.

Explanation

Benthic food plays a large role in the diet of several North Sea predators and in future sampling programmes, information on the diet of these predators may be of value to BEWG though it does not improve estimates of the amount of fish consumed. WGSAM therefore asks BEWG to consider whether determining benthos in stomach contents to species or species groups would provide a significant value to BEWG.

IBTSWG and WGBIFS: WGSAM requests IBTSWG and WGBIFS to assist in developing a manual for stomach samplings, comment on whether the sampling can be carried out in practice, whether the manual is sufficiently clear, what the expected

number of stomachs collected during the exercise would be (given expected length distributions and catch rates), what the demands would be regarding additional personnel and vessel time, whether the exchange format and labels suggested in Annex 5 are appropriate and sufficient and to design a structure for incoming datasets from future sampling campaigns.

Explanation

Providing advice on natural mortalities relies heavily on ecosystem models capable of evaluating how the effects of fishing and environmental change are spread through the ecosystem by complex foodweb interactions. The heart of all of these models is information on who-eats-who and how much. The last comprehensive investigation of species interactions in the North Sea and Baltic was conducted 20 years ago and is unlikely be representative of what is now a very different ecosystem. In order to assure that the multispecies and ecosystem models provide reliable predictions, WGSAM considers that they should be calibrated with up to date information.

WGSAM therefore suggests that under the auspices of ICES, the process of collecting food composition data on existing surveys should be initiated. To assure that the stomach data sampled are subsequently used in the multispecies and ecosystem models, it is essential that a standardized sampling protocol is followed as was the case in previous large-scale stomach sampling exercises (ICES 1991). However, since these exercises were performed new knowledge of the statistical properties of stomach content analyses has been gained. The annex attached to WGSAM 2010 (Annex 5) is a first draft of an updated sampling manual and is intended to be the start of the communication with IBTSWG and WGBIFS on the details of the sampling procedure as requested by PGCCDBS.

Annex 5: Report on Key Run for the North Sea Ecopath with Ecosim Ecosystem Model, 1991–2007

Key run summary sheet

Area	North Sea
Model name	Ecopath with Ecosim
Type of model	Foodweb compartment
Run year	2011
Species/ Groups	68 functional groups
Time range	1991–2007
Time-step	Monthly
Area structure	Model covers North Sea ICES division IVa,b and c. No spatial aspects are modelled.
Stomach data	1991 year of the stomach and others
Purpose of key run	Describing changes in the North Sea ecosystem and providing model data
Model changes since last key run	First key run
Output available at	http://www.ices.dk/workinggroups/ViewWorkingGroup.aspx?ID=193
Further details in	Report of the Working Group on Multispecies Assessment Methods 2011, ANNEX 5

About this annex

This annex describes the updates to the parameterization of the North Sea Ecopath with Ecosim model and its calibration to time-series data 1991–2007. The contents have been presented and discussed at the ICES WG Multispecies Assessment Model in October 2011 and are published as an Annex to that report. The report and output data files are made available via the WGSAM webpage and also via the Cefas website. Plans for periodic updates are in discussion. The contents Table and text provides details of the key associated spreadsheet file, allowing users to access key input and output data. An inventory of all output and supporting data files are given in Appendix 5A. The model has a component describing the modelling of fleets and economics, which uses data reported in the 2008 Annual Economic Report (EU 2008)[1] to define the ex-vessel price (Euro/tonne) of each species to each fleet and the cost and revenue of each modelled fleet. However, because the necessary expertise for reviewing this component was not present in 2011, it is omitted from this Annex. Details are reported in Heymans *et al.* (2011, and SAMs report 166).

[1] The AER report contains economic data from all EU Member States collected under the framework of the Data Collection Regulation (Council Regulation (EC) No. 1543/2000).

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North Sea Ecopath model

2.1 Updated model

The North Sea Ecopath model published by Mackinson and Daskalov, (2007), and Mackinson *et al.* (2009) was updated to:

- include multi-stanza representation of life stages for 5 species
- reflect current information on catches, discards and
- include fleet economics, in particular the amount of subsidies.

Details of the updates to catch profiles and economic data are published in Heymans *et al.* (2011). Additional details of the changes are reported below.

2.1.1 Multi-stanza representation of life stages

Multi-stanza groups were used to represent life-history stages and associated ontogenetic changes in feeding for 5 important commercial fish species; cod, whiting, haddock, saithe and herring. Each species was represented with 2 stages, adult and juvenile.

Multi-stanza representation tracks monthly cohorts of each stanza and makes species dynamics more realistic by allowing for explicit consideration of density- and risk-dependent growth and how this influences stock–recruitment relationships (and its impacts on sustainable fishing rates). It also provides consideration of stanza-specific habitat use in Ecospace. For each stanza, the mortality rates (M0, predation, fishing) and diet composition are assumed to be similar for individuals within each stanza (e.g. larvae having high mortality and feeding on zooplankton, juveniles having lower mortality and feeding on benthic insects, adults having still lower mortality and feeding on fish).

The multi-stanza representation improves upon the previous 'split pool' representation used in the North Sea model (Mackinson and Daskalov, 2007). Walters (pers comm.) says that with multi-stanza representation "*the internal calculations of survivorship and biomass are done in monthly age steps, so as to allow finer resolution than one year in the stanza biomass and mortality structure (e.g. larval and juvenile stanzas that last only one or a few months).*" In doing so, the seasonal effects of production of juveniles are spread evenly over the annual cycle. The important implication of this for users to note is that the biomass of juveniles as estimated by the model does not reflect the actual biomass in $t.km^{-2}$. Instead, the biomass of Juveniles in the model becomes a "*de-seasonalised proxy variable*" (Walters, pers comm.). Spreading the production of juveniles across the year avoids the problems with having to represent a seasonal production of food and feeding of juveniles in that short period of time. In general, the model biomass of juveniles might be expected to be around $1/12^{th}$ of real biomass. When moving from 'split-pool' to multi-stanza representation this means that predation mortality of juveniles in the model can be much too high, thus requiring reductions to the proportion of juvenile stanzas in the diet of their predators.

Further details on multi-stanza representation in Ecosim and Ecospace are described in Christensen and Walters (2004), Walters and Martell (2004), Christensen *et al.* (2005) and Walters *et al.* (2010).

Implementation of the multi-stanza representation

Implementation required specification of the parameters for each stanza (Table 2.1), followed by minor changes to the diet compositions and discards of juvenile stanzas, such that the predation and fishing mortalities on each stanza were consistent with the data and model described in Mackinson and Daskalov, (2007).

Table 2.1. Multi-stanza parameters.

Multi–stanza name	Cod	Whiting	Haddock	Saithe	Herring
Vbert K (Nsea)	0.23	0.32	0.19	0.07	0.38
Recruit power	1	1	1	1	1
BA/B	0	0	0	0	0
Age at maturity (Nsea)	3.5	1.5	2.5	4.5	2
Wmat (g) (Nsea)	3800	150	400	1500	115
Mean weight of adult (g) (Nsea)	6500	250	619	3000	183
Winf (g) FishBase	17610	8150	2728	54733	230
Wmat/Winf	0.22	0.02	0.15	0.03	0.5
Ref	WGNSSK 08	WGNSSK 08	WGNNSK 08	WGNNSK 08	HAWG 2009

2.1.2 Landing and discard profiles of the fisheries

The proportion of the landings and discards of each species taken by each fleet, as reported by STECF (Scientific, Technical and Economic Committee for Fisheries) from 2003 to 2007, was used to update the distribution of landings and discards among the 12 modelled fleets. The STECF does not resolve the catch information to different age groups. Thus, for functional groups split into adult and juvenile components in the model (cod, *Gadus morhua*; whiting, *Merlangius merlangus*; haddock, *Melanogrammus aeglefinus*; saithe, *Polachius virens*; and herring, *Clupea harengus*), the distribution of the catch to landings and discards was maintained as in the original 1991 model (Mackinson and Daskalov, 2007). This division is made based on data from discard sampling trips undertaken from 1994–2007. The result of the re-profiling of the distribution of catches is that the model maintains the fishing mortalities of each species in 1991, and hence mass-balance, but is better suited to address the future policy questions addressed here because it reflects the present day fleet structure more accurately.

2.1.3 Fishing fleet structure

The Data Collection Framework (DCF) provides the basis for this mapping since it is used to define the fleet structures used in both the AER reports and ecosystem model (Table 2.2, (Table S1 in Heymans *et al.*, 2011)). The mapping is however, not a perfect one, with some differences in the fleet descriptions used by the AER, DCF and ecosystem model still remaining. Where AER fleets did not have a direct link to a fleet in the model, the associated catch compositions were examined and used to assign the AER fleet to its corresponding model fleet.

It should be noted that this fleet structure, and associated socio-economic considerations, were not reviewed by WGSAM, which focused its evaluation of this key-run on

the biological responses to fisheries removals, environmental effects, and foodweb dynamics. This description of fleets was retained in the annex to facilitate interpretation the results of this EwE work. Further review of this material would be required for use in advising on several other socio-economic issues, requiring expertise not present in WGSAM this year. Additional information on associated socio-economics (e.g. revenue, costs, employment) can be found in SAMS report 166 (Heymans *et al.*, 2011).

Table 2.2. AER and Ecopath model fleet groups (where classification was uncertain main species caught is given).

Gear type	Size groups used
BEAM TRAWLS	
BEL_Beam trawl	12m - 24m and 24m - 40m
DEU_Beam trawl	24m - 40m
DNK_Beam trawl	24m - 40m
FRA_Beam trawl	0m - 12m, 12m - 24m and 24m - 40m
GBR_Beam trawl	0m - 12m, 12m - 24m, 24m - 40m and >40m
NLD_Beam trawl	0m - 12m, 12m - 24m, 24m - 40m and >40m
DEMERSAL TRAWL and DEMERSAL SEINER	
BEL_Demersal trawl and demersal seiner	24m - 40m
DEU_Demersal trawl and demersal seiner	0m - 12mm, 12m - 24m and 24m - 40m
DNK_Demersal trawl and demersal seiner	0m - 12m
FRA_Demersal trawl and demersal seiner	0m - 12m, 12m - 24m, 24m - 40m and >40m
GBR_Demersal trawl and demersal seiner	24m - 40m and >40m
NLD_Demersal trawl and demersal seiner	0m - 12m, 12m - 24m and 24m - 40m
SWE_Demersal trawl and demersal seiner	24m - 40m
DREDGES	
DNK_Dredges 0m - 12m	0m - 12m and 12m - 24m
FRA_Combining mobile and passive gears	0m - 12m and 12m - 24m

FRA_Dredges	0m - 12m, 12m - 24m and 24m - 40m
FRA_Other passive gears	0m - 12m
FRA_Polyvalent mobile gears	0m - 12m and 12m - 24m
GBR_Dredges	0m - 12m, 12m - 24m, 24m - 40m and >40m
NLD_Dredges	0m - 12m, 12m - 24m, 24m - 40m and >40m
DRIFT and FIXED NETS	
BEL_Drift nets and fixed nets	12m - 24m
DEU_Drift nets and fixed nets	12m - 24m
DEU_Passive gears	0m - 12m
FRA_Drift nets and fixed nets	0m - 12m, 12m - 24m, 24m - 40m and >40m
FRA_Polyvalent passive gears	0m - 12m
GBR_Drift nets and fixed nets	0m - 12m, 12m - 24m, 24m - 40m and >40m
NLD_Polyvalent passive gears	0m - 12m
SWE_Drift nets and fixed nets	12m - 24m
GEARS USING HOOKS	
FRA_Gears using hooks	0m - 12m, 12m - 24m and 24m - 40m
GBR_Gears using hooks	0m - 12m, 12m - 24m and 24m - 40m
SWE_Gears using hooks	12m - 24m
NEPHROPS TRAWLERS	
DNK_Demersal trawl and demersal seiner 12m - 24m	12m - 24m
FRA_Demersal trawl and demersal seiner 12m - 24m	12m - 24m
GBR_Demersal trawl and demersal seiner 0m - 12m	0m - 12m and 12m - 24m
SWE_Demersal trawl and demersal seiner 0m - 12m	0m - 12m and 12m - 24m
Non Active Vessels	
DEU_Non Active Vessels	0m - 12m, 12m - 24m, 24m - 40m and >40m

GBR_Non Active Vessels	0m - 12m, 12m - 24m, 24m - 40m and >40m
NLD_Non Active Vessels	0m - 12m, 12m - 24m, 24m - 40m and >40m
SWE_Non Active Vessels	0m - 12m and 12m - 24m
OTHER METHODS	
DNK_Combining mobile and passive gear	0m - 12m and 12m - 24m
DNK_Polyvalent passive gears	0m - 12m and 12m - 24m
FRA_Combining mobile and passive gears	24m – 40m
FRA_Other mobile gears	0m - 12m and 12m - 24m
FRA_Other passive gears	12m - 24m
FRA_Polyvalent passive gears	12m - 24m
GBR_Combining mobile and passive gears	0m - 12m and 12m - 24m
GBR_Polyvalent mobile gears	0m - 12m, 12m - 24m, 24m - 40m and >40m
GBR_Polyvalent passive gears	0m - 12m, 12m - 24m and 24m - 40m
NLD_Other passive gears	0m - 12m, 12m - 24m and 24m - 40m
NLD_Polyvalent passive gears	12m - 24m and 24m – 40m
SWE_Passive gears	0m - 12m
PELAGIC TRAWLS and SEINE	
DNK_Pelagic trawls and seiners	12m - 24m, 24m - 40m and >40m
FRA_Pelagic trawls and seiners	0m - 12m, 12m - 24m, 24m - 40m and >40m
FRA_Polyvalent mobile gears	24m - 40m
GBR_Pelagic trawls and seiners	0m - 12m, 12m - 24m, 24m - 40m and >40m
NLD_Pelagic trawls and seiners	0m - 12m, 12m - 24m, 24m - 40m and >40m
SWE_Pelagic trawls and seiners	12m - 24m, 24m - 40m and >40m
POTS	

FRA_Pots and traps	0m - 12m
FRA_Pots and traps	12m - 24m
GBR_Pots and traps	0m - 12m, 12m - 24m and 24m - 40m
SHRIMP TRAWLERS	
DEU_Beam trawl 0m - 12m	0m - 12m and 12m - 24m
DNK_Beam trawl 12m - 24m	12m - 24m

2.2 Updated results Tables

[All data Tables are given in the File: NSea 1991_Key Run_Ecosystem Indices_Ecopath.xlsx]

Changes to the model to include multi-stanza representation, landings, discards and economics resulted in the following changes to the model representation reported in Mackinson and Daskalov, (2007).

Input data

- estimates of diet proportions
- specification of multi-stanza parameters
- landings and discards allocated to each fleet
- price of each species to each fleet
- proportions of fixed and variable costs of fishing

Output estimates

- consumption estimates (output)
- predation mortality rates (output)

3. Time-series data

[Data from 1991–2007 are given in the file: NSea Time series 1991_2007_csv DATA & GRAPHS.xlsx]

Data on fisheries and environmental variables covering the period 1950–2007 were collated and used in empirical analyses and for calibrating biomass dynamics of the North Sea ecosystem model. The aim of the data collation was to obtain quantitative information concerning as many functional groups in the model as possible, such that the behaviour of each group was subject to constraints justified by available data.

The data comprise stock assessment data, biological survey data, fishing mortalities, catch and fishing effort data, resulting in a complete dataset of 266 variables, 114 of which are used in the key run presented here (see Section 5). The data were used in two ways; (i) as a constraint in the fitting procedure, and (ii) to guide choices. The treatment of the data is described in detail in Mackinson (in prep), Appendix 2 and summarized below.

Tables 3.1 and 4.1 summarize the data sources and describes the strategy for how they were applied in model fitting. Where multiple relative biomass time-series existed, only one time-series per functional group was chosen to fit to; this being the most complete and or the one for which evidence from correlations showed it to be most strongly correlated with driver series. E.g. Station M1 for Dove time-series had a stronger correlation with environmental time-series than stn P.

Most relative biomass time-series were available up to 2007. In cases where they were not, the series was filled up to 2007 by maintaining the last value as a constant, under the weak, but not unreasonable assumption that it is often true that the last year value is the best predictor of the next year (i.e. strong autocorrelation in time-series). An example is the last year for plankton groups.

Additional descriptions of the data sources used are given below.

1.1 Data sources

Fisheries Independent Monitoring Surveys – International Bottom Trawl Surveys

http://datras.ices.dk/Home/Default.aspx

Fisheries Landings

ICES Statlant

ICES collates international databases on fisheries landings, which includes landings of fish and shellfish from 20 countries, for each species at the spatial resolution of an ICES division (Figures 1–3). In the North Sea, there are 3 Divisions, IVa,b,c. The Coordinating Working Party on Fishery Statistics (CWP) organizes the collection of these statistics under the STATLANT programme. ICES have published these data in the Bulletin Statistique des Pêches Maritimes from 1903 to 1987, and from 1988 onwards in ICES Fisheries Statistics. Until 2009, the STATLANT database only contained data from 1973. However, ICES is working on making its data more available and this year, electronic data back to 1950 became available. However, using it requires combining results from one database with another. The database provides a comprehensive catalogue of reported landings for 223 North Sea species. Unfortunately they are not broken down by gear type. The landings database is accessed and manipulated using an FAO database query tool called Fishstat. Fishstat can also be used to query the FAO international fisheries landings data collated by themselves.

http://www.ices.dk/fish/CATChSTATISTICS.asp

Scientific and Technical and Economic Committee for Fisheries – STCF database 1991 and STECF 2008

The 1991 STCF North Sea database contains port landings data and some economic data for the top 27 commercially important species in the North Sea. Data come from eight countries and are divided into 58 fleets (Lewy *et al.*, 1992; Vinther and Thomsen, 1992). Discard values are already included for five species: cod, haddock, whiting, plaice and sole. This one-off database is published, but not widely available. It has been superseded by the data collected by member states under the EU Data Collection Regulations. This includes data on landings, discard and economics for each fleet segment in each country and is collated by STECF to address questions from the commission, to where requests for the data may be sent. STECF encourages member states to make the data available to ICES expert groups for analyses. Data from STECF up to 2007 are used in the fitting.

Stock assessments

Single species stock assessment

ICES member states undertake numerical assessments of the status of stocks in the North Sea. A variety of models based on the principals of Virtual Population Analysis are used. Details of individual stock assessments are provided in ICES working group reports (WGNSSK in the case of the North Sea) and summarized by region in ICES advice. In 2009, the input and output files of ICES stock assessments for each region became available online for the first time, containing data on: Recruitment, Spawning-stock biomass (SSB), Total Biomass, Landings and Mean Fishing mortality.

http://www.ices.dk/datacentre/StdGraphDB.asp

Multispecies Vitual Population Analysis (MSVPA), SMS 2008 Key run.

Time-series from the SMS Key run published in WGSAM 2008 were used in the model fitting

http://www.ices.dk/workinggroups/ViewWorkingGroup.aspx?ID=193

UK National Marine Monitoring Programme for Benthos

The National Marine Monitoring Programme was initiated in the late 1980s to coordinate marine monitoring in the UK between a number of organizations. The NMMP aims to detect long-term trends in the quality of the marine environment, to ensure consistent standards in monitoring, to establish appropriate protective regulatory measures, to coordinate and optimize marine monitoring in the UK, and to provide a high quality key dataset for key variables. Specific non-disturbed sites around the UK are sampled annually to gain an understanding of background changes in benthic communities. The data is collated by Cefas and in the North Sea includes two sites extending back to 1993.

Dove Benthos time-series

The Dove Marine Laboratory benthos time-series includes information on the relative abundance of benthos species from two sites off the Northumberland coast, sampled twice yearly by the University of Newcastle since 1971. Contact: ben.wigham@newcastle.ac.uk

Plankton monitoring data

Continuous Plankton Recorder (CPR) data on zooplankton and phytoplankton in the North Sea were provided by SAHFOS (Priscilla Licandro, David Johns).

Environmental indices

Numerous sources for environmental time-series data already exist on the Internet. Some provide series of direct observation measurements such as temperature and salinity, while other provide composite indices (e.g. NAO, Gulf Stream Index, AMO). Table 3.1 identifies those used in the model parameterization. A key resource for environmental time-series data is Climate Diagnostics Center, National Oceanographic and Atmospheric Administration of the US Department of Commerce (www.cdc.noaa.gov).

http://www.esrl.noaa.gov/psd/data/climateindices/list/

Nutrient and chemistry

Data from research cruises undertaken since 1903 to present are available from the ICES datacentre to download or upon specific request. Data were collated for winter

surveys from 1903 onwards and were used in correlation analyses (Mackinson, in prep) and in model fitting.

http://ecosystemdata.ices.dk/

Other information

Many of the information sources provide data for different regions. ICES is making great efforts to make its data assets publicly available via an online warehouse, so it is worth checking there frequently for updates. http://ecosystemdata.ices.dk/

Table 3.1. Metadata summary Table. Full metadata given in Appendix 2. (from Mackinson in prep). NOTE: Only data from 1991 onwards are used in the model calibration reported here. Longer time-series are used in the empirical analyses and in preparation for calibration of the 1973 model representation (see Mackinson and Daskalov).

Data type	Dates (min to max)	Database/Model	Used for	Source
Catch	1950–2007	ICES Statlant	Fitting catch / display	
Catch	1963–2003	MSVPA 2005 Key run	Fitting catch	WGMSNS 2006
Catch	1963–2008	Single species (SS) stock assessments	Fitting catch	ICES WG reports
cpue	1984–2003	Cephalopods, Shrimp and Nephrops	Fitting catch	Cephalo-pods (Gra-ham Pierce, pers. comm.), WGPAN04 (Pandalus), WGNSSK08 (Nephrops)
Relative biomass	1983–2006	International Bottom Trawl Survey (IBTS)	Fitting biomass	ICES datras
Relative biomass	1977–2008	IBTSQ1 survey	Fitting biomass	ICES datras
Relative biomass	1963–2008	SS assessment	Fitting biomass	ICES WG reports
Relative biomass	1963–2003	MSVPA 2005 Key run	Fitting biomass	WGMSNS 2006
Relative biomass	1963–2007	Stochastic Multispecies Model 2008 (Vinther pers. comm)	Fitting biomass	WGSAM 2009

Data type	Dates (min to max)	Database/Model	Used for	Source
cpue	1984–2003	Cephalopods, Shrimp and Nephrops	Fitting biomass	Cephalo-pods (Pierce pers. comm.), WGPAN04 (Pandalus), WGNSSK08 (Nephrops)
Relative biomass	1958–2007	Continuous Plankton Recorder abun-dance and biomass	Fitting biomass	Licandro, Johns SAHFOS
F	1963–2007	SS assessment	Driver	ICES WG reports
F	1963–2003	MSVPA 2005 Key run	Driver	WGMSNS 2006
Z	1963–2003	MSVPA 2005 Key run	Fitting mortality	WGMSNS 2006
Relative biomass	1993–2006	National Maring Monitoring Programme - Benthos	Fitting biomass	Cefas Uni-corn data-base
Relative biomass	1971–2006	Dove Laboratory benthos time-series	Fitting biomass	Newcastle and Liver-pool Univer-sity, in the public do-main at Bristish Oceano-graphic Data Centre.
Fishing effort, by gear	1973–2008	SS assessment	Driver	ICES WG reports, and pers. comms.
Environ-NAO Win-ter Index	1950–2007	http://www.cgd.ucar.edu/cas/jhurrell/indices.html	Driver	Web
Environ-AMO	1950–2007	http://www.esrl.noaa.gov/psd/data/timeseries/AMO/	Driver	Web
Environ-AMO smoothed (USGS)	1950–2007	http://www.esrl.noaa.gov/psd/data/timeseries/AMO/	Driver	Web

Data type	Dates (min to max)	Database/Model	Used for	Source
Environ-Gulfstream Index	1950–2007	http://web.pml.ac.uk/gulfstream/data.htm	Driver	Web
Environ-Faroe/Shet flux (Q1)	1950–2007	Skogen, M., and Søiland, H. 1998. A user's guide to NORWECOM v2.0. The NORWegian ECOlogical Model system. Institute of Marine Research, Bergen. Technical report Fisken og Havet, 18/98. 42 pp.	Driver	Publication
Environ-l Marsdiep T	1950–2007	Data from van Aken 2003, as in MacKenzie and Schiedek, 2007	Driver	Publication
Environ-Marsdiep Salinity	1950–2007	Data from van Aken 2003, as in MacKenzie and Schiedek, 2007	Driver	Publication
Environ-HADISST	1950–2007	North Sea Hadley Centre data from Rayner *et al.*, 2003, as in MacKenzie and Schiedek, 2007	Driver	Publication based on Hadley Centre data
Environ-Atlantic tripole	1950–2007	http://www.esrl.noaa.gov/psd/data/climateindices/list/#Atlantictripole	Driver	Web
Environ-Windspeed	1950–2007	Llope, M *et al.* 2009. Effects of environmental conditions on the seasonal distribution of phytoplankton biomass in the North Sea. Limnol. Oceanogr., 54(2), 2009. Data supplied by author.	Driver	Llope pers comm.
Environ-Nutrients	1903–2010	ICES oceanographic survey data, on request. http://www.ices.dk/ocean/	Driver	Neil Holdsworth ICES data manager

3.2 Relative biomass

Only those series used in fitting the model are shown

Survey data fish, benthos and plankton (Figure 3.1, 3.2).

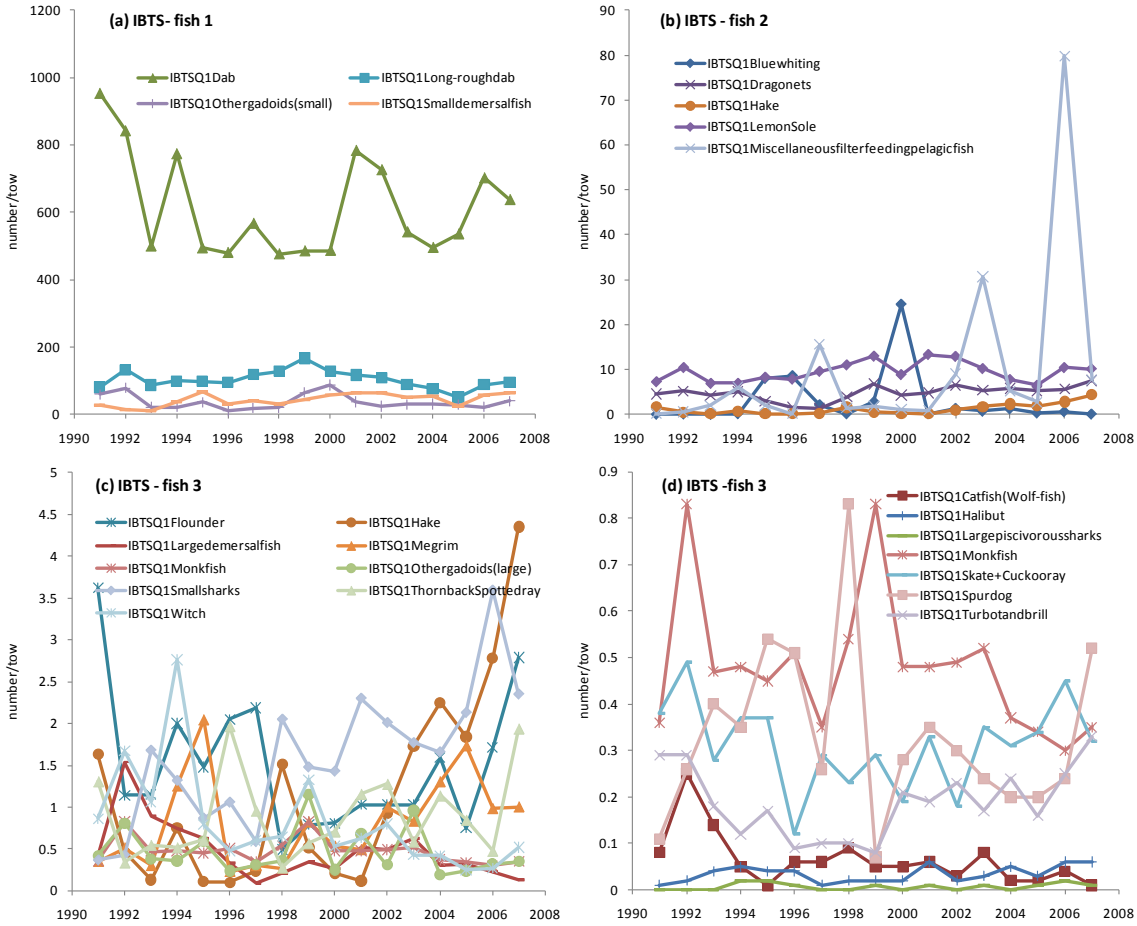


Figure 3.1. IBTS survey data used as relative biomass series in model fitting.

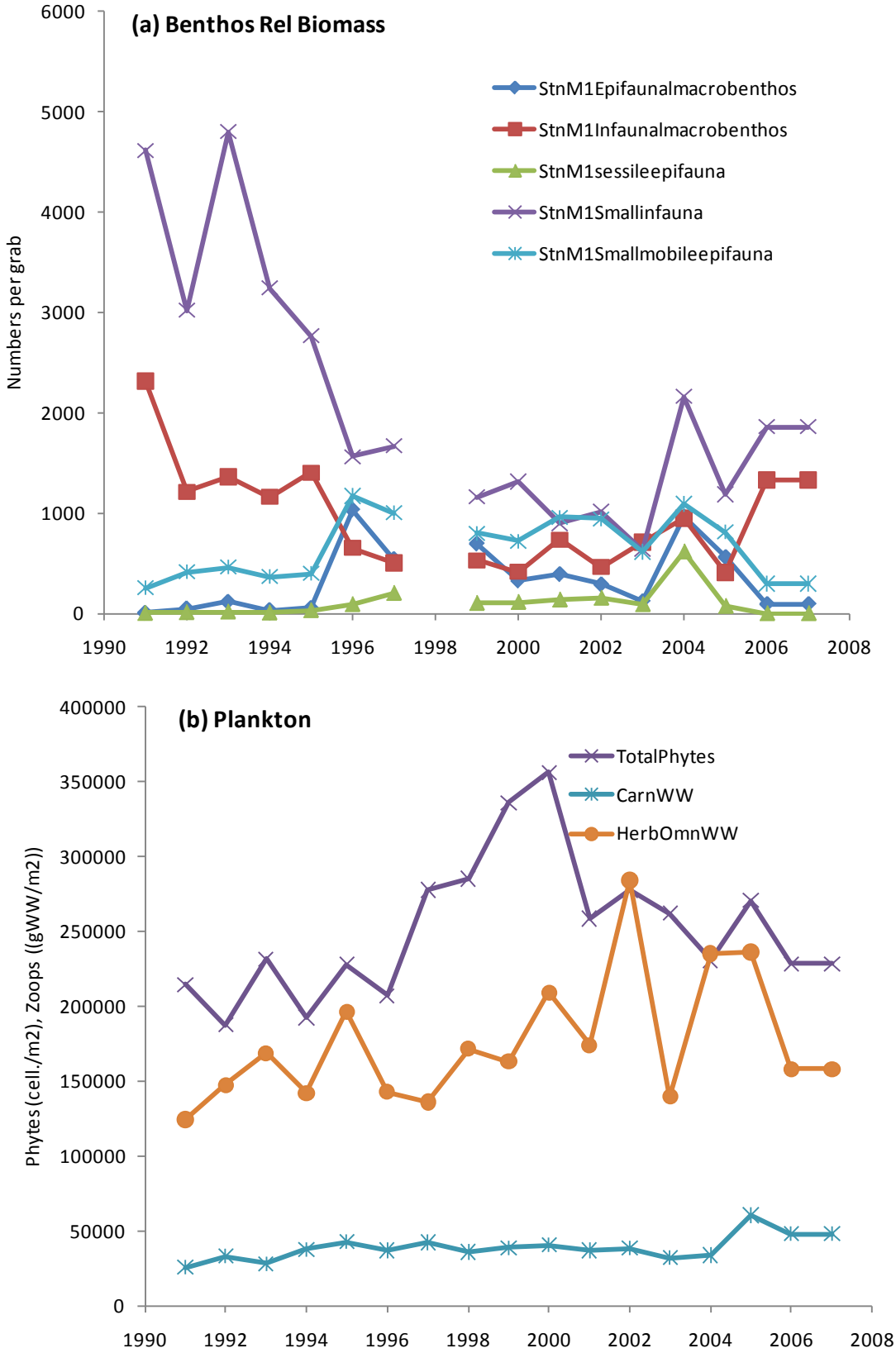
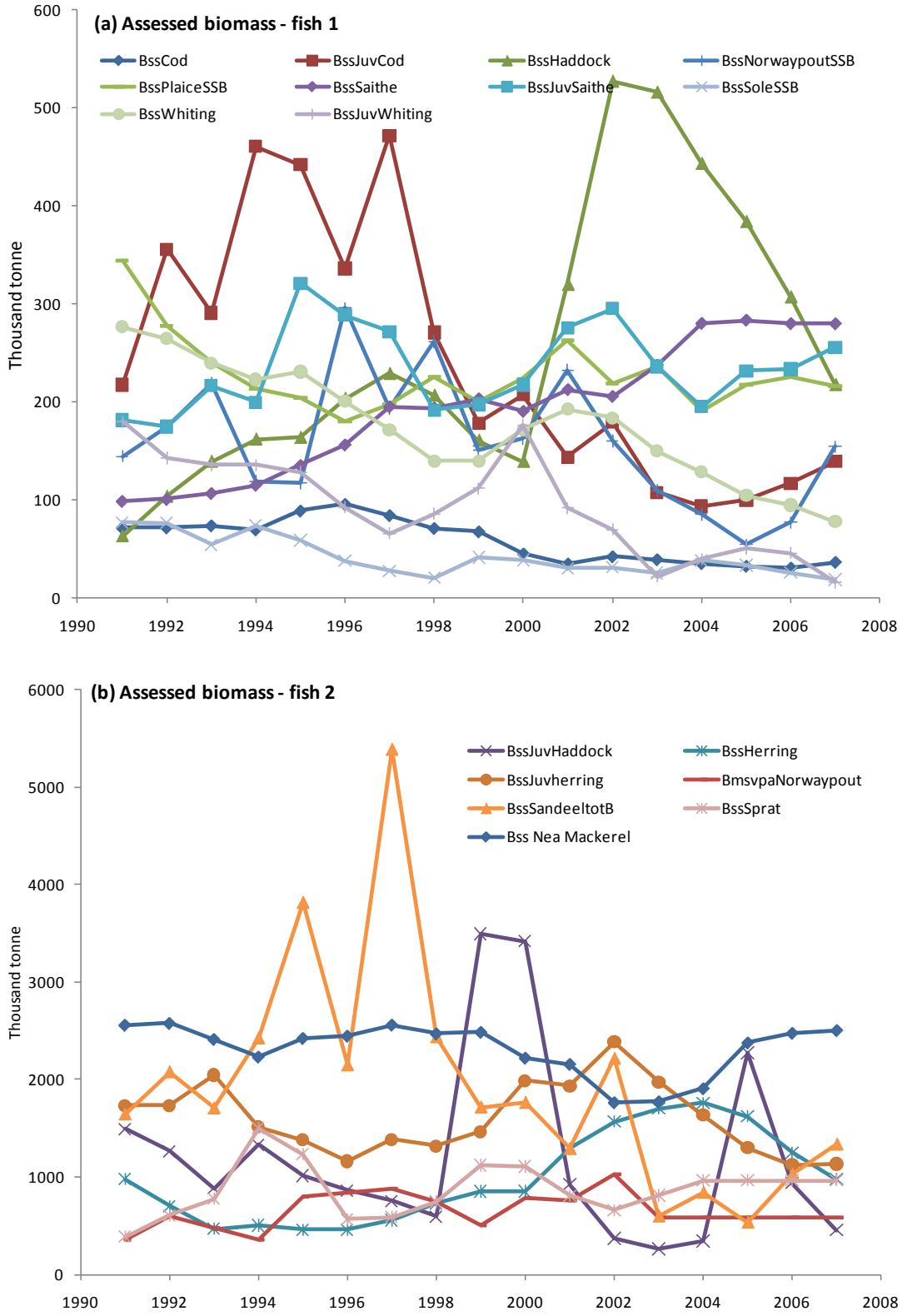


Figure 3.2 (a). Benthos time-series (Dove Marine Laboratory data) and (b) Plankton data (CPR) as indices of relative biomass in model fitting.



3.2.2. Stock assessment – single species and multispecies biomass (Figure 3.3).

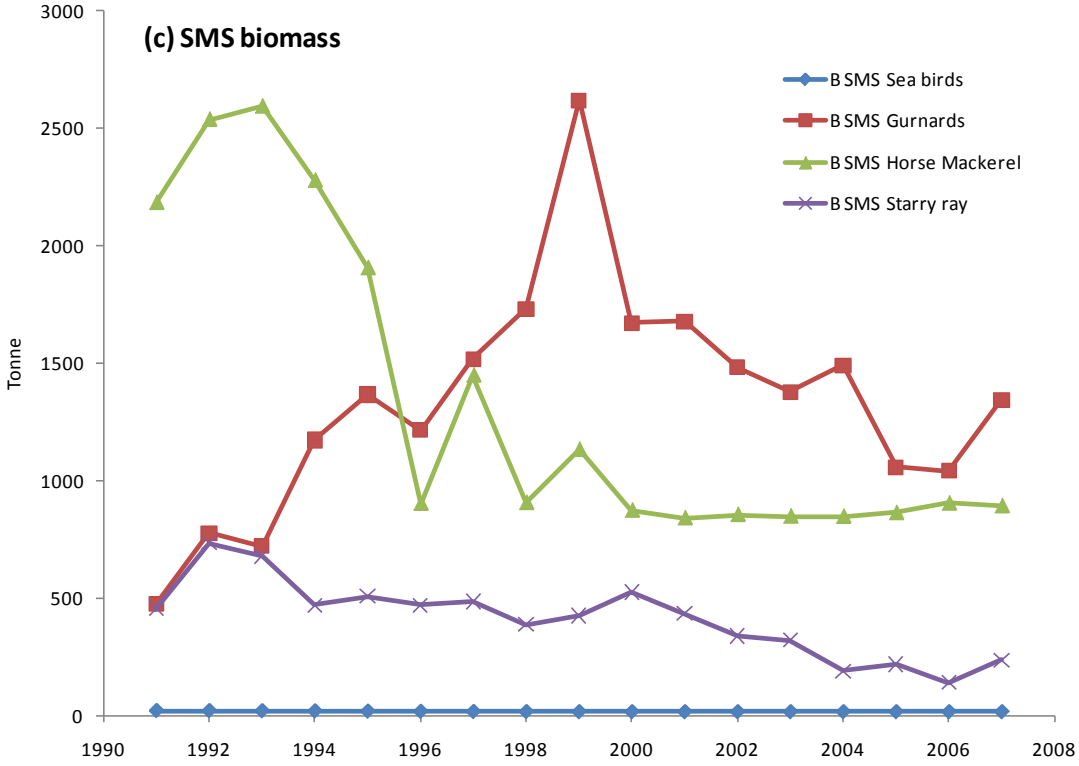


Figure 3.3. Single species (a and b) and multispecies (c) stock assessment data used as relative biomass series in model fitting.

Fishing

3.3.1 Fishing mortality (Figure 3.4)

For assessed species, fishing mortality time-series were taken from the ICES single-species assessment data. For non-assessed groups, Ecosim calculates a fishing mortality time-series for each group by taking the partial fishing mortality estimates for each group-fleet combination that is defined in the ecopath model base year (1991 here), then multiplying them by time-series data on the relative effort of each fleet. This results in a partial fishing mortality time-series for each group-fleet pair, which is summed to provide a time-series of fishing mortality on each group. Thus, although relative effort series are used as 'input' to Ecosim, it is the calculated fishing mortality on each group that is used to drive the model.

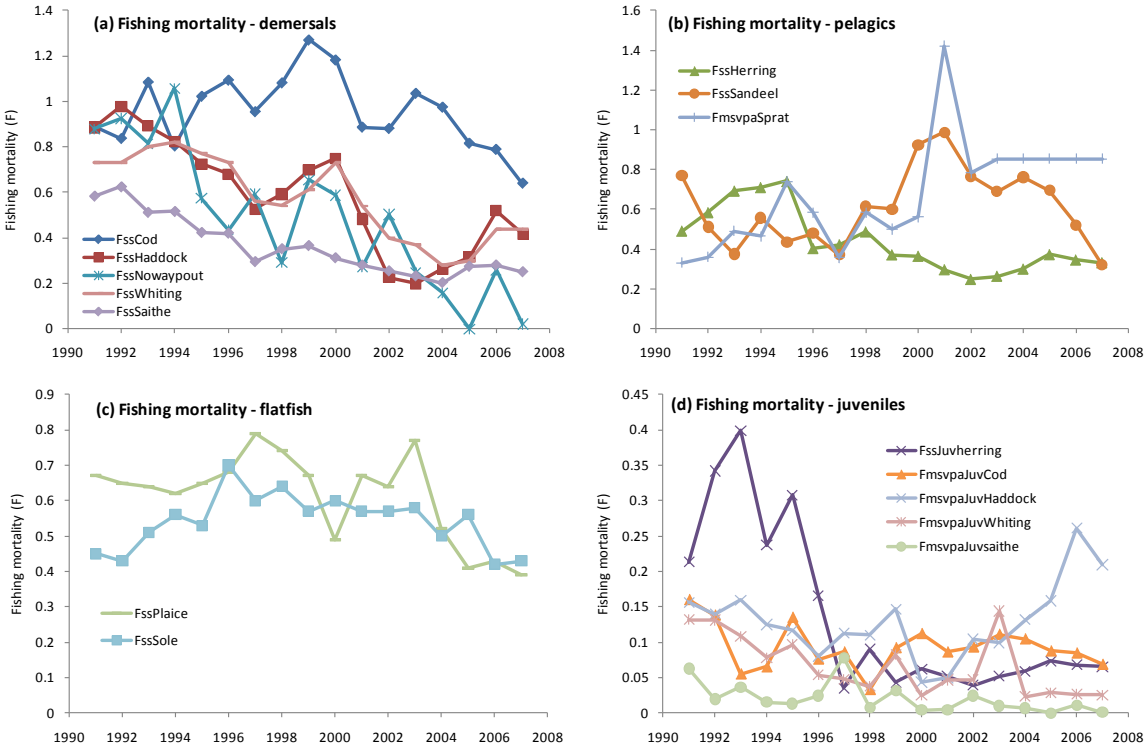


Figure 3.4. Fishing mortality data used to drive changes in the model.

3.3.2 Fishing Effort data– stock assessment tuning data (Figure 3.5, 3.6)

Trends in effort of the tuning fleets used in ICES stock assessments were obtained from WG assessment reports (Table 3.2) and mapped to fleet categories based on the Data Collection Framework. The effort trends (Figure 3.5, 3.6) are consistent with that published in STECF SGRST report 2009. In recent years, assessment procedures for some stocks have moved away from using the effort of commercial fleets to tune stock assessments, in favour of using fisheries-independent survey data. The outcome of this is that the time-series are no longer routinely reported in ICES WG assessment reports and must be sought directly from sources in each country. Note, for example that effort data are not reported for fleets targeting *Nephrops* and shrimp since 2003 and Scottish Trawls since 2004.

Where effort data is not available in recent years (*Nephrops* and shrimp since 2003; Pelagic fish in 2005 and 2007), the value of the previous year was maintained. No information of changes in effort is available for the remaining gears, so effort is assumed to have been stable since the base year (1991).

Table 3.2. Sources for effort data categorized in to DCF fleets.

DCF Fleet	Period available	Source and notes
Demersal trawl and seine	1978–2008	(WGNNSK08, 2008), including fleets SCOSEI_IV, SCOLTR_IV, ENGTRL_IV, ENGSEI_IV, FRATRB_IV, FRATRO_IV, NORTRL_IV, GER_OTB_IV
Beam trawl	1979–2007	(WGNNSK08, 2008), including fleets NL_BT_EFF,UK_BT_EFF
Pelagic trawl and seine	1987–2006	(WGNNSK08, 2008) fleet NOR_DEN_NPOUT_EFF
Nephrops trawl	1981–2004	(WGNSSK06, 2006), summed over Nephrops functional units
Shrimp trawl	1984–2003	(WGPAN, 2005), Pandulus – total international effort in ICES div IV.

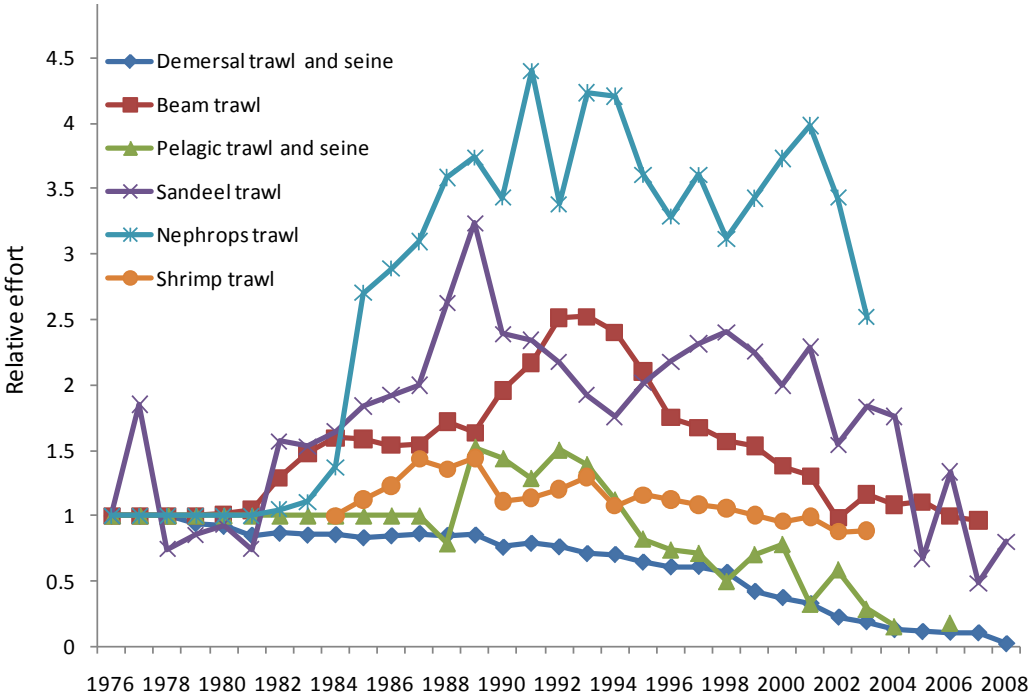


Figure 3.5. Relative effort of principal North Sea fleets since 1976. Note: see Table 3.1 for description of aggregation of fleet categories used.

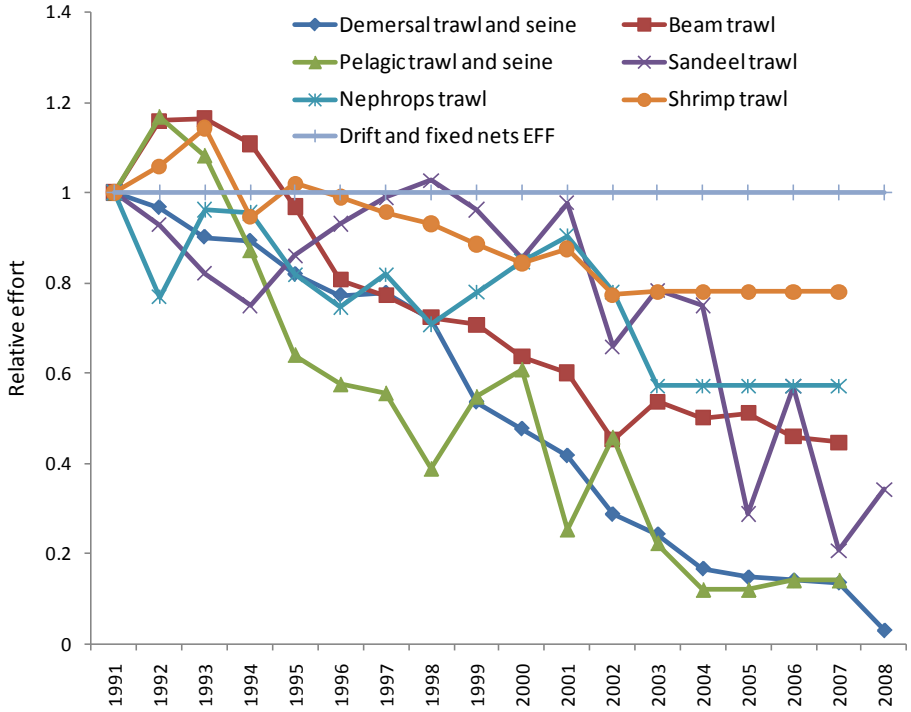
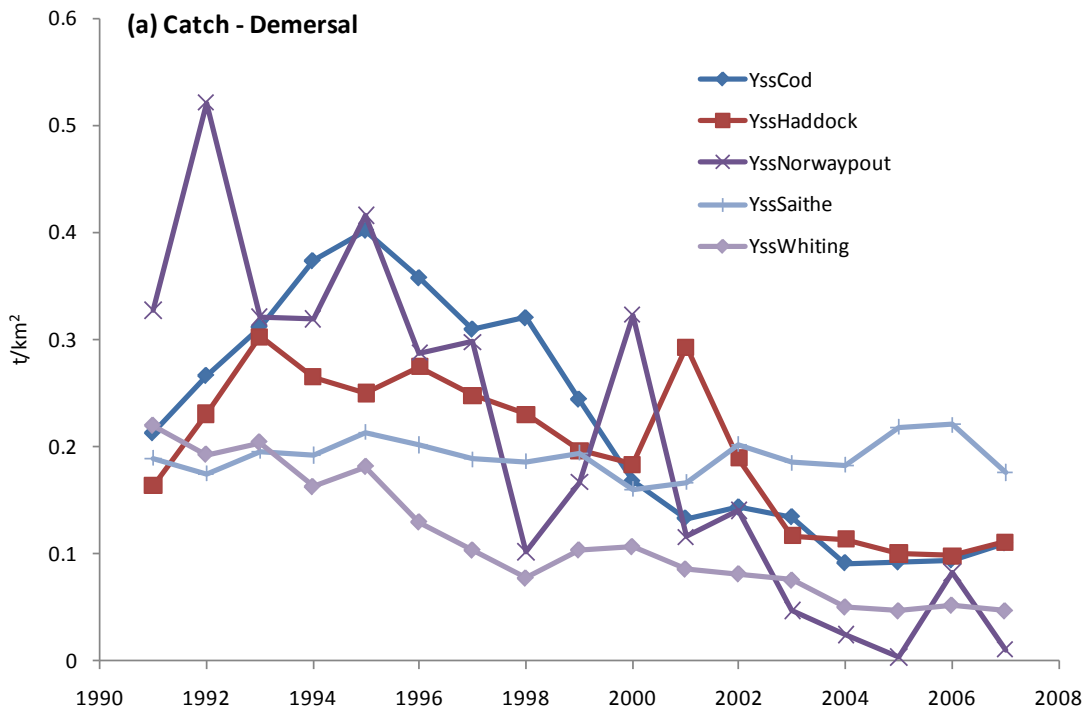
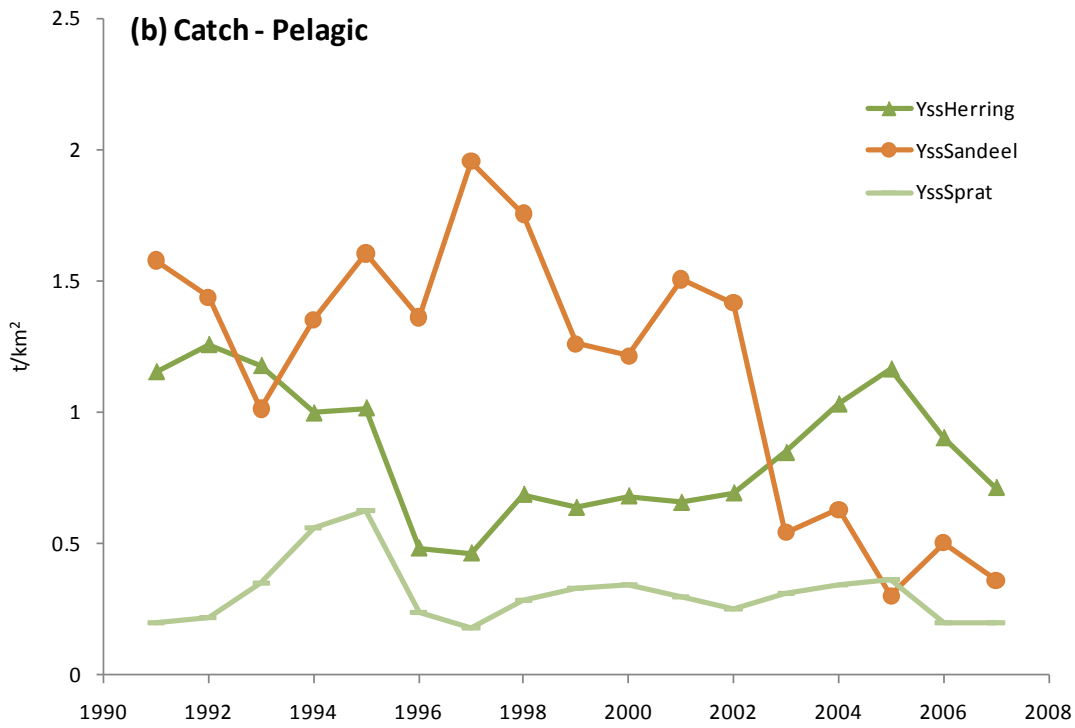


Figure 3.6. Relative effort of principal North Sea fleets since 1991 and used as input to the model. Note: see Table 3.1 for description of aggregation of fleet categories used.

3.3.3 Catches (Figure 3.7, 3.8)

For assessed species, catches are taken from single species assessment reports and for non-assessed species from the ICES Statlant database.


(a) Catch - Demersal
t/km²
0.6
0.5
0.4
0.3
0.2
0.1
0
1990
1992
1994
1996
1998
2000
2002
2004
2006
2008
YssCod
YssHaddock
YssNorwaypout
YssSaithe
YssWhiting


(b) Catch - Pelagic
t/km²
2.5
2
1.5
1
0.5
0
1990
1992
1994
1996
1998
2000
2002
2004
2006
2008
YssHerring
YssSandeel
YssSprat

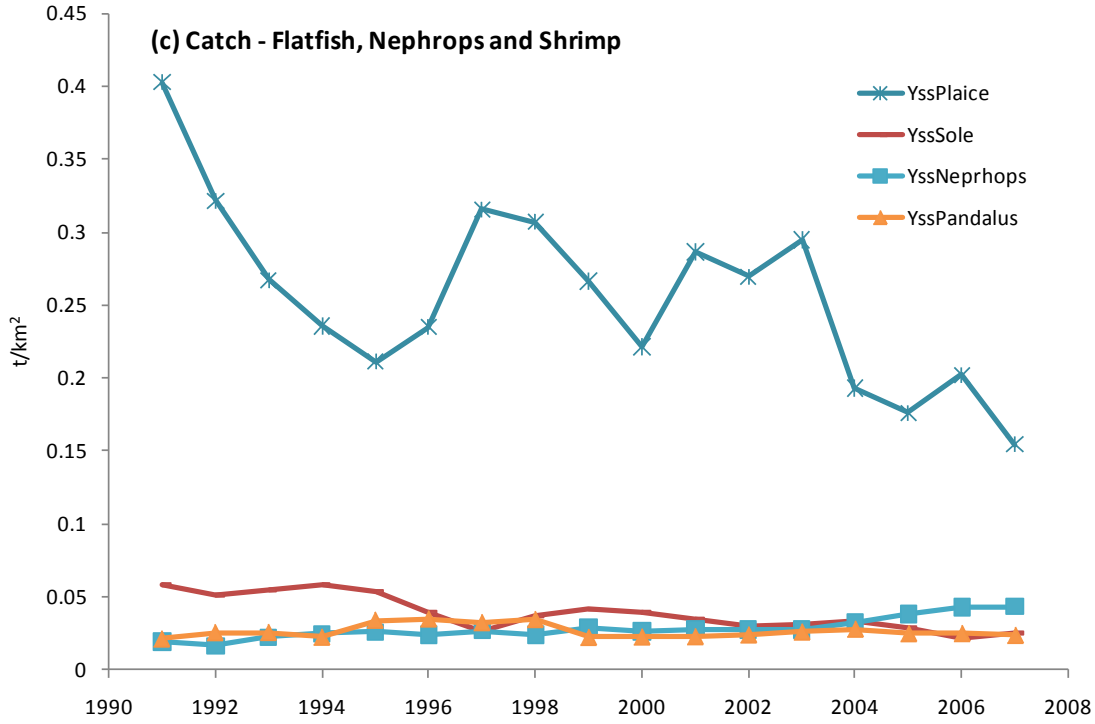


Figure 3.7. Catch data from single species assessments for assessed species used as in model fitting.

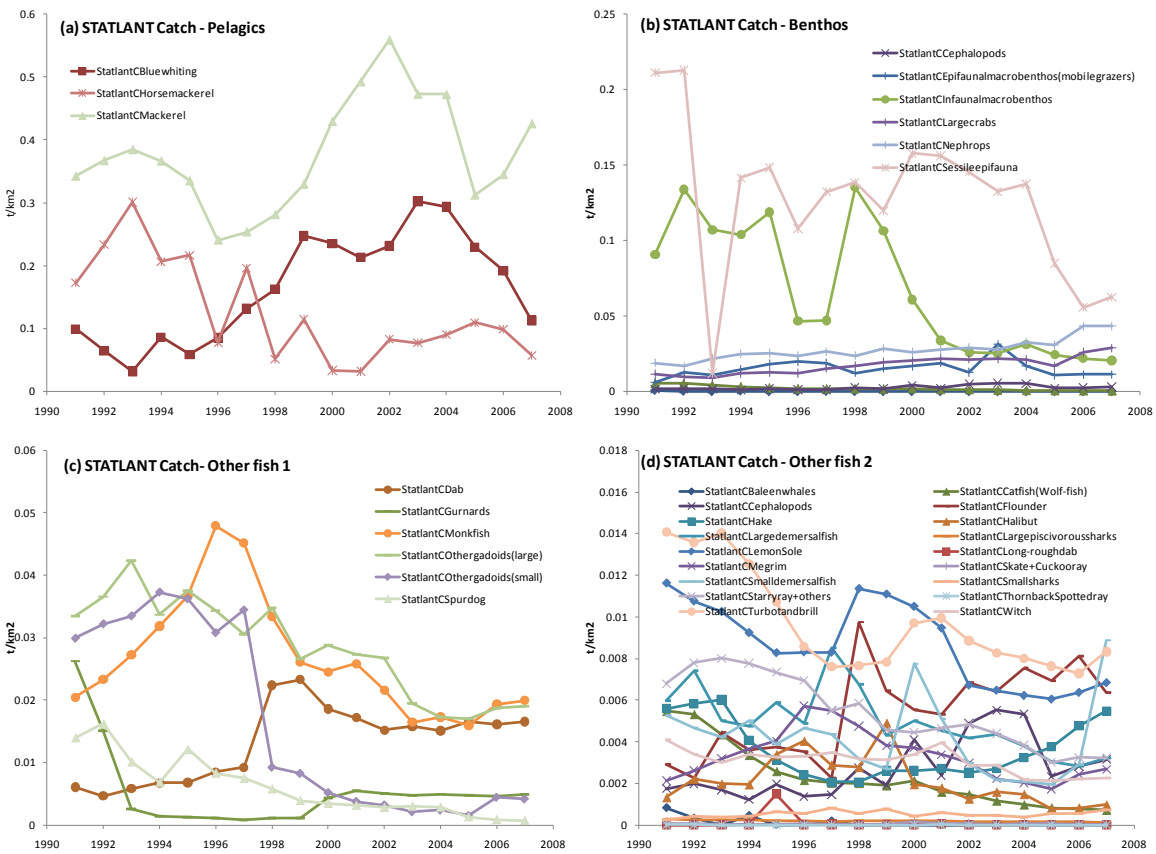
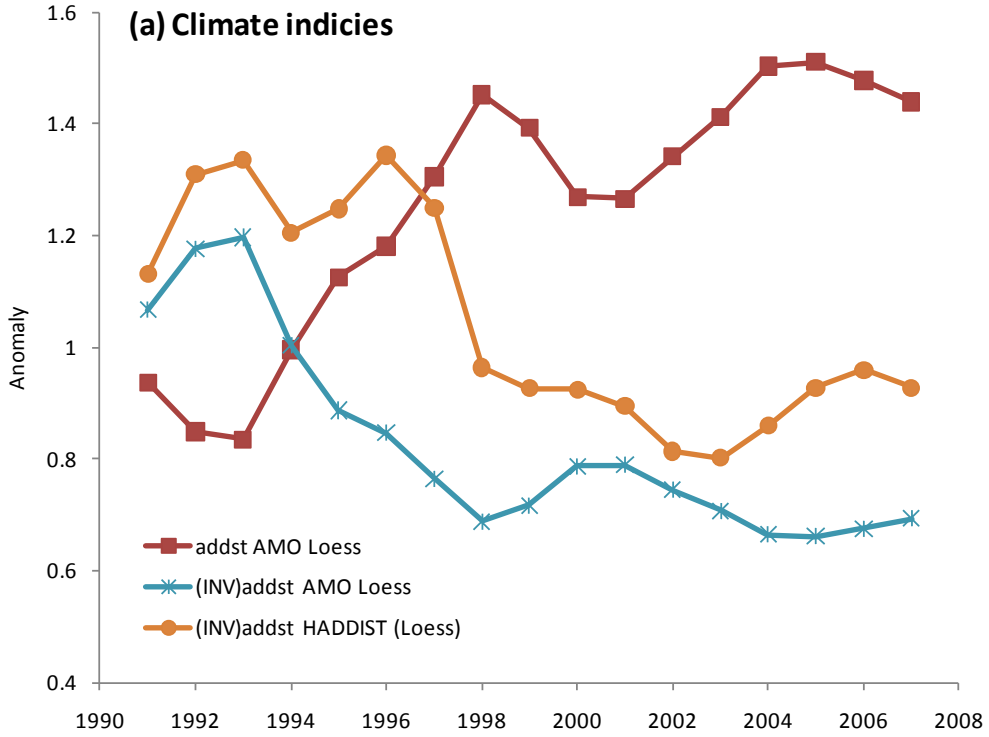


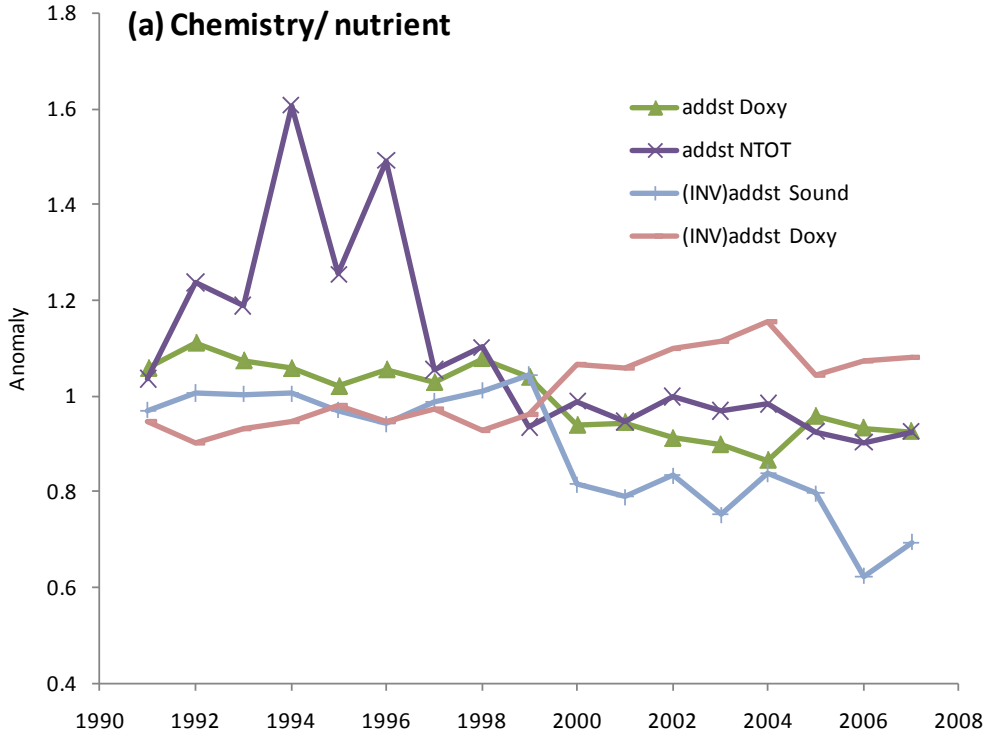
Figure 3.8. Catch data from ICES STATLANT for non-assessed species used as in model fitting.

3.4 Environment (Figure 3.9)

Correlation analyses were performed to identify relationships between pressure and state variables and to help identify redundancies in pressure variables used to drive the model. Details are provided in Mackinson (in prep). The analysis was broken down into two distinct periods. The first covered the data period used in the model simulations 1991–2007, the second corresponded to the full length of the dataset, 1950–2007, where longer term patterns could be investigated. Taking account of the delayed influence that changes in oceanography and primary production are expected to have on higher trophic levels, fish recruitment and biomass time-series data were lagged by one year. The effect of the lags was tested and confirmed that stronger correlations were more prevalent when the lag was included. The 1950–2007 correlation matrix comprises c. 14,000 values, yielding insight in to long-term patterns of change in the North Sea.

Examination of alternative parameterizations performed during calibration of the model, together with evidence from the correlation analysis of empirical data revealed links between environmental indices, changes in primary production and species biomass trends across a broad range of functional groups. This evidence, reported in detail in Mackinson (in prep) was used in calibrating the model. Specifically, environmental data were used to define a forcing function describing bottom–up changes in primary production (PP anomaly) and to modify specific trophic interactions in some functional groups (see 4.2.3). The PP anomaly applied in the model (Figure 3.9c) is a composite function using 3 indices, (Hadley SST, AMO, total nitrogen), and was scaled using an estimated PP anomaly from Ecosim.


(a) Climate indicies
Anomaly
1.6
1.4
1.2
1
0.8
0.6
0.4
1990
1992
1994
1996
1998
2000
2002
2004
2006
2008
addst AMO Loess
(INV)addst AMO Loess
(INV)addst HADDIST (Loess)


(a) Chemistry/ nutrient
Anomaly
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
1990
1992
1994
1996
1998
2000
2002
2004
2006
2008
addst Doxy
addst NTOT
(INV)addst Sound
(INV)addst Doxy

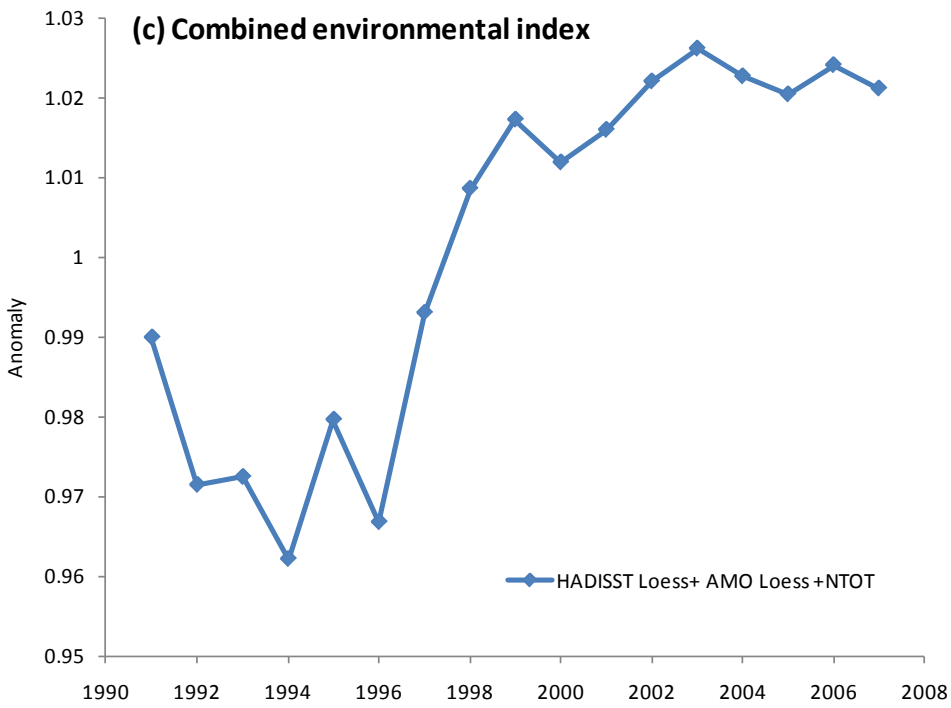


Figure 3.9. Environmental indices used as drivers in model fitting (a) climate indices – AMO and Hadley Sea surface temperature, (b) nutrient chemistry – dissolved oxygen, total nitrogen and sound, (c) combined index used as primary production forcing anomaly – Hadley sea surface temperature+AMO+ total nitrogen.

4. Ecosim fit to data

[File: EwE key run Calibration and testing file.xlsx]

4.1 Approach overview / strategy

The procedural strategy to fitting the model to observation data comprised of 7 elements described.

1) Selection of time-series data and investigation of the relationships among pressure variables used to force changes in the model (fishing end environmental forcing) and the trends in time-series of response variables (e.g. biomass; see Section 3). Data used in fitting were also selected and prioritized on the basis of their perceived 'quality'. Table 4.1 describes the data selection and rationale.

2) An initial evaluation of the sensitivity of each functional group to changes in vulnerability (the main parameter estimated by Ecosim during the fitting process) was carried out and used to guide the selection of groups whose vulnerability parameters would be estimated during fitting. Groups with sensitive vulnerabilities were only chosen for estimation if time-series observation data were also available. NOTE: It was found that the principle groups that had sensitive vs. (all low) were those for which good quality time-series data were available.

3) Fishing mortality, fishing effort, a primary production anomaly and trophic forcing functions were used to drive the model (see Section 3).

4) Running the fitting procedure in Ecosim to estimate vulnerability parameters. Ecosim uses a Marquardt non-linear search algorithm with 'trust region modification' of the Marquardt steps to search for vulnerabilities that improve the fit of model predictions to time-series o**b**servations.

5) Evaluating the quality of the model fit to data, both globally and for each individual functional group using goodness-of-fit and AICc to account for differences in the number of parameters estimated by the model.

6) Test the performance of alternative setup specifications regarding the effects of weighting time-series, initialization of vulnerabilities (not reported here but included in Mackinson, in prep) and application of selected environmental forcing functions to consumer groups where supported by empirical evidence (Section 4.2).

7) Evaluate the dynamics of multi-stanza groups and where necessary make adjustments to parameters estimated by the non-linear fitting routine to ensure (i) stock–recruitment relationships display compensatory behaviours of Beverton–Holt type form and that the relative steepness of the slopes is consistent with evidence from other data, (ii) predictions of F_{msy} for each group are credible according to known levels of fishing mortality that have been recorded.

Table 4.1. Selection and prioritization of data used in model calibration 1991–2007 (from Mackinson in prep).

Variable	Priority data	Secondary data
Pressure variables		
Fishing mortality	Estimates from single species assessment models were prioritized because they are accepted and applied in fisheries management.	Estimates from multispecies models (SMS 2008, MSVPA 2005,) used when single species assessments were not available.
Environmental	Primary production anomaly based on empirical data (see Mackinson in prep)	
State variables		
Relative biomass data	Biomass estimates from single species stock assessment models. The advantage of assessment data is that the high variability that arises from spatial and temporal patchiness observed in surveys and causing misleading trends is avoided.	Surveys were used for non assessed species. They provide a useful overall index but can display high interannual variability due to patchiness in survey data. Estimates from multispecies models (SMS 2008, MSVPA 2005,) were used for some groups when single species assessments were not available (Figure3.3).
Catch data	Single species assessment data where possible because in many instances it includes an estimate of discarded fish and thus more closely reflects removals.	For non assessed species, ICES (Statlant) landings data are used.

4.2 Testing different setup specifications

4.2.1 Evaluating the goodness-of-fit

Sums of squared deviations were used to evaluate how well model predictions corresponded with observed time-series. SS were calculated both for individual functional groups and for all groups combined. Where model parameterization tests involved estimating a different number of parameters (e.g. intermediate steps where primary production anomalies and vulnerabilities were estimated simultaneously), Akaike's information criteria score for small sample sizes (AICc) was calculated. (see Mackinson in prep for tests of alternative hypotheses)

4.2.2 Time-series weighting

The influence on the model fit of weighting time-series data to represent differences in the data quality or reliability was evaluated in terms of the qualitative representation of the patterns in biomass trends. Quantitative evaluation is difficult because the weighting changes the absolute values of the SS. The findings were that:

Only the fit for whiting improved when additional weight was applied to the assessed species. Others performed worse, supporting previous findings that the dynamics of whiting in the model are contrary to other key species such as cod and haddock (Mackinson *et al.*, 2009)

There was no difference between applying x10 and x100 weighting.

Weighting only the relative biomass series made a bigger difference but the performance was not as good as equal weighting throughout.

Not including juveniles in the fitting, changes only the result for juvenile herring and nothing else. The default, equal weighting of 1 performed best.

The outcome of the weighting test was that the best overall qualitative fit of the model to data were achieved when each observation dataset was given the same weight, thereby having an equal influence on the overall constraint.

4.2.3 Direct forcing of consumers feeding behaviour

Analysis of the relationship among trends in relative biomass and environmental time-series (Mackinson in prep) revealed empirical support that direct environmental forcing, represented through climate indices and nutrients, might strongly influence the dynamics of some species (Table 4.2). The effects of including direct environmental forcing on the consumption behaviour of some species was evaluated. Specifically, an evaluation was made of how model predictions of each functional group matched observations when direct environmental forcing was applied to modify trophic relationships in different ways. Those that provided the best fit of predictions to data were retained and are applied in the key-run (Table 4.2).

Table 4.2. Direct environmental drivers applied to specific functional groups.

Group #	Name	Main Environmental correlate (Strength and direction of correlation in brackets – see Mackinson in prep for details)	Main Nutrient correlate	Forcing code #ref	Applied to consumer mode with lowest score
13	Juvenile Cod(0–2, 0–40cm)	HADISST North Sea(Strong-)	DOXY(umol/l)(Mod+), NTOT(umol/l)(Mod+)	F28,F24,F25	Vuln
14	Cod (adult)	HADISST North Sea(Mod-)	DOXY(umol/l)(Mod+)	F28, F24	v and arena
15	Juvenile Whiting (0–1, 0–20cm)		DOXY(umol/l)(Weak+)	F24	vuln
16	Whiting (adult)	AMO smoothed (USGS)(Mod-)	Sound(Mod-)	F26,F29	vuln
18	Haddock (adult)	AMO smoothed (USGS)(Mod+)	DOXY(umol/l)(Mod-)	F20,F30	vuln
20	Saithe (adult)	AMO smoothed (USGS)(Strong+)	Sound(Mod+)	F20,F24	vuln
23	Norway pout	HADISST North Sea(Weak-)	DOXY(umol/l)(Weak+)	F28,F24	Search rate
28	Herring (juvenile 0, 1)	HADISST North Sea (+Weak)		F22	Arena
29	Herring (adult)	HADISST North Sea(Mod+)	DOXY(umol/l)(Mod-)	F22,F30	v and arena
38	Sole	AMO smoothed (USGS)(Mod-)	NTOT(umol/l)(Weak+)	F26,F25	non/search

4.3 Ecosim parameters estimated

4.3.1 Vulnerabilities

Vulnerabilities were estimated by Ecosim during the fitting routine, giving the best fit of the model predictions to data. They were subsequently modified manually to meet the need to ensure that the model behaviour adequately represented the dynamics of multi-stanza groups and predictions for MSY (see comments in next sections). Vulnerabilities estimated during model fitting and applied in the key run are given in Table 4.3. The default value of 2 was used for all other groups (see Mackinson in prep for tests of this assumption).

Table 4.3. Vulnerabilities estimated and applied in the key-run model

Number	Group	Key Run v's
13	Juvenile Cod(0–2, 0–40cm)	1.2
14	Cod (adult)	10
15	Juvenile Whiting (0–1, 0–20cm)	10
16	Whiting (adult)	1.2
17	Juvenile Haddock (0–1, 0–20cm)	1.15
18	Haddock (adult)	1.8
19	Juvenile Saithe (0–3, 0–40cm)	1.4
20	Saithe (adult)	10
23	Norway pout	6
27	Gurnards	2
28	Herring (juvenile 0, 1)	1.15
29	Herring (adult)	3
30	Sprat	2
31	Mackerel	3
33	Sandeels	10
34	Plaice	2.2
38	Sole	1.65
51	Carnivorous zooplankton	1
52	Herbivorous and Omnivorous zooplankton (copepods)	15

4.3.2 Multi-stanzas, stock recruitment and MSY

Following the approach reported in Mackinson *et al.* (2009), careful attention was given to ensuring that the dynamics of multi-stanza groups were stable and produced credible emergent stock recruitment relationships and for which the relative degree of compensation (relative slopes) among different species was consistent with the evidence from data (Figure 4.3). This is a particularly important part of 'calibrating' the model since the degree of compensation in recruitment has a strong influence upon the predictions of MSY, with more conservative predictions arising with lower compensation (SR slopes closer to 1) .

Predictions of MSY for non multi-stanza groups were examined, and where considered unfeasible, manual changes were made to the feeding time factors of multi-stanza groups (see Section 5) and the vulnerabilities estimated during the automatic fitting procedure.

Both of the steps described here generally serve to worsen the model fit to data. Prettier fits can be achieved by applying the vulnerabilities estimated by the fitting routine. However, although they make for pretty fits to data, they do not provide the basis for credible predictions. We must accept a model whose fit is less convincing but whose behaviour is more consistent with the evidence and 'expectations' from fisheries science.

Plots of the emergent SR relationship for each multi-stanza are given in Figure 4.4, which shows the results when a 'V' shaped fishing pattern is applied simultaneously to the multi-stanza groups by combining individual 'V'-shaped mortality patterns designed to drive the stocks biomass through high and low values. Tests show that

the relative compensation in recruitment among species is consistent with that shown in Figure 4.3; the order of least to most compensation being saithe, cod, whiting, herring, haddock.

Equilibrium yield and biomass plots for key commercial species and fleets are given in Figure 4.5 and 4.6.

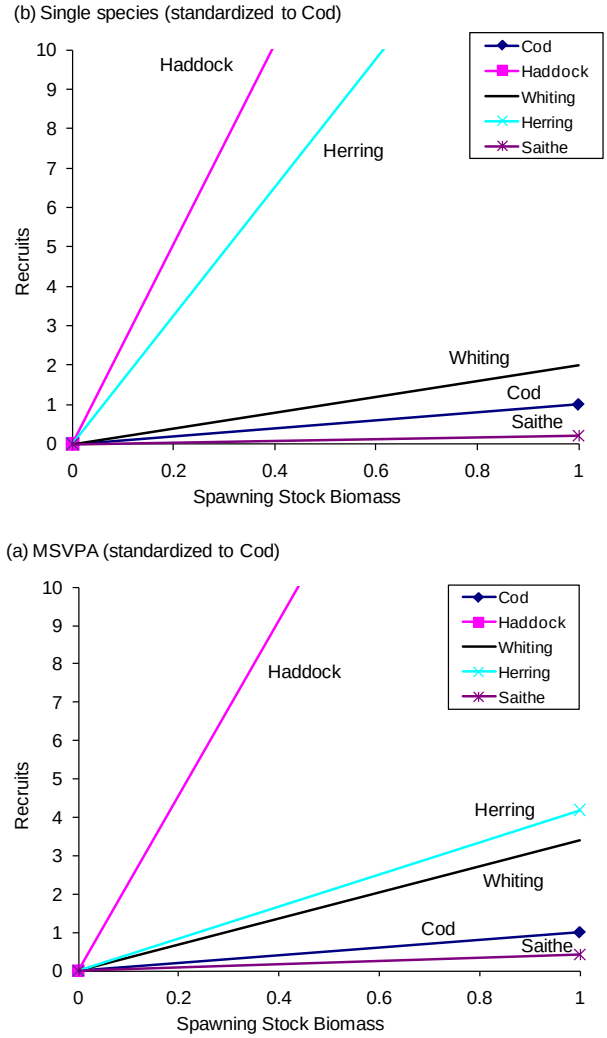


Figure 4.3. Comparison of the relative degree of compensatory response among species represented as multi-stanza groups in the model. Lines indicate the slope of the 50percentile R/SSB relative to that of cod (set to 1): (a) data from the MSVPA 2005 key run; (b) data from single-species assessments. (From Mackinson *et al.*, 2009).

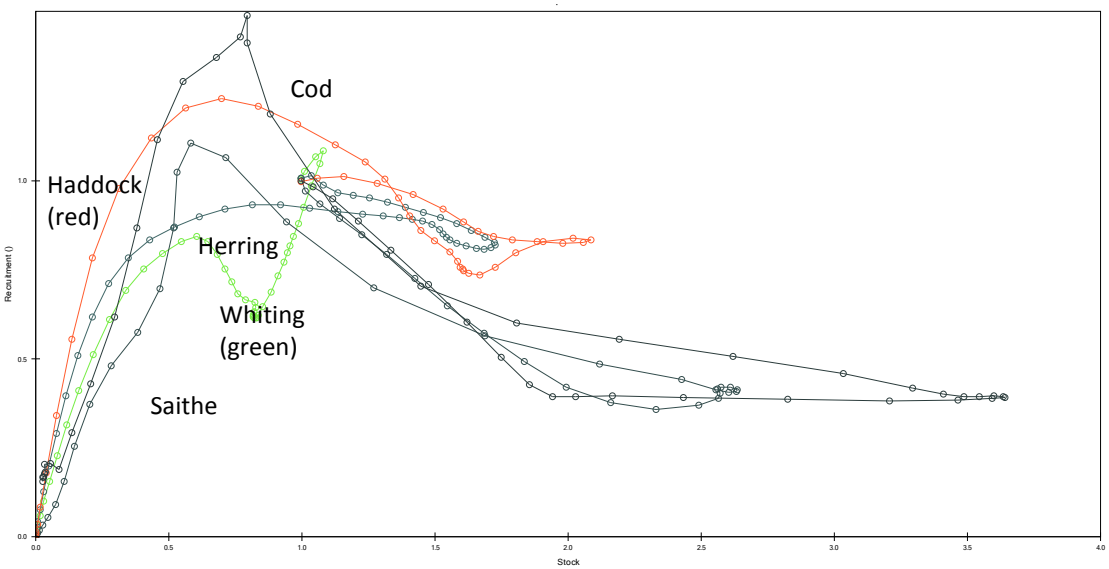
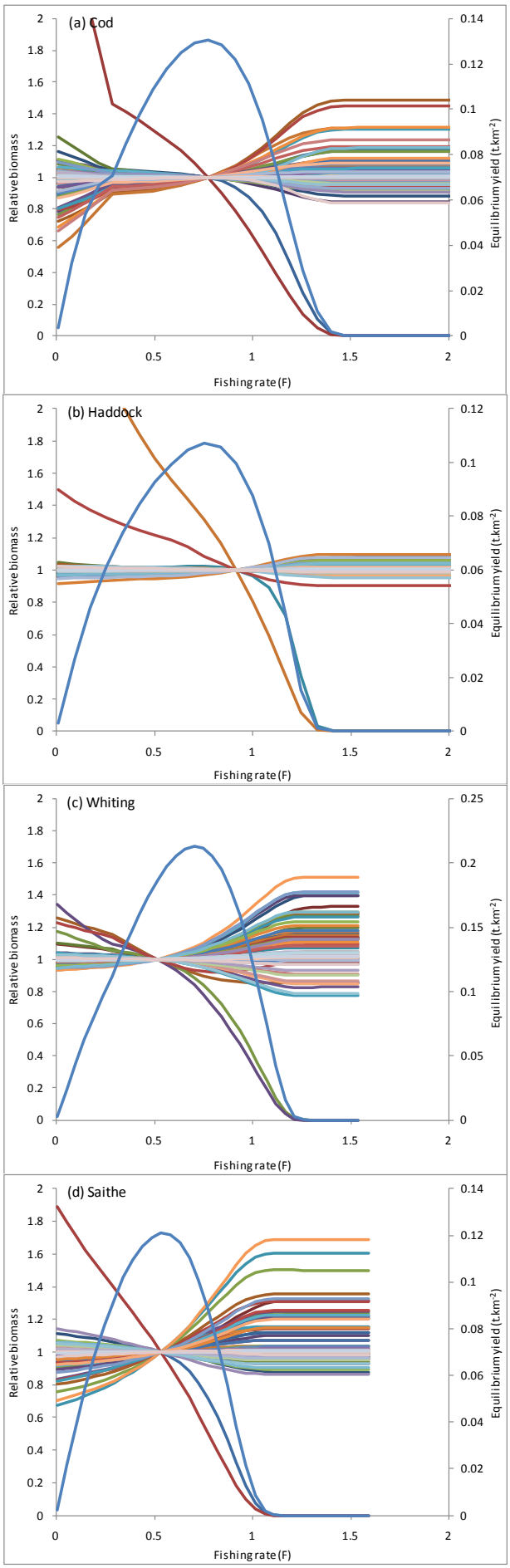
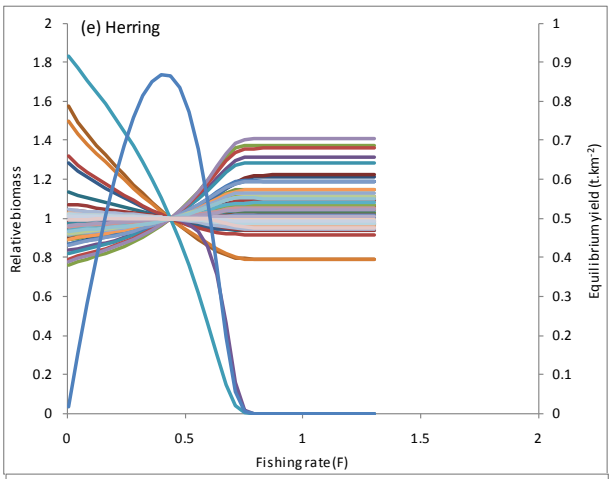
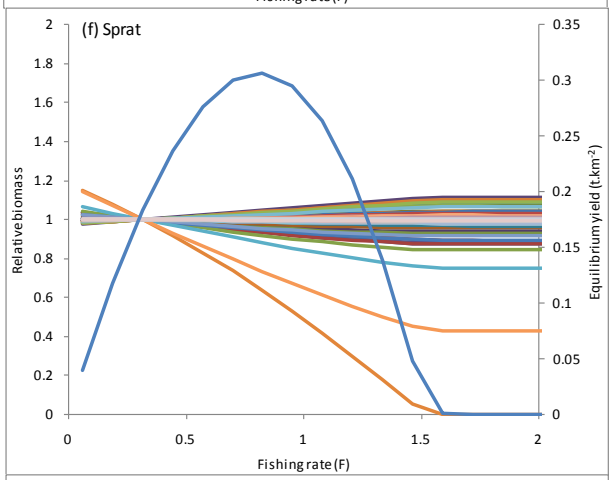
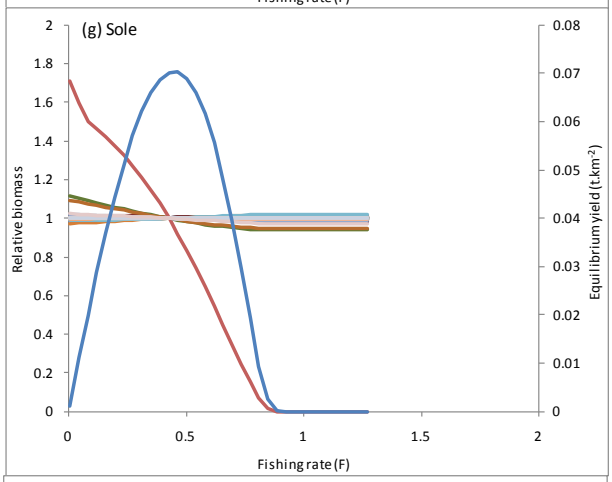


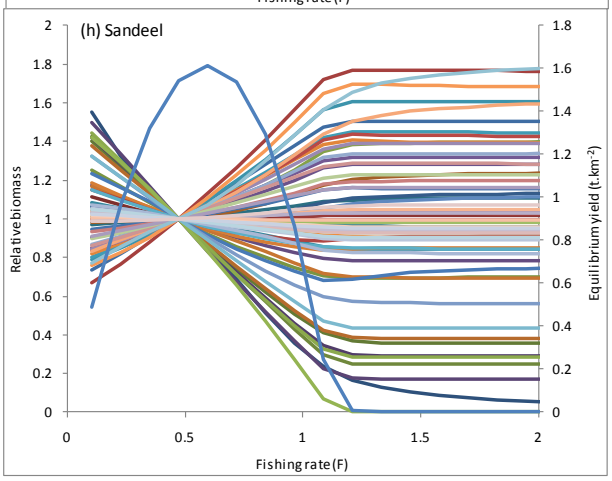
Figure 4.4. Emergent SR relationships from the key-run model when 'V'-shaped fishing pattern are applied to all multi-stanza groups simultaneously. x-axis: Spawning stock, y-axis: recruits.

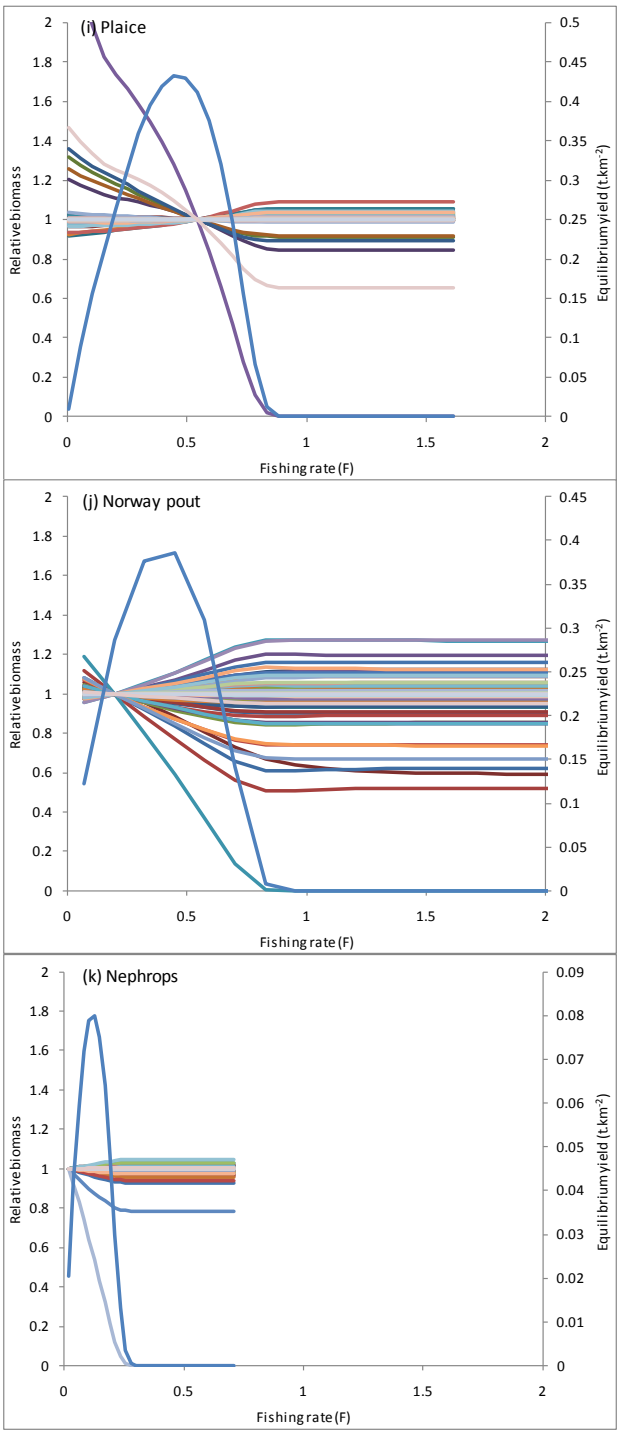

(a) Cod
(b) Haddock
(c) Whiting
(d) Saithe
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)


(e) Herring
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)


(f) Sprat
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)


(g) Sole
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)


(h) Sandeel
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)


(i) Plaice
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)
(j) Norway pout
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)
(k) Nephrops
Relative biomass
Equilibrium yield (t.km-2)
Fishing rate (F)

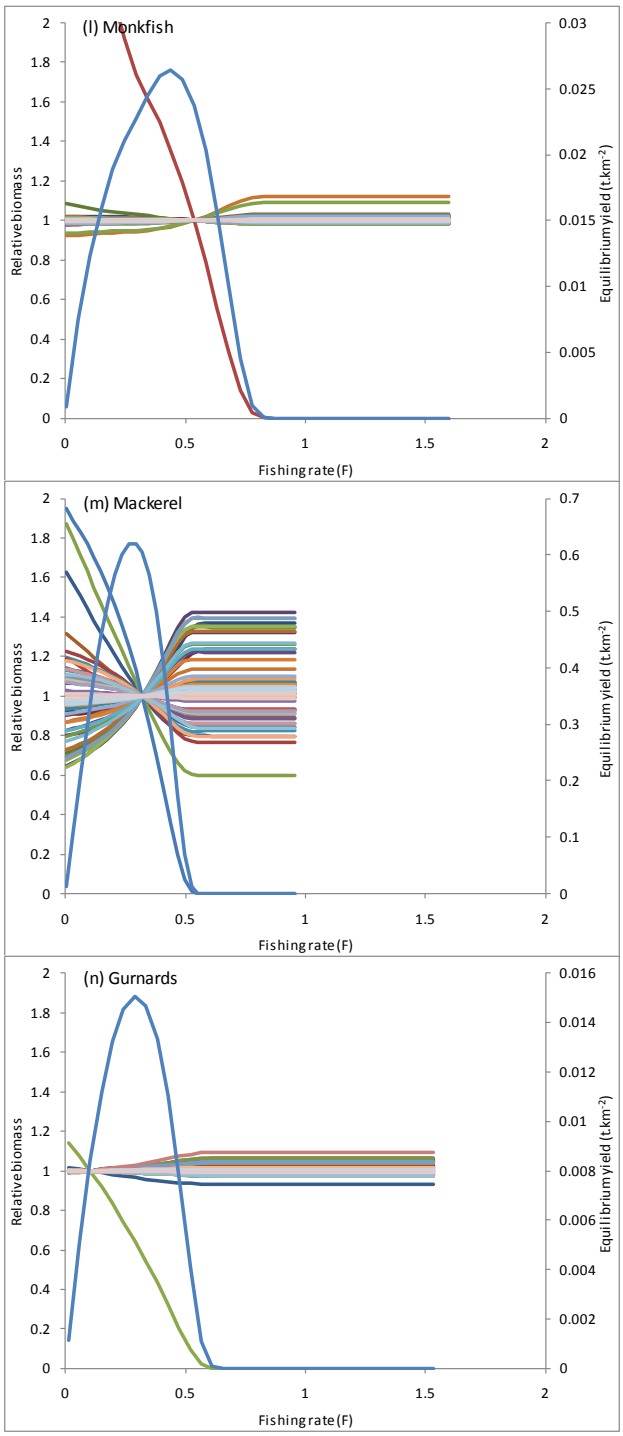
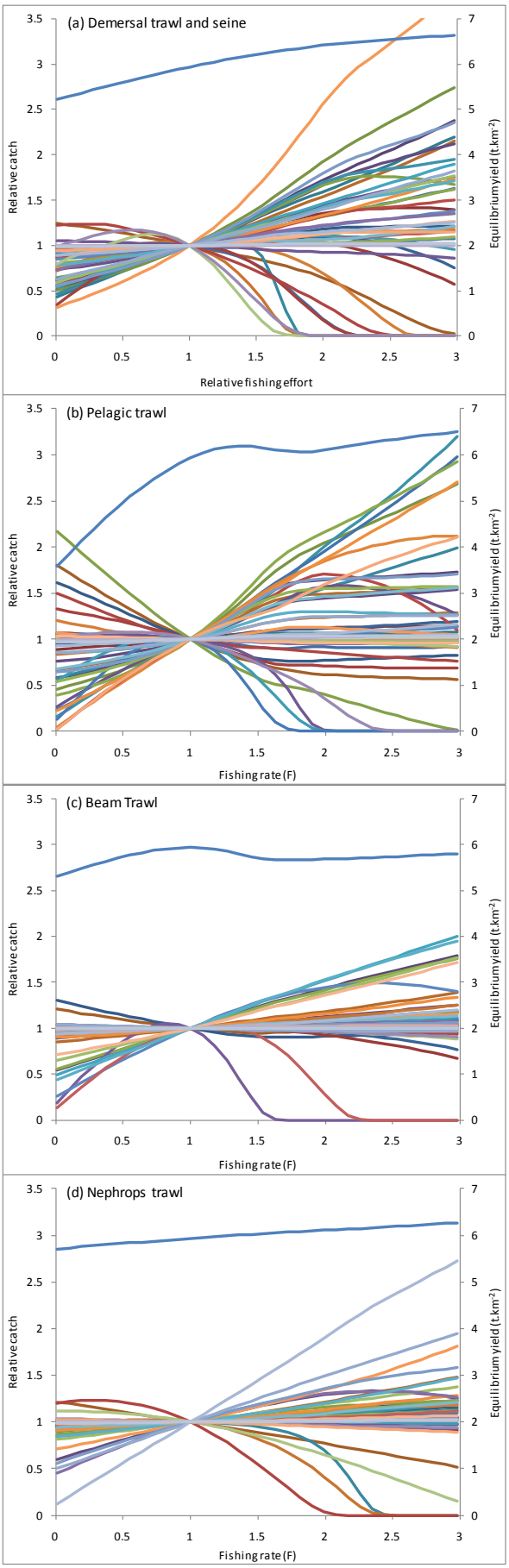
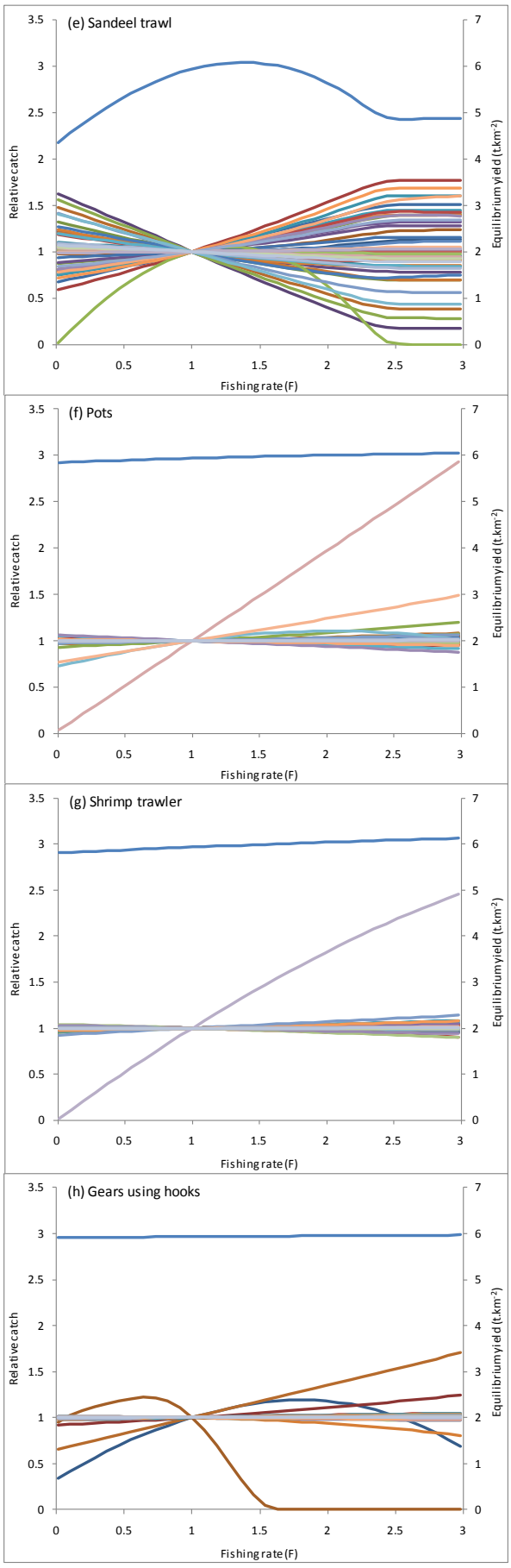
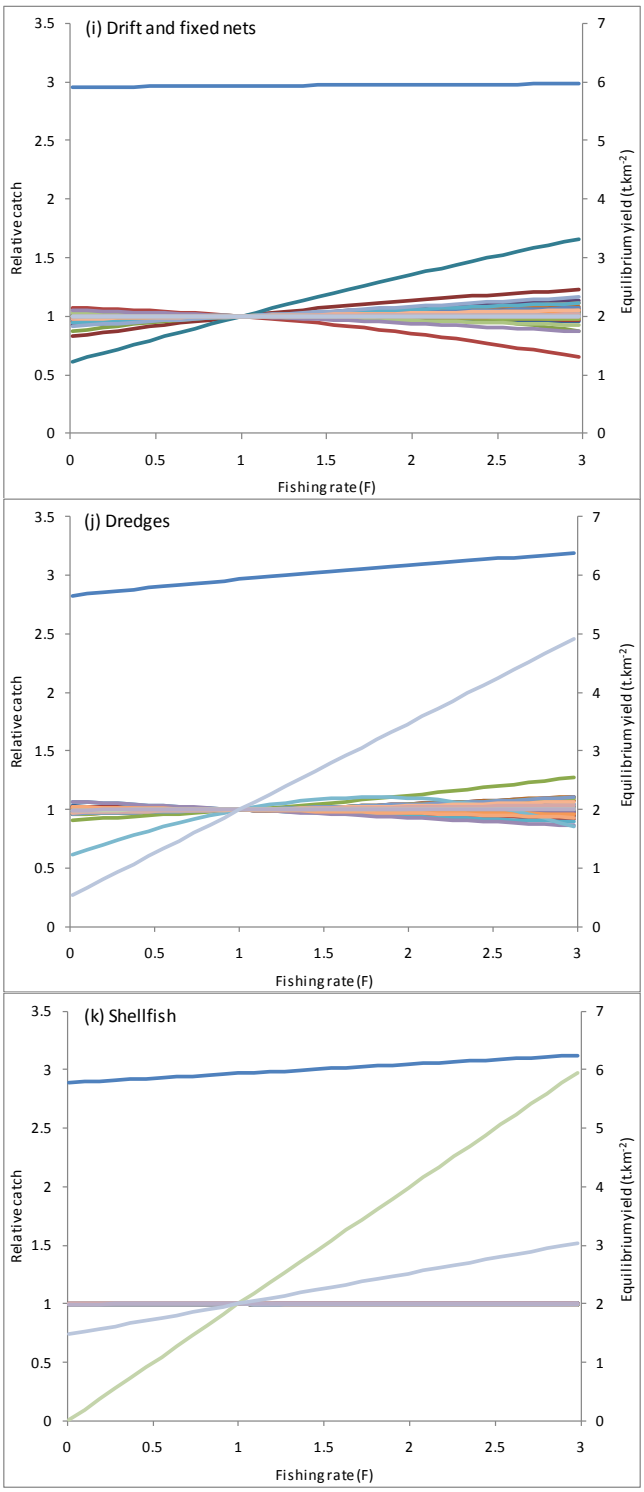


Figure 4.5. Equilibrium yield curves for key commercial species (blue), superimposed over relative biomasses of all other groups in the system (coloured lines).


(a) Demersal trawl and seine
Relative catch
Equilibrium yield (t.km-2)
Relative fishing effort
(b) Pelagic trawl
Relative catch
Equilibrium yield (t.km-2)
Fishing rate (F)
(c) Beam Trawl
Relative catch
Equilibrium yield (t.km-2)
Fishing rate (F)
(d) Nephrops trawl
Relative catch
Equilibrium yield (t.km-2)
Fishing rate (F)


(e) Sandeel trawl
(f) Pots
(g) Shrimp trawler
(h) Gears using hooks
Relative catch
Equilibrium yield (t.km-2)
Fishing rate (F)
0
0.5
1
1.5
2
2.5
3
3.5
4
5
6
7


(i) Drift and fixed nets
(j) Dredges
(k) Shellfish
Relative catch
Equilibrium yield (t.km-2)
Fishing rate (F)
0
0.5
1
1.5
2
2.5
3
3.5
0
1
2
3
4
5
6
7

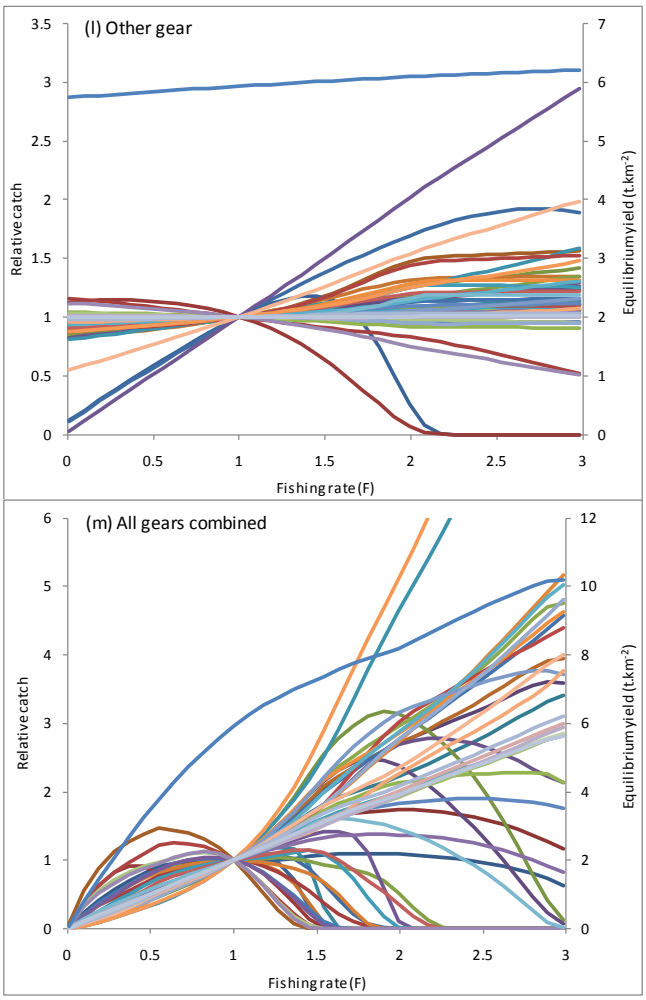


Figure 4.6. Equilibrium yield curves for the fishing fleets.

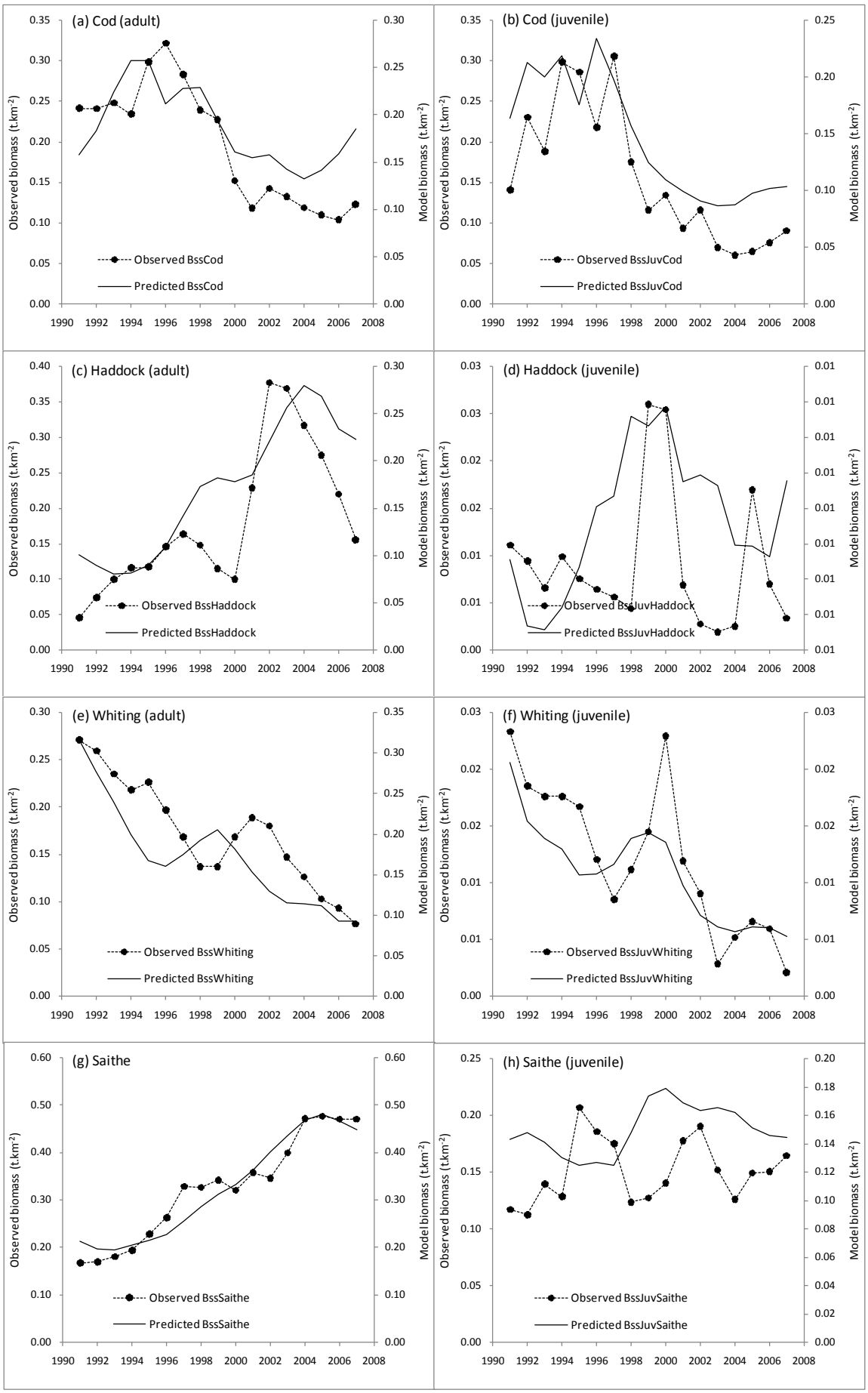
4.3.3 Adult-juvenile feeding time factors

The feeding time factor determines how fast organisms adjust feeding times so as to stabilize consumption rate per biomass. Adjustments to the feeding time factors of multi-stanza groups were made alongside changes to vulnerability to ensure credible dynamics (see Section 4.32). The resultant values, used in the key run, represent implicitly the assumptions that (i) adults have a fast time response (value 1), which causes reduction in vulnerability to predation rather than increased growth rate when/if food density increases, (ii) juveniles have a slow time response (value 0.1), which means that when food is available, changes in consumption per biomass result in more rapid growth rate changes, but at the risk of higher exposure to predation. Feeding time factors for all other groups were set to 0.5, testing of alternative values having revealed no improvements in the fit.

4.4 Plots of model predictions and observed data

4.4.1 Biomass, Catch and Mortality for assessed fish

The fit of model predictions to observed data for selected key species of interest, including all those for which vulnerability was estimated during model fitting, are shown in Figures 4.6, 4.7, 4.8.


(a) Cod (adult)
Observed BssCod
Predicted BssCod
(b) Cod (juvenile)
Observed BssJuvCod
Predicted BssJuvCod
(c) Haddock (adult)
Observed BssHaddock
Predicted BssHaddock
(d) Haddock (juvenile)
Observed BssJuvHaddock
Predicted BssJuvHaddock
(e) Whiting (adult)
Observed BssWhiting
Predicted BssWhiting
(f) Whiting (juvenile)
Observed BssJuvWhiting
Predicted BssJuvWhiting
(g) Saithe
Observed BssSaithe
Predicted BssSaithe
(h) Saithe (juvenile)
Observed BssJuvSaithe
Predicted BssJuvSaithe
Observed biomass (t.km-2)
Model biomass (t.km-2)

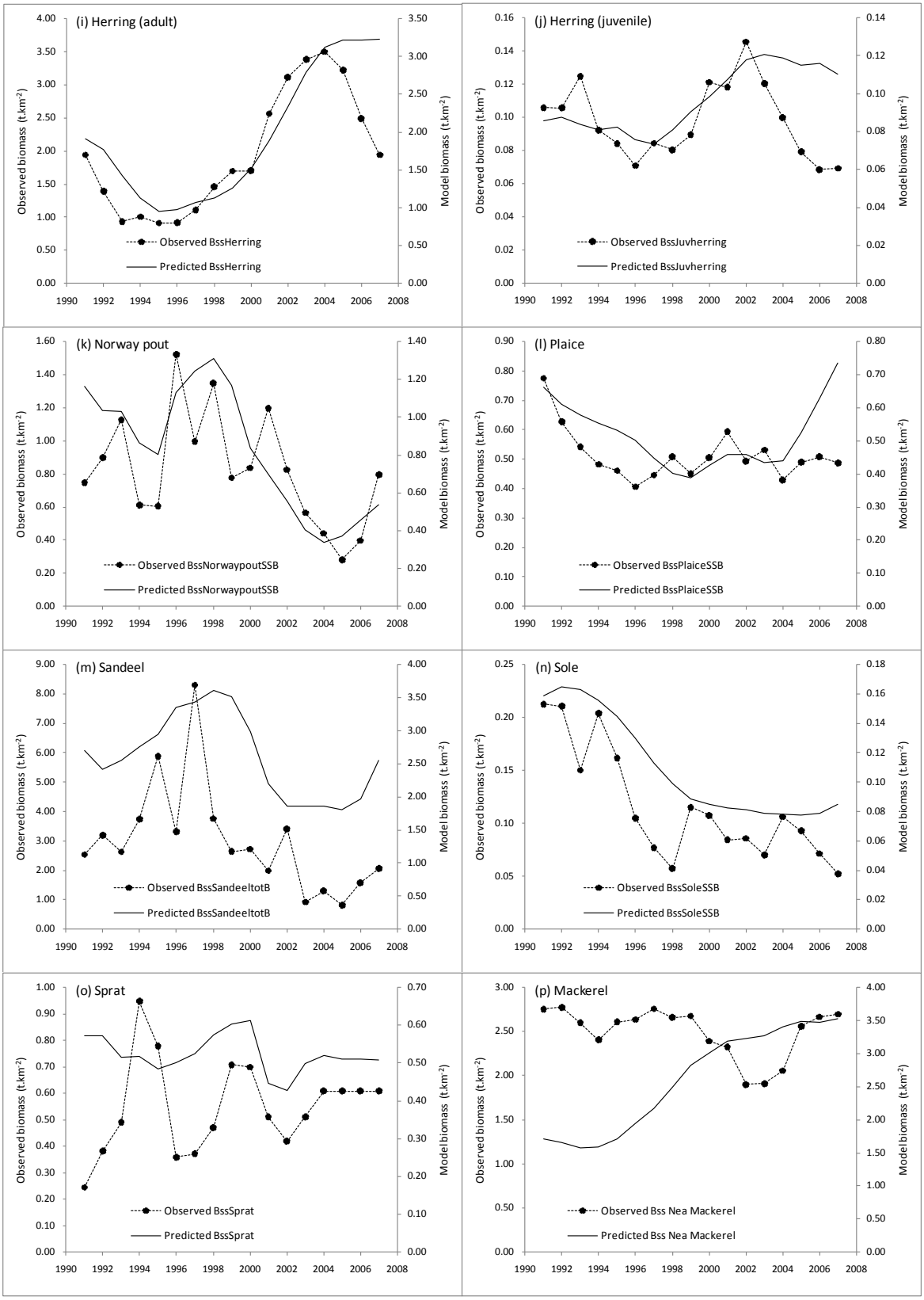
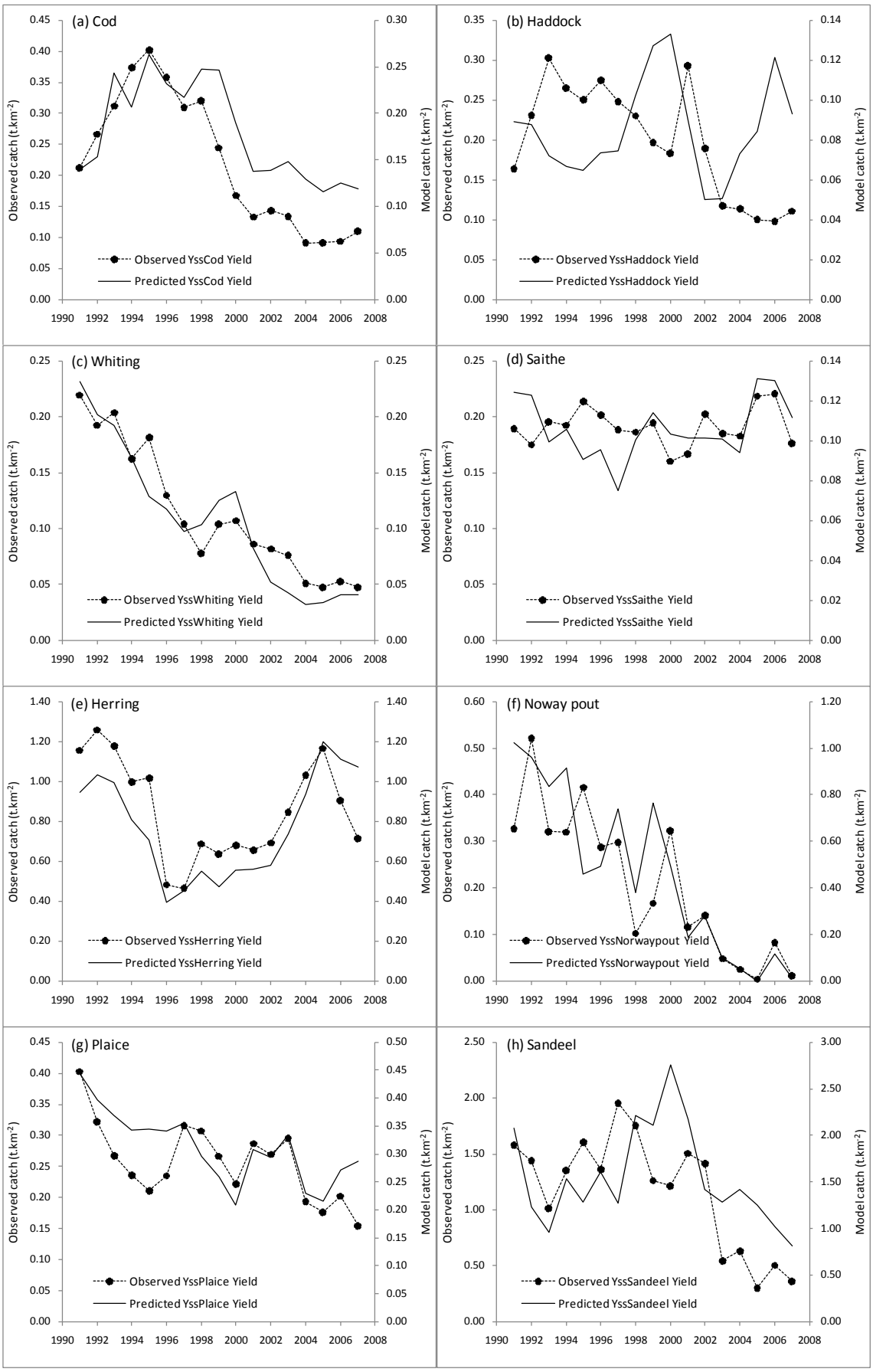


Figure 4.6. Relative biomass plots - observed and model predicted. 'Observed' data are derived from single species stock assessments.


(a) Cod
Observed YssCod Yield
Predicted YssCod Yield
(b) Haddock
Observed YssHaddock Yield
Predicted YssHaddock Yield
(c) Whiting
Observed YssWhiting Yield
Predicted YssWhiting Yield
(d) Saithe
Observed YssSaithe Yield
Predicted YssSaithe Yield
(e) Herring
Observed YssHerring Yield
Predicted YssHerring Yield
(f) Noway pout
Observed YssNorwaypout Yield
Predicted YssNorwaypout Yield
(g) Plaice
Observed YssPlaice Yield
Predicted YssPlaice Yield
(h) Sandeel
Observed YssSandeel Yield
Predicted YssSandeel Yield
Observed catch (t.km-2)
Model catch (t.km-2)

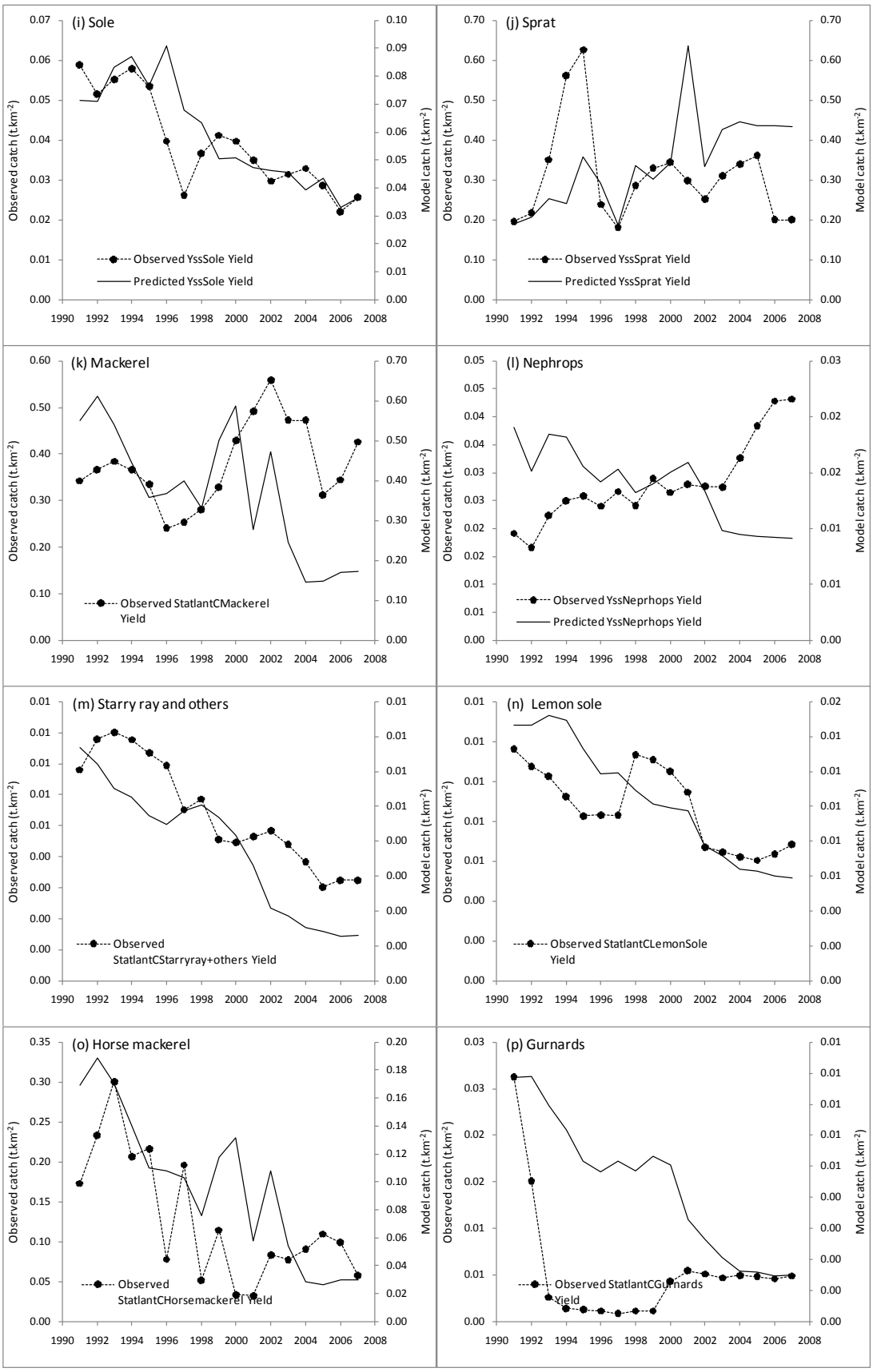
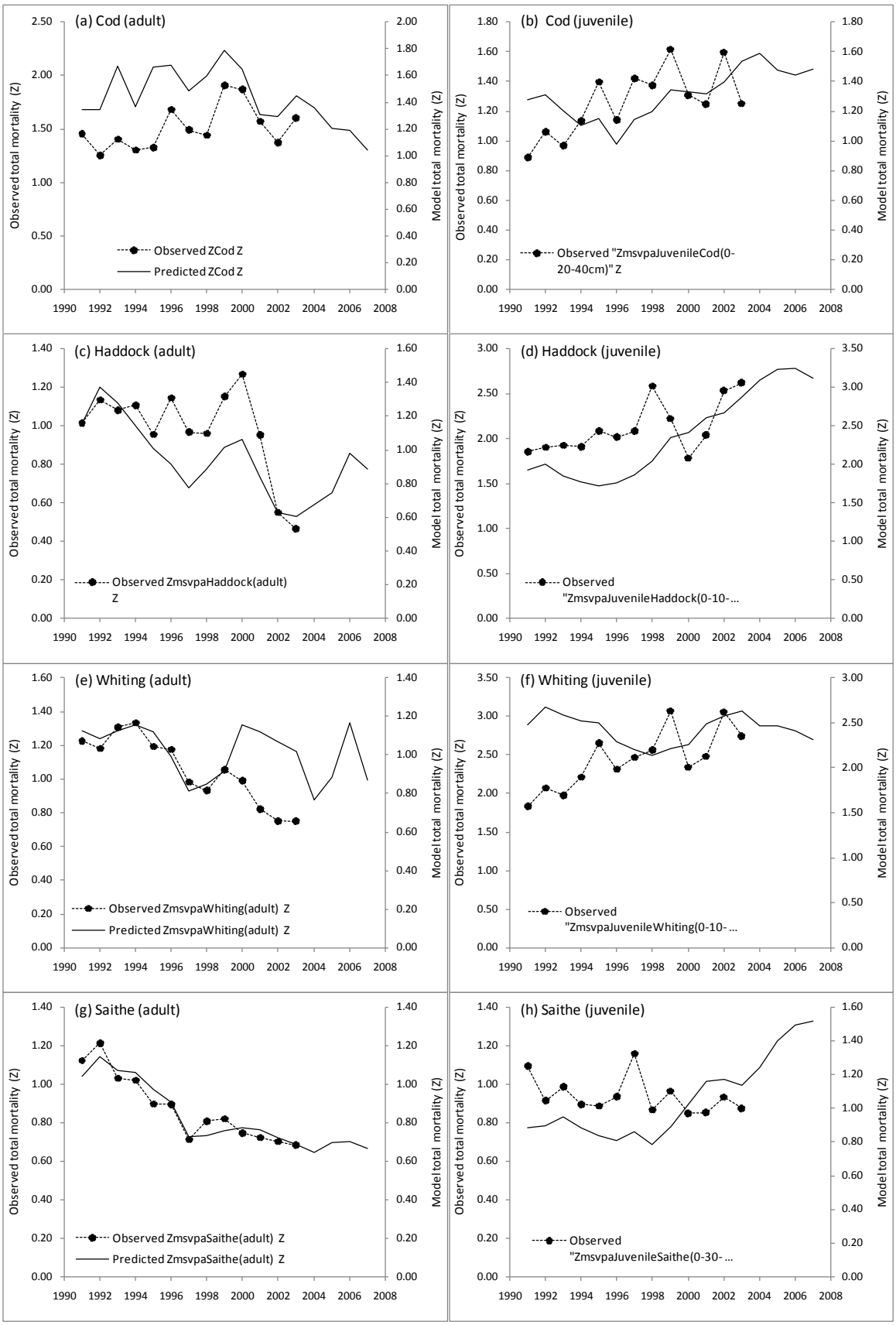


Figure 4.7. Catch plots - observed and model predicted. 'Observed' catch data are derived from single species stock assessments where available, otherwise from Statlant (for Mackerel, Starry ray, Lemon sole, Horse mackerel, gurnards).


(a) Cod (adult)
Observed ZCod Z
Predicted ZCod Z
(b) Cod (juvenile)
Observed "ZmsvpaJuvenileCod(0-20-40cm)" Z
(c) Haddock (adult)
Observed ZmsvpaHaddock(adult) Z
(d) Haddock (juvenile)
Observed "ZmsvpaJuvenileHaddock(0-10-...
(e) Whiting (adult)
Observed ZmsvpaWhiting(adult) Z
Predicted ZmsvpaWhiting(adult) Z
(f) Whiting (juvenile)
Observed "ZmsvpaJuvenileWhiting(0-10-...
(g) Saithe (adult)
Observed ZmsvpaSaithe(adult) Z
Predicted ZmsvpaSaithe(adult) Z
(h) Saithe (juvenile)
Observed "ZmsvpaJuvenileSaithe(0-30-...
Observed total mortality (Z)
Model total mortality (Z)

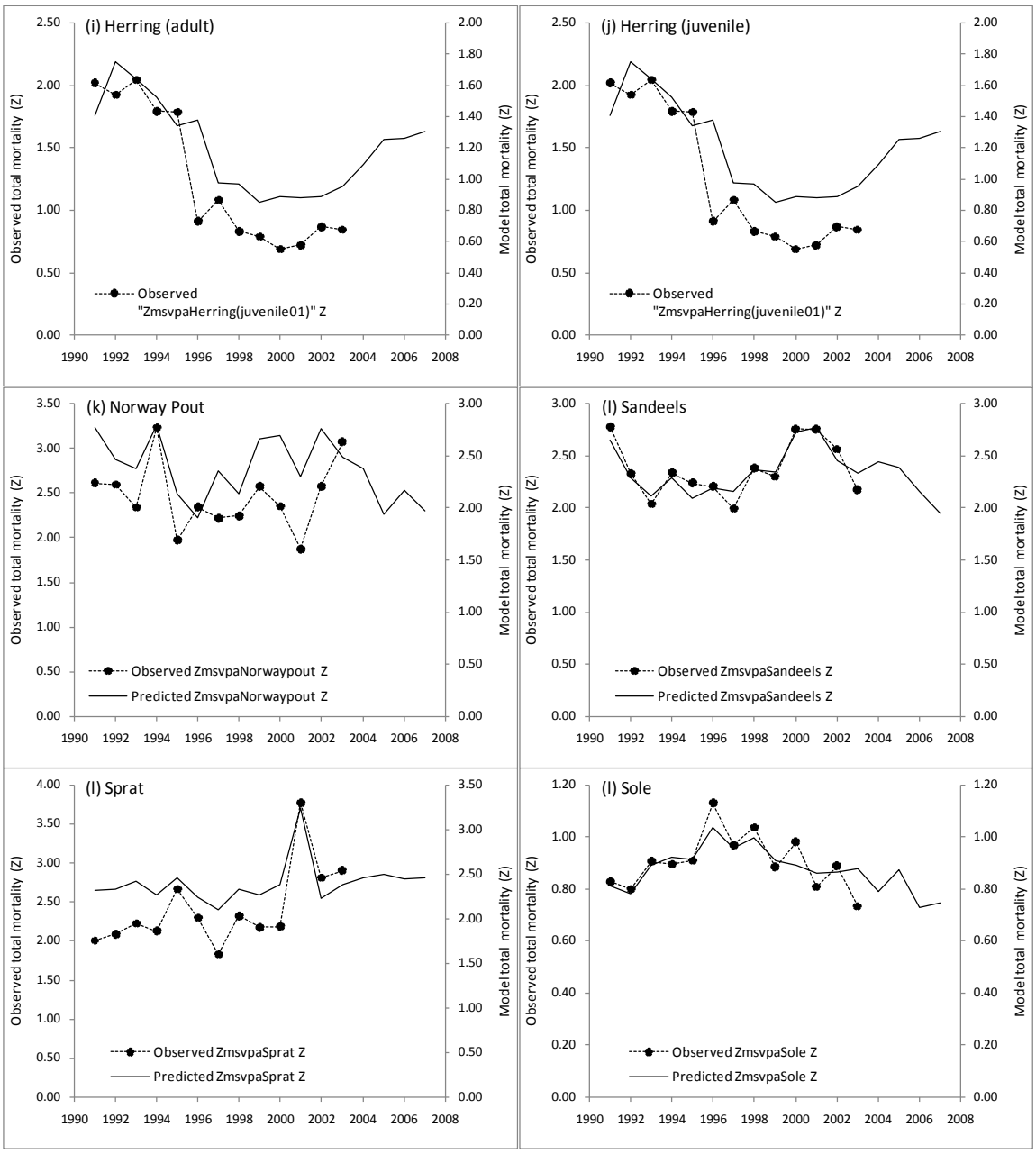
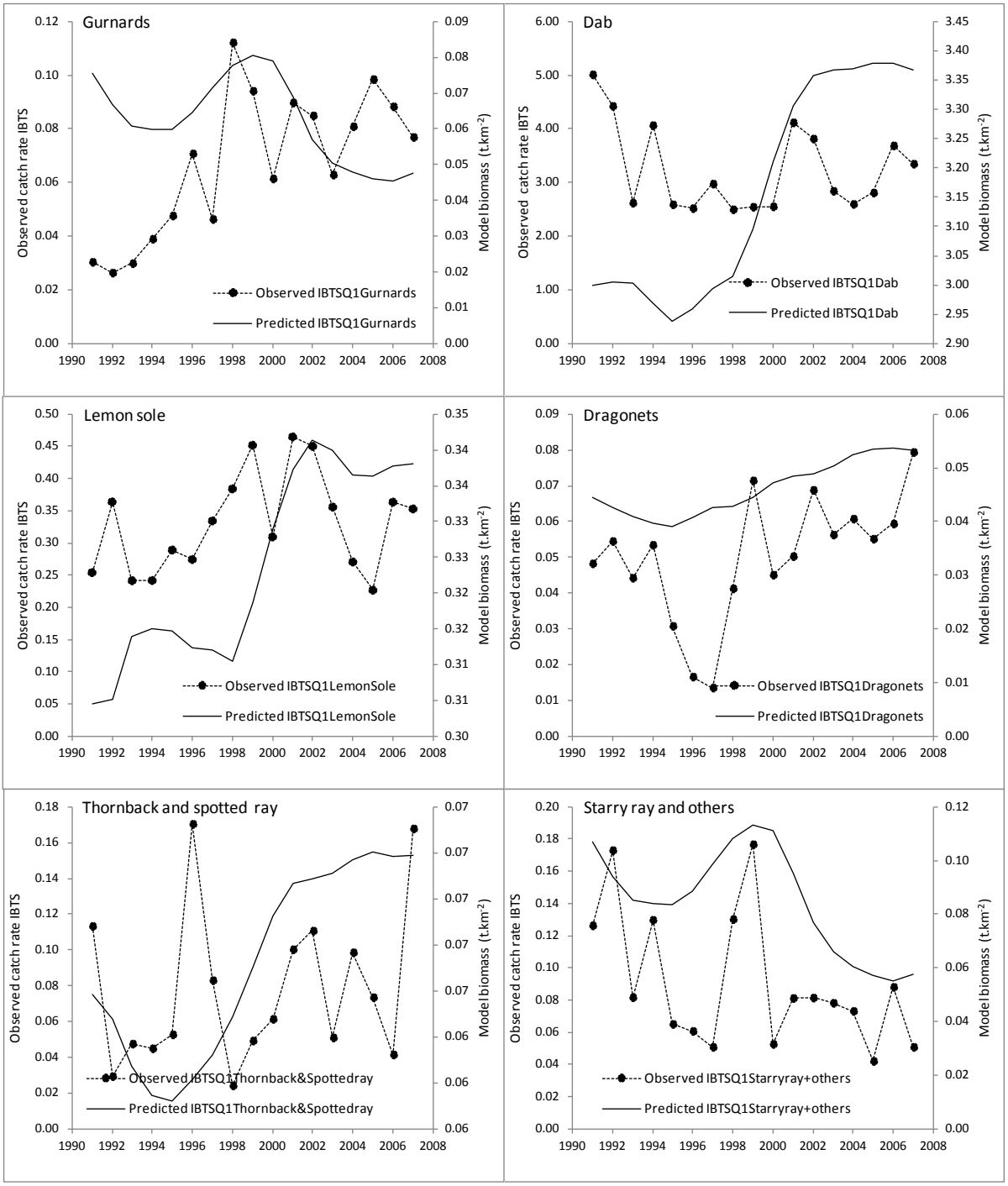


Figure 4.8. Total mortality plots - observed and model predicted. 'Observed' catch data are derived from MSVPA 2005 key run.

4.4.2 Relative biomass for non-assessed fish, benthos and plankton (Figure 4.9)



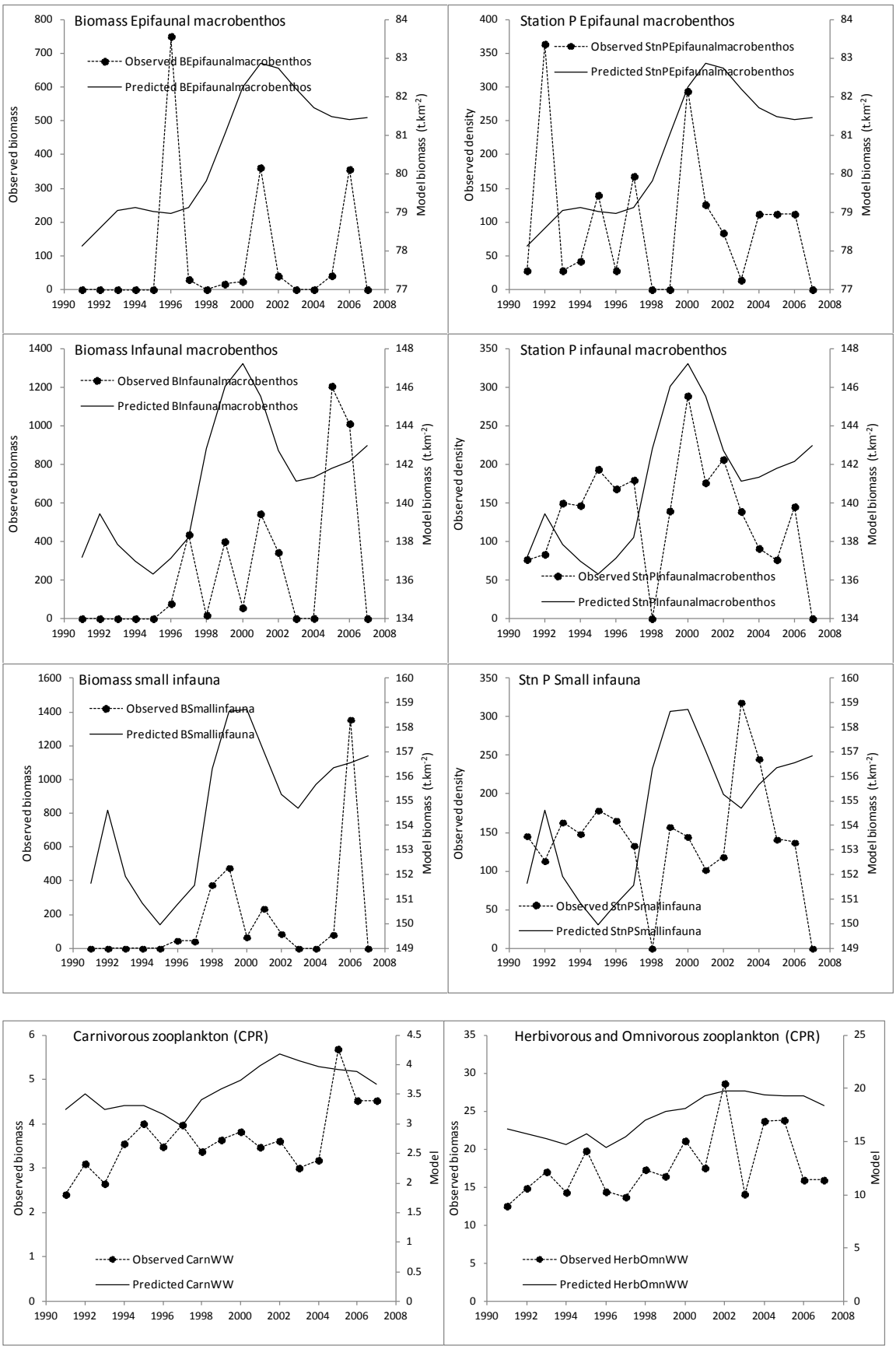


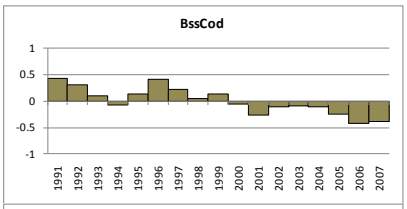
Figure 4.9. Relative biomass plots - observed and model predicted for non-assessed fish, benthos and zooplankton based on survey data.

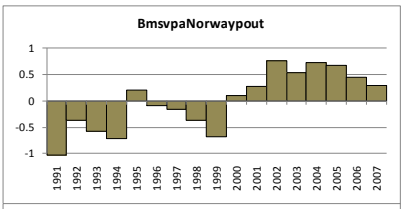
4.4.3 Equilibrium yield and biomass for fish and fleets

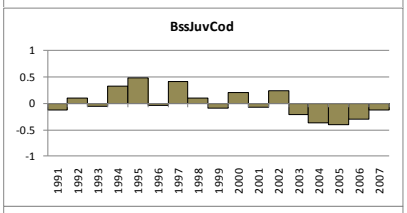
Equilibrium yield and biomass plots for key commercial species and fleets are given in Figures 4.5 and 4.6.

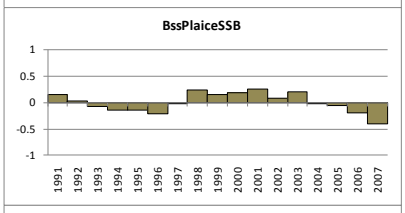
4.5 Fitting diagnostics

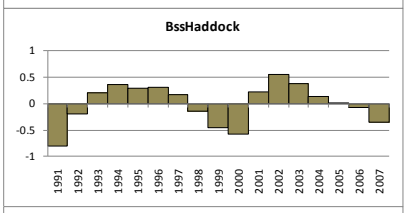
Plots of sum of squares residuals of model predictions to observation data are given in Figures 4.10, 4.11 for key species of interest, including all those for which vulnerability was estimated during model fitting. The rank order of SS values contributing to the total SS (Table 4.4) shows that the key species of interest in the model fitting score high in the ranking (where low SS score rank highly). This demonstrates that good model fits are achieved for these data, and that fitting data for the juvenile groups is more difficult due to their high variability. Fits of data for gurnard catch and biomass were poor (see also Figure 4.7, 3.9), but the quality of data for gurnards is highly uncertain.

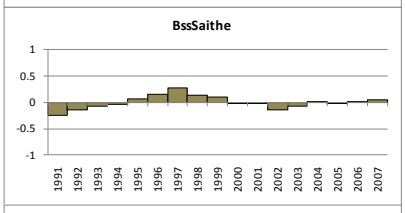

BssCod

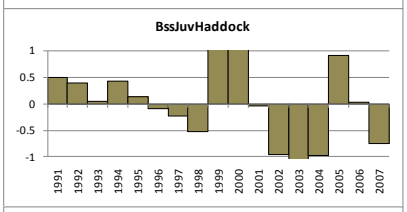

BmsvpaNorwaypout

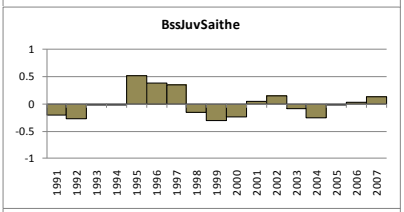

BssJuvCod

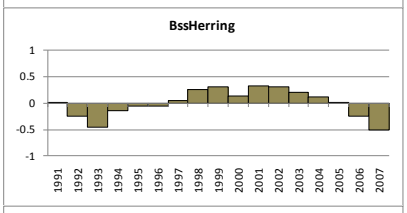

BssPlaiceSSB

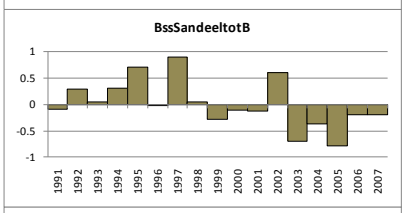

BssHaddock

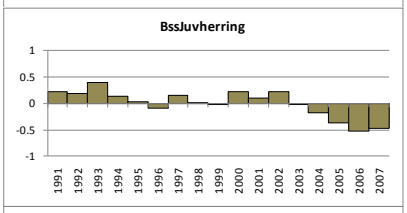

BssSaithe

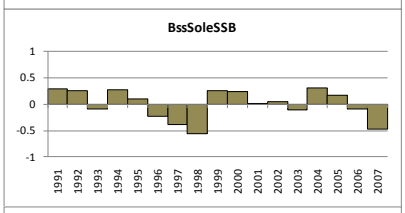

BssJuvHaddock

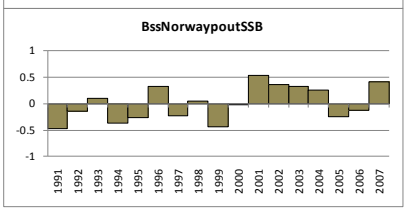

BssJuvSaithe

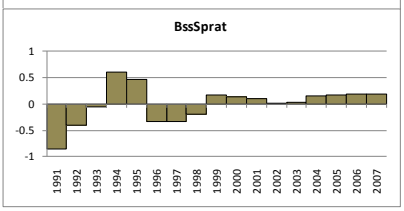

BssHerring


BssSandeeltotB


BssJuvherring


BssSoleSSB


BssNorwaypoutSSB


BssSprat

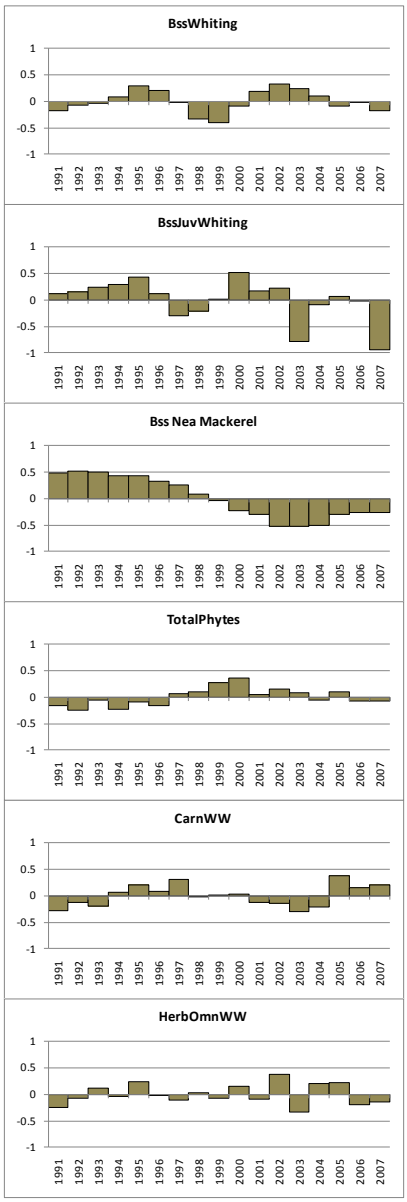
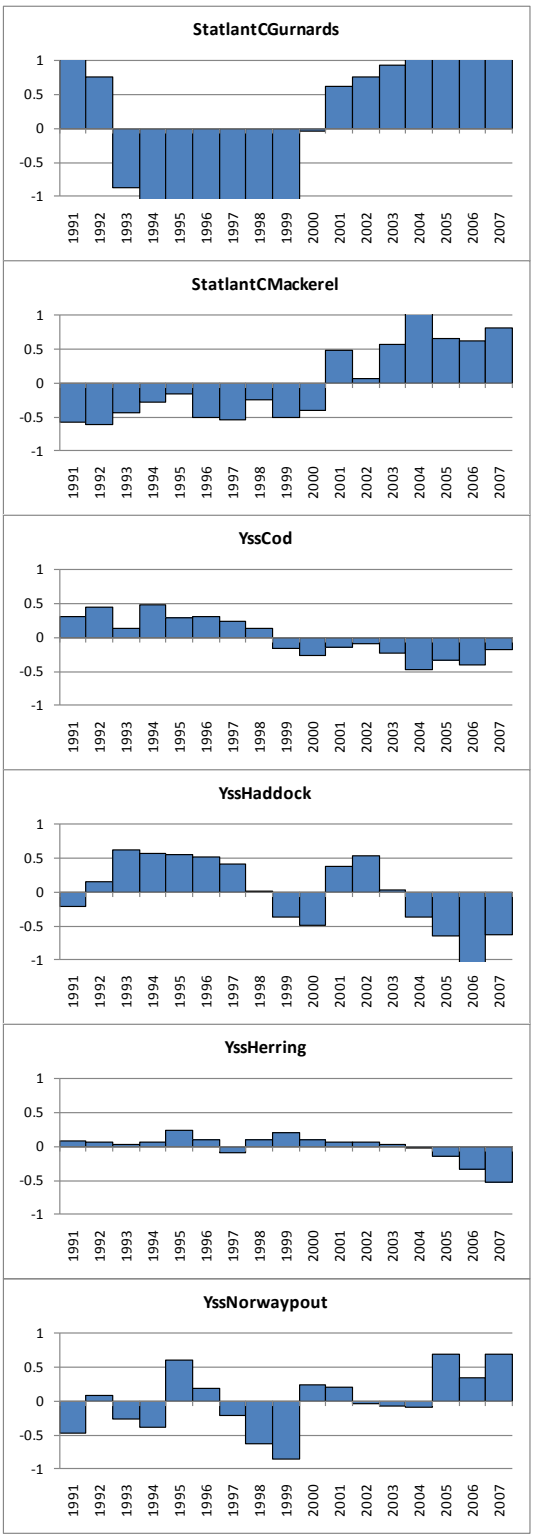


Figure 4.9. Residuals for relative biomass plots.


StatlantCGurnards
StatlantCMackerel
YssCod
YssHaddock
YssHerring
YssNorwaypout
1
0.5
0
-0.5
-1
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007

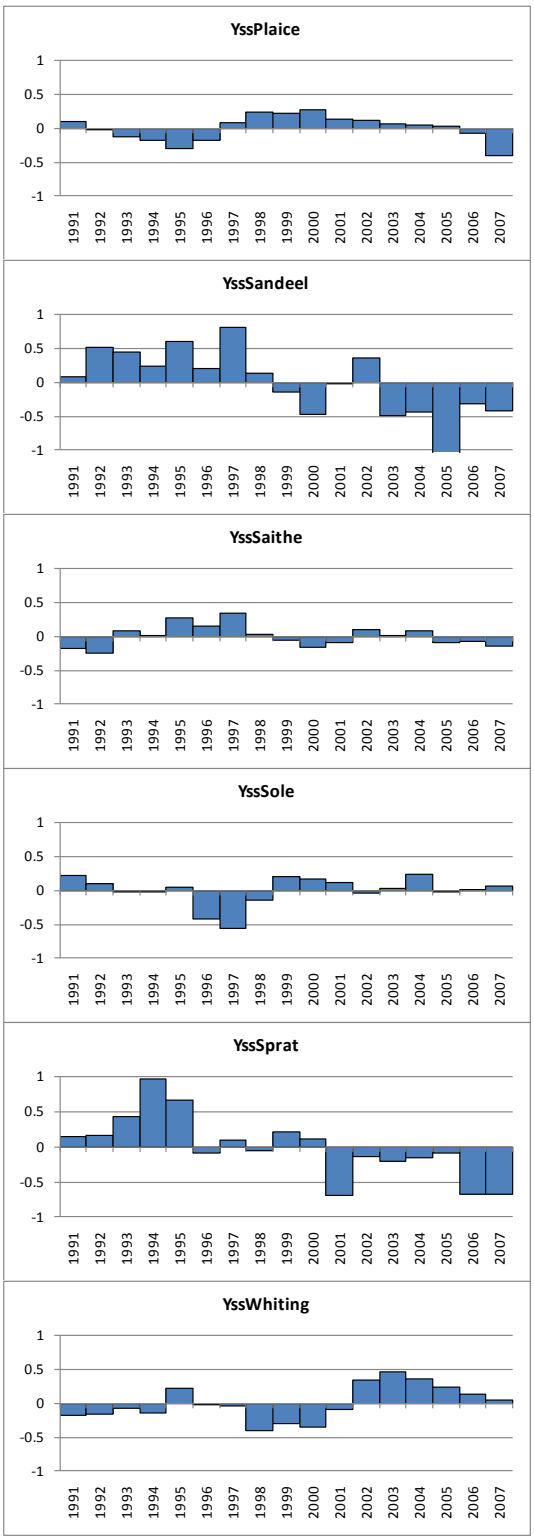


Figure 4.10. Residuals for catch plots

[Note: Graphs on Residual Analysis and Rank SS tab in file: EwE key run Calibration and testing file.xlsx]

Table 4.4. Rank order sums of squares contributions of each functional group to the overall model fit.

Functional group	SS for KEY Run	Parameter estimated in fitting (1=yes)
Baleen whales	0.00	
Seabirds	0.18	
Phytoplankton	0.44	
Herbivorous and Omnivorous zooplankton (copepods)	0.58	1
Carnivorous zooplankton	0.66	1
Saithe (adult)	0.69	1
Shrimp	0.77	
Seals	1.15	
Plaice	1.23	1
Juvenile Saithe (0–3, 0–40cm)	1.35	1
Lemon sole	1.50	
Juvenile Cod(0–2, 0–40cm)	1.54	1
Herring (juvenile 0, 1)	1.67	1
Herring (adult)	1.75	1
Sole	2.02	1
Whiting (adult)	2.20	1
Cod (adult)	2.72	1
Juvenile Whiting (0–1, 0–20cm)	2.77	1
Dragonets	2.87	
Large crabs	2.95	
Starry ray + others	3.22	
Large piscivorous sharks	3.78	
Small mobile epifauna (swarming crustaceans)	3.96	
Turbot and brill	4.76	
Nephrops	4.94	
Norway pout	5.06	1
Sprat	5.23	1
Small infauna (polychaetes)	5.26	
Other gadoids (large)	5.81	
Haddock (adult)	6.99	1
Small demersal fish	7.01	
Sandeels	7.11	1
Witch	7.52	
Mackerel	7.70	1
Small sharks	7.87	
Monkfish	9.19	
Juvenile Haddock (0–1, 0–20cm)	9.25	1
Thornback and Spotted ray	9.52	
Halibut	10.36	
Skate + cuckoo ray	10.65	
Flounder	11.66	

Functional group	SS for KEY Run	Parameter estimated in fitting (1=yes)
Infaunal macrobenthos	12.53	
Megrim	12.74	
Dab	13.06	
Horse mackerel	13.48	
Other gadoids (small)	13.84	
Large demersal fish	17.12	
Hake	20.11	
Catfish (Wolf-fish)	22.35	
Squid and cuttlefish	22.76	
Spurdog	24.71	
Gurnards	25.86	1
Long-rough dab	30.04	
Epifaunal macrobenthos (mobile grazers)	31.50	
Sessile epifauna	51.92	
Miscellaneous filterfeeding pelagic fish	56.67	
Blue whiting	81.24	
Grand Total	**625.82**	

5. Key run specification and setup

[File: EwE 5_6 Key Run Specification file.xlsx]

Details of the model setup-up to reproduce the model key run are provided in Table 5.1. Specific details of the key-run model settings, parameters and residual analysis are given in the file 'EwE 5_6 Key Run Specification file.xlsx'

Table 5.1. Definition of the model setup required to reproduce the key run.

Ecopath version	Database name	Model name	Time-series file name	Ecosim scenario name	Fishing	PP force	PP forcing function #	Sums of squares	No.vs.searched	number of time-series fitted to
EwE 5	EwE5_NorthSea 1991_Multistanza _New Profiles_2008 fit_Key Run.mdb	NorthSea1991_v7 SUBSIDIES	EwE5_NSea Time-series 1991_2007_July 2011_Key Run.csv	Key Run1_Fitting F and Obs PP	On	HADISST Loess+AMO Loess_NTOT	11	616.11 (when time-series read from database), 625.41 (when time-series read from file)	19	114
EwE 6	NorthSea1991_Key Run.eweaccdb		NSea Time-series 1991_2007_July 2011_Key Run.csv	Key Run1_Fitting F and Obs PP	On	HADISST Loess+AMO Loess_NTOT	11	625.82 (with time-series from database)	19	114

6. Outputs Ecosystem indicator trends

All output files of ecosystem metrics describing the state of the North Sea ecosystem in 1991, and changes from 1991–2007 in system and community level indicators are given in 2 data files:

- **NSea 1991_Key Run_Ecosystem Indices_Ecopath.xls** - which includes the outputs metrics from the balanced Ecopath model and thus serve as 'state' descriptors
- **NSea 1991_Key Run_Time Series Indicators.xlsx** - which includes plots of the indicators derived from the calibrated Ecosim key run.

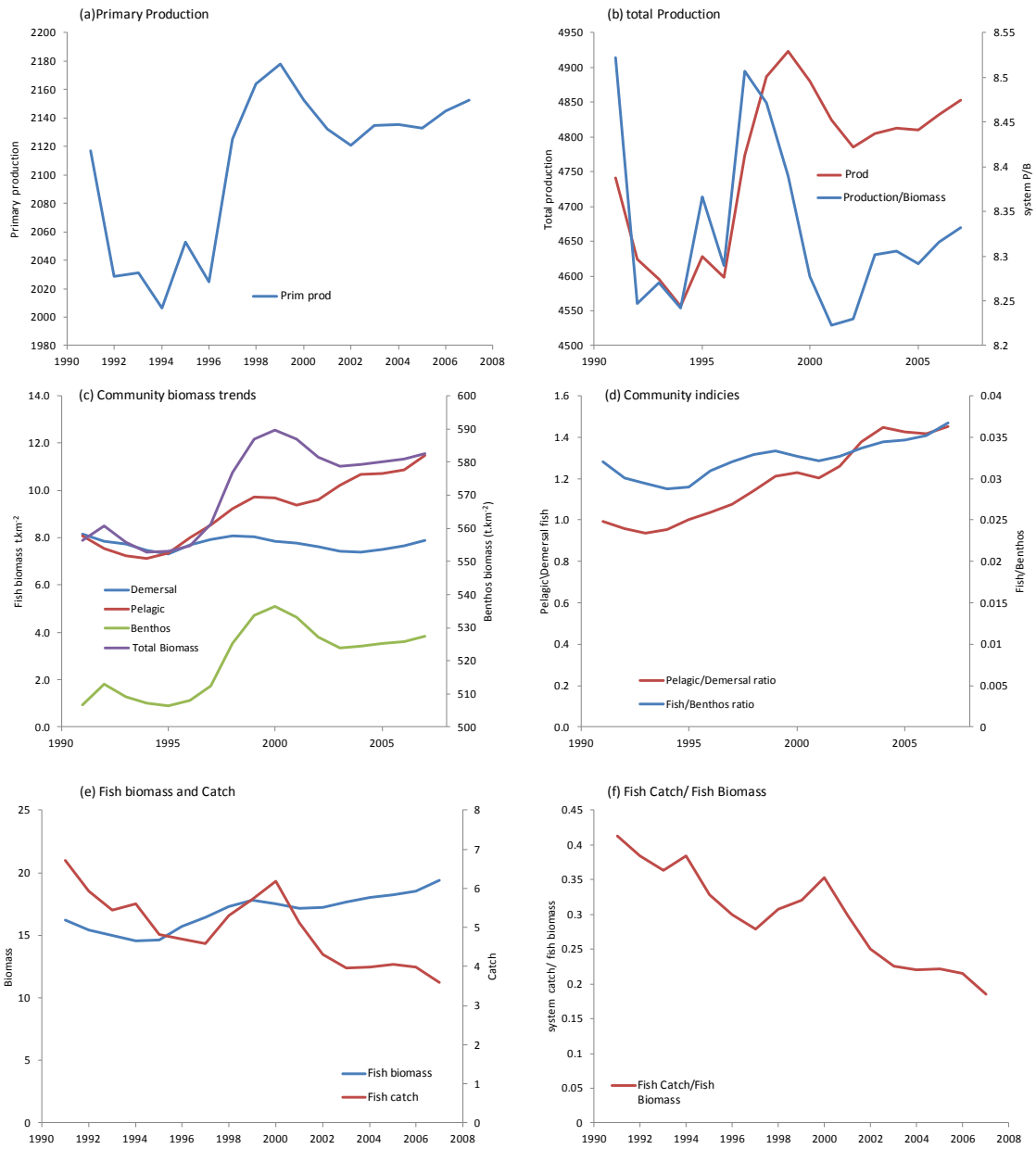
Changes in selected system and community indicators are shown in Figure 6.1. Referring to the figure panels, these include: *[Note: descriptions of ecosystem indicators referring to g and h are taken (with permission) from Tomczak et al., 2011.]*

- Primary production
- Total production of all groups in the system
- Trends in total system biomass and biomass of demersal fish, pelagic fish and benthos
- Community indices – Demersal/ Pelagic fish and Fish/Benthos
- Fish biomass and catch
- Total catch/biomass – as a measure of the overall fishing pressure
- Trophic level of the catch – catch weighted by trophic level – representing the mean TL of the catch, often referred to as the Marine Trophic Index. TL of catch captures how removal of top predatory fish results in catches dominated by small, lower TL species. It is expected to decrease with fishing (Pauly *et al.*, 1998, Shannon *et al.*, 2009). It has been the subject of controversy.
- Trophic level of the biomass – TL weighted by biomass – representing the mean TL of the system. Two indices are given; one that includes only assessed fish, the other including all groups.
- Overhead flow and proportion of flow to detritus.
- Overhead is the fraction of a system's capacity and it indicates the system's energy in reserve (Monaco and Ulanowicz, 1997). Overhead is divided into import, export, dissipation and internal flows (Ulanowicz, 2000), and the internal flow overhead seems to be the best indicator of the distribution of energy flow among the ecosystem pathways (Heymans *et al.*, 2007). Christensen (1995) links the system overhead to ecosystem stability and Heymans *et al.* (2003) proposed Ohv as an index of system resilience.
- The proportional flow to detritus measures the proportion of total trophic flows that flow into the detritus box (t/km^2/year). It has been proposed that as fishing impact increases, this indicator increases due to disruption of energy paths in the foodweb (Shannon *et al.*, 2009).
- Finns cycling index and mean path length.

The Finn's cycling index (Finn, 1976) is the proportion of the total system throughput that is recycled in the system. According to Monaco and Ulanowicz (1997), cycling is considered to be an important indicator of an ecosystem's ability to maintain its structure and integrity through positive feedback and is used as an indicator of stress

(Ulanowicz, 1986) and systems maturity (Christensen, 1995, Vasconcellos *et al.*, 1997). FCI is an indicator of the recovery time of an ecosystem through development of routes to conserve nutrients (Vasconcellos *et al.*, 1997). An increase in the FCI would mean the system would recover faster from a perturbation, whereas a system would be expected to take longer to recover (lower FCI) when it is in a more degraded state.

The mean path length (MPL) accounts for the number of functional groups involved in a flow of matter (Finn, 1976) and represents the average number of groups that an inflow or outflow passes through (Finn, 1980). The MPL is expected to decrease with fishing (Shannon *et al.*, 2009).



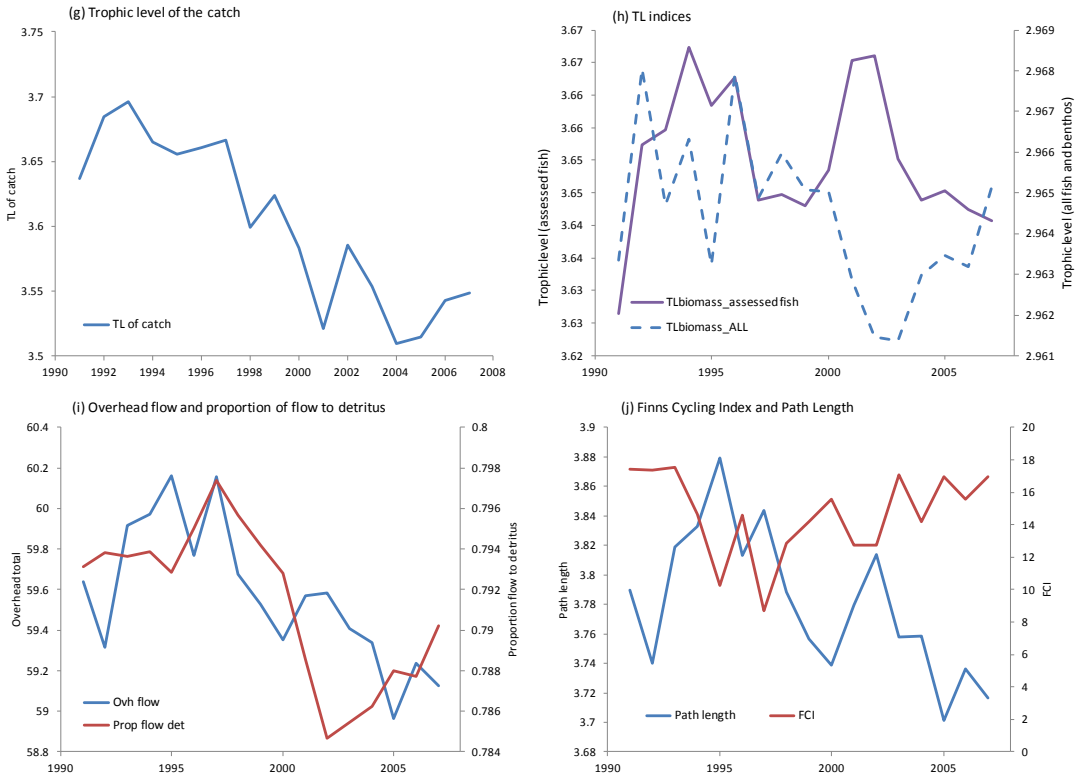


Figure 6.1. Ecosystem indicators derived from the model key run.

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APPENDIX 1: File Inventory

Ecopath 5			
Type	**Name**	**Description**	**USE**
Database	EwE5_NorthSea 1991_Multistanza_New Profiles_2008 fit_Key Run.mdb	Ecopath 5 database	
Ecopath file	NorthSea1991_v7 SUBSIDIES	Ecopath file	
Ecosim scenario	Key Run1_Fitting F and Obs PP	Ecosim scenario	
Time-series file	EwE5_NSea Time series 1991_2007_July 2011_Key Run.csv	Time series file for forcing and fitting data (can also be loaded directly from the database)	
Outputfiles	EwE5NorthSea.csv	Monthly for each individual functional group: Biomass, fishing rate. If the run is performed with the 'indices' on, the additional data include monthly Tlcatch, FIB, Kemptons, TL for each functional group	
	EwE5NorthSea_gear.csv	Monthly for each fished functional group-gear combination: catch	
	EwE5NorthSea_value.csv	Yearly values for each gear: effort, weight caught, value of catch, fishing costs. Plus, yearly for each functional group: landings and value	
	Allfit_Biomass.csv	For each relative biomass time series (type 0), yearly values for the time series data side-by-side with the model predictions for biomass	Plotting fits to time series
	Allfit_Catches.csv	For each catch time series (type 6), yearly values for the time series data side-by-side with the model predictions for catch	Plotting fits to time series
	Allfit_Mortality.csv	For each total mortality (z) time series (type 5), yearly values for the time series data side-by-side with the model predictions for	Plotting fits to time series

		biomass	
		Equilibrium	
	EwE5NorthSea1991_v7 SUBSIDIES_MSY.csv	Predictions of equilibrium plots included cross species impacts (if required). Analysis will only include the non-hidden groups selected. And, you can choose to freeze the biomass of other groups or let them vary in response to changes in F on the target group. Definitions: Fbase = baseline ecopath F, FMSY= F at MSY single species (but with interactions on), CmsySS= Catch MSY Single species (if all other groups held constant), CmsyMS= catch MSY multispecies (Ecosystem-scale MSY estimate (MSY for group if all other groups were harvested at Fmsy), ValueSS=,ValueSSbase =? File also contains data matrix for a bar-chart, with each row of the chart representing 10% reduction in F from Fmsy for one group, and the bars across this row representing resultant change in equilibrium yield for each of the other harvested groups.	Equilibrium analysis MSY
	Haddock equil_NorthSea.csv (NOTE: file produced only for the selected species then needs renaming)	For each F value, biomass of each FG and catch (named 'harvest') of the selected group	Equilibrium analysis - single species MSY
	Network.csv [only output if you do it without PPR]	Monthly network indices pertaining to the Ecosim run, except PPR	Ecosystem indices
Ecopath 6			
Database	NorthSea1991_Key Run.eweaccdb	The Ecopath and Ecosim model file	
Ecopath file		N/A (in EwE6 you can only have one model in one database)	
Ecosim scenario	Key Run1_Fitting F and Obs PP	Ecosim scenario	

Time series file	NSea Time series 1991_2007_July 2011_Key Run.csv	Time series file for forcing and fitting data (can also be loaded directly from the database)	
Outputfiles (Results Extractor)	FunctGroup(D19-8-2011)(T9-36-31).xlsx	For each FG - yearly values of biomass, catch, predation mortality, fishing mortality, and integrated biomass over the period. NOTE: Functional groups listed alphabetically	
	Fisheries(D19-8-2011)(T9-36-31).xlsx	Landings, discards and total catch per fleet for each prey, plus effort by year for each fleet	
	Indicators(D19-8-2011)(T9-36-31).xlsx	Outputs for Ecopath initial parameters in the file used, including: Basic estimates, fisheries catch quantities and value, search rates, electivity, predator and prey overlap matrices, respiration, consumption, fishing mortalities, predation mortality and mortality coefficients, key indices	
Output files (Ecosim)	EwE6-Ecosim_annual_biomass.csv	Yearly biomass for each FG	
	EwE6-Ecosim_annual_cons_biom.csv	Yearly Q/B values for each FG	
	EwE6-Ecosim_annual_feedingtime.csv	Yearly feeding time factor for each FG	
	EwE6-Ecosim_annual_mortality.csv	Yearly Total Mortality (Z) for each FG	
	EwE6-Ecosim_annual_predation.csv	Yearly predation mortality on the selected (viewed group) by its predators	
	EwE6-Ecosim_annual_prey.csv	Yearly proportion of each prey in the diet of the selected predator	
	EwE6-Ecosim_annual_TL.csv	Yearly TL for each FG	
	EwE6-Ecosim_annual_weight.csv	Yearly average weight for each FG	
	EwE6-Ecosim_annual_yield.csv	Yearly catch for each FG	
	NorthSea1991_v7 SUBSIDIES_Allfit_Biomass.csv	For each relative biomass time series (type 0), yearly values for the time series data side-by-side with the model predictions for	Plotting fits to time series

		biomass	
	NorthSea1991_v7 SUBSIDIES_Allfit_Catches.csv	For each catch time series (type 6), yearly values for the time series data side-by-side with the model predictions for catch	Plotting fits to time series
	NorthSea1991_v7 SUBSIDIES_Allfit_Mortality.csv	For each total mortality (z) time series (type 5), yearly values for the time series data side-by-side with the model predictions for biomass	Plotting fits to time series
	EwE6-NA_monthly_IndicesWithoutPPR.csv	Monthly values of key network indicators (except PPR)	Ecosystem indicators
	EwE6-NA_annual_IndicesWithoutPPR.csv	Yearly values of key network indicators (except PPR)	Ecosystem indicators
Excel support files			
Location	Name	Description	
Key Run 2011	NSea 1991_Key Run SNAPSHOT.xls	All inputs for Ecopath and Ecosim required to 'rebuild' a database by cutting and pasting from Excel. Plus all the key output diagnostics from Ecopath such as mortalities, consumptions and ecosystem metrics	
Key Run 2011	NSea 1991_Key Run_Ecosystem Indices_Ecopath.xlsx	All OUTPUTS from Ecopath for describing the state of the Ecosystem, foodweb, flows, fisheries, predator–prey interactions. Lots of Tables	

Key Run 2011	Key Run_Time Series Indicators.xlsx	Time series of network indicators for the Key run, compiled from EwE6-NA_monthly_IndicesWithoutPPR.csv and (for FIB) Network.csv (EwE 5 output)	
Key Run 2011	EwE 5_6 Key Run Specification file.xlsx	Setup run setting and Key run SS, Search weightings, time series data, forcing function codes, fitted vulnerabilities for EwE5 and 6, Group infor params, Residual Analysis plots for key run	
Key Run 2011	EwE key run Calibration and testing file.xlsx	File used during preparation and calibration to time series data. Includes setup and templates for evaluating residuals, testing plots and group info settings, SR parameters and exploring any aspects of the fitting processes.	
Key Run 2011	NSea 1991_Key Run SNAPSHOT.xls	All inputs for Ecopath and Ecosim required to 'rebuild' a database by cutting and pasting from Excel. Plus all the key output diagnostics from Ecopath such as mortalities, consumptions and ecosystem metrics	
Key Run 2011	Fitting Plots.xlsx	File for pasting the Ecopath output files and creating graphs, including biomass, catch, equilibrium plots. Includes output from MSY routine.	

Key Run 2011	Fitting Plot_Template_previous.xlsx	File used previous for plotting the data used in evaluating MSY in the "Mixed Fishery or Ecosystem conundrum" paper. May be useful for future MSY work	
Key Run 2011	Scaling Environmental data_Solver_new scaling.xlsx	File used for scaling observed environmental indicators to the scale of the primary production anomaly generated by EwE. It also includes combined anomalies scaled to the PP anomaly. Use solver to make the fits. Also used to convert anomaly months to years and do a correlation on the EwE pp anomaly and observed environmental indices. Only data for Key run fit	
Key Run 2011	Stock recruitment plots_V2.xls	Plots of relative slopes of R/SSB and SR series for those commercial species defined as multi-stanza groups in the model	

Appendix 2: Metadata summary Table

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
Baleen whales	**1**								1950-2007								
Toothed whales	**2**																
Seals	**3**			**1963 - 2003**													
Seabirds	**4**			1963 - 2003	**1963 - 2007**												
Juvenile sharks	**5**																
Spurdog	**6**					**1977 - 2008**		1983-2006	**1950-2007**								

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
Large piscivorous sharks	**7**					**1977-2008**		1983-2006	**1950-2007**								
Small sharks	**8**					**1977-2008**		1983-2006	**1950-2007**								
Juvenile rays	**9**																
Starry ray + others	**10**			1963-2003	**1963-2007**	1977-2008		1983-2006	**1950-2007**								
Thornback and Spotted ray	**11**					**1977-2008**		1983-2006	**1950-2007**								
Skate	**12**					**1977-2008**		1983-2006	**1950-2007**								
Juvenile Cod(0-2, 0-	**13**		**1963-2007**	1963-	1963-		1982-						**1963-2003**		1963-		

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
40cm)				2003	2007		2007 (IBTS_Q1_IV)								2003		
Cod (adult)	**14**	y	**1963-2008**	1963-2003	1963-2007	1977-2008	1982-2007 (IBTS_Q1_IV)	1983-2006	1950-2007	1963-2003	**1963-2007**		**1963-2003**	**1963-2007**	1963-2003		
Juvenile Whiting (0-1, 0-20cm)	**15**		**1980-2007**	1963-2003	1963-2007		1967-2008 (IBTS_Q1)						**1963-2003**		1963-2003		
Whiting (adult)	**16**	y	**1980-2008**	1963-2003	1963-2007	1977-2008	1967-2008 (IBTS_Q	1983-2006	1950-2007	1963-2003	**1980-2007**		**1963-2003**	**1980-2007**	1963-2003		

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
							1)										
Juvenile Haddock (0-1, 0-20cm)	**17**		**1963-2007**	1963-2003	1963-2007		1982-2008 (IBTS_Q1)						**1963-2003**		1963-2003		
Haddock (adult)	**18**	y	**1963-2008**	1963-2003	1963-2007	1977-2008	1982-2008 (IBTS_Q1)	1983-2006	1950-2007	1963-2003	**1963-2007**		**1963-2003**	**1963-2007**	1963-2003		
Juvenile Saithe (0-3, 0-40cm)	**19**		**1967-2007**	1963-2003	1963-2007		1984-2004 (IBTS_Q1)						**1963-2003**		1963-2003		
Saithe (adult)	**20**	y	**1967-2008**	1963-	1963-	1977-	1984-	1983-2006	1950-2007	1963-	**1967-**		**1963-2003**	**1967-**	1963-		

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
				2003	2007	2008	2004 (IBTS_Q1)			2003	**2007**			**2007**	2003		
Hake	**21**					**1977-2008**		1983-2006	**1950-2007**								
Blue whiting	**22**					**1977-2008**		1983-2006	**1950-2007**								
Norway pout	**23**	y	**1983-2008**	1963-2003	1963-2007	1977-2008	1970-2008 (IBTS_mean over surveys)	1983-2006	1950-2007	1963-2003	**1983-2007**		**1963-2003**	**1983-2008**	1963-2003		
Other ga-	**24**					**1977**		1983-	**1950-**								

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
doids (large)						**-2008**		2006	**2007**								
Other gadoids (small)	**25**					**1977-2008**		1983-2006	**1950-2007**								
Monkfish	**26**					**1977-2008**		1983-2006	**1950-2007**								
Gurnards	**27**			1963-2003	**1963-2007**	1977-2008		1983-2006	**1950-2007**								
Herring (juvenile 0, 1)	**28**		**1963-2007**	1963-2003	1963-2007		1984-2004 (Acoustic)						**1963-2003**	**1963-2008**	1963-2003		
Herring (adult)	**29**	y	**1963-2007**	1963-2003	1963-2007	1977-2008	1984-2004	1983-2006	1950-2007	1963-2003	**1963-2007**		**1963-2003**	**1963-2008**	1963-2003		

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
							(Acoustic)										
Sprat	**30**		**1983-2004**	1963-2003		1977-2008	1984-2005 (IBTS_Q1)	1983-2006	1950-2007	1963-2003	**1987-2006**		**1963-2003**	**1984-2003**	1963-2003		
Mackerel	**31**		**1980-2008**	1963-2003	1963-2007	1977-2008		1983-2006	**1950-2007**								
Horse mackerel	**32**			1963-2003	**1963-2007**	1977-2008		1983-2006	**1950-2007**								
Sandeels	**33**	y	**1983-2007**	1963-2003	1963-2007	1977-2008	1976-2008 (DAN_NOR_	1983-2006	1950-2007	1963-2003	**1983-2008**		**1963-2003**	**1983-2008**	1963-2003		

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
							SAN_cpue)										
Plaice	**34**	y	**1963-2008**	1963-2003	1963-2007	1977-2008	1970-2007 (DFS)	1983-2006	1950-2007	1963-2003	**1963-2007**		**1963-2003**	**1963-2007**	1963-2003		
Dab	**35**					**1977-2008**		1983-2006	**1950-2007**								
Long-rough dab	**36**					**1977-2008**		1983-2006	**1950-2007**								
Flounder	**37**					**1977-2008**		1983-2006	**1950-2007**								
Sole	**38**	y	**1963-2008**	1963-2003	1963-2007	1977-2008	1970-2007	1983-2006	1950-2007	1963-2003	**1963-2007**		**1963-2003**	**1963-2007**	1963-2003		

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
							(SNS)										
Lemon sole	**39**					**1977-2008**		1983-2006	**1950-2007**								
Witch	**40**					**1977-2008**		1983-2006	**1950-2007**								
Turbot and brill	**41**					**1977-2008**		1983-2006	**1950-2007**								
Megrim	**42**					**1977-2008**		1983-2006	**1950-2007**								
Halibut	**43**					**1977-2008**		1983-2006	**1950-2007**								
Dragonets	**44**					**1977-**		1983-									

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
						2008		2006									
Catfish (Wolffish)	**45**					**1977 - 2008**		1983-2006	**1950-2007**								
Large demersal fish	**46**					**1977 - 2008**		1983-2006	**1950-2007**								
Small demersal fish	**47**					**1977 - 2008**		1983-2006	**1950-2007**								
Miscellaneous filterfeeding pelagic fish	**48**					**1977 - 2008**		1983-2006	**1950-2007**								
Squid and cuttlefish	**49**						1980 - 2002	1983-2006	1950-2007			**1980-2004 (UK stats)**					

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
Fish larvae	**50**																
Carnivorous zooplankton	**51**						1958-2006										
Herbivorous and Omnivorous zooplankton (copepods)	**52**						1958-2007										
Gelatinous zooplankton	**53**																
Large crabs	**54**							1983-2006	**1950-2007**								
Nephrops	**55**	y					1984-2003	1983-2006	1950-2007			**1982-2007**					
Epifaunal macroben-	**56**						1993-		**1950-2007**								

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
thos (mobile grazers)							2006, 1971-2006										
Infaunal macrobenthos	**57**						1993-2006, 1971-2006		**1950-2007**								
Shrimp	**58**	y					1984-2003		1950-2007			**1970-2007**					
Small mobile epifauna (swarming crustaceans)	**59**						1971-2006	1983-2006									
Small infauna (polychaetes)	**60**						1993-										

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
							2006										
Sessile epifauna	**61**						1971 - 2006		**1950-2007**								
Meiofauna	**62**																
Benthic microflora (incl. Bacteria, protozoa))	**63**																
Planktonic microflora (incl. Bacteria, protozoa)	**64**																
Phytoplankton	**65**						1958 - 2006										
Detritus - DOM -water	**66**																

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
column																	
Detritus - POM - sediment	**67**																
Discards	**68**																
NAO Winter Index																	1950-2007
AMO																	1950-2007
AMO smoothed (USGS)																	1950-2007
Gulfstream Index																	1950-2007
Faroe/Shetflux (Q1)																	1950-2007

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
Marsdiep T																	1950-2007
Marsdiep Salinity																	1950-2007
HADISST North Sea																	1950-2007
Atlantic tripole																	1950-2007
Windspeed																	1950-2007
Demersal trawl and seine																**1973-2008**	
Beam trawl																**1973-2007**	

			Fitting data (used is in bold)											**Forcing data avail-**			
			Relative biomass						**Catch**				**Total mortality (Z)**	**Fishing mortality**			**Environmental variable**
GROUPS	**Pool code**	Stock assessment?	B_SS assess-sess-ment	B_MSVPA	SMS 2008	B_IBTSQ1 Survey	B_IBTS Survey	B_IBTS Survey	Stat-lantC	Y_msvpa	Y_SS	Other	Z_msvpa	F_SS	F_msvpa	**Relative effort**	
Pelagic trawl and seine																**1973 - 2007**	
Sandeel trawl																**1973 - 2008**	
Nephrops trawl																**1973 - 2003**	
Shrimp trawl																**1984 - 2003**	