

# Ecosystem-based fisheries management increases catch and carbon sequestration through recovery of exploited stocks: the western Baltic Sea case study

## **Supplementary Materials**

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## Study area

Geographical regions represented by the western Baltic Sea (WBS) model are Great and Little Belt, Kiel Bight, Bay of Mecklenburg and the Arkona Basin until west of the Bornholm Basin (ICES subdivisions 22 and 24); all NATURA 2000 areas in the German Exclusive Economic Zone (EEZ) are included. These NATURA 2000 areas comprise the Fehmarn Belt, Kadetrinne, western Rönnebank, Adlergrund and Pomeranian Bay with the Oderbank. The Pomeranian Bay is a designated EU bird protection area.

## Materials and methods

### Trophic groups represented in the model of the WBS ecosystem

In the Ecopath with Ecosim (EwE) model of the WBS ecosystem, the following 18 trophic groups are represented: (1) harbour porpoises, (2) seals, (3) seabirds, (4) adult cod, (5) juvenile cod, (6) flatfish, (7) other demersal fish, (8) herring, (9) sprat, (10) other pelagic fish, (11) pelagic macrofauna, (12) benthic macrofauna, (13) benthic meiofauna, (14) zooplankton, (15) bacteria/microorganisms, (16) phytoplankton, (17) benthic producers, and (18) detritus/DOM.

Table S1. List of species included in the “seabirds”, “other demersal fish” and “other pelagic fish” compartments; the symbol † indicates flatfish.

Family	Species
Seabirds	
Alcidae	<i>Alca torda</i> , <i>Cephus grylle</i> , <i>Uria aalge</i>
Anatidae	<i>Aythya marila</i> , <i>Clangula hyemalis</i> , <i>Melanitta fusca</i> , <i>Melanitta nigra</i> , <i>Mergus serrator</i> , <i>Somateria mollissima</i>
Gaviidae	<i>Gavia stellata</i> , <i>Gavia arctica</i>
Laridae	<i>Hydrocoloeus minutus</i> , <i>Larus argentatus</i> , <i>Larus canus</i> , <i>Larus fuscus</i> , <i>Larus marinus</i> , <i>Larus ridibundus</i> , <i>Rissa tridactyla</i>
Phalacrocoracidae	<i>Phalacrocorax carbo</i>
Podicipedidae	<i>Podiceps auritus</i> , <i>Podiceps cristatus</i> , <i>Podiceps grisegena</i>
Procellariidae	<i>Fulmarus glacialis</i>
Sternidae	<i>Sterna hirundo</i> , <i>Sterna paradisaea</i> , <i>Sterna sandvicensis</i>
Sulidae	<i>Morus bassanus</i>
Other demersal fish	
Agonidae	<i>Agonus cataphractus</i>
Ammodytidae	<i>Ammodytes marinus</i> , <i>Ammodytes tobianus</i> , <i>Hyperoplus lanceolatus</i>
Anguillidae	<i>Anguilla anguilla</i>

Bothidae †	<i>Arnoglossus laterna</i>
Callionymidae	<i>Callionymus lyra</i> , <i>Callionymus maculatus</i>
Cottidae	<i>Myoxocephalus scorpius</i>
Cyclopteridae	<i>Cyclopterus lumpus</i>
Esocidae	<i>Esox lucius</i>
Gadidae	<i>Melanogrammus aeglefinus</i> , <i>Merlangius merlangus</i> , <i>Pollachius virens</i> , <i>Pollachius pollachius</i> , <i>Trisopterus esmarkii</i> , <i>Trisopterus minutus</i>
Gasterosteidae	<i>Gasterosteus aculeatus</i> , <i>Spinachia spinachia</i>
Gobiidae	<i>Gobius niger</i> , <i>Neogobius melanostomus</i> , <i>Pomatoschistus minutus</i> , <i>Pomatoschistus microps</i>
Gunnelidae	<i>Pholis gunnellus</i>
Lotidae	<i>Enchelyopus cimbrius</i> , <i>Gaidropsarus vulgaris</i> , <i>Molva molva</i>
Merlucciidae	<i>Merluccius merluccius</i>
Moronidae	<i>Dicentrarchus labra</i>
Mullidae	<i>Mullus surmuletus</i>
Percidae	<i>Gymnocephalus cernua</i> , <i>Perca fluviatilis</i>
Petromyzontidae	<i>Lampetra fluviatilis</i>
Pleuronectidae †	<i>Glyptocephalus cynoglossus</i> , <i>Hippoglossoides platessoides</i> , <i>Hippoglossus</i> <i>hippoglossus</i> , <i>Microstomus kitt</i>
Salmonidae	<i>Salmo salar</i>
Scophthalmidae †	<i>Lepidorhombus whiffiagonis</i> , <i>Zeugopterus punctatus</i>
Scorpaenidae	<i>Liparis liparis liparis</i>
Soleidae †	<i>Buglossidium luteum</i> , <i>Solea solea</i>
Stichaeidae	<i>Leptoclinus maculatus</i> , <i>Lumpenus lampretaeformis</i>
Syngnathidae	<i>Syngnathus acus</i> , <i>Syngnathus rostellatus</i> , <i>Syngnathus typhle</i>
Trachinidae	<i>Trachinus draco</i>
Triglidae	<i>Chelidonichthys lucerna</i> , <i>Eutrigla gurnardus</i>
Zoarcidae	<i>Zoarces viviparus</i>
Other pelagic fish	
Clupeidae	<i>Alosa fallax</i> , <i>Sardina pilchardus</i>
Atherinidae	<i>Atherina presbyter</i>
Belonidae	<i>Belone belone</i>
Engraulidae	<i>Engraulis encrasicolus</i>
Osmeridae	<i>Osmerus eperlanus</i>
Salmonidae	<i>Salmo trutta</i>
Percidae	<i>Sander lucioperca</i>
Scombridae	<i>Scomber scombrus</i>
Carangidae	<i>Trachurus trachurus</i>

## Data sources

Principal data sources were: FishBase<sup>1</sup>, SeaLifeBase<sup>2</sup>, ICES databases, ICES Advice documents, ICES Working Group Reports, ICES Stock Summaries, DATRAS BITS, HELCOM, ecosystem models from other Baltic Sea areas (see Table S2), relevant literature (articles and grey literature) and personal communications by expert colleagues.

Table S2. Selection of ecosystem models for the Baltic Sea with indication of publication year, area and objectives. Grey fields indicate model areas. Objective of models targeting fisheries are in bold; text in italics informs on the methods applied. The model type (MT) column clarifies whether mass-balanced (MB) and/or other (O) approaches are applied. Further descriptions of the models are in Eero et al. (2021) and Korpinen et al. (2022).

MT	Author	Year	Baltic Sea area modelled											Objectives	
			Baltic Proper	Western Baltic Sea	Central Baltic Sea	Puck Bay	Curonian Spit	Lithuanian Coast	Gulf of Riga	Pärnu Bay	Gulf of Finland	Bothnian Sea	Öre Estuary	Gulf of Bothnia	
O	Karlson et al.	2020													Testing - by use of <i>partial least square regressions</i> - whether the physiological status of consumers (grey seal, cod, herring, sprat and a benthic isopod) can be explained by food web structure and prey food value
MB	Opitz and Froese	2019													<b><i>EwE model for the analysis and attenuation of impacts of commercial fisheries on the ecosystem and its components by comparison of different fisheries scenarios</i></b>
MB	Bauer et al.	2019a													Potential future ecological states of the Baltic Sea ecosystem under five scenarios based on an <i>EwE spatial food web model</i> that was forced by a <i>physical-biogeochemical model</i>
MB & O	Bauer et al.	2019b													<b>Model uncertainty and simulated multispecies fisheries management advice in the Baltic Sea</b>

<sup>1</sup> [www.fishbase.org](http://www.fishbase.org)

<sup>2</sup> [www.sealifebase.org](http://www.sealifebase.org)

MB & O	Kulatska et al.	2019													Reconstruction of the predator-prey dynamics of cod, herring, and sprat to predict changes in the Baltic cod diet by use of an <i>age-length structured multispecies model</i> using <i>Gadget</i>
O	Maldonado et al.	2019													Analysis of the Baltic Sea food web by use of <i>Dynamic Bayesian Networks</i> to examine potential unobserved processes affecting the ecosystem and predicting some (hidden) variables of interest
MB & O	Bauer et al.	2018													<b>Potential of eutrophication management to affect future commercial fishing in the central Baltic Sea analyzed by use of an <i>EWE spatial-temporal framework</i> that was forced by a <i>coupled physical-biogeochemical model</i></b>
O	Bossier et al.	2018													<b>Baltic implementation of the <i>spatially explicit end-to-end Atlantis ecosystem model</i> linked to two external models for the exploration of the different pressures on the marine ecosystem</b>
O	Uusitalo et al.	2018													Fitting of a series of <i>Dynamic Bayesian Networks</i> with different hidden variable structures to the Baltic Sea food web
O	Jacobsen et al.	2017													<b>Efficiency of fisheries in five large marine ecosystems (including the Baltic Sea), with respect to yield and ecosystem impact by use of a novel calibration of <i>size-based ecosystem models</i></b>
O	Norrström et al.	2017													<b>Definition of a multi-species-MSY to be solved through the “<i>game theory concept of Nash equilibrium</i>” with two solutions for the Baltic Sea</b>
O	Gårdmark et al.	2015													Identification (based on field data) of mechanisms underlying alternative stable states caused by predator-prey interactions by use of <i>theory on size-structured community dynamics</i>

O	Lindegren et al.	2014													Metacommunity-perspective on source-sink dynamics of Baltic Sea fish stocks using a <i>spatially disaggregated statistical food web model</i>
O	Gårdmark et al.	2013													<b>Simulation of the long-term response of eastern Baltic cod to future fishing and climate change by use of a “biological ensemble modelling approach”</b>
MB	Tomczak et al.	2013													Simulation of long term-dynamics from 1974 to 2006 (i.e. including a regime shift) with a <i>food web model</i> forced by climate and fishing
MB & O	Niiranen et al.	2013													<i>Multi-model approach</i> that studies how the concurrent action of climate, nutrient load and cod fishing affects the biomass of main trophic groups in the ecosystem
O	Casini et al.	2012													Understanding effects of flows across systems
MB	Tomczak et al.	2012													<b><i>Food web model</i> to simulate and understand trophic interactions and matter flows over time from primary producers to top predators including fisheries</b>
O	Teschner et al.	2010													Effects of oxygen deficiency on Baltic cod consumption rates and their impact on stock size estimates; the analysis was based on <i>multi-species models</i>
O	Lindegren et al.	2009													Reconstruction of the history of the eastern Baltic cod stock using a <i>statistical food web model</i> for the Baltic Sea
MB	Tomczak et al.	2009							Coast						Comparative analysis of carbon flow networks in southeastern coastal ecosystems of the Baltic Sea
O	Van Leeuwen et al.	2008													Cod-sprat interaction based on a <i>stage-structured biomass model</i> , paying special attention to the size-dependent prey preferences of differently sized cod
MB	Hansson et al.	2007													<b>Management of Baltic Sea fisheries under contrasting conditions of production and predation</b>
MB	Sandberg	2007													Analysis across ecosystems of pelagic food web structure and processes in three main basins of the Baltic Sea

MB	Sandberg et al.	2004													Quantitative assessment of the relative significance of terrigenous dissolved organic substances as carbon source for secondary producers (e.g. bacteria) and as structuring factor for the pelagic food web in the Gulf of Bothnia
MB	Harvey et al.	2003													<b>Evaluating interactions between fisheries and the food web</b>
MB	Sandberg et al.	2000													Re-evaluating carbon flows in food webs of the Baltic Sea using a <i>mass-balance approach</i>
O	Horbowy	1996													Estimating dynamics of cod, herring, and sprat stocks from 1982 to 1992 by use of a <i>multispecies stock-production model</i>
MB	Jarre-Teichmann	1995													Analysis of the seasonal energy budget and significance of interspecific control mechanisms in the central Baltic Sea
O	Rudstam et al.	1994													Overview on evidence for and possible consequences of top-down control in the pelagic ecosystem of the Baltic Sea
MB	Wulff and Ulanowicz	1989													Descriptions of structural and functional relationships on a system level in two strongly exploited marine systems, the Baltic Sea and the Chesapeake Bay
MB	Elmgren	1984													Overview on main biological energy flows

## Preparation of inputs for the basic EwE model

The following sections provide details on basic parameters used to set up the 1994 Ecopath model. All input data for (1) biomass (B), (2) production/biomass (P/B) ratios, (3) consumption/biomass (Q/B) ratios, (4) diet composition (DC) and (5) ecotrophic efficiency (EE) are in Tables S7-S9 of the *Results* section. Assessment of input data robustness and reliability with respect to general ecological and fishery principles is performed through the application of pre-balancing (PREBAL) diagnostics (Link, 2010).

### Biomass (B)

Because biomass of a trophic group is far more ecosystem-specific than physiological parameters like production and consumption (see e.g. Lohbeck et al., 2015), realistic biomass values are therefore of paramount importance for a model aimed to closely represent flows of matter in a specific ecosystem. In the following it will be explained how biomass values for trophic groups were calculated. Wet or fresh weight was transformed into carbon by applying a conversion factor of 10:1. If not otherwise stated, values represent biomass in ICES subdivisions (SDs) 22 and 24.

Harbour porpoises: Wet weight (WW) in German coastal waters of the Baltic Sea was derived from indications by A. Gilles (pers. comm.) and Viquerat et al. (2014).

Seals: Seals are top predators in the WBS ecosystem and are represented by two species: harbour seal (*Phoca vitulina*, Phocidae) and grey seal (*Halichoerus grypus*, Phocidae). The wet weight biomass (WWB) was derived from a trophic model by Harvey et al. (2003) representing ICES SDs 25-29 and 32 from 1974 to 2000.

Seabirds: For the 50 seabird species occurring in the WBS ecosystem (Helsinki Commission<sup>3</sup>), WWB was derived from abundance estimates of 27 bird species (listed in Table S1) in different zones (EEZ, coastal and offshore zones of Schleswig-Holstein and Mecklenburg-Western Pomerania) of the German Baltic Sea (information made available by colleagues from ECOLAB, FTZ Büsum<sup>4</sup>). Estimates refer to the first decade of the 21st century. The number of individuals was multiplied by mean WW of species. Mean WW values were obtained through internet queries, mostly consulting Wikipedia<sup>5</sup>. For each species, total weight was divided by total area size (in m<sup>2</sup>); information on area size (in km<sup>2</sup>) was kindly made available by colleagues from ECOLAB, FTZ Büsum.

Adult cod: Cod >35 cm represents adults of the western Baltic cod stock. The cut-off length of 35 cm between adults and juveniles represents the official EU minimum landing size of

<sup>3</sup> [www.helcom.fi](http://www.helcom.fi)

<sup>4</sup> [www.ftz.uni-kiel.de/de/forschungsabteilungen/ecolab-oekologie-mariner-tiere](http://www.ftz.uni-kiel.de/de/forschungsabteilungen/ecolab-oekologie-mariner-tiere)

<sup>5</sup> <https://de.wikipedia.org/>



cod in the Baltic Sea after 2014<sup>6</sup> and roughly matches the length at 50% maturity (Figure S1). Biomass was based on data for the western Baltic cod stock from ICES (2020a,b). To obtain biomass in gWW m<sup>-2</sup>, 1994 spawning stock biomass (SSB) was divided by area size for ICES SDs 22-24 (44,746 km<sup>2</sup>) thus assuming homogeneous density (ICES, 2020a,b).

Juvenile cod: Cod ≤ 35 cm includes western Baltic cod juveniles. Biomass was calculated by the multi-stanza routine in Ecopath based on adult cod biomass. The multi-stanza routine result was adapted to an external value of biomass for juvenile cod. The external value was calculated as the difference between total stock biomass (TSB) and SSB (ICES, 2020a,b) in 1994. To obtain biomass in gWW m<sup>-2</sup>, the difference was then divided by area size for SDs 22-24 (44,746 km<sup>2</sup>).

Flatfish: Five commercially important flatfish species included brill (*Scophthalmus rhombus*, Scophthalmidae), dab (*Limanda limanda*, Pleuronectidae), flounder (*Platichthys flesus*, Pleuronectidae), plaice (*Pleuronectes platessa*, Pleuronectidae) and turbot (*Scophthalmus maximus*, Scophthalmidae). The total stock biomass in SDs 22 and 24 represents the sum of individual biomasses in 1994 of the five species. Wet weight biomass was calculated from DATRAS BITS CPUE data<sup>7</sup> separately for dab, flounder, plaice, turbot and brill. The biomass was quantified starting from the number of individuals per length class from CPUE, which was multiplied by weight per individual per length class obtained by species-specific length-weight relationships (LWRs). To obtain biomass in gWW m<sup>-2</sup>, the total biomass was divided by area size in SDs 22 and 24 (42,228 km<sup>2</sup>).

Other demersal fish: Original biomass for this group was calculated from DATRAS BITS CPUE data for demersal fish excluding cod and the five species in the flatfish compartment (percentage of flatfish biomass was about 13.5% compared to total group biomass). The number of individuals per length class from CPUE was multiplied by weight per individual per length class obtained from species-specific LWRs. WW was then converted into carbon weight. The biomass value of 0.0436 gC m<sup>-2</sup> obtained from DATRAS BITS represented only a restricted number of demersal fish occurring in the western Baltic Sea. Therefore, it was suspected to be too low to satisfy the energetic requirements of predators (including extraction by the fishery). A minimum value for biomass necessary to satisfy extraction by predators and the fishery was obtained during the balancing.

Herring: Biomass (expressed as gWW m<sup>-2</sup>) in 1994 for the stock of western Baltic spring-spawning (WBSS) herring was obtained by dividing TSB (ICES, 2019a,b, 2020c,d) by the size of SDs 20-24 (102,288 km<sup>2</sup>). The report of the working group for herring assessment

<sup>6</sup> [www.ices.dk/sites/pub/Publication%20Reports/Stock%20Annexes/2019/cod.27.22-24\\_SA.pdf](http://www.ices.dk/sites/pub/Publication%20Reports/Stock%20Annexes/2019/cod.27.22-24_SA.pdf)

<sup>7</sup> [www.ices.dk/data/data-portals/Pages/DATRAS.aspx](http://www.ices.dk/data/data-portals/Pages/DATRAS.aspx)

(ICES, 2020c) provides indications about TSB of herring in SDs 20-24. Biomass in SDs 22 and 24 was estimated assuming the same average annual density in all five SDs.

Sprat: ICES (2020e) treats sprat in the Baltic Sea as a single stock; an estimate of biomass in SDs 22 and 24 was obtained as follows. First, the ratio between catch and SSB (i.e. the exploitation rate, ExplR) in SDs 22 and 24 was assumed to be the same as in SDs 22-32, an area for which complete data on catches and biomasses were available. Such ratio was calculated for SDs 22-32 during the entire time series (1994-2019; ICES, 2019c, 2020e). The average of the ratios between catches in SDs 22 and 24 vs. catches in SDs 22-32 from 2001 onward (ICES, 2019c) was used to estimate catches in SDs 22 and 24 during 1994-2000, starting from catches in the total area (SDs 22-32). The ratio between catches in SDs 22 and 24 vs. ExplR allowed estimating SSB from 1994 to 2019 in SDs 22 and 24.

Other pelagic fish: Original biomass was calculated from DATRAS BITS CPUE. For species in the DATRAS database identified to be pelagic (Table S1), the number of individuals per length class from CPUE data was multiplied by weight per individual per length class obtained from species-specific LWRs. The resulting biomass ( $0.00349 \text{ gC m}^{-2}$ ) was too low to satisfy requirements of predators and extraction by the fishery. This value was suspected to underestimate strongly the real biomass of pelagic fish since DATRAS BITS surveys are made with bottom trawls targeting demersal species. An assessment of the minimum biomass needed to keep  $EE < 1$  was obtained during the EwE balancing process.

Pelagic macrofauna: Biomass is an average of biomasses in Harvey et al. (2003,  $0.133 \text{ gC m}^{-2}$ ) and Jarre-Teichmann (1995,  $0.27 \text{ gC m}^{-2}$ ) for Baltic Proper.

Benthic macrofauna: Fresh weight was read off an unpublished benthic macrofauna figure for ICES SDs 22 and 24 provided by M. Zettler, IOW Warnemünde.

Benthic meiofauna: M. Zettler from IOW Warnemünde (pers. comm.) provided an estimate of fresh weight.

Zooplankton: Total biomass represents the sum of biomasses for macro- (mainly mysids), meso- and microzooplankton. Biomass of each group corresponds to the mean of a range from a series of published trophic models (see Table S2 for an overview). For conversion of WW into carbon a factor of  $1 \text{ gC} = 12.07 \text{ gWW}$  and  $1 \text{ gWW} = 0.0828 \text{ gC}$  was applied.

Bacteria/microorganisms: B represents the average of a range ( $0.21\text{-}0.42 \text{ gC m}^{-2}$ ) from a series of published trophic models (see Table S2 for an overview).

Phytoplankton: Biomass represents the average of a range ( $1.01\text{-}3.312 \text{ gC m}^{-2}$ ) from a series of published models (see Table S2 for an overview). For conversion of WW into carbon a factor of  $1 \text{ gC} = 12.07 \text{ gWW}$  and  $1 \text{ gWW} = 0.0828 \text{ gC}$  was applied.

Benthic producers: Biomass represents a rough estimate of a range ( $0.02\text{-}0.0214 \text{ gC m}^{-2}$ ) from several published models (see Table S2 for an overview), which mainly describe the

Baltic Proper (Jarre-Teichmann, 1995; Sandberg et al., 2000) and the entire Baltic Sea (Wulff and Ulanowicz, 1989 with data adopted from Elmgren, 1984). A very high value of  $65.74 \text{ gC m}^{-2}$ , based on estimates of macroalgae production for the whole Baltic Sea, was also taken into account (Bergström, 2012).

Detritus/DOM: Biomass corresponds to the average of a range (680-885  $\text{gC m}^{-2}$ ) from Sandberg et al. (2000) for Baltic Proper and from Wulff and Ulanowicz (1989) for the whole Baltic Sea.

## Production/biomass ratios (P/B)

Production refers to the growth of biomass by a group over the period considered, entered as P/B per year ( $\text{y}^{-1}$ ) and transformed into absolute flows ( $\text{gC m}^{-2} \text{y}^{-1}$ ) by Ecopath. Under the condition assumed for the construction of mass-balance models, the total mortality  $Z$  is equal to production over biomass (Allen, 1971). Hence,  $Z$  was used for trophic groups in case no P/B ratio was available. Below, the origin of P/B inputs is described individually for each trophic group.

Harbour porpoises: P/B was adopted from Table 3 (i.e.  $Z$  for harbour porpoises) in Araújo and Bundy (2011).

Seals: P/B corresponds to the lower limit (0.095-0.1  $\text{y}^{-1}$ ) from Harvey et al. (2003) and Mackinson and Daskalov (2007).

Seabirds: P/B represents the mean of the range (0.3-7.027  $\text{y}^{-1}$ ) in Tomczak et al. (2009) for five coastal ecosystems in the southern and southeastern Baltic Sea.

Adult cod: For P/B, total mortality  $Z$  was used.  $Z$  (1.47  $\text{y}^{-1}$ ) is the sum of natural mortality  $M = 0.20 \text{ y}^{-1}$  (ICES, 2020a) and fishing mortality in 1994, calculated as  $F_{1994} = -\log_e(1 - C_{1994}/B_{1994}) = 1.27 \text{ y}^{-1}$ .  $B_{1994}$  stands for SSB and  $C_{1994}$  for total catches of adults, both values for SDs 22 and 24 in 1994 (ICES, 2020a,b).

Juvenile cod: For P/B, total mortality  $Z$  was used. Natural mortality  $M$  in 1994 for juvenile cod was  $0.44 \text{ y}^{-1}$  (ICES, 2020a) while fishing mortality  $F$  was calculated using the formula  $F_{1994} = -\log_e(1 - C_{1994}/B_{1994}) = 0.32$ ;  $B_{1994}$  and  $C_{1994}$  refer to the juvenile stock biomass and catch of juvenile cod in 1994 (ICES, 2020a,b). Therefore,  $Z = M + F = (0.44 + 0.32) \text{ y}^{-1} = 0.76 \text{ y}^{-1}$ .

Flatfish: P/B corresponds to the weighted mean of  $0.85 \text{ y}^{-1}$  for plaice and  $0.86 \text{ y}^{-1}$  for turbot and brill; P/B values were adopted from Table 3.3 in Mackinson and Daskalov (2007).

Other demersal fish: A P/B of  $0.64 \text{ y}^{-1}$  was calculated with data from Table 3.3 in Mackinson and Daskalov (2007).

Herring: P/B corresponds to the weighted mean of  $0.8 \text{ y}^{-1}$  for adult and  $1.31 \text{ y}^{-1}$  for juvenile herring; P/B values were adopted from Table 3.3 in Mackinson and Daskalov (2007).

Sprat: An average P/B value of  $1.5 \text{ y}^{-1}$  was calculated from data in Table 3.3 of Mackinson and Daskalov (2007) and compared to values in published trophic models of the Baltic Sea (an overview in Table S2). The new value is higher but still comparable with the P/B value of  $1.1 \text{ y}^{-1}$  calculated from data for P and B in Elmgren (1984) for the entire Baltic Sea.

Other pelagic fish: P/B was adopted from Jarre-Teichmann (1995) for “other pelagic fish”.

Pelagic macrofauna:  $P/B = [3.3, 7.5] \text{ y}^{-1}$ ; benthic macrofauna:  $P/B = [0.32, 1.41] \text{ y}^{-1}$ ; benthic meiofauna:  $P/B = [4.1, 6.17] \text{ y}^{-1}$ ; bacteria/microorganisms:  $P/B = [143, 149] \text{ y}^{-1}$ ; phytoplankton:  $P/B = [87.5, 151.6] \text{ y}^{-1}$ . All P/B values calculated for these trophic groups represent the average from two published models (Jarre-Teichmann, 1995; Harvey et al., 2003).

Zooplankton: P/B values for macro-, meso-, and microzooplankton (weighted for differing production) were used to calculate a mean P/B. P/B values for each group correspond to the mean of a range from published trophic models (see Table S2 for an overview).

Benthic producers: P/B was adopted from two predecessor models (Wulff and Ulanowicz, 1989; Jarre-Teichmann, 1995).

## Consumption/biomass ratios (Q/B)

Consumption is the intake of food by a trophic group over the time period considered. In Ecopath, it is entered as the ratio of consumption over biomass (Q/B) per year. Absolute consumption computed by Ecopath is then a flow expressed in  $\text{gC m}^{-2} \text{ y}^{-1}$ . Below, origin of Q/B inputs are described individually for each trophic group.

Harbour porpoises: Q/B is based on information in Andreassen et al. (2017).

Seals: Q/B is the mean for grey seal and harbour seal. Q/B for both species were calculated based on individual weight and daily food intake obtained from the German Oceanographic Museum<sup>8</sup> and Wikipedia<sup>9</sup>. The Q/B ( $\text{y}^{-1}$ ) employed here is at the upper limit of food intake since maximum weight and maximum food intake were used for the calculation.

Seabirds: Q/B is the mean of a range  $[5, 14.41] \text{ y}^{-1}$  in Tomczak et al. (2009) for five coastal ecosystems in the southern and southeastern Baltic Sea.

Q/B values for all fish groups except for other pelagic fish and sprat were retrieved from Table 3.3 in Mackinson and Daskalov (2007). An updated Q/B value of  $2.7 \text{ y}^{-1}$  for juvenile

<sup>8</sup> [www.deutsches-meeresmuseum.de/wissenschaft/infothek/artensteckbriefe](http://www.deutsches-meeresmuseum.de/wissenschaft/infothek/artensteckbriefe)

<sup>9</sup> [www.wikipedia.org](http://www.wikipedia.org)

cod ( $\leq 35$  cm) was calculated by the multi-stanza routine (Figure S1) of EwE using both total mortality  $Z = 1.47 \text{ y}^{-1}$  and  $Q/B = 1.5 \text{ y}^{-1}$  (see Table A.9 in Funk, 2017) of adult cod and  $P/B$  of juvenile cod. The value calculated by the routine was compared for reliability assessment with  $Q/B$  ratio of juvenile cod from Mackinson and Daskalov (2007). The ratio computed by the multi-stanza routine was finally compared with original  $Q/B$  values for juvenile cod.  $Q/B$  for juvenile cod is thus a trade-off between literature values and multi-stanza routine logic.  $Q/B$  value for flatfish is the weighted (by consumption) mean of  $3.68 \text{ y}^{-1}$  for dab,  $3.2 \text{ y}^{-1}$  for flounder,  $2.78 \text{ y}^{-1}$  for plaice and  $2.2 \text{ y}^{-1}$  for turbot.  $Q/B$  for herring is the weighted (by consumption) mean of  $4.34 \text{ y}^{-1}$  for adult and  $5.63 \text{ y}^{-1}$  for juvenile herring.

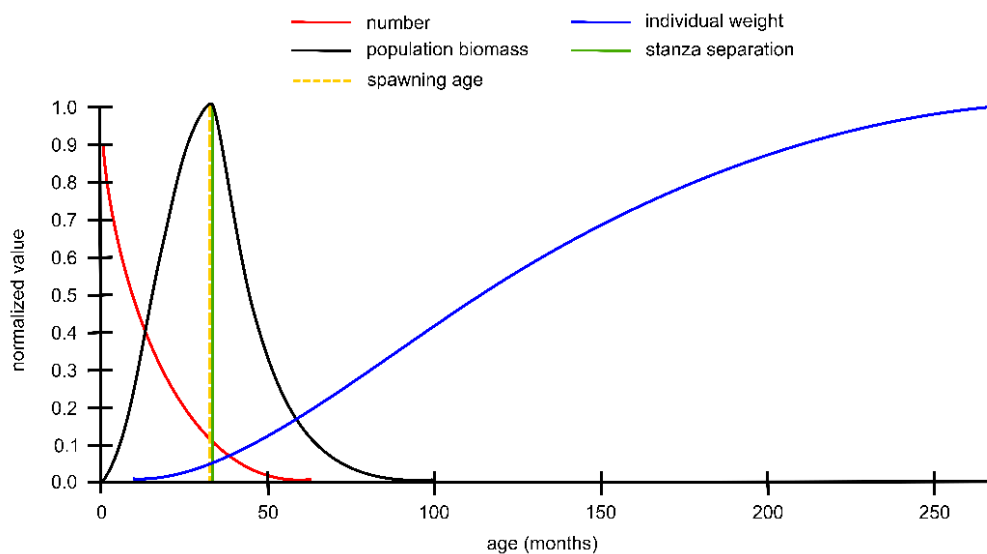


Figure S1. Multi-stanza representation of cod, with population split into juvenile and adult individuals. Parameters used:  $K = 0.15$  (annual rate at which the asymptotic length/weight is approached, from the von Bertalanffy growth function - VBGF), recruitment power = 1 (default value; recruitment power expresses the degree of density dependence in juvenile survival for individuals outside the modelled area), and  $W_{\text{maturity}}/W_{\text{inf}} = 0.0379$ . The ratio  $W_{\text{maturity}}/W_{\text{inf}}$  was calculated with  $L_m = 38$  cm (average value specific for cod in SDs 22 and 24; source: FishBase),  $L_{\text{inf}} = 110$  cm (source: FishBase), and a species-specific LWR with  $a = 0.00692$  and  $b = 3.08$  (source: FishBase; see also Froese et al., 2014).

Sprat: Consumption and biomass data for the Baltic Sea from Elmgren (1984) were used to calculate a  $Q/B$  of  $7.66 \text{ y}^{-1}$ .

Other pelagic fish: A  $Q/B$  of  $2.85 \text{ y}^{-1}$  was adopted from Jarre-Teichmann (1995).

Ratios for pelagic macrofauna [ $10.6, 25$ ]  $\text{y}^{-1}$ , benthic macrofauna [ $9.5, 13$ ]  $\text{y}^{-1}$  and benthic meiofauna [ $31.17, 33.9$ ]  $\text{y}^{-1}$  represent the average of values from two published models (Jarre-Teichmann, 1995; Harvey et al., 2003).

Zooplankton: The  $Q/B$  represents the average for macro-, meso- and microzooplankton (weighted for differing consumptions). The  $Q/B$  value for each group corresponds to the mean of a range from published trophic models for the Baltic Sea (see Table S2 for details).

Bacteria/microorganisms: The Q/B represents the average from two published models (355  $y^{-1}$  in Jarre-Teichmann, 1995 and 248  $y^{-1}$  in Harvey et al., 2003 for Baltic Proper).

### Non-assimilated part of the food (NA)

To quantify correctly flows of matter within the WBS ecosystem, an estimate of the fraction of the food that is not assimilated by a group is needed as input. NA is directed towards the detritus pool. Table S3 below shows for each trophic group the fraction of food ingested per year ( $y^{-1}$ ) that is not assimilated.

Table S3. Fraction of food ingested per year by trophic group that is not assimilated.

Group name	Unassimilated consumption	Sources
Harbour porpoises	0.150	Used same as for seals
Seals	0.150	Harvey et al. (2003) for ICES SDs 25-29 and 32, for years 1974-2000
Seabirds	0.200	Default
Adult cod	0.185	Mean of indications in Jarre-Teichmann (1995) and Harvey et al. (2003)
Juvenile cod	0.185	Mean of indications in Jarre-Teichmann (1995) and Harvey et al. (2003)
Flatfish	0.185	Used same as for cod
Other demersal fish	0.175	Mean of indications in Jarre-Teichmann (1995) and Sandberg et al. (2000, adopted from Elmgren, 1984 and Wulff and Ulanowicz, 1989)
Herring	0.230	Mean of indications in Jarre-Teichmann (1995) and Harvey et al. (2003)
Sprat	0.230	Mean of indications in Jarre-Teichmann (1995) and Harvey et al. (2003)
Other pelagic fish	0.175	Mean of indications in Jarre-Teichmann (1995) and Sandberg et al. (2000, adopted from Elmgren, 1984 and Wulff and Ulanowicz, 1989)
Pelagic macrofauna	0.195	Mean of indications in Jarre-Teichmann (1995) and Harvey et al. (2003)
Benthic macrofauna	0.465	Mean of indications in Jarre-Teichmann (1995), Sandberg et al. (2000) and Harvey et al. (2003)
Benthic meiofauna	0.350	Mean of indications in Jarre-Teichmann (1995), Sandberg et al. (2000) and Harvey et al. (2003)
Zooplankton	0.300	Mean of indications in Jarre-Teichmann (1995), Sandberg et al. (2000) and Harvey et al. (2003)
Bacteria and microorganisms	0.100	Mean of indications in Sandberg et al. (2000) and Harvey et al. (2003)

## Diet composition (DC)

In trophic networks, the feeding interactions link together the different groups. They must be entered for all groups except for primary producers and detritus. Diet composition (DC) is expressed in percentages of volume or weight and should sum up to 1 for each trophic group. This section describes origin of information of feeding preferences for all consumers in the WBS model.

Harbour porpoises: Information on DC in the western Baltic Sea was retrieved from Tables 6 and 8 in Andreassen et al. (2017).

Seals: DC for this group was adapted from data read off Table 13 and Figures 20-21 in Gilles et al. (2008). Origin of data is mainly from North Sea individuals. Diet info of grey seals in the central Baltic Sea (Lundström et al., 2007) was also considered.

Seabirds: DC was inferred from quantitative, semi-quantitative, and qualitative information on food and feeding of seabirds in the Baltic Sea (Mendel et al., 2008) and weighted by abundance of the species in the study area.

Adult cod: DC for the WBS was retrieved from Appendix Tables in Funk (2017). Prey groups were adapted to WBS model groups.

Juvenile cod: DC is based on data in Zalachowski (1985) for the southern Baltic Sea from 1977 to 1981 and published models from the 1980s and 1990s for Baltic Proper and eastern Baltic Sea (see Table S2). Data from Funk (2017) were not used, since “fish” food was not specified. Values from both sources are comparable for zooplankton and macrobenthos as food items.

Flatfish: DC for flounder, dab, plaice, turbot and brill were retrieved from the Table 3.4 in Mackinson and Daskalov (2007) and weighted (by consumption) to calculate the mean.

Other demersal fish: DC was quantified using feeding preferences of (other) demersal fish from published models (Jarre-Teichmann, 1995; Sandberg et al., 2000; Sandberg, 2007).

Herring: DC was characterized using adult and juvenile herring data from published models (Elmgren, 1984; Rudstam, 1994; Jarre-Teichmann, 1995; Harvey et al., 2003; Sandberg, 2007).

Sprat: DC was defined using feeding preferences of sprat from published models (Elmgren, 1984; Rudstam, 1994; Jarre-Teichmann, 1995; Harvey et al., 2003; Sandberg, 2007).

Other pelagic fish: DC was adapted from “other pelagic fish” data in Sandberg et al. (2000, 2007).

Pelagic macrofauna: DC was composed from data on pelagic macrofauna feeding in Jarre-Teichmann (1995) and Harvey et al. (2003). An assumed 5% for cannibalism was included (based on personal observation by S. Opitz: e.g. *Cyanea sp.* feeding on *Aurelia aurita*).

DCs for benthic macrofauna, benthic meiofauna and bacteria/microorganisms were defined based on feeding habits of these groups as shown in published models (Jarre-Teichmann, 1995; Sandberg et al., 2000; Harvey et al., 2003; Sandberg, 2007).

Zooplankton: DC was quantified using data on feeding preferences of macro-, meso-, and microzooplankton from published models (Jarre-Teichmann, 1995; Sandberg et al., 2000; Harvey et al., 2003; Sandberg, 2007). DC of Zooplankton was weighted for consumption of components.



## PREBAL diagnostics

PREBAL diagnostics were introduced by Link (2010) to verify that network models are built using sound input data that respond to general ecological and fishery principles (e.g. the trophic chain decline of compartments' biomass due to thermodynamic constraints). These diagnostics must be applied before balancing of the network models and prior to executing dynamic applications, with the objective of increasing the robustness and reliability of the outcomes generated. Diagnostics provide guidelines grouped in five criteria. First, the assessment of biomasses across taxa and trophic levels. Second, the analysis of biomass ratios between trophic groups. Third, the evaluation of vital rates' changes displayed by taxa along the trophic chain. Fourth, the inspection of vital rates' ratios and their trends with respect to trophic levels. Fifth, the quantification of total production and removal as a function of trophic groups and across the trophic chain. Model verification according to the guidelines provided by PREBAL diagnostics allows a detailed model review. It may help finding potential pitfalls and weaknesses in model design and assembly. Heymans et al. (2016) recommend using PREBAL when constructing EwE models that target ecosystem-based management.

In this study, PREBAL diagnostics were applied to check and visually inspect biomasses, vital rates, consumption, production and respiration across taxa and trophic levels for the 1994 Ecopath network model (Figure S2). This step is needed because the main objective of the present work was comparing the performance of alternative fisheries management scenarios. In particular, the goal was quantifying the benefits of ecosystem-based fisheries management (EBFM) vs. business as usual (BAU). Null hypothesis was that ecosystem-based fisheries management (EBFM) represents the best solution to (1) rebuild heavily exploited fish stocks, (2) maintain long-term sustainable yields of fishery, (3) preserve the populations of charismatic top predators, (4) increase the resilience to warming, and (5) intensify carbon sequestration.

### **Criterion #1 – Biomasses across taxa and trophic levels**

The biomass of trophic compartments spans over about six orders of magnitude, from the seals ( $0.00005 \text{ gC m}^{-2}$ ; trophic level 4.413) to the benthic macrofauna ( $41 \text{ gC m}^{-2}$ ; trophic level 2.013). These changes in biomass are realistic; they fall within the limits of 5-7 orders of magnitude (see Link, 2010) and reflect dissipations along the trophic chain. The linear slope of the relationship ( $\log_{10}$  scale) is about 20% (Figure S2a) and represents a sound value when fishes feeding on zooplankton are at trophic levels 3-4 (Sommer et al., 2018). Overall, no major deviations were identified compared to the linear model fitted to describe biomass decline as a function of trophic levels. Benthic macrofauna standing stock exceeds this trend line, and the relatively slow turnover rate of species composing this trophic group

(e.g. *Mytilus edulis*; Franz et al., 2019) may justify such a condition. Seals' standing stock lies instead below the trend line, a consequence of the fact that highest abundance of seals occurs in the Kattegat (HELCOM, 2018). Grey seal is the main species of the corresponding group in our model and attained historical minimum population size in the Baltic during the 1980s; then, it started increasing from early 1990s to mid-2000s (Harding et al., 2007).

### **Criterion #2 – Biomass ratios**

The general expectation is that the biomass of prey is larger than that of predators (Link, 2010). The same principle should hold for comparisons among trophic levels although high turnover rates of planktonic species may lead to less striking biomass differences (Sommer et al., 2018). The following ratios were calculated: (1) mesozooplankton/phytoplankton = 0.323; (2) pelagic fish/mesozooplankton = 0.695; (3) marine mammals/pelagic fish = 0.014; (4) demersal fish/benthic macroinvertebrates = 0.004; (5) pelagic and demersal fish/total macroinvertebrates = 0.015. All ratios were below 1, which stands for biomasses of higher trophic level consumers being less than biomasses of their resources. Ratios with phytoplankton and mesozooplankton at denominator were larger than those involving non-planktonic organisms. Such finding is not surprising because of the high phytoplankton and mesozooplankton productivity compared to low standing stock biomasses (see Table S7). Macroinvertebrates are a relevant food source for the western Baltic Sea, and the declines in the biomasses of organisms along the grazing chain "phytoplankton – mesozooplankton – forage fish – carnivorous fish" are likely to result in a shift towards detritus-based trophic chains. For instance, during the last decades of excessive fish extraction, the stock biomass of herring significantly decreased, forcing the western Baltic cod to shift parts of its dietary needs towards feeding of the common shore crab *Carcinus maneas* (Funk et al., 2021).

### **Criterion #3 – Vital rates across taxa and trophic levels**

Consumption, production and respiration rates are presented as ratios to the biomass (Q/B, P/B and R/B) and expressed as a function of trophic levels (Figure S2b). These ratios exhibit non-monotonic declining trends with increasing trophic levels (Figure S2b) due to presence of homeotherms towards the top of the trophic chain (compartments indicated by numbers 15–17 on the x-axis of Figure S2b: harbour porpoises, seabirds, and seals). Trends found for vital rates match the expectations summarized by Link (2010).

### **Criterion #4 – Biomass and vital rate ratios**

First, the trend shown by ratios between compartments' biomass and total ecosystem net primary production, NPP ( $502.518 \text{ gC m}^{-2} \text{ y}^{-1}$ ) was assessed (Figure S2c, light blue bars).

Second, changes in the ratios between compartments' production and NPP were inspected (Figure S2c, pink bars). Values were plotted vs. trophic levels in descending order, from top predators to primary producers. Biomass/NPP ratios span over seven orders of magnitude (the smallest value was found for seals while the largest one was obtained for benthic macrofauna) and are lower than 1. Production/NPP ratio attains its minimum for seals while phytoplankton is the upper bound; all P/NPP ratios are well below 1. For the sake of clarity, B/NPP and P/NPP ratios are visualized on a  $\log_{10}$  scale after multiplication of raw values by  $10^8$  and  $10^9$ , respectively. Production/NPP ratios show that extremely low amounts of total net productivity are used by species at the top of the trophic chain in the western Baltic Sea. Low P/B ratios found for the high trophic level consumers suggest that small size of fish stocks and low biomass of marine mammals cannot be attributed to the lack of primary production. Inefficient transfers of energy due to the excessive extraction of forage fish such as herring (ICES, 2020c,d) are the most plausible cause to explain the low valorization of primary productivity. The western Baltic Sea ecosystem shows in fact negligible amounts of primary productivity converted into catches (3%; see Table S11) compared to the 24-35% efficiency exhibited by other coastal, marine ecosystems (Pauly and Christensen, 1995). Comparisons between the average P/B ratios of specific trophic groups confirm that species at the bottom of the trophic chain are characterized by larger vital rates than those found for species towards the top of the trophic hierarchy: (1) mesozooplankton/phytoplankton = 0.639; (2) pelagic fish/phytoplankton = 0.007; (3) pelagic fish/mesozooplankton = 0.012; (4) marine mammals/pelagic fish = 0.156; (5) demersal fish/benthic invertebrates = 0.250. Finally, production/consumption (P/Q) ratios were computed to verify that, for each trophic group, production does not exceed its consumption (Figure S2d). In agreement with the second principle of thermodynamics, P/Q ratios (i.e. gross efficiency) must be less than 1 for all trophic groups, a condition met for the WBS model. Lowest efficiencies were detected for marine mammals.

### **Criterion #5 – Total production and removals**

In ecosystem models, it is relevant to verify that the relative contribution of compartments to total consumption (Q) and sum of production (P) and respiration (R) declines with trophic levels. First, the trend of the scaled sum of (P + R) confirmed that initial estimates of vital rates are reasonable (Figure S2e, pink bars). Second, scaled Q also decreases in response to increasing trophic levels (Figure S2e, light blue bars). Pelagic macrofauna, sprat and seals show lower relative contributions than expected (Figure S2e). Deviations can be caused by (1) scarcity of data about jellyfish, which might have affected the quantification of pelagic macrofauna biomass, (2) spatial distribution of sprat, with SSB in SDs 22 and 24 amounting to 4% of total SSB in the entire Baltic Sea, and (3) highest abundance of seals found in the Kattegat rather than in the WBS (HELCOM, 2018).

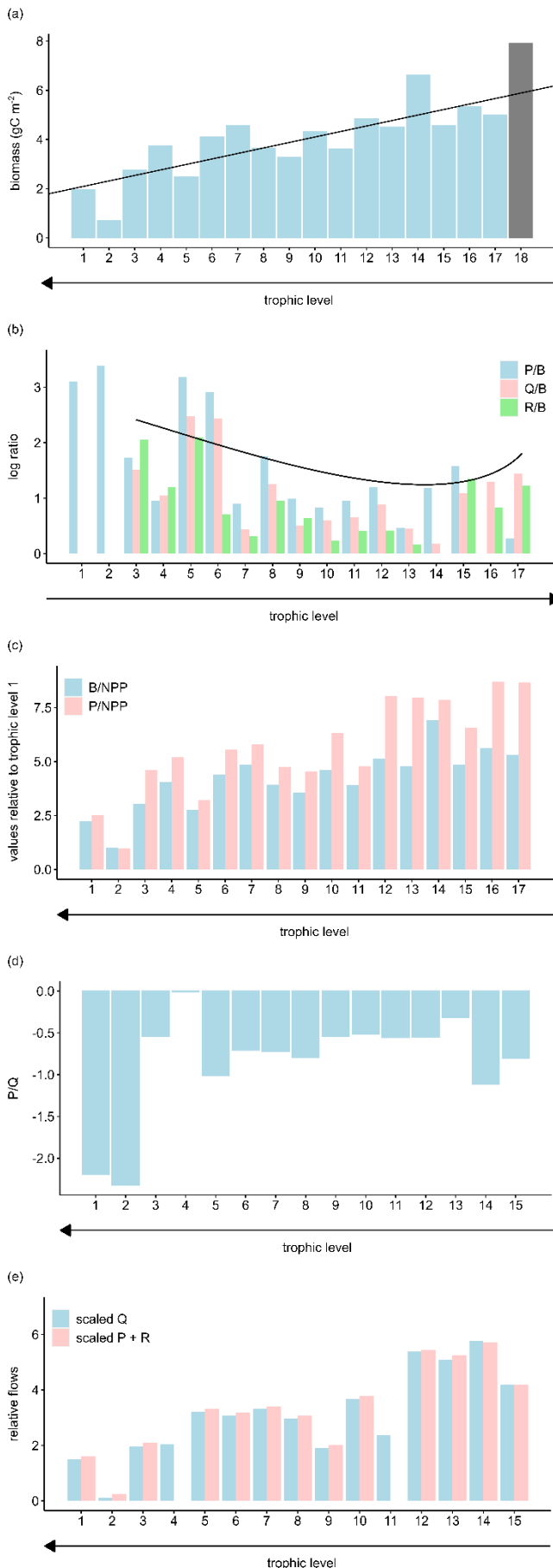


Figure S2. PREBAL diagnostics carried out before balancing the 1994 Ecopath model. They show trends of (a) biomasses; (b) vital rates; (c) vital rates compared to net primary production; (d) production to consumption ratio; (e) consumption and “production plus respiration”, scaled to total values in the full ecosystem. All charts visualize the trends with respect to compartments’ trophic levels (Table S7) in descending order, except for (b). In all charts, the y-axis illustrates  $\log_{10}$ -transformed values, eventually rescaled to avoid negative numbers (e.g. seals have the smallest biomass in the model,  $5 \times 10^{-5} \text{ gC m}^{-2}$ , and all biomass values were then multiplied by  $10^5$  to have numbers larger than 0 after  $\log_{10}$ -transformation). No rescaling was applied for log-10 values in chart (d) to get an image analogous to the original generated using the EwE software tool. These charts correspond to criteria 1 (a), 3 (b), 4 (c, d) and 5 (e) as described by Link (2010).

Correspondence between numbers on the x-axis and trophic groups is the following: 1-harbour porpoises; 2-seals; 3-seabirds; 4-adult cod; 5-other pelagic fish; 6-sprat; 7-herring; 8-other demersal fish; 9-flatfish; 10-pelagic macrofauna; 11-juvenile cod; 12-zooplankton; 13-bacteria/microorganisms; 14-benthic macrofauna; 15-benthic meiofauna; 16-phytoplankton; 17-benthic producers; 18-detritus.

Reversed order applies to chart (b): x-axis starts with 1-phytoplankton and ends with 17-harbour porpoises.

Abbreviations: Q – consumption; P – production; R – respiration; NPP – net primary production; B – biomass.

## Fishery

Main objective of this study was to analyse the impact of commercial fisheries on the WBS ecosystem and to explore improved fisheries management options. To assemble reliable model inputs of fishery extractions was therefore of paramount importance. Fishery data used for the base model representing the WBS in 1994 are in Tables S4-S5. Time series used as a benchmark for model fitting in response to fishery (1994-2019) are in Figure S3.

The “fishery” in our WBS models is divided into a pelagic and a demersal fleet, recreational fishery and bycatch/IUU (illegal, unreported and unregulated) fishery. Origin of inputs for landings, bycatch and discards are described below (see also section *Data Sources* above). If not stated otherwise, fishery data represent values for ICES SDs 22 and 24 during 1994. Original catch, landing and discard values were transformed into  $\text{gWW m}^{-2} \text{y}^{-1}$  by dividing weight (in tons) by area size ( $42,228 \text{ km}^2$ ). Landings, bycatch and discard values in  $\text{gWW m}^{-2} \text{y}^{-1}$  were transformed into carbon by applying a conversion factor of 10:1.

### Pelagic fleet landings

Herring: Commercial landings of western Baltic spring-spawning herring were retrieved by consulting ICES (2019a,b, 2020c,d) documents, which allowed quantifying total amounts in SDs 22 and 24 from 1994 to 2019.

Sprat: Commercial landings in SDs 22 and 24 were available from 2001 onwards (ICES, 2019d). An average ratio of about 0.037 for commercial landings in SDs 22 and 24 vs. commercial landings in SDs 22-32 during years 2001-2018 was calculated based on time series of landings available separately for each subdivision, from 22 to 32 (ICES, 2019c,d). To obtain an estimate of commercial landings for years 1994-2000 in SDs 22 and 24 the ratio of 0.037 was applied to the total landings in SDs 22-32 (ICES, 2019c,d). Finally, the time series was completed with 2019 data (ICES, 2020e).

Other pelagic fish: Landings of “other fish” were read off Appendix Tables in Rossing et al. (2010) for Germany and Denmark. The mean for both countries for years 2003 to 2007 was used to calculate commercial landings. The total amount was divided into two equal parts for “other pelagic fish” and “other demersal fish”.

### Demersal fleet landings

Adult cod: Commercial landings for the western Baltic cod stock in SDs 22-24, during 1994 and along the entire time series up to 2019, were obtained by consulting ICES (2020a,b) documents. They were quantified by multiplying landings in numbers by the mean annual weight-at-age (Table 2.3.15 at page 153 of ICES, 2020a) and by the proportion of mature

individuals at age. The commercial landings were transformed into  $\text{gWW m}^{-2} \text{y}^{-1}$  by dividing total weight (in tons) by area size ( $44,746 \text{ km}^2$ ). The density of landings in SDs 22-24 was estimated assuming homogeneous distribution. Catches of juvenile cod were calculated as for adult cod, using ICES (2020a) data; the proportions of juveniles per age class were computed by subtracting the fractions of mature individuals per age class from 1.

Flatfish: Catches in SDs 22 and 24 were estimated assuming uniform distribution. ICES (2019e,f) documents provided data on commercial landings for flounder in SDs 22 and 24. Since separate values for SDs 24 and 25 were available only from 2000 onwards, the fraction of catches in SD 24 was expressed as function of total landings in SDs 24-25 by use of a generalized linear model (GLM). Total landings for dab and plaice in SDs 22 and 24 were obtained from ICES (2019d,g). Commercial landings for turbot and brill in SDs 22 and 24 were retrieved from ICES (2019d). Landings of the five species were added up to represent joint flatfish landings. Finally, these landings were updated for 2019 using the relative change observed for plaice from 2018 to 2019 (ICES, 2020f).

Other demersal fish: Landings of “other fish” were read off Appendix Tables in Rossing et al. (2010) for Germany and Denmark. Mean for both countries for years 2003 to 2007 was used to calculate the landings. The total amount was divided into two equal parts for other pelagic fish and other demersal fish since Rossing et al. (2010) provided only very limited information on species composition and their respective shares in “others”. They merely stated for Denmark that recreational catches of “others” included garfish (*Belone belone*), sea trout (*Salmo trutta trutta*), northern pike (*Esox lucius*) and a group of miscellaneous “finfishes”. Separate landing data for salmon were added to other demersal fish.

Benthic macrofauna: According to various authors (e.g. Kaiser et al., 2006; Queiros et al., 2006), the impact of bottom trawling on size of standing stock varies according to substrate and adaptation to natural disturbance level. According to findings of Dr. Thomas Brey (AWI, Bremerhaven, Germany, 2019) biomass does not change (in contrast to turnover rate) but no quantitative data are available to date. A very low estimate of 0.1% of annual production of benthic macrofauna was therefore used as model input, with the intention to show that there is an effect of bottom trawling on the demersal community.

### **Recreational fishery landings**

If not stated otherwise, catch values for recreational fishery used for the WBS model originate from Appendix Tables for Germany and Denmark in Rossing et al. (2010).

Adult cod: The amount of recreational fishery in SDs 22-24 was quantified subtracting commercial landings and discards from total catches (ICES, 2020a). These latter were determined by multiplying total catches in numbers by the mean annual weight-at-age and by the proportion of mature individuals at age. Such a value, expressed in tons, was then

transformed into  $\text{gWW m}^{-2} \text{ y}^{-1}$  through division by area size (44,746  $\text{km}^2$ ), resulting in a homogeneous density.

Flatfish: A GLM was constructed employing data from 1994 to 2007 (Rossing et al., 2010) to determine fractions of recreational catches as a function of catches in the period 2008-2019. Model results ranged from 3.7% in 1995 to 9.0% in 2009. These percentages were applied to catches of flatfish for estimates of recreational fishery after 2007.

Other demersal fish: Recreational fishery was calculated as 5% of catches in SDs 22 and 24. The value for “salmon” (in Rossing et al., 2010) was added in proportion to catches.

Herring: Average fraction of recreational fishery vs. total catches in Denmark and Germany during the years 1994-2007 was calculated using the data from Rossing et al. (2010). Such average ratio amounted to 2.7% and was applied to the entire time series of catches (1994-2019) to estimate the amounts of recreational fishery in SDs 22 and 24.

Sprat: Recreational fishery in SDs 22 and 24 is negligible both in Denmark and in Germany (Rossing et al., 2010; ICES, 2018) and was therefore set to 0 for the entire time series.

Other pelagic fish: 6% of “other pelagic fish” landings in SDs 22 and 24 may be attributed to this type of fishery (deduced from Rossing et al., 2010).

### **Bycatch/IUU fishery landings**

Most input values used for fish compartments in the WBS model originate from Appendix Tables for Germany and Denmark in Rossing et al. (2010); the values, expressed as  $\text{gWW m}^{-2} \text{ y}^{-1}$ , refer to SDs 22 and 24.

Harbour porpoises, seals and birds are caught as bycatch of the fishery in fixed nets/traps. To date, reliable quantitative information on bycatch of marine mammals and seabirds range from scarce to non-existent for the WBS. Therefore, information from nearby regions were also used to obtain preliminary estimates.

Harbour porpoises: Data from Table 2 in Scheidat et al. (2008) were used to estimate the percentage of harbour porpoise bycatch in the southwestern Baltic Sea. Because of the unavailability of proper time series, bycatch in fixed nets and traps was set to be constant. Excluding minimum and maximum values reported, bycatch amounted to about 4% of the population size; the value used in the model is 4.2%. van Beest et al. (2017) developed a spatially explicit individual-based model for inner Danish waters and found bycatch to be in the range of 1.2% to 5.02%, which corroborates the value used in our study.

Seals: The Finnish Game and Fisheries Research Institute (2013) estimated a bycatch rate of 7.7-8.4% of grey seal population size in the eastern Baltic Sea, while estimates of annual population growth rates for grey and harbour seals ranged from 3.5% to 9.4% for different

periods and locations. A study by Vanhatalo et al. (2014) suggests that >2000 seals are bycaught in the eastern Baltic, representing at least 90% of the total bycatch in the Baltic Sea. Based on these figures, we concluded that 10% of annual population production being bycaught in fixed nets and traps would be a conservative estimate for the WBS model.

Seabirds: According to various authors (Zydelis et al., 2009, 2013; Bellebaum et al., 2012), a rough estimate of 100,000-200,000 water birds are drowning annually in the North Sea and Baltic Sea, of which the great majority refers to the Baltic Sea. Derived from this information, a preliminary estimate of 0.25% of annual production was entered into the model to represent bycatch of seabirds in fixed nets.

Adult cod and juvenile cod: The contribution of bycatch/IUU fishery landings is embedded in the commercial landing data of western Baltic cod stock (i.e. included in the assessment since 1994; see ICES, 2020a,b).

Flatfish: The entire data series for 1994-2007 in Rossing et al. (2010) was used to construct a GLM expressing fractions of bycatch and IUU fishery as a function of catches. The function was applied to estimate the fractions of bycatch/IUU fishery with respect to catches after 2007. Fractions ranged from 5.40% in 1994 to 10.86% in 2016.

Other demersal fish: The assumed value of 27% corresponds to the average of all fish groups based on data in Rossing et al. (2010) in SDs 22 and 24.

Herring: Bycatch/IUU fishery landings for western Baltic spring-spawning herring are low (Oceana, 2012). Values of a recent ICES advice document (ICES, 2020d) were compared with figures of the period 1994-2007 (Rossing et al., 2010). Bycatch/IUU fishery along the entire time series (1994-2019) was estimated as 2% of catches.

Sprat: The average ratio between bycatch/IUU landings and catches for years 1994-2007 (Rossing et al., 2010) was used to estimate bycatch/IUU landings after 2007.

Other pelagic fish: The assumed value of 27% corresponds to the average of all fish groups based on data in Rossing et al. (2010) in SDs 22 and 24.



Table S4. Commercial pelagic and demersal fleet landings, recreational landings and bycatch/IUU landings (seals, birds and harbour porpoises in gillnets and entangling nets) in ICES SDs 22 and 24 during 1994; landings are expressed as gC m<sup>-2</sup> y<sup>-1</sup>. Values in *italics* were calculated based on figures in Rossing et al. (2010).

Group name	Pelagic fleet	Demersal fleet	Recreational fishery	bycatch/IUU fishery	Total extracted by fishery	Total extracted by fishery (%)
Seals				4.75E-07	4.75E-07	0.0001
Seabirds				0.0020	0.0020	0.5073
Harbour porpoises				3.74E-05	3.74E-05	0.0095
Adult cod		0.0360	0.0043		0.0403	10.2223
Juvenile cod		0.0156	0.0026		0.0182	4.6165
Flatfish		0.0192	<i>0.0010</i>	<i>0.0052</i>	0.0254	6.4428
Other demersal fish		<i>0.0309</i>	<i>0.0017</i>	<i>0.0083</i>	0.0409	10.3744
Herring	0.1529		<i>0.0041</i>	<i>0.0031</i>	0.1601	40.6100
Sprat	0.0252			<i>0.0068</i>	0.0320	8.1169
Other pelagic fish	<i>0.0300</i>		<i>0.0017</i>	<i>0.0081</i>	0.0398	10.0954
Pelagic macrofauna						
Benthic macrofauna		0.0355			0.0355	9.0047
Benthic meiofauna						
Zooplankton						
Bacteria/microorganisms						
Phytoplankton						
Benthic producers						
Detritus/DOM						
Total	0.2081	0.1372	0.0154	0.0335	0.3942	100.0000

### Pelagic fleet discards

All values for pelagic fleet discards were read off Appendix Tables in Rossing et al. (2010) for Germany and Denmark.

Herring and sprat: Discards of herring and sprat are considered negligible by ICES (2018, 2020d,e) in contrast to the estimates by Rossing et al. (2010) for Germany and Denmark. Discards in 1994 were quantified as percentages of catches from both countries with 4.90% for herring and 10.66% for sprat.

Other pelagic fish: Total amount for “other fish” in Rossing et al. (2010) was divided into two equal parts for other pelagic fish and other demersal fish. Hence, a value corresponding to 12% of catches of this group was entered for discards in 1994.

### Demersal fleet discards

Demersal fleet discards were either read off Appendix Tables in Rossing et al. (2010) for Germany and Denmark or derived from figures in official ICES documents.

Adult cod and juvenile cod: Discards during 1994 were estimated using data from ICES (2020a). For adult cod, they were quantified by multiplying the discards in numbers by the mean annual weight-at-age and by the proportion of mature individuals at age. In the case of juvenile cod, the same approach was adopted and proportions of juveniles per age class were computed by subtracting fractions of mature individuals per age class from 1. Finally, the values in  $\text{gC m}^{-2} \text{y}^{-1}$  were obtained by dividing amounts in tons by the area of SDs 22-24 ( $44,746 \text{ km}^2$ ), assuming homogeneous density.

Flatfish: Discards in SDs 22 and 24 were estimated using ICES data and assuming uniform distribution. Partial time series for discards of flounder were available for SDs 22 and 24 (ICES, 2019e,f). Discards in 1994 were determined applying GLMs constructed separately for each subdivision and expressing discards as a function of commercial landings. Discards of dab were available for Denmark and Germany during years 2012-2018 (ICES, 2019d). Discards in 1994 were then estimated by using the average ratio between discards and commercial landings for the period 2012-2018. Discards of plaice in SDs 22 and 24 were estimated from partial time series (ICES, 2019d,g). They were expressed as a function of commercial landings by use of GLMs, which were applied to quantify 1994 discards. Mean ratio of discards vs. commercial landings of turbot in the years 2012-2018 (ICES, 2019d) was used to determine discards during 1994 in SDs 22 and 24. Also for brill, mean ratio of discards vs. commercial landings in the years 2012-2018 (ICES, 2019d) was used to quantify the amount of discards in 1994 in SDs 22 and 24. Discards of all five species in 1994 were added up to determine the total for the compartment.

Other demersal fish: Total amount for "other fish" in Rossing et al. (2010) was divided into two equal components for other pelagic fish and other demersal fish. During 1994, discards corresponded to 12% of catches of this group.

Table S5. Fishery discards in ICES SDs 22 and 24 in 1994, expressed as  $\text{gC m}^{-2} \text{y}^{-1}$ . Values in *italics* were calculated based on figures in Rossing et al. (2010).

Group name	Pelagic fleet	Demersal fleet	Recreational fishery	Bycatch /IUU fishery	Total
Seals	no info	no info	no info	no info	
Seabirds	no info	no info	no info	no info	
Harbour porpoises	no info	no info	no info	no info	
Cod					
Adult cod		0.0012	no info	no info	0.0012
Juvenile cod		0.0024	no info	no info	0.0024
Flatfish		0.0141	no info	no info	0.0141
Other demersal fish		0.0038	no info	no info	0.0038
Herring	0.0075		no info	no info	0.0075
Sprat	0.0027		no info	no info	0.0027

Other pelagic fish	0.0037		no info	no info	0.0037
Pelagic macrofauna			no info	no info	
Benthic macrofauna			no info	no info	
Benthic meiofauna			no info	no info	
Zooplankton			no info	no info	
Bacteria/microorganisms			no info	no info	
Phytoplankton			no info	no info	
Benthic producers			no info	no info	
Detritus/DOM			no info	no info	
Total	0.0139	0.0215			0.0353

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## Data pedigree

Quality of model inputs is an important issue when judging the results of a modelling exercise. Qualitative ranking of model inputs – named here “data pedigree” – was prepared and is presented below. The first part of Table S6 provides ranking definitions applied in the second part to classify quality of model inputs.

Table S6. Quality pedigree of model inputs; LP = low precision, MP = medium precision, HP = high precision, sim. = similar, syst. = system.

### Part 1: Ranking definitions

Rank	Biomass	Rank	Production/Biomass (P/B)	Rank	Consumption/Biomass (Q/B)	Rank	Diet	Rank	Catch
1	Sampling locally, HP	1	Same species, sim. syst., HP	1	Same species, sim. syst., HP	1	Sampling, same syst., HP	1	Local data, HP
2	Sampling locally, MP	2	Sim. species, sim. syst., HP	2	Sim. species, sim. syst., HP	2	Sampling, sim. syst., HP	2	Local data, MP
3	Sampling locally, LP	3	Same species, sim. syst., LP	3	Same species, sim. syst., LP	3	Sampling, same syst., LP	3	Local data, LP
4		4	Sim. species, sim. syst., LP	4	Sim. species, sim. syst., LP	4	Sampling, sim. syst., LP	4	
5	From other model	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.
6	Estimated by Ecopath	6	Estimated by Ecopath	6	Estimated by Ecopath	6	Estimated by Ecopath	6	Estimated by Ecopath
7	Estimated by authors	7	Estimated by authors	7	Estimated by authors	7	Estimated by authors	7	Estimated by authors

## Part 2: Qualitative ranking of model parameter inputs

Group name		Biomass		P/B		Q/B		Diet		Catch
Seals	5	From other model	3	Same spec., sim. syst., LP	3	Same species, sim. syst., LP	2	Sampling, sim. syst., HP	7	Estimate*
Seabirds	3	Sampling locally, LP	4	Sim. species, sim. syst., LP	4	Sim. species, sim. syst., LP	3	Sampling, same syst., LP	7	Estimate*
Harbor porpoises	1	Sampling locally, HP	3	Same spec., sim. syst., LP	3	Same species, sim. syst., LP	1	Sampling, same syst., HP	3	Local data, LP
Adult cod	2	Sampling locally, MP	1	Same species, sim. syst., HP	1	Same species, sim. syst., HP	1	Sampling, same syst., HP	1	Local data, HP
Juvenile cod	2	Sampling locally, MP	6	Estimated by Ecopath	1	Same species, sim. syst., HP	1	Sampling, same syst., HP	1	Local data, HP
Flatfish	2	Sampling locally, MP	1	Same species, sim. syst., HP	1	Same species, sim. syst., HP	5	From other model for sim. syst.	1	Local data, HP
Other demersal fish	6	Estimated by Ecopath	4	Sim. species, sim. syst., LP	4	Sim. species, sim. syst., LP	5	From other model for sim. syst.	3	Local data, LP
Herring	2	Sampling locally, MP	1	Same species, sim. syst., HP	1	Same species, sim. syst., HP	5	From other model for sim. syst.	1	Local data, HP
Sprat	2	Sampling locally, MP	1	Same species, sim. syst., HP	1	Same species, sim. syst., HP	5	From other model for sim. syst.	1	Local data, HP
Other pelagic fish	6	Estimated by Ecopath	4	Sim. species, sim. syst., LP	4	Sim. species, sim. syst., LP	5	From other model for sim. syst.	3	Local data, LP
Pelagic macrofauna	5	From other model	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.		
Benthic macrofauna	1	Sampling locally, HP	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.	3	Local data, LP
Benthic meiofauna	1	Sampling locally, HP	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.		
Zooplankton	5	From other model	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.		
Bacteria/microorganisms	5	From other model	5	From other model for sim. syst.	5	From other model for sim. syst.	5	From other model for sim. syst.		
Phytoplankton	5	From other model	5	From other model for sim. syst.						
Benthic producers	5	From other model	5	From other model for sim. syst.						
Detritus/DOM	5	From other model	5	From other model for sim. syst.						

\* See text on bycatch/IUU fisheries above

## Balancing process

Flows based on original model inputs did not always balance, i.e. consumption by certain system elements exceeded production of their prey.

Imbalances of model inputs, originating from prey groups with EEs >1, were then balanced by applying the following strategies:

Increasing the biomass of a group by (1) immigration or (2) letting the EwE software tool estimate the minimum biomass needed to balance the extraction by predators (including fisheries) by entering a limiting EE of 0.99. In cases when (1) and (2) were not applicable, "import" of the respective food item by the predator was assumed. Furthermore, (3) small shifts in diet between food items served to eliminate "questionable" food requirements (i.e. derived from published models) or to smooth out initial inputs. Hereafter, the balancing process is described in detail for each trophic group. Start and end EE values for all trophic groups and shifts within the diet matrix are listed in the Input-Output Tables S7-S9.

Juvenile cod: Excess predation pressure by its main predator, the harbour porpoises (30% of their diet), was shifted to "import" under the assumption that if not enough juvenile cod is available within the boundaries of WBS, harbour porpoises obtain this prey elsewhere by roaming in a neighbouring system (e.g. Kattegat, Skagerrak). Consumption by herring was viewed as "questionable"; it was reduced from 1.7% to 0% and shifted to zooplankton, as this presumably referred to planktonic eggs and larvae of cod. For the same reason, the very small share (i.e. 0.25%) of juvenile cod in the diet of benthic macrofauna was set to 0% and shifted to benthic macrofauna (cannibalism).

Other demersal fish: Strategy (2) was applied since the initial biomass ( $0.043 \text{ gC m}^{-2}$ ) that was estimated from DATRAS BITS data was too low to satisfy the food requirements of all predators (including fishery). The original value based on catch data in DATRAS represents only 24% of the necessary biomass calculated by EwE (from less than 50% of fish species known to occur in the western Baltic Sea). Table S7 in the Results of this document shows that for other demersal fish the minimum biomass needed for satisfying food requirements of predators should be in the range of  $0.177 \text{ gC m}^{-2}$ .

Other pelagic fish: For the same reason as for other demersal fish, strategy (2) was applied to this group; start value from DATRAS BITS was  $0.003 \text{ gC m}^{-2}$ . This value, based on catch data in DATRAS, represents only 0.7% of the necessary biomass calculated by EwE. There are good arguments to view critically the biomass values obtained through DATRAS BITS, foremost, because the methods applied are not laid out for catching pelagic fish. Table S7 shows that for other pelagic fish the minimum biomass needed to satisfy food requirements of predators should be in the range of  $0.435 \text{ gC m}^{-2}$ .

Sprat: Predation by herring was considered questionable and may refer to larvae as part of macrozooplankton. It was set to 0% and shifted to zooplankton. Benthic macrofauna predation was reduced from 0.3 to 0.02% and shifted to benthic macrofauna (cannibalism).

Pelagic macrofauna: Predation by herring was set to 0 since *Mysis* sp. in our model forms part of macrozooplankton instead of pelagic macrofauna as in previously published models that were source of information on herring diet (see Table S2).

Benthic meiofauna: A 90% reduction of this group in the diet of benthic macrofauna decreased EE of benthic meiofauna to 0.824. The missing amount in the diet composition of benthic macrofauna was shifted to detritus/DOM.

Bacteria/microorganisms: Consumption by zooplankton was reduced from 22.8% to 15% and shifted to zooplankton (cannibalism) and detritus/DOM. Cannibalism within this group was reduced from 19.8% to 15% and shifted to detritus/DOM. Final EE of 0.926 is <1 but still very high.

Phytoplankton: A slight reduction of grazing pressure by benthic macrofauna, zooplankton and bacteria/microorganisms resulted in a small reduction of EE from 0.974 to 0.964. This value is still very high and should be in the range of 0.4 to 0.6. Standing stock biomass of 2.16 gC m<sup>-2</sup> for phytoplankton was adopted from models published for other parts of the Baltic Sea (Table S2). More recent values for years 1990 to 1997 (Thamm et al., 2004) for the WBS were even lower and in the range of 1.5 gC m<sup>-2</sup>.

Benthic producers: Consumption by benthic macrofauna was reduced from 1.25% to 1% and shifted to detritus/DOM. The resulting EE is 0.020.

## Dynamic modelling of different fishery management scenarios

This part of the study was performed in two steps.

The first step included the construction of a basic model and fitting of its outputs to time series of fishing yield (primary target) and stock size from external sources for the period 1994 to 2019. Time series used are depicted in Figure S3.

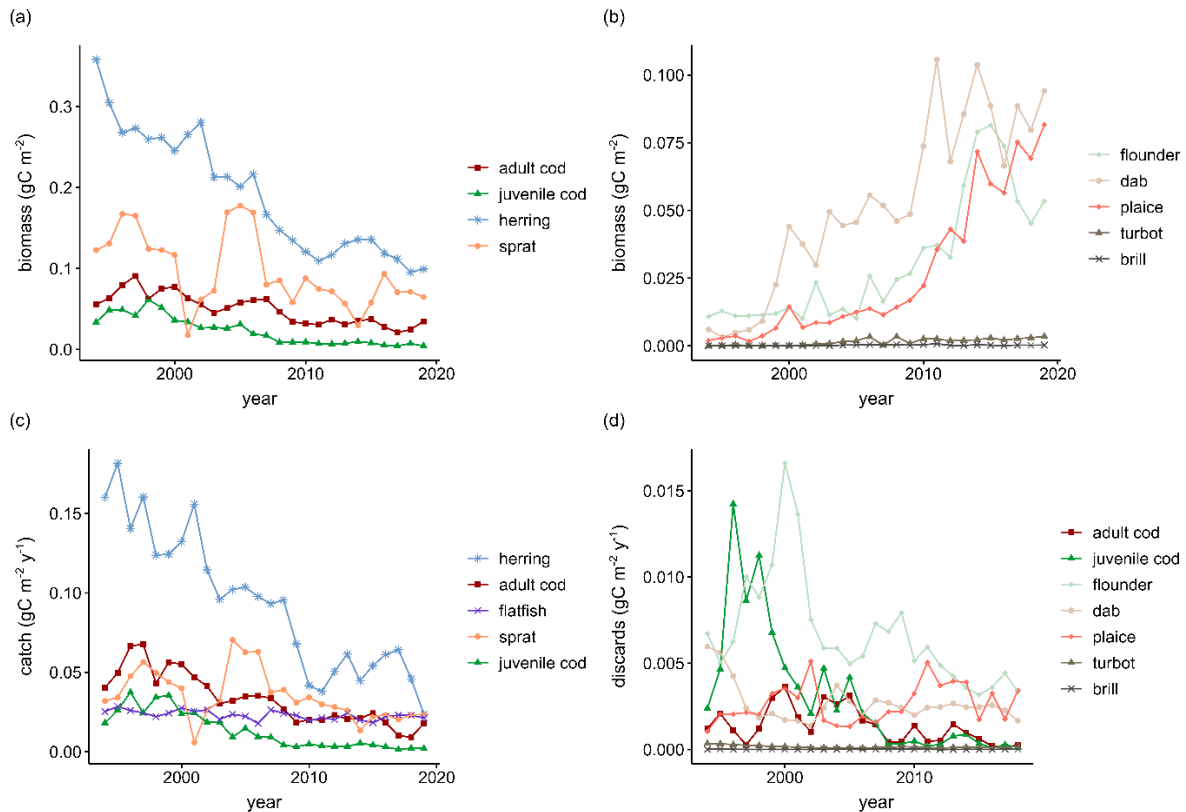


Figure S3. Trends of biomasses (gC m<sup>-2</sup>), catches and discards (gC m<sup>-2</sup> y<sup>-1</sup>) in SDs 22 and 24 from 1994 to 2019 for important commercial species in the WBS ecosystem. All data were extracted and deduced either from publications by ICES or from Rossing et al. (2010). Population sizes are shown as spawning stock biomass (SSB; adult cod and sprat), juvenile stock biomass (JSB; juvenile cod) or total stock biomass (TSB; herring and flatfish). They illustrate the trends for (a) cod, herring and sprat and (b) flatfishes. Plot (c) displays total catches (commercial landings, recreational fishery, and bycatch/IUU fishery) for the main commercial fish species and plot (d) displays discards of cod and five flatfish species. Note different scales of y-axis in plots.

Using the static 1994 Ecopath model as a starting point, model runs were executed after loading time series of biomass, catch and calculated fishing mortality (for each year  $i$ ,  $F_i = -\log_e[1 - C_i/B_i]$ ). Data illustrated the trends of important commercial fish stocks such as cod, herring, sprat and several flatfish species, lumped into a flatfish compartment as described in the section *Trophic groups represented in the model of the WBS ecosystem* (Figure S3 for biomass, catch and discards of important commercial fish stocks from 1994 to 2019). Purpose of this step was checking whether model predictions reflected fisheries during the



past prior to applying Ecosim to fisheries scenarios reaching far into the future. Calculated fishing mortalities were the driving parameters employed to simulate changes in catch and stock size. Trends simulated with the original model matched well real data (sum of squares of the fitting is equal to 67.17) and the model did not require any adjustments.

The second step involved model-supported calculation of possible impacts that changes in fishing pressure may exert on all trophic groups in the WBS ecosystem. The objective was to assess what fisheries management option responds to different goals. First, achieving the best productivity of the most economically important species in the German EEZ and the German NATURA 2000 areas of the western Baltic Sea. Second, obtaining highest levels of catch with the least negative impacts on the ecosystem, as specified by the CFP ( $F < F_{MSY}$ ) and the MSFD ( $B > B_{MSY}$ ). Third, ensuring the healthy size and age structure of the stocks. Fourth, preserving food web elements that safeguard long-term abundance and reproduction. The following future scenarios were tested covering the period 2020 to 2050 (medium-term predictions) and 2020-2100 (long-term scenario to allow all trends attaining a new steady state):

1. *No Fishing*: Stock development with all fishing activities cancelled.
2. *Business as usual*: Stock development under fishing pressure equal to the average of fishing mortalities exerted during last five years (2015-2019).
3.  $F_{MSY}$ : Stock development when fishing pressure is reduced (if previously higher) or raised (if previously lower) to a value where fishery yield is sustainably at or slightly below the maximum level (if available).
4. *Half  $F_{MSY}$* : Stock development when fishing pressure is reduced to (if previously higher) or raised (if previously lower) to 50%  $F_{MSY}$ .
5. *EBFM*: No fishing on juvenile cod, herring and sprat fished at 50%  $F_{MSY}$  and other stocks fished at 80%  $F_{MSY}$ .

## Results

### Starting situation in 1994 represented by the static model

Input-Output Tables for the static model and results of the balancing process are presented in Tables S7-S9.

Table S7. Basic model parameters before and after balancing. In **bold** = values reduced during balancing; in ***bold italics*** = values increased during balancing; values in grey fields = biomasses estimated by EwE.

Group #	Group name	Trophic level	Biomass end (gC m <sup>-2</sup> )	Biomass start (gC m <sup>-2</sup> )	Z (y <sup>-1</sup> )	P/B (y <sup>-1</sup> )	Q/B (y <sup>-1</sup> )	Net migration (gC m <sup>-2</sup> y <sup>-1</sup> )	Ecotrophic efficiency start	Ecotrophic efficiency end
0	Recreational fishery	4.415								
0	Pelagic fleet	4.447								
0	Bycatch/IUU fishery	4.443								
0	Demersal fleet	4.035								
1	Seals	4.413	5.00E-05			0.095	20.000		0.058	<b><i>0.100</i></b>
2	Seabirds	3.742	0.0057			3.565	12.282		0.050	<b><i>0.100</i></b>
3	Harbour porpoises	4.424	0.0009			0.180	28.000		0.052	<b><i>0.233</i></b>
4	Adult cod	3.531	0.0559		1.470		1.500		0.508	0.508
5	Juvenile cod	3.193	<b><i>0.0665</i></b>	0.040	0.758		<b><i>2.719</i></b>		0.550	0.550
6	Flatfish	3.225	0.0191			0.928	3.257	-0.0245	2.336	<b><i>0.951</i></b>
7	Other demersal fish	3.431	<b><i>0.1770</i></b>	0.043		0.640	3.950		4.186	<b><i>0.990</i></b>
8	Herring	3.434	0.3583			0.860	4.500		0.609	0.609
9	Sprat	3.441	0.1228			1.500	7.660		0.706	0.706
10	Other pelagic fish	3.517	<b><i>0.4350</i></b>	0.003		0.280	2.850		66.025	<b><i>0.990</i></b>
11	Pelagic macrofauna	3.214	0.2015			5.400	17.800		0.182	<b><i>0.257</i></b>
12	Benthic macrofauna	2.013	41.0000			0.865	11.250		0.085	<b><i>0.102</i></b>
13	Benthic meiofauna	2.004	0.3600			5.135	32.500		8.809	<b><i>0.824</i></b>
14	Zooplankton	2.471	0.6970			76.690	271.360		0.640	<b><i>0.849</i></b>
15	Bacteria/microorganisms	2.176	0.3150			146.000	301.000		1.346	<b><i>0.926</i></b>
16	Phytoplankton	1.000	2.1610			120.000			0.974	<b><i>0.964</i></b>
17	Benthic producers	1.000	1.0000			234.000			1.190	<b><i>0.020</i></b>
18	Detritus/DOM	1.000	782.5000					-212.4000	0.738	<b><i>0.533</i></b>
	Total		829.4758							

Table S8. Diet composition of trophic groups in the WBS ecosystem before (b, grey columns) and after (a) balancing. In **bold** = values reduced during balancing; in ***bold italics*** = values increased during balancing.

Group #	Prey\predator	1	2	3b	3a	4	5	6	7	8b	8a	9	10	11	12b	12a	13	14b	14a	15b	15a
1	Seals																				
2	Seabirds																				
3	Harbour porpoises																				
4	Adult cod			0.009	0.009																
5	Juvenile cod	0.104	0.030	0.301	<b>0.033</b>	0.043	0.003	0.00031		0.0170	<b>0.000</b>				2.5E-05	<b>0.0000</b>					
6	Flatfish	0.077	0.010			0.013															
7	Other demersal fish	0.145	0.230	0.390	0.390	0.193	0.036	0.10714	0.011				0.0035								
8	Herring	0.434	0.070	0.260	0.260	0.071	0.012	0.00016													
9	Sprat		0.050	0.040	0.040	0.048	0.016	0.00110	0.120	0.0345	<b>0.000</b>				0.0027	<b>0.0002</b>					
10	Other pelagic fish	0.240	0.060				0.040		0.093				0.0005								
11	Pelagic macrofauna		0.030					0.03672		0.0035	<b>0.000</b>		0.0770	0.05							
12	Benthic macrofauna		0.310			0.613	0.812	0.82289	0.576	0.0630	0.063	0.0002	0.0320		0.0045	<b>0.0060</b>					
13	Benthic meiofauna							0.00080							0.0352	<b>0.0032</b>	0.004				
14	Zooplankton		0.030			0.019	0.073	0.03087	0.200	0.8770	<b>0.932</b>	0.9798	0.8870	0.75	0.0025	0.0026		0.152	<b>0.20</b>		
15	Bacteria/ microorganisms																	0.228	<b>0.15</b>	0.198	<b>0.15</b>
16	Phytoplankton													0.20	0.2222	0.2220		0.554	<b>0.55</b>	0.469	<b>0.45</b>
17	Benthic producers														0.0125	<b>0.0100</b>					
18	Detritus/DOM		0.050				0.008			0.0050	0.005	0.0200			0.7204	<b>0.7560</b>	0.996	0.066	<b>0.10</b>	0.333	<b>0.40</b>
	Import		0.130	0.000	<b>0.268</b>																
	Total	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

#### Notes

Import seabirds: part of diet covered by freshwater and terrestrial organisms

Import harbour porpoises: juvenile cod from neighboring marine areas

Table S9. Consumption by trophic groups and extraction by fisheries (gC m<sup>-2</sup> y<sup>-1</sup>) in the WBS ecosystem after balancing. In **bold** = values reduced during balancing, in **bold italics** = values increased during balancing. Fishery catches in *italics* were calculated based on figures in Rossing et al. (2010). Values ">0" stand for positive values but <0.00009.

Group #	Prey\predator	Seal	Seabirds	Harbour porpoises	Adult cod	Juvenile cod	Flatfish	Other demersal fish	Herring	Sprat	Other pelagic fish	Pelagic macrofauna	Benthic macrofauna	Benthic meiofauna	Zooplankton	Bacteria microorganisms	Detritus/DOM	Pelagic fleet	Demersal fleet	Recreational fishery	Bycatch/IUU fishery	Discards
1	Seals																0.0002				>0	
2	Seabirds																0.029				0.002	
3	Harbour porpoises																0.005				>0	
4	Adult cod			0.0002													0.056		0.036	0.004		0.001
5	Juvenile cod	0.0001	0.002	<b>0.001</b>	0.004	0.001	>0		<b>0</b>								0.056		0.016	0.003		0.002
6	Flatfish	>0	0.001		0.001												0.012		0.019	<i>0.001</i>	<i>0.005</i>	0.014
7	Other demersal fish	0.0001	0.016	0.010	0.016	0.007	0.007	0.008			0.004						0.123		<i>0.031</i>	<i>0.002</i>	<i>0.008</i>	<i>0.004</i>
8	Herring	0.0004	0.005	0.006	0.006	0.002	>0										0.491	0.153		<i>0.004</i>	<i>0.003</i>	<i>0.007</i>
9	Sprat		0.004	0.001	0.004	0.003	>0	0.084	<b>0</b>								0.271	0.025			<i>0.007</i>	<i>0.003</i>
10	Other pelagic fish	0.0002	0.004			0.007		0.065			0.001						0.218	<i>0.030</i>		<i>0.002</i>	<i>0.008</i>	<i>0.004</i>
11	Pelagic macrofauna		0.002				0.002		<b>0</b>		0.096	0.179					1.508					
12	Benthic macrofauna		0.022		0.051	0.147	0.051	0.402	0.102	0.0002	0.040		<b>2.768</b>				246.328		0.036			
13	Benthic meiofauna						>0						<b>1.476</b>	0.047			4.421					
14	Zooplankton		0.002		0.002	0.013	0.002	0.140	<b>1.503</b>	0.922	1.101	2.690	1.199		<b>37.825</b>		64.789					
15	Bacteria/microorganisms														<b>28.369</b>	<b>14.222</b>	12.881					
16	Phytoplankton											0.717	102.490		<b>104.018</b>	<b>42.667</b>	9.428					
17	Benthic producers												<b>4.613</b>				229.388					
18	Detritus/DOM		0.004			0.001			0.008	0.019			<b>348.705</b>	11.653	<b>18.912</b>	<b>37.926</b>						
	Import		0.009	<b>0.007</b>																		
	Sum	0.001	0.070	0.025	0.084	0.181	0.062	0.699	1.612	0.941	1.241	3.587	461.250	11.700	189.124	94.815	570.004	0.208	0.137	0.015	0.033	0.035

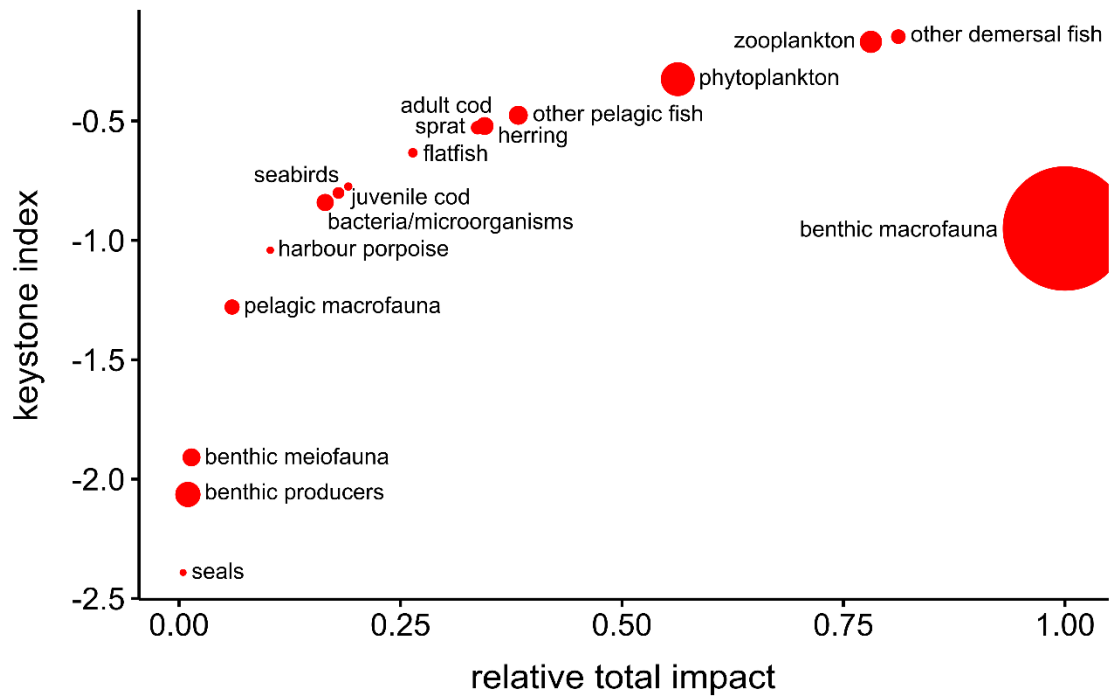


Figure S4. Strength of impact of trophic groups on the WBS food web. Herring shows the highest impact for a single species group, caused by feeding low in the food web and thus transporting matter from the bottom of the trophic chain to predators high in the food web (low-trophic level species with high impact on the food web). Dot size is proportional to the biomass of the trophic group (see Libralato et al., 2006 for a theoretical background of the keystone concept).

## Uncertainty analysis

The qualitative classification provided by the data pedigree (Table S6) was employed to quantify the degree of uncertainty associated to initial biomasses and parameters (i.e. P/B and Q/B ratios). An approach analogous to the one adopted by Corrales et al. (2017) was applied, which consists in assigning coefficients of variation reflecting the reliability of model inputs. The coefficients of variation listed in Table S10 served to set the limits that constrained the 99 Monte Carlo simulations run for each fisheries management scenario. The procedure allowed quantifying non-parametric confidence intervals associated to time series of stocks and catches simulated with the reference Ecosim model (i.e. 2% and 98% percentiles were used as lower and upper bounds). The same ensemble of simulation runs was applied to compare the performance of BAU vs. EBFM under multi-stressors scenarios, thereby ensuring any differences between the treatments could be attributed to alternative fishery management options rather than being due to different parameters combinations. The Ecosampler routine (Steenbeek et al., 2018) made possible comparing the EBFM and BAU scenarios using the same set of simulation runs.

Table S10. Coefficients of variation used to quantify the uncertainty of biomasses and input parameters in the balanced 1994 Ecopath model of the western Baltic Sea; NA values indicate input data not considered when executing Monte Carlo simulations.

Group name	Biomass	P/B	Q/B
Seals	0.4	0.2	0.2
Seabirds	0.4	0.4	0.4
Harbor porpoises	0.1	0.2	0.2
Adult cod	0.1	0.1	0.1
Juvenile cod	NA	0.1	NA
Flatfish	0.1	0.1	0.1
Other demersal fish	NA	0.4	0.4
Herring	0.1	0.1	0.1
Sprat	0.1	0.1	0.1
Other pelagic fish	NA	0.4	0.4
Pelagic macrofauna	0.4	0.4	0.4
Benthic macrofauna	0.1	0.4	0.4
Benthic meiofauna	0.1	0.4	0.4
Zooplankton	0.4	0.4	0.4
Bacteria/microorganisms	0.4	0.4	0.4
Phytoplankton	0.4	0.4	NA
Benthic producers	0.4	0.4	NA

# Dynamic modelling of different fishery management scenarios

## Exploring ecosystem development under different fisheries scenarios

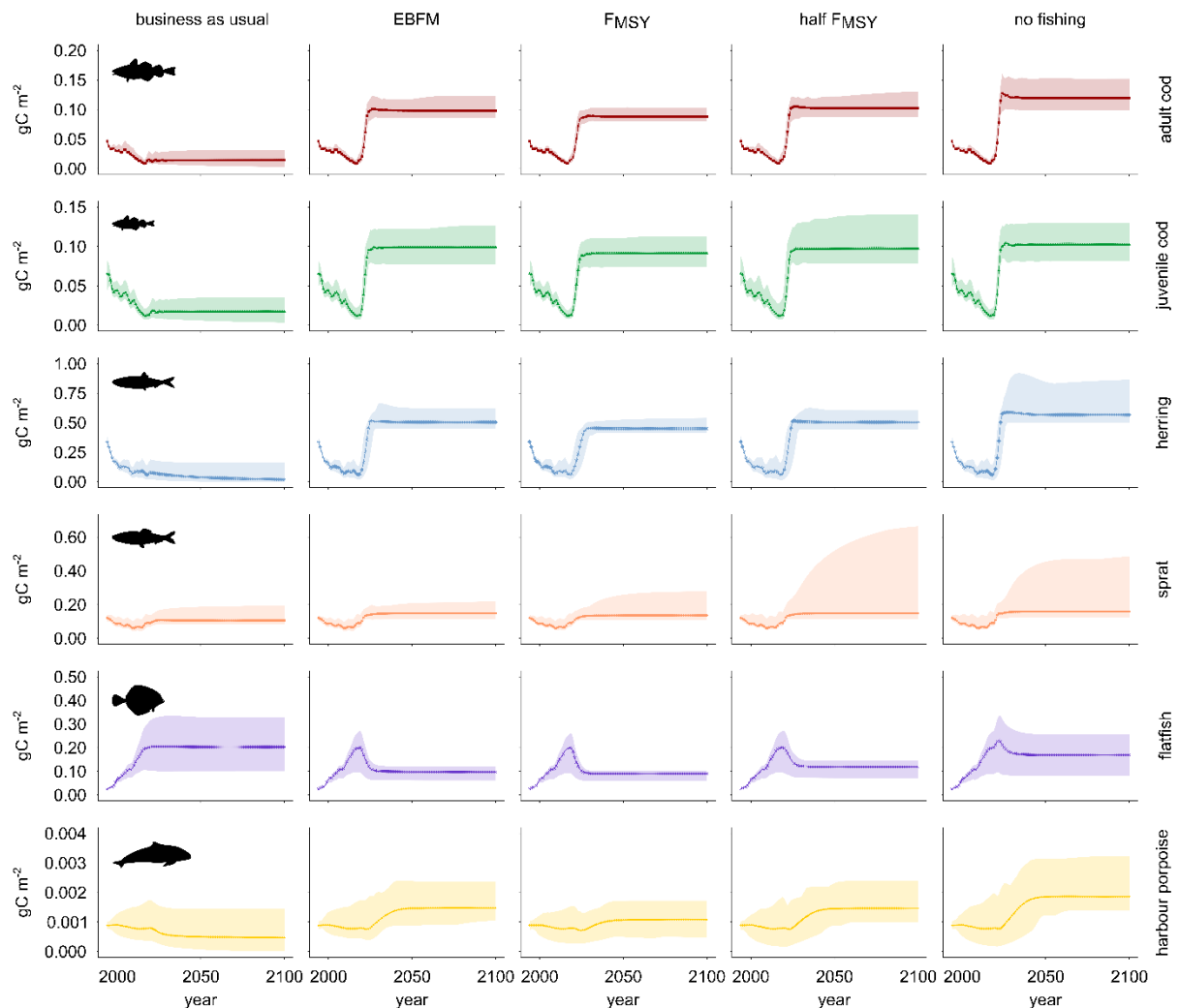


Figure S5. Stock biomass projections up to 2100. Predictions were made considering alternative fishing management scenarios (shown in columns). Each row illustrates fish stock trends, except for the last one where changes in the biomass of harbour porpoise are visualized (for this last group, data were not sufficient to perform any hindcasting; see Figure 3). Solid lines depict the trends found when running Ecosim starting from the 1994 reference Ecopath model. Shaded areas delimit 2% and 98% percentiles calculated with the outcomes of Monte Carlo simulations. These charts show that reference trends found for fish groups with simulations up to 2050 (Figure 4) do not substantially deviate from the steady state attained in 2100, with the exception of the uncertainty associated to forage fish under certain scenarios (e.g. sprat under the half  $F_{MSY}$  fishing regime).

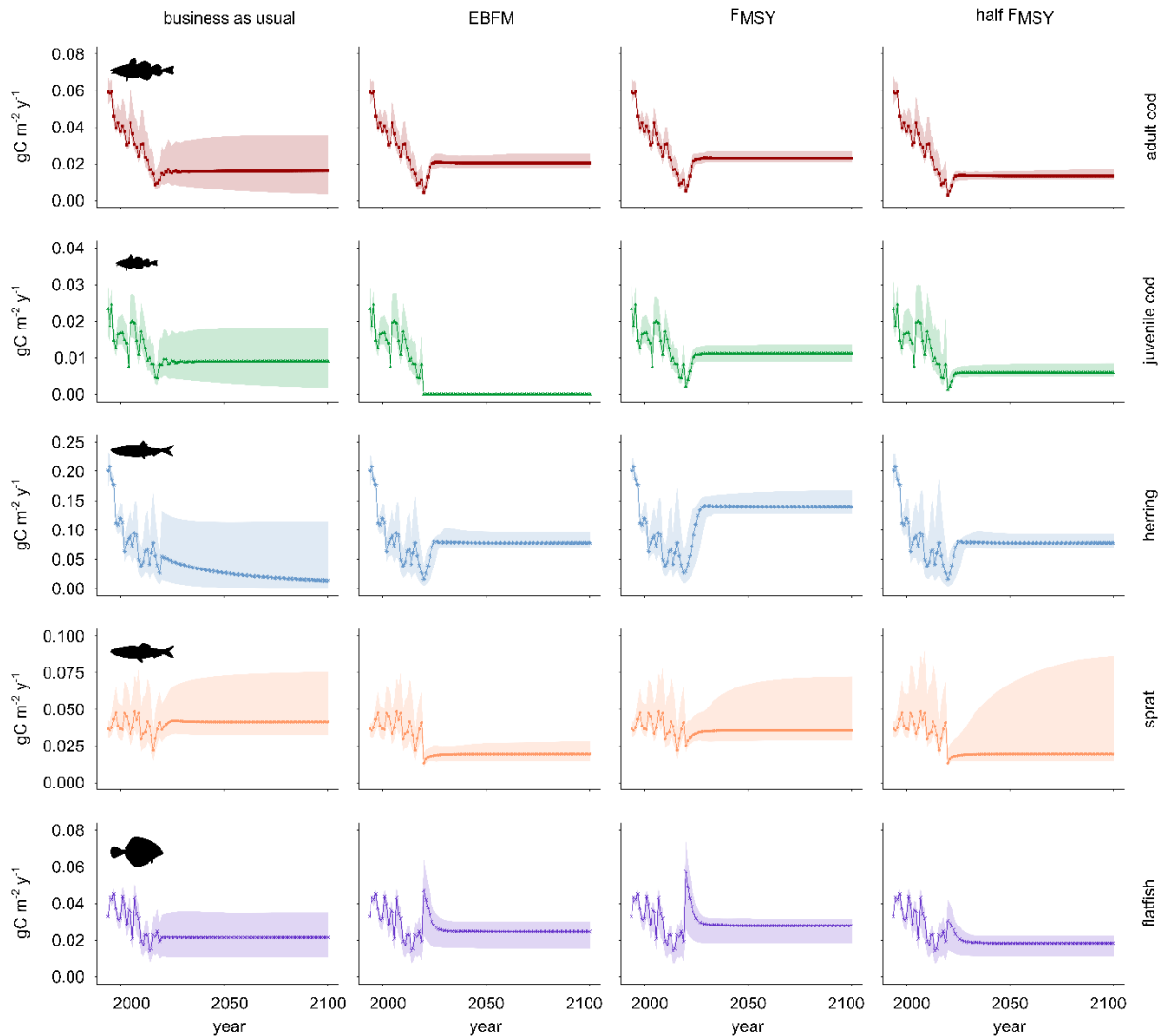


Figure S6. Catch projections up to 2100. Predictions were made considering alternative fishing management scenarios (in columns). The no-fishing scenario is omitted because plots of all trophic groups would have displayed absence of any catch starting from 2020. Each row illustrates catch trends for the main commercial targets (cod, herring, sprat and flatfish). For these trophic groups, the reliability of the Ecosim model was assessed through hindcasting, using official ICES data as a benchmark (Figure 3). Solid lines depict the trends found when running Ecosim starting from the 1994 reference Ecopath model. Shaded areas delimit 2% and 98% percentiles calculated with the outcomes of Monte Carlo simulations. These charts show that reference trends found with simulations up to 2050 (Figure 5) do not substantially deviate from the steady state attained in 2100, with the exception of the uncertainty associated to forage fish under certain scenarios (e.g. sprat under the half F<sub>MSY</sub> fishing regime).



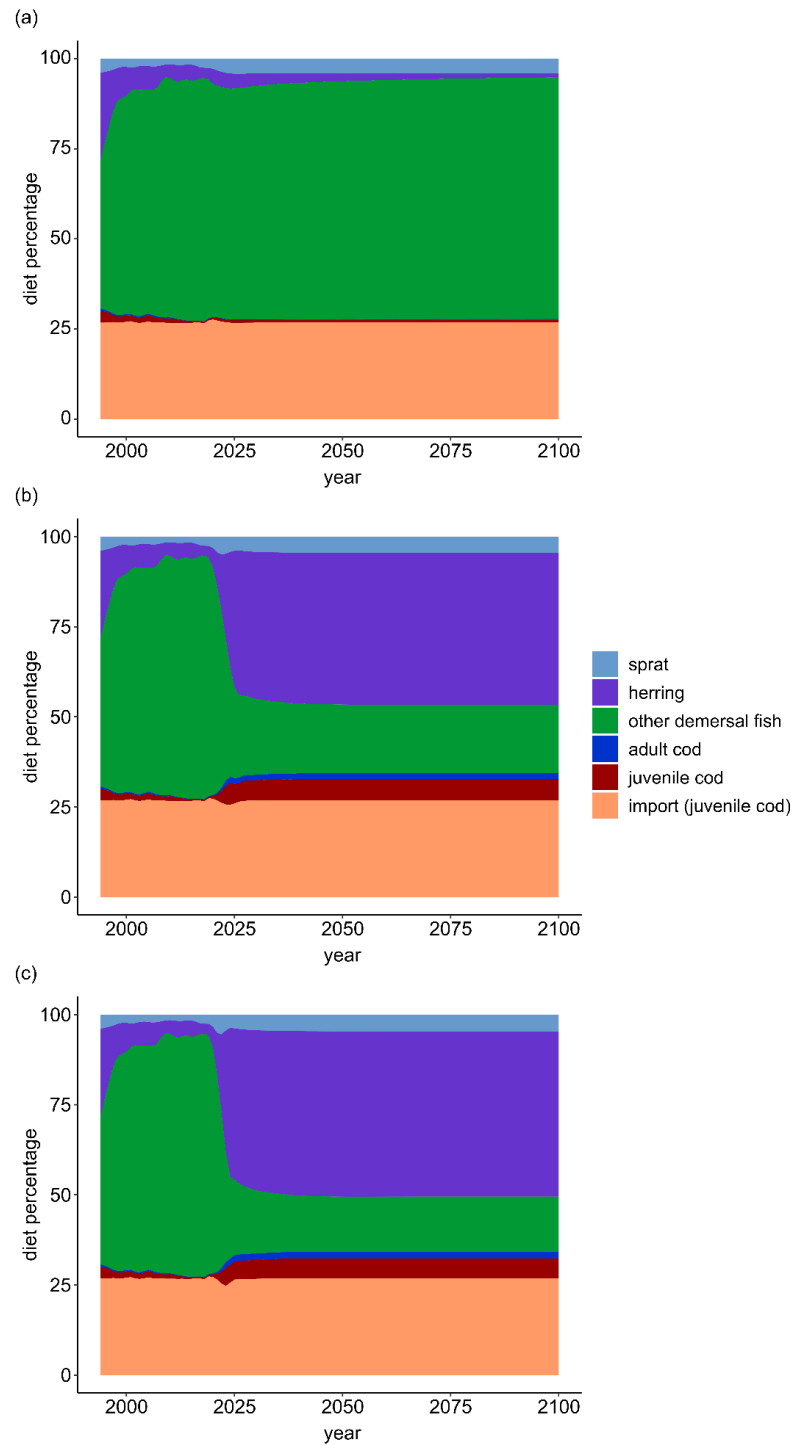


Figure S7. Diet composition of harbour porpoises in the WBS ecosystem under scenarios: (a) business as usual, (b) Ecosystem-Based Fisheries Management and (c) no fishing.

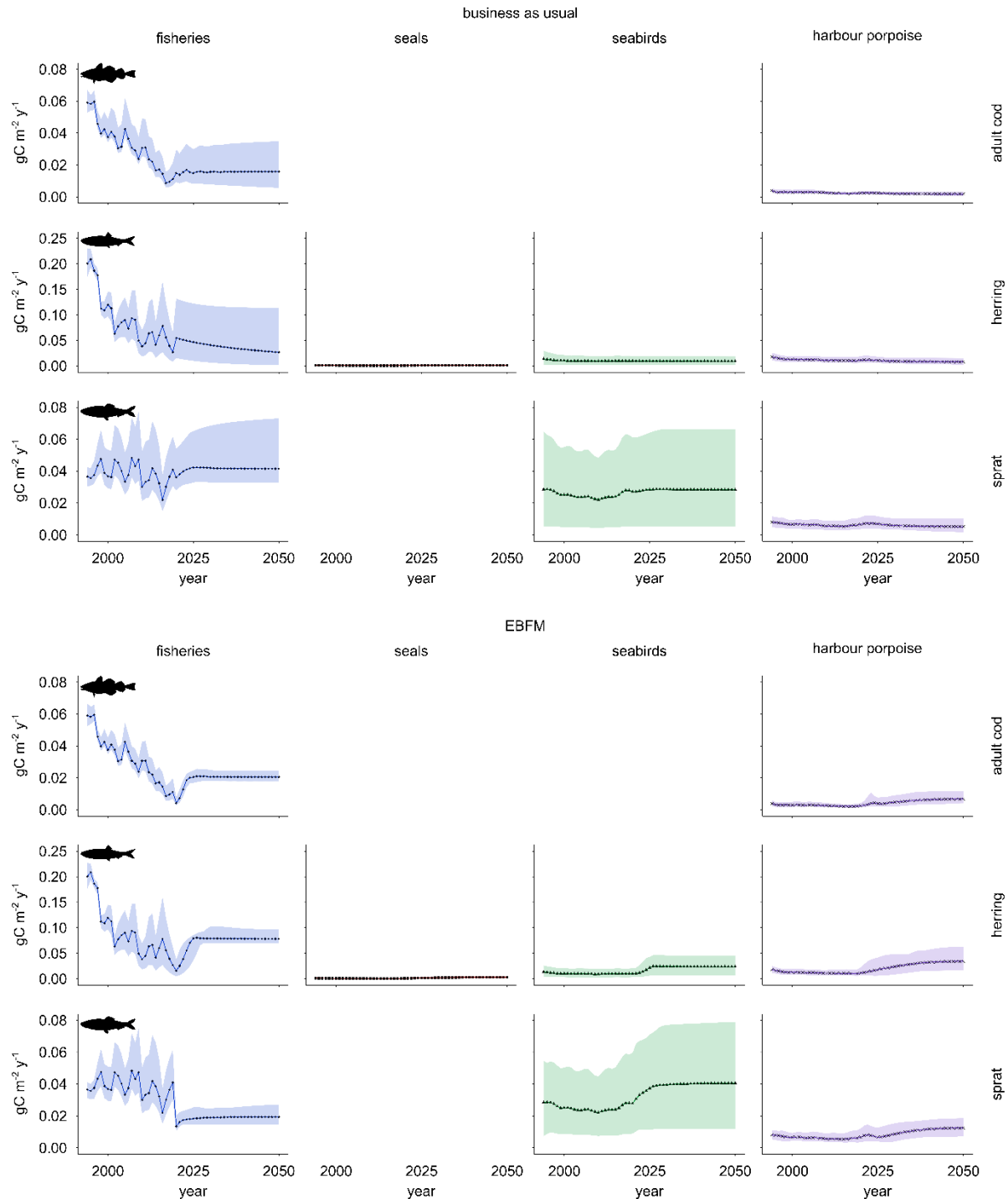


Figure S8. Annual extraction of commercially important fish by fishers and top predators. In the upper part, charts refer to the business as usual (BAU) scenario while the lower part of the image informs about trends under ecosystem-based fisheries management (EBFM). Columns report catches (blue) and predation by seals (red), seabirds (green) and harbour porpoises (violet). Solid lines were simulated starting from the 1994 reference Ecopath model while shaded areas delimit non-parametric confidence intervals. Interval limits are 2% and 98% percentiles, identified using the results of uncertainty analysis obtained with Monte Carlo simulations. Absence of charts indicate fish species in the rows that are not consumed by the corresponding predator. Fishers often consider harbor porpoises, seals and seabirds as competitors for target species such as cod, herring, and sprat. The results presented here dismiss such a viewpoint, except for seabirds feeding over sprat. However, this finding holds under both fishing scenarios (the potential competition of seabirds is not specific to EBFM) and vast uncertainty is associated to the consumption level.

### Primary production required by components of the WBS ecosystem

The relationship between primary production required (PPR) and total primary production (totPP) is expressed by the ratio PPR/totPP (%). Decreasing values from 1994 to 2019 and further on under business-as-usual management (BAU) indicate an increasing mismatch between PP available and use by system elements, nourishing the hypothesis that due to increasing release of non-point source pollutants and/or shrinking fish stocks – provoked mainly by overfishing, particularly of low-trophic level species like herring and sprat, – eutrophication in the western Baltic Sea ecosystem might be increasing. Under the scenario ecosystem-based fisheries management (EBFM), results of the model show to what extent the situation may be reversed compared to conditions in 2019.

Table S11. PPR/totPP (%) by commercially most relevant fish stocks in the WBS.

<b>Group name/year</b>	<b>1994</b>	<b>2019</b>	<b>2100 BAU</b>	<b>2100 EBFM</b>
Adult cod	0.687	0.458	0.465	0.569
Flatfish	1.571	0.559	0.573	0.483
Herring	0.698	0.169	0.056	0.324
Sprat	0.046	0.053	0.055	0.026
<b>Total</b>	<b>3.002</b>	<b>1.239</b>	<b>1.149</b>	<b>1.402</b>

Table S12. Catches ( $\text{gC m}^{-2} \text{y}^{-1}$ ) and biomass ( $\text{gC m}^{-2}$ ) of selected groups in the western Baltic Sea. Comparison between start (2019) and end (2100) year of fishery scenarios.**Catches**

Scenario & year	Seals	Seabirds	Harbour porpoises	Adult cod	Juvenile cod	Flatfish	Other demersal fish	Herring	Sprat	Other pelagic fish
2019	8.1014E-07	3.0833E-03	3.3564E-05	0.01121	0.00823	0.01918	0.05377	0.02687	0.04098	0.04230
BAU 2100	1.3221E-06	2.6795E-03	1.9796E-05	0.01603	0.00912	0.02154	0.05105	0.01351	0.04154	0.04175
EBFM 2100	1.8015E-06	2.7776E-03	6.2058E-05	0.02047	0.00000	0.02449	0.03655	0.07825	0.01937	0.04974
F <sub>MSY</sub> 2100	1.4166E-06	2.4776E-03	4.5528E-05	0.02292	0.01113	0.02801	0.03693	0.13956	0.03555	0.04980
50% F <sub>MSY</sub> 2100	1.9418E-06	2.8489E-03	6.1586E-05	0.01329	0.00593	0.01839	0.03639	0.07821	0.01940	0.04997
BAU x-fold	1.632	0.869	0.590	1.430	1.108	1.123	0.949	0.503	1.014	0.987
EBFM x-fold	2.224	0.901	1.849	1.826	0.000	1.277	0.680	2.913	0.473	1.176
F <sub>MSY</sub> x-fold	1.749	0.804	1.356	2.044	1.353	1.461	0.687	5.195	0.868	1.177
50% F <sub>MSY</sub> x-fold	2.397	0.924	1.835	1.185	0.720	0.959	0.677	2.911	0.473	1.181

**Biomass**

Scenario & year	Seals	Seabirds	Harbour porpoises	Adult cod	Juvenile cod	Flatfish	Other demersal fish	Herring	Sprat	Other pelagic fish
2019	8.5278E-05	8.6454E-03	7.9913E-04	0.01552	0.01325	0.20116	0.21755	0.06978	0.09107	0.43742
BAU 2100	1.3917E-04	7.5132E-03	4.7132E-04	0.01442	0.01746	0.20320	0.20214	0.01940	0.10651	0.41861
EBFM 2100	1.8963E-04	7.7882E-03	1.4776E-03	0.09842	0.09893	0.09873	0.14475	0.50482	0.14898	0.49866
F <sub>MSY</sub> 2100	1.4911E-04	6.9469E-03	1.0840E-03	0.08816	0.09127	0.09037	0.14625	0.45019	0.13674	0.49930
50% F <sub>MSY</sub> 2100	2.0440E-04	7.9880E-03	1.4663E-03	0.10221	0.09716	0.11864	0.14412	0.50461	0.14923	0.50095
no fishing 2100	2.7702E-04	9.2963E-03	1.8614E-03	0.11921	0.10248	0.16920	0.14216	0.56721	0.15943	0.50375
BAU x-fold	1.632	0.869	0.590	0.929	1.318	1.010	0.929	0.278	1.170	0.957
EBFM x-fold	2.224	0.901	1.849	6.340	7.465	0.491	0.665	7.234	1.636	1.140
F <sub>MSY</sub> x-fold	1.749	0.804	1.356	5.679	6.886	0.449	0.672	6.451	1.502	1.141
50% F <sub>MSY</sub> x-fold	2.397	0.924	1.835	6.584	7.331	0.590	0.662	7.231	1.639	1.145
no fishing x-fold	3.248	1.075	2.329	7.679	7.733	0.841	0.653	8.128	1.751	1.152
EBFM/no fishing	0.685	0.838	0.794	0.826	0.965	0.584	1.018	0.890	0.934	0.990

## Multi-stressors' scenarios

Ecosim simulations were run considering fishing as the most relevant stressor altering fish stocks and influencing the entire ecosystem. The influence on the ecosystem is both direct (e.g. harbour porpoise mortality because of bycatches) and indirect (e.g. decline of herring stock, which leads to lower amounts of energy available for high trophic level consumers). The choice of focusing on fish extraction is motivated by overfishing being the main threat to the integrity of fish stocks (Dulvy et al., 2021). Other stressors have been in fact shown to exert negative impacts on fish populations but their relevance is limited compared to overfishing. For instance, eutrophication may delay the onset of a regime shift triggered by fishing due to transitory prevalence of bottom-up forcing (Bodini et al., 2018). Climate change as well may threat fish populations and usually manifests over stocks eroded by overfishing, i.e. those with lower resilience (Möllmann et al., 2021; Froese et al., 2022).

The objectives of this section are (1) describing the choice made for modelling the impact of changes in phytoplankton biomass and ocean warming, (2) explaining the construction of alternative models that include the eastern Baltic cod in SD 24, and (3) illustrating the outcomes of models that implement the effect of additional stressors either in isolation or with a fully factorial design.

### Construction of models with multi-stressors' factors

#### **Phytoplankton biomass changes**

Excess nutrient load caused by stream runoffs from agricultural areas is a major threat to Baltic Sea ecosystem health and results in 97% of its basin being affected by eutrophication (HELCOM, 2018). In recent years, a decrease of nitrogen inputs and constant amounts of phosphorous discharges have been reported for the western Baltic Sea (Kuss et al., 2020), without resulting in major improvements of water quality (HELCOM, 2018).

Nutrient enrichment sustains phytoplankton primary productivity, leading to an increase of phytoplankton biomass. Indirect consequences include increased levels of water turbidity, which reduces the colonization depth of macroalgae and seagrasses, and the expansion of oxygen minimum zones, which is caused by organic matter degradation. Low-oxygen areas harm sea bottom invertebrates' survival and impair fish recruitment by increasing egg and larvae mortality. However, quantifying mortality variations triggered by eutrophication is complicated. Impacts of changes in nutrient concentrations were then modelled here in the form of their most direct effect, which is via modulation of phytoplankton concentration. Two scenarios were implemented, varying phytoplankton biomass by  $\pm 25\%$  compared to the reference model. These changes are realistic and do not exceed the variability observed in the western Baltic Sea during the 2000s (Henriksen, 2009). Phytoplankton biomass was used together with fishing mortality as forcing factor for simulations.

## Ocean warming

Increasing water temperature influences marine organisms in a multifaceted way. For instance, it may shift the phenology of species thus altering trophic interactions (Aberle et al., 2012), or may modify the spatial distribution of taxa (Csapó et al., 2021). All species in an ecosystem are exposed to the effects of climate change but the way they respond is non-homogeneous. Therefore, it is complicated ranking species to identify what aspects of an ecosystem model should be modified to account for impacts triggered by ocean warming. Here, the decision was to change either parameters regulating the dynamics or the stock biomass of the three main commercial targets, i.e. western Baltic cod, western Baltic spring-spawning herring, and sprat. Modulation of phytoplankton biomass described under the previous section might reflect also responses to warming (see Wasmund et al., 2019). Other changes might have been implemented to model a reduction of biomass and nutritional value of copepods (Garzke et al., 2016) and an increase of jellyfish biomass in response to warming (Haraldsson et al., 2013). Criteria adopted to model the impact that warming has on the western Baltic cod, herring and sprat are described in the remainder of this section. Three levels of temperature increase (+1, +2 and +3 °C) were modelled.

Western Baltic cod: Voss et al. (2012) showed for cod that the larval window of opportunity (WOO), an index measured in days and positively related to the chances of larvae survival, varies as a function of August temperatures at 30-40 m depth. This index determines the number of days during which larvae have to establish successful feeding prior to depleting all energy reserves. Longer WOOs translate into higher chances of survival.

Since the main spawning area of the western Baltic cod is located in the Arkona Basin (Eero et al., 2019), the average temperature at 30-40 m depth in years 1994-2019 was extracted using the Baltic Sea Physics Reanalysis<sup>10</sup> dataset for the coordinates [55, 55.5]°N and [13, 15]°E. The average mean temperature of 10.4 °C served to calculate the average WOO of cod in the period 1994-2019 through the equation shown in Figure 6 of Voss et al. (2012). This average temperature was progressively increased by 1 to 3 °C for determining the length of WOO under warming (Table S13). Relative changes of WOO length compared to the 1994-2019 reference period were calculated and used to estimate the total mortality of juvenile cod over the years 2020-2100. Such total mortality was applied as forcing factor to run ocean warming scenarios. The relative WOO change was divided by 80 (years) and evenly distributed to increase by small fractions the total mortality of each year.

Herring: Polte et al. (2021) showed that the day of winter onset (DWO; threshold set at 4°C) has an effect on the N20 larvae index and GERAS 1-wr index. The N20 index informs about larvae abundance and estimates the number of larvae that reach 20 mm length thus surviving major early life stages mortality. The GERAS 1-wr index informs about juveniles

<sup>10</sup> Product identifier: BALTICSEA\_REANALYSIS\_PHY\_003\_011 – accessed from: <https://www.copernicus.eu/en>

abundance (i.e. 1-year-old juveniles identified through first winter ring detected in otoliths using the microscope) and shows a 1-year lag response compared to N20 larvae.

Polte and colleagues (2021) found that a 1-day delay of winter onset decreases by 1.3% the abundance of 1-year-old juveniles of the following year. Such an information was used to establish an explicit connection between changes in average SST winter temperature and juveniles' abundance. First, DWO in the years 1994-2019 was identified. Second, a linear model expressing DWO as a function of the average SST during November-December ( $DWO \sim 31.166 + 9.975 \cdot SST$ ;  $F_{1,24} = 35.48$ ,  $p < 0.001$ ,  $R^2 = 0.580$ ) was assembled. The temperature was retrieved from the Baltic Sea Physics Reanalysis<sup>10</sup> for the Greifswald Bay at coordinates [54.10, 54.20]°N and [13, 13.70]°E. Third, an average DWO during 1994-2019 was calculated. Fourth, variations in the DWO due to warming were quantified and, indirectly, their impact on recruitment (i.e. the GERAS 1-wr index) was determined (Table S13). Finally, relative decreases of juveniles' survival compared the reference scenarios served to scale total herring mortality in the period 2020-2100. The global relative change was then divided by 80 (years) and its contribution evenly distributed along the entire time series. The increase of total mortality caused by warming was quantified with a procedure analogous to the one presented for the western Baltic cod. Finally, such total mortality was applied as forcing factor to run Ecosim under warming scenarios.

Sprat: Differently from the western Baltic cod, which displays a monotonic downward trend of larval WOO in response to raising temperatures, the larval WOO of sprat shows an initial positive response to warmer temperatures (Voss et al., 2012). For sprat, WOO quantifies the number of days until eye pigmentation, a time frame correlated with maximum survival without feeding. Mildly warmer temperatures than those presently recorded might benefit sprat in the WBS, in contrast to what is reported for western Baltic cod and herring.

Following Voss et al. (2012), WOO of sprat was derived using the average temperature in the layer 0-10 m depth, from April 15th to May 15th. Temperatures were extracted from the Bornholm Basin at coordinates [54.75, 56]°N and [14.75, 17]°E. Previous works have in fact described this area as the most relevant for sprat reproduction (Dickmann et al., 2007). Temperature data were retrieved from the Baltic Sea Physics Reanalysis<sup>10</sup> dataset. Average temperature over the period 1994-2019 was 5.8 °C, corresponding to a WOO of about 10.4 days (Table S13). The average temperature was progressively increased from 1 to 3 °C to determine the length of WOO under warming. Relative changes of WOO length compared to 1994-2019 were projected over the years 2020-2100. Updated time series of SSB were the forcing factor applied to run ocean warming scenarios. Relative WOO change was divided by 80 (years) and distributed evenly across the 2020-2100 time series. This procedure resulted in annually increasing SSB of sprat.

Table S13. Relative variations of total mortality (Z) and spawning stock biomass (SSB). These changes were quantified for western Baltic cod and herring (Z), and for sprat (SSB). Reference temperatures were calculated over the period 1994-2019, (1) during August at 30-40 m depth (western Baltic cod), (2) in the months of November and December at 1.5 m depth (i.e. SST; herring), and (3) from April 15th to May 15th at 0-10 m depth (sprat). Reference areas are (1) Arkona Basin (western Baltic cod), (2) Greifswald Bay (herring), and (3) Bornholm Basin (sprat). WOO – window of opportunity; DWO – day of winter onset.

<b>Western Baltic cod</b>			
Scenario	Temperature	WOO	Relative Z increase
Reference	10.4 °C	13.113	
+1 °C	11.4 °C	11.842	0.097
+2 °C	12.4 °C	10.693	0.185
+3 °C	13.4 °C	9.656	0.264
<b>Herring</b>			
Scenario	Temperature	DWO	Relative Z increase
Reference	5.34 °C	85	
+1 °C	6.34 °C	95	0.130
+2 °C	7.34 °C	105	0.260
+3 °C	8.34 °C	115	0.390
<b>Sprat</b>			
Scenario	Temperature	WOO	Relative SSB increase
Reference	5.34 °C	10.397	
+1 °C	6.34 °C	12.163	0.170
+2 °C	7.34 °C	12.623	0.214
+3 °C	8.34 °C	11.622	0.118

## Construction of models with eastern Baltic cod

Mixing of western and eastern Baltic cod has been reported for the SD 24 but hydrographic conditions make such subdivision unsuitable for reproduction of eastern Baltic cod (Hüssy et al., 2016). The recruitment of eastern Baltic cod in SD 24 is impaired due to low salinity, which causes eggs to sink, and low oxygen zones (oxygen levels  $<2 \text{ ml O}_2 \text{ l}^{-1}$ ), which result in developmental failure and mortality of eggs (Wieland et al., 1994). The likelihood of this last threat to occur is enhanced by bottom contact of eggs due to low salinity. It is clear that the potential contribution to fishery of the eastern Baltic cod in our ecosystem model of the WBS cannot be considered by simply adding migratory input to the existing western Baltic cod compartment. This latter stock presents in fact stock-recruitment dynamics and metabolic parameters different from those of the eastern Baltic cod. In this section, details on a new eastern Baltic cod compartment added to the 1994 Ecopath model and used for Ecosim simulations are provided.

Mixing proportions in SD 24 were deduced based on otoliths from the German BITS survey in quarter 4 (ICES, 2019h; Figure 5 at page 237). These data were available for the period 1994-2017. The 1994-2017 average was calculated to complete the time series and assign



mixing proportions to the last two years. Spawning stock biomass of the western Baltic cod (WBC) in SD 24 was known and proportions of the two stocks in SD 24 allowed quantifying SSB of eastern Baltic cod (EBC):  $SSB_{EBC,24} = \text{prop}_{EBC,24} \cdot SSB_{WBC,24} / \text{prop}_{WBC,24}$ , with the term “prop” indicating the proportion of either WBC or EBC in SD 24. The eastern Baltic cod compartment includes the biomass of mature adults only because no recruitment occurs for this stock in the WBS. Catch levels of the eastern Baltic cod were instead retrieved from ICES (2021; see Table 8 at pages 6-7). Conversions of the wet weight stock biomass and catch in SD 24 were obtained using the same calculations presented earlier for the western Baltic cod. Stocks and catches of the eastern Baltic cod were then expressed as  $\text{gC m}^{-2}$  and  $\text{gC m}^{-2} \text{ y}^{-1}$ , respectively. Forcing factor for simulations was the exploitation rate, given by the ratio of catches/SSB. Finally, the diet of the eastern Baltic cod was assumed to be the same as the one of the western Baltic cod while metabolic parameters employed to assemble the initial, steady-state Ecopath model changed ( $P/B = 0.920$ ,  $Q/B = 3.500$ , and unassimilated consumption = 0.185; see Table 1 in Tomczak et al. 2012). Following the inclusion of the eastern Baltic cod, the total sum of squares for the hindcasting obtained applying the updated version of the Ecosim model was 72.84 (eastern Baltic cod: SSB = 3.157; catches = 3.157).

## Outcomes of simulations with multi-stressors' factors

### **Phytoplankton biomass changes and ocean warming (base model)**

In most simulations, the base model predicted larger stocks (Figure 6) and higher catches (Figure 7) for western Baltic cod and herring under EBFM than in presence of BAU. Sprat biomass was used as forcing factor to simulate warming and variations in SSB and catch can be assessed only when phytoplankton biomass changed by  $\pm 25\%$  to reflect different concentrations of nutrients. Under these circumstances, EBFM outperforms BAU for stock depletion but shows lower catches. Flatfish is the only group for which BAU ensures stocks that are larger and yields that are higher compared to the EBFM. These results confirm the trends found using the reference scenario (i.e. with fish extraction alone used as a forcing factor). They highlight that EBFM is the key to maintain ecosystem resilience in the spite of warming, in particular for the stocks of western Baltic cod and herring.

### **Phytoplankton biomass changes, ocean warming and eastern Baltic cod**

Eastern Baltic cod uses the SD 24 as feeding ground and additional simulations were carried out to include its migration to and fishery in WBS. When stressors are modelled in isolation (Figure S9 for stocks and Figure S10 for catches) or in combination (Figure S11), models with eastern Baltic cod confirm EBFM as the most efficient strategy of fishery management compared to BAU. These findings do not contradict previous conclusions based on models

with western Baltic cod stock only and urge the need of a drastic and abrupt change in the management of cod fishery in the western Baltic Sea.

### **Alternative scenarios and the fate of cod stocks and fishery**

A last exercise concerns calculating total biomass and catch of cod in the WBS, irrespective of stock identity. Results shown in Figure S12 confirm the expectation that under EBFM the total stock biomass is larger than in presence of BAU. Trends displayed by the catches are aligned with those of the stocks. Even this analysis confirms EBFM being a viable strategy to preserve a degree of ecosystem resilience that allows buffering negative consequences triggered by ocean warming and changes in nutrients concentrations.

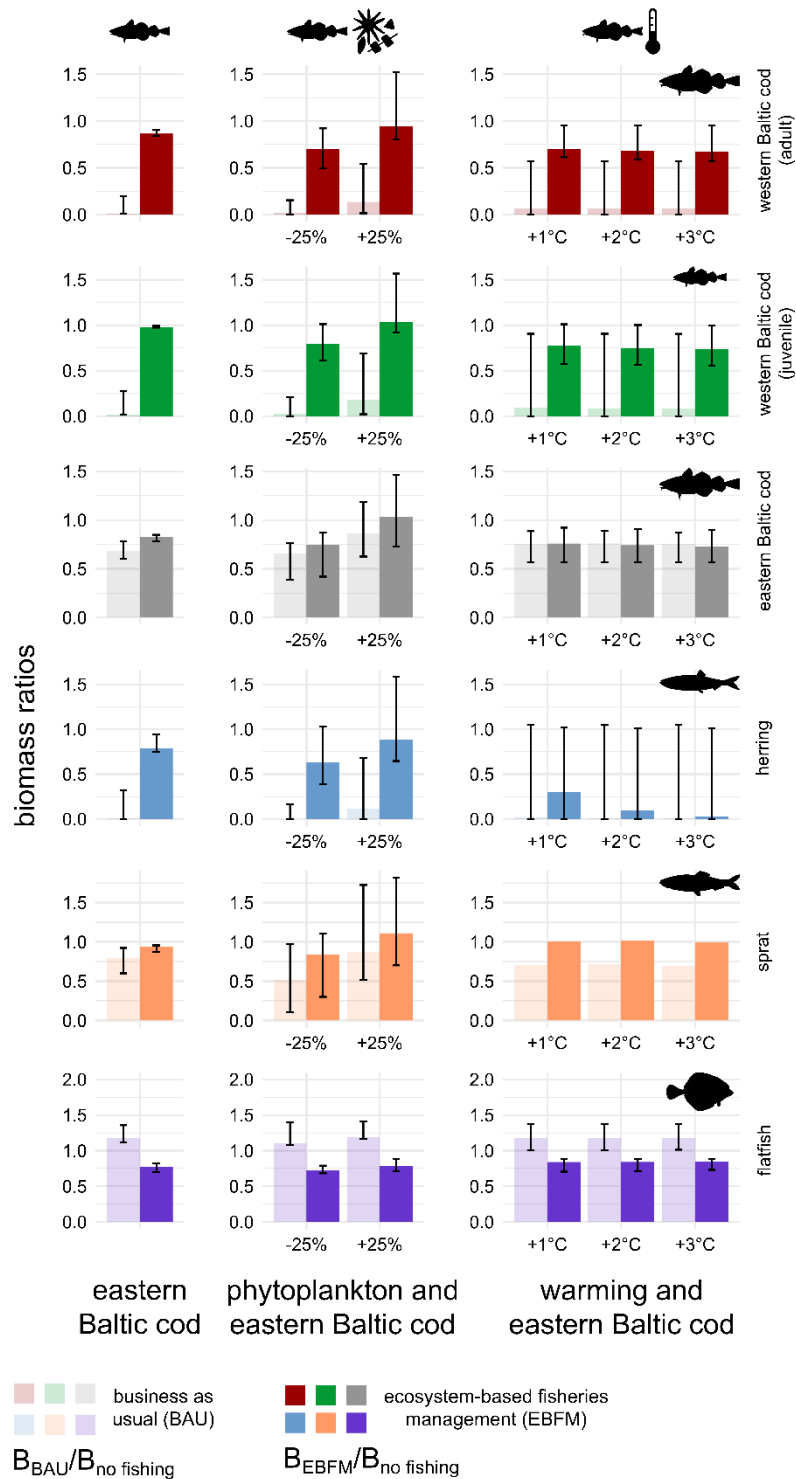


Figure S9. Biomass ratios under the impact of fish extraction alone or in combination with another stressor (either phytoplankton biomass changes or ocean warming). All simulations were performed by considering the presence of eastern Baltic cod in SD 24. Each bar shows the ratio between the biomass under BAU (transparent) or EBFM (opaque) and the no-fishing scenario biomass, all quantified using reference runs. A ratio equal to 0.5 indicates  $B_{MSY}$ . Biomasses were computed as the average of values estimated for the last 20 years of each simulation (2081-2100). Error bars were built using the outcomes of Monte Carlo randomizations; they illustrate 2% and 98% percentiles as lower and upper bounds, respectively. Under all warming scenarios, sprat bar plots do not have error bars because its stock biomass was used as forcing factor.

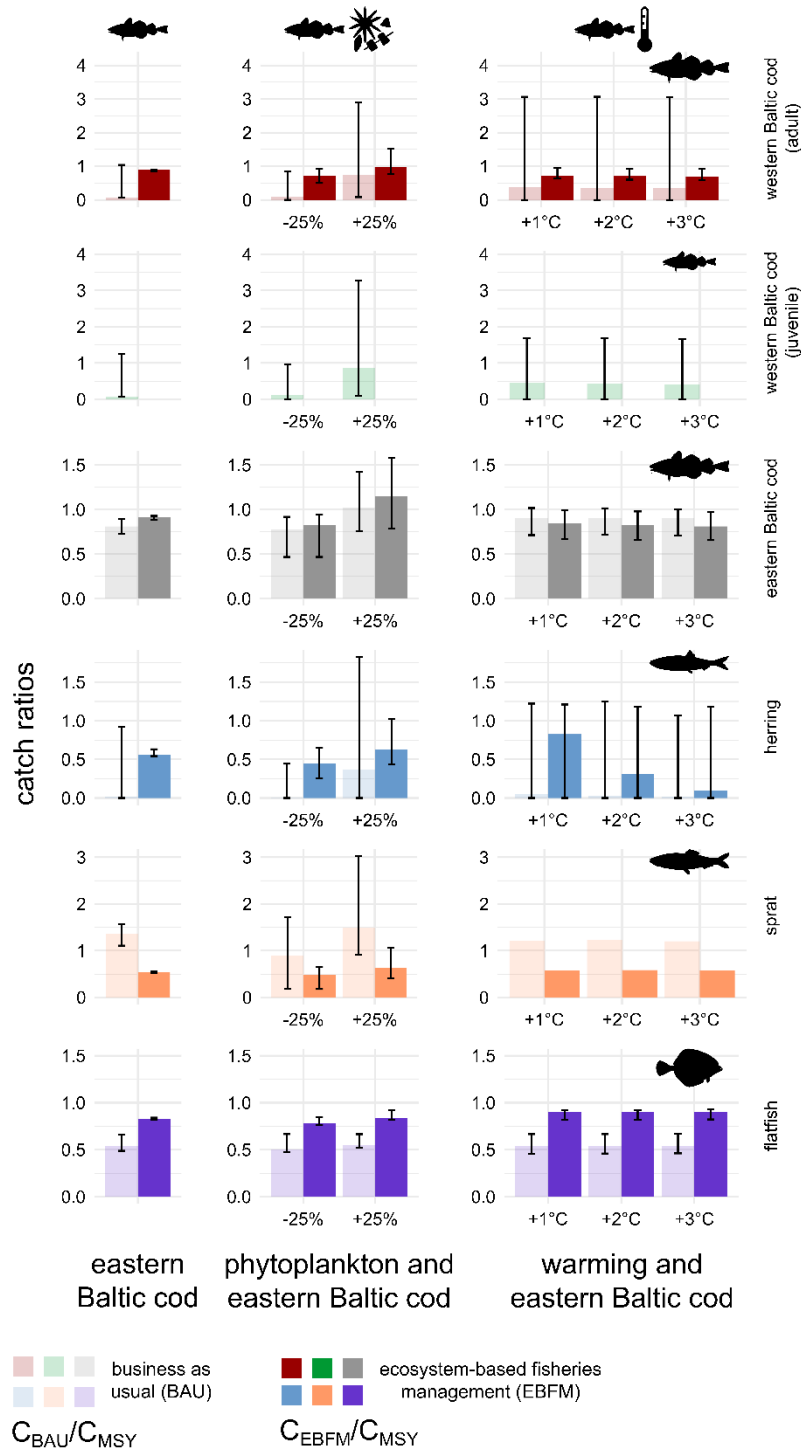


Figure S10. Catch ratios under the impact of fish extraction alone or in combination with another stressor (either phytoplankton biomass changes or ocean warming). All simulations accounted for the presence of eastern Baltic cod in SD 24. Each bar illustrates the ratio between the catch under BAU (transparent) or EBFM (opaque) and  $F_{MSY}$  catch, all quantified using reference runs. Ratios equal to 1 indicate the  $C_{MSY}$  level. Catches were computed as the average of values estimated for the last 20 years of each simulation (2081-2100). Error bars were built using the corresponding Monte Carlo randomizations; they visualize 2% and 98% percentiles as lower and upper bounds, respectively. Under all warming scenarios, sprat bar plots do not have error bars because its stock biomass was used as forcing factor.

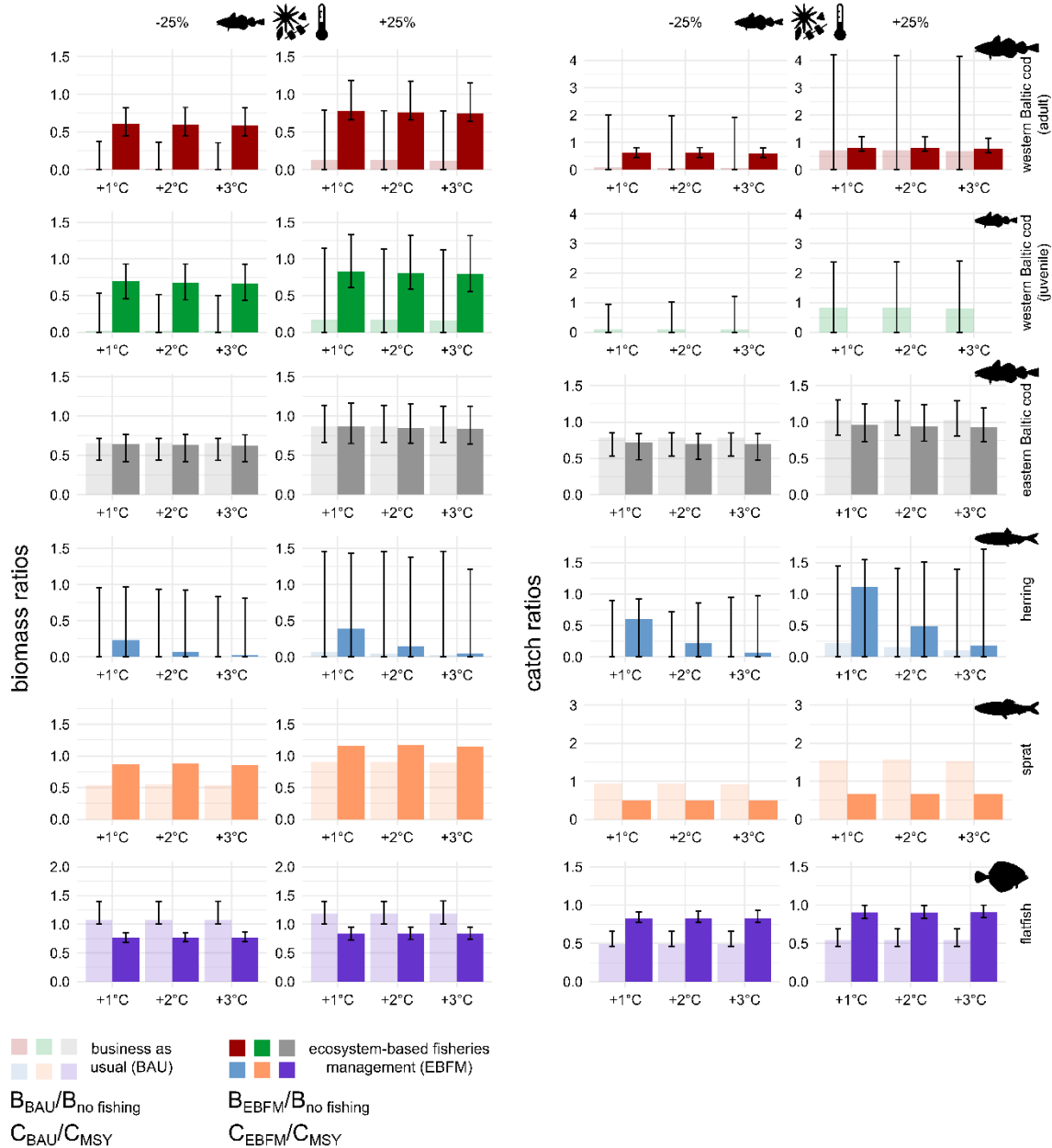


Figure S11. Biomass and catch ratios under simultaneous impacts of fishery, phytoplankton biomass changes and ocean warming. All simulations accounted for the presence of eastern Baltic cod in SD 24. Bars on the left show ratios between the biomass under BAU (transparent) or EBFM (opaque) and the no-fishing scenario biomass, quantified using reference runs. A ratio equal to 0.5 indicates  $B_{MSY}$ . Bars on the right illustrate ratios between the catch under BAU (transparent) or EBFM (opaque) and  $F_{MSY}$  catch, all quantified using reference runs. Ratios equal to 1 indicate the  $C_{MSY}$  level. Stock biomasses and catches were computed as the average of values estimated for the last 20 years of each simulation (2081-2100). Error bars were built using the corresponding Monte Carlo randomizations and visualize 2% and 98% percentiles as lower and upper bounds, respectively. Sprat bar plots do not have error bars because its stock biomass was used as forcing factor.

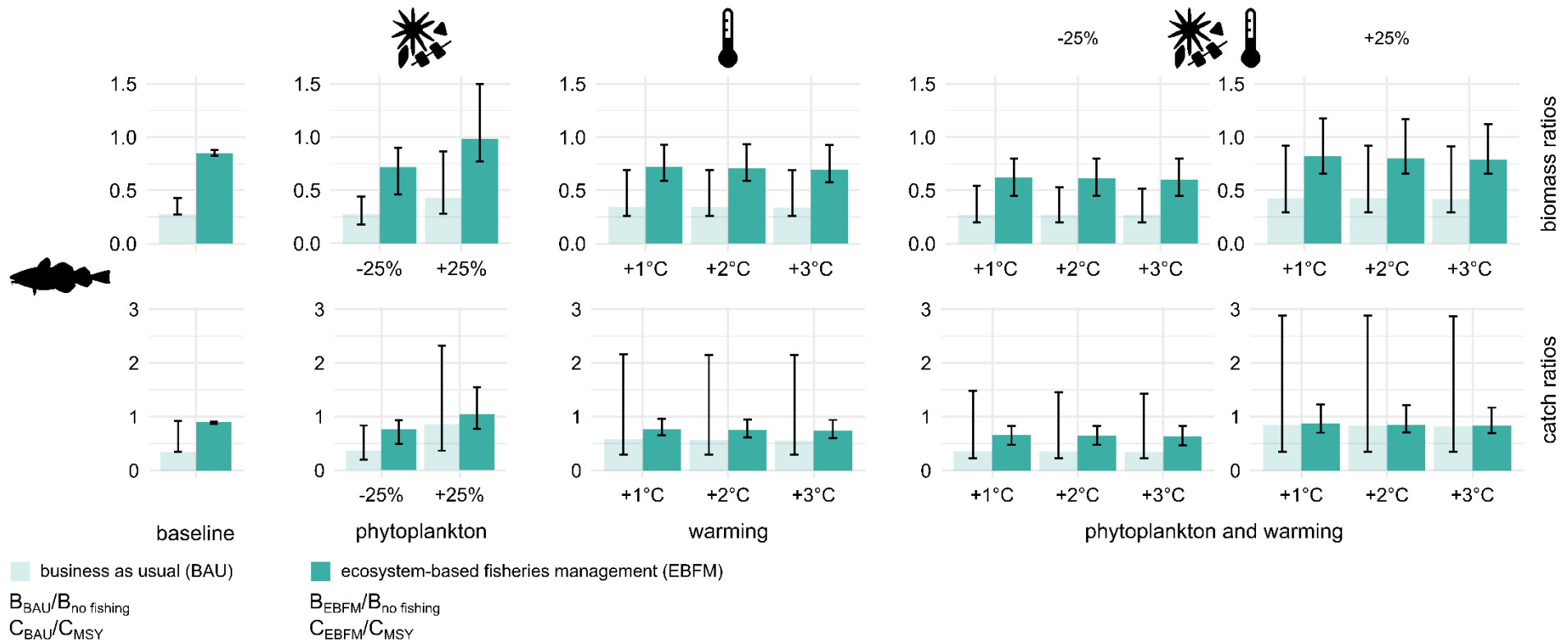


Figure S12. Changes in biomass and catch ratios of eastern and western Baltic cod together, under the impact triggered by fisheries plus changes in phytoplankton biomass, ocean warming, or both in combination. All simulations accounted for the presence of eastern Baltic cod in SD 24. Bars in the upper part report the ratios between the biomass under BAU (transparent) or EBFM (opaque) and the no-fishing scenario biomass, all quantified using reference runs. Ratios equal to 0.5 indicate  $B_{MSY}$ . Bars in the lower part illustrate the ratio between the catch under BAU (transparent) or EBFM (opaque) and  $F_{MSY}$  catch, all quantified using reference runs. Ratios equal to 1 indicate the  $C_{MSY}$  level. Biomasses and catches were computed as the average of values estimated for the last 20 years of each simulation (2081-2100). Error bars were built using the corresponding Monte Carlo randomizations and show 2% and 98% percentiles as lower and upper bounds, respectively.

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