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FEATURE ARTICLE

Regional trade-offs from multi-species maximum sustainable yield (MMSY) management options

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ABSTRACT: The maximum sustainable yield (MSY) is, theoretically, the largest yield that can be taken from a single species' stock over an indefinite period. Formulation of strategic MSY management goals is, however, complicated by the need to move beyond biological single-species considerations. Interactions among species necessitate multi-species (MMSY) definitions, incorporating ecological, economic and social considerations. We developed an ecological–economic model of the Baltic Sea, simulating stock dynamics of interacting populations of cod *Gadus morhua*, herring *Clupea harengus* and sprat *Sprattus sprattus*. We investigated a set of different strategic management options. These likely, yet non-formalized experiments evaluate and illuminate alternative regional trade-offs. We computed multi-species maximum economic yield (MMEY) under certain ecological constraints, with profits as a performance indicator. An unconstrained profit-maximizing management strategy would lead to a highly profitable cod fishery in a cod-dominated ecosystem. Concurrent sprat stock size (and profits) would be low, falling below ecological precautionary reference points. Consideration of ecological constraints on minimum stock sizes leaves a range of strategies, including the change from a cod-dominated to a more clupeid-dominated system. The regional distribution of profits depends on the management. Therefore, adjustment payments or other forms of compensation might be needed to achieve a concordant agreement on strategic multi-species management goals.

KEY WORDS: Equity · Baltic Sea · Distribution · Relative stability · Profits · Economic optimization · Bio-economic model

Managers must be able to wear different hats in order to achieve sustainability in multi-species fisheries.

Photos: Imme Schmidt

INTRODUCTION

Successful and commonly accepted fisheries management rules are a key to sustainability. Worldwide, approximately 500 million people are directly dependent on fisheries for earning their livelihood (FAO 2012). A growing world population, in combination with an increasing coastal population, is likely to further exacerbate problems linked to poor management of marine resources. Regulations are missing or are limping behind; partly because basic ecological and economic conditions for the relevant fishery are not understood. Even in Europe, fisheries management is still focused on single species, based on the natural sciences, but ignoring species interactions or any social and economic considerations. Consequently, in its latest evaluation of the European Common Fisheries Policy (CFP) the European Commission concluded that the CFP policy had failed and

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needed substantial revision (EC 2009). In the 2009 Green Paper on the reform of the CFP, the maximum sustainable yield (MSY) concept was included as a principle, accounting for the global imperative to manage fish stocks sustainably. Achieving this goal is complicated by the lack of a common interpretation of 'sustainability' and 'yield' and by the fact that achieving a theoretical long-term 'maximized' yield for one stock may detrimentally affect other stocks and may result in unwanted ecosystem, economic or social outcomes. Especially in systems with strong predator–prey links, either top down or bottom up, management decisions taken for one stock will inevitably influence the other stock(s). Rebuilding stocks of large predators like cod might negatively affect future profits from the corresponding prey-fish fishery, as the prey stocks will be depleted by the abundant predator. Different interpretations or prioritizations of 'yield' will therefore result in different long-term management goals, e.g. steering a system towards maximum yield in terms of biomass (usually prioritizing forage fish) is adverse to maximum yield in terms of profit (usually prioritizing large predatory fish). Unconstrained optimization for any given target might result in unacceptable situations, as defined by legally binding ecosystem indicators, e.g. the Good Environmental Status (GES) within the EU Marine Strategy Framework Directive (MSFD), or stock levels may fall below precautionary biomass levels. Therefore, the feasible 'space of possible solutions' might be narrower than originally perceived. However, even in a reduced decision space, decisions on trade-offs have to be explicitly made.

Current reforms of the CFP include new management measures, more regional structures and a more participatory and open process. However, some principles still seem to be 'carved in stone'. One principle which is not subject to discussion is the 'principle of relative stability'. According to this principle, the Baltic countries hold fixed shares of the quotas for cod *Gadus morhua*, herring *Clupea harengus* and sprat *Sprattus sprattus*. Therefore, the absolute catch amounts may differ between years depending on the stock status, but the percentage distribution of total allowable catches (TACs) to countries does not.

In this study we explored strategic management goals in a multi-species set-up and investigated the regional effects. We use the example of the Baltic Sea to show that inflexibility in the distribution of catch shares to countries, as constrained by the principle of relative stability, can lead to regional inequity in the distribution of future profits.

The fish community in the central Baltic Sea is dominated by cod, herring and sprat. The fishery mainly consists of single-species fisheries. However, the fish stocks are closely connected by strong ecological inter-connections between species (Köster & Möllmann 2000), as cod preys on both herring and sprat (Lewy & Vinther 2004). Thus, fluctuations in the size of the cod stock are related to considerable changes in natural mortality rates of sprat and juvenile herring. Under optimal management, the cod fishery would be the most profitable fishery by far (Nieminen et al. 2012, Quaas et al. 2013). The combination of high fishing pressure and environment-driven low recruitment success led to a decrease of the cod spawning stock biomass (SSB) from almost 700 000 to 100 000 t from 1983 to 1992, increasing shortly thereafter, but reaching a record low level in 2005 (ICES 2012). This strong decrease in the cod stock and a concurrent increase in the sprat stock resulted in a change from a cod-dominated system to a sprat-dominated system. In recent years the eastern Baltic cod recovered, like a number of other Northeast Atlantic stocks (Fernandes & Cook 2013), to a spawning stock biomass of >200 000 t in 2011 (ICES 2012). The recovery was due to a combination of improved recruitment and the implementation of a cod long-term management plan in 2006 (EC 2007). This plan was aimed at rebuilding the full reproductive capacity of the stock and resulted in better compliance and a substantially reduced fishing mortality ($F = 0.3$). A major difference compared to the previous management strategy is that inter-annual changes in TAC, both in terms of reductions and increases, are limited to a maximum of 15 %.

In June 2011, the European Commission and its member states agreed that the Baltic cod plan should be replaced by a Baltic multi-species management plan that would account for major species interactions. The Baltic Regional Advisory Council (RAC) also expressed support for such an approach. Accordingly, a number of expert groups were initiated, dealing e.g. with defining the methods of multi-species stock assessment (see Rindorf et al. 2013 for an overview). The scientific basis is formed by earlier multi-species works (Gislason 1999) showing that single- and multi-species reference points are different and that it is impossible to define a 'safe' level of biomass without taking changes in species interactions into account. The inclusion of first-order interactions is needed for medium-term management purposes (Collie & Gislason 2001). In the case of the Baltic Sea this applies to predation mortality induced by the cod stock. Socio-economic considerations are often neglected in the terms of reference for the expert groups and are only treated in subsequent analysis. We think that this needs to be changed, as useful management targets can only be achieved by including more detailed socio-economic analysis of the fisheries (Gislason 1999).

Using 2006 as our base year (i.e. the year of adoption of the cod management plan), we undertake model experiments to investigate 4 hypothetical long-term management goals and their outcome in terms of ecology (stock sizes), economy (total profits) and social aspects (regional distribution of profits): (1) an unconstrained economic optimization (maximizing profits) of the 3 species system, (2) optimization of the cod fishery's present value, while respecting a precautionary biomass level of sprat, (3) optimization of the sprat fishery, while maintaining a precautionary biomass level of cod and (4) a simulation of the current cod management plan. We show that a change back to a cod-dominated system is economically highly profitable as an aggregate, but not all countries would actually benefit from this change in an equal way. Therefore, compensation might be needed to avoid inequity.

We developed and applied a combined 3-species, age-structured ecological–economic model, which takes cod predation on 2 clupeid species into account. We used 4 scenarios (Table 1) to investigate the distribution of country-specific future profits.

MATERIALS AND METHODS

Ecological–economic modeling

Our model is an extension of the single-species age-structured fishery model of Tahvonen (2009) and Tahvonen et al. (2013), similar in scope to that of Nieminen et al. (2012). We use the subscript $i \in \{C,S,H\}$ for the cod (C), sprat (S) and herring (H) fisheries. The fishing profit for the cod fishery in year t is:

$$\pi_C = \sum_{s=1}^{8} p_C(s) w_C(s) \{1 - e^{(-F_C(t))}\} q_C(s) x_C(s,t) - c_C F_C(t) \quad (1)$$

Here we use $x_C(s,t)$ to denote stock numbers of age s in year t,$p_C(s)$ for age-specific prices, $w_C(s)$ for age-specific weights and $q_C(s)$ for age-specific relative catchabilities. Instantaneous fishing mortality is $F_C(t)$, and the cost function is of the Spence (1974) type, where c_C is a cost parameter (as in Quaas et al. 2012). Sprat and herring i = S, H are modeled as schooling fisheries (Tahvonen et al. 2013), with profits:

$$\pi_i = (p_i - c_i) \sum_{s=1}^{8} w_i(s) \{1 - e^{(-F_i(t))}\} q_i(s) x_i(s,t) \quad (2)$$

where p_i is the market price (which is assumed to be independent of age) and c_i is the constant marginal cost of harvest. For each fishery i = S, H, C we consider a representative fisherman's intertemporal utility from fishing income:

$$V_i = \sum_{t=0}^{\infty} \rho^t \frac{\pi_i^{1-\eta}}{1-\eta} \quad (3)$$

where ρ is the discount factor and η is the representative fisherman's aversion against inter-annual income fluctuations.

The higher η is, the more a constant income stream over time is preferred. Such a desire for relative constancy is reflected in several management plans for European fish stocks (e.g. Baltic cod; EC 2007), which have been agreed upon by a broad range of stakeholders, including fishermen. It is expressed, for example, as a requirement that TACs shall not change by more than a certain percentage between 2 subsequent years (15% in the case of Baltic cod).

The objective is to maximize a weighted sum of the intertemporal utilities ($E_{(t)}$) of the representative fishermen of all 3 fisheries:

$$\max_{E_C(t),\, E_S(t),\, E_H(t)} \{\lambda_C V_C + \lambda_S V_S + \lambda_H V_H\} \quad (4)$$

This model set-up allows changing the weights $\lambda_i > 0$ to model different strategic management goals or constraints. In the case of unconstrained economic optimization we take $\lambda_C = \lambda_H = \lambda_S = 1$, which means that all 3 fisheries have equal weight in the management optimization. Fishing mortalities may not be negative, i.e. $F_i(t) \geq 0$ in all cases. The age-structured multi-species population dynamics are described as follows. SSB of species i in year t are given by:

Table 1. Management Scenarios 1 to 4, using different sets of input or optimized fishing mortalities (F). SSB: spawning-stock biomass

Scenario	Objective
1	Unconstrained economic optimization
2	Optimization for profits from cod fishery, while respecting a precautionary SSB (B_{PA}) of 570 000 t of sprat (ICES 2013)
3	Optimization for profits from sprat fishery, while respecting a cod B_{PA} of 88 000 t (ICES 2013)
4	Simulation of the agreed long-term management plan for cod and subsequent optimization for clupeids

$$\mathrm{ssb}_i(t) = \sum_{s=1}^{8} w_i(s)\gamma_i(s)x_i(s,t) \tag{5}$$

where $\gamma_i(s)$ is used to denote age-specific maturities. Population dynamics of the stock of species i are given by:

$$x_i(s,t+1) = \varphi_i\,\mathrm{ssb}_i(t)\mathrm{e}^{[-\beta_i\,\mathrm{ssb}_i(t)]} \quad \text{for } s = 1 \tag{6}$$

$$x_i(s,t+1) = \alpha_i(s-1)(1-q_i(s)\{1-\mathrm{e}^{[-F_i(t)]}\})x_i(s-1,t) \quad \text{for } s = 2,\ldots,7$$

$$x_i(s,t+1) = \alpha_i(7)(1-q_i(s)\{1-\mathrm{e}^{[-F_i(t)]}\})x_i(7,t) + \alpha_i(8)(1-q_i(8)\{1-\mathrm{e}^{[-F_i(t)]}\})x_i(8,t) \quad \text{for } s = 8$$

This formulation implies that fishing and natural mortality are sequential, and was chosen as it is standard in resource dynamics literature (Tahvonen 2009). Changing the model to address fishing and natural mortality as competitive causes of death would slightly affect cost and catch, but not population dynamics.

For cod and herring we assume stock-recruitment functions of the Ricker (1954) type; for sprat we assume a Beverton-Holt type (Beverton & Holt 1957). Age-specific survival rates are:

$$\begin{aligned} \alpha_{\mathrm{C}}(s) &= \mathrm{e}^{(-M_{\mathrm{C}}(s))} && \text{for cod} \\ \alpha_{\mathrm{S}}(s,t) &= \mathrm{e}^{(-M_{\mathrm{S1}}(s)-M_{\mathrm{S2}}(s)\mathrm{ssb}_{\mathrm{C}})} && \text{for sprat} \\ \alpha_{\mathrm{H}}(s,t) &= \mathrm{e}^{(-M_{\mathrm{H1}}(s)-M_{\mathrm{H2}}(s)\mathrm{ssb}_{\mathrm{C}})} && \text{for herring} \end{aligned} \tag{7}$$

which are constant for cod. Residual (M_{i1}) and predation (M_{i2}) mortality estimates for the different age classes of herring and sprat are based on regression analysis, using SMS (Lewy & Vinther 2004) output on mortality for different stock sizes of cod. Predation mortality is almost linearly dependent on the cod stock biomass for a wide range of stock states (Tahvonen et al. 2013). This shortcut in the calculation of M_{i2} values was used to reduce model complexity and implies a dependency of predation mortality on both predator and prey abundance.

Data and estimation of model parameters

Age-specific weights $w_i(s)$ and maturities $\lambda_i(s)$ were taken from the ICES (2012) assessment reports for the 3 stocks, using the mean values from 2002 to 2006.

Age-specific catchabilities were estimated based on mean age-specific fishing mortalities for the years 2002 to 2006, as reported in ICES (2012), with $q_{\mathrm{A}} = 1$ for the age class with the highest mortality by normalization. In the case of reaching $q_{\mathrm{A}} = 1$ for an age class <8, it was kept constant for the older age classes, as it is meant to reflect mesh-size selection (Table 2).

Natural mortalities for the herring and sprat age classes were calculated dependent on the size of the cod stock. Estimates are based on a stochastic multi-species model (SMS: Lewy & Vinther 2004) and are reported in Table 3. The parameters for the stock-recruitment functions are given in Table 4.

For cod we used age-specific European reference prices, which are the lowest prices at which imports of cod of specific weight classes, sprat, or herring into the European Union are allowed (EC 1999, 2009), see Table 5. The cost parameter for cod is c_{C} = 55.2 million euros (Quaas et al. 2012). To estimate prices and cost parameters for sprat, we used price data and data on variable fishing costs for the Swedish (years 2002 to 2008) and Polish (years 2005 to 2008) pelagic trawler and seiner fleets from STECF (2011), which led to p_{S} euros kg^{-1} for the price and c_{S} euros kg^{-1} for the cost parameter. Similarly, for herring, we used STECF (2011) data for the Danish, Estonian, Finnish, Polish and Swedish trawler and seiner fleets (years 2002 to 2008), which gave estimates of p_{H} euros kg^{-1} for the price and c_{H} euros kg^{-1} for the cost parameter. For the representative fisherman's aversion to inter-annual income fluctuations, we assumed $\eta = 0.25$. The discount rate was set at 0 and 5%, respectively.

Table 2. *Gadus morhua* (C), *Clupea harengus* (H), *Sprattus sprattus* (S). Parameters used in the ecological–economic model. Values for maturity, weight and catchability are from the ICES (2012) standard assessment, using mean values from 2002 to 2006; numbers at age (corrected for spawning time) are from ICES (2012) for 2006

Age	Numbers in 2006 (millions)			Maturity			Weight (g)			Catchability		
class	C	H	S	C	H	S	C	H	S	C	H	S
1	196.555	11597	60816	0	0	0.17	80	11	52	0	0.28	0.27
2	131.041	5123	23884	0.13	0.7	0.93	179	20	84	0.11	0.44	0.49
3	122.411	5519	60692	0.36	0.9	1	511	25	96	0.42	0.66	0.79
4	52.298	5919	19240	0.83	1	1	838	31	105	0.81	0.82	0.85
5	15.187	1713	3179	0.94	1	1	1204	37	111	1	0.97	1
6	3.546	1105	1519	0.96	1	1	1796	43	113	1	0.96	1
7	0.714	830	1510	0.96	1	1	2596	48	111	1	1	1
8	0.383	789	1959	0.98	1	1	4068	53	113	1	1	1

Table 3. *Gadus morhua* (C), *Clupea harengus* (H), *Sprattus sprattus* (S). Natural mortality estimates used in ecological–economic modeling. Residual and predation mortality estimates in the multi-species (interaction) case are based on regression analysis, using stochastic multi-species output on mortality for different stock sizes of cod

	Mortality 'no interaction'			Residual mortality (M_1)			Predation mortality (M_2) coefficient		
Age class	C	H	S	C	H	S	C	H	S
1	0	0.2	0.2	0	0.1702028	0.1317657	–	3.324×10^{-4}	8.740×10^{-4}
2	0.2	0.2	0.2	0.2	0.1727799	0.1366770	–	2.312×10^{-4}	7.076×10^{-4}
3	0.2	0.2	0.2	0.2	0.1778390	0.1317657	–	0.448×10^{-4}	6.737×10^{-4}
4	0.2	0.2	0.2	0.2	0.1878390	0.1317657	–	0.448×10^{-4}	6.737×10^{-4}
5	0.2	0.2	0.2	0.2	0.1878390	0.1317657	–	0.448×10^{-4}	6.737×10^{-4}
6	0.2	0.2	0.2	0.2	0.1878390	0.1317657	–	0.448×10^{-4}	6.737×10^{-4}
7	0.2	0.2	0.2	0.2	0.1878390	0.1317657	–	0.448×10^{-4}	6.737×10^{-4}
8	0.2	0.2	0.2	0.2	0.1878390	0.1317657	–	0.448×10^{-4}	6.737×10^{-4}

Numerical optimization

To determine the optimal management, while paying regard to any given constraints in the management scenarios, we solved the optimization problem numerically. For this, the dynamic optimization was performed using the interior-point algorithm of the Knitro (Version 8.1) optimization software with Matlab (R2012b), as well as AMPL.

Transition dynamics

The transition path (i.e. short-term effects) towards the long-term management goal might be crucial for acceptance of that goal by the fisheries. Even if the long-term goal is accepted, the transition dynamics (e.g. fishing restrictions) might confront the fisheries with severe problems due to anticipated short-term income losses. In addition to the long-term steady state, we investigated the short-term (2006 to 2012) transition dynamics under each scenario. Yearly fishery-specific and total profits were calculated.

Regional distribution of profits

The distributional effects of the 4 management scenarios were calculated by simulating the interacting stock dynamics, associated species-specific catch and cost data and finally country-specific future profits. Country-specific quota allocation followed the relative stability principle. Overall annual profits were calculated as the sum of profits of all 3 fisheries. To illustrate the long-term distributional effect we chose the reference year 2030, i.e. we chose a year after initial transition dynamics would have stabilized.

RESULTS

Stock development scenarios

Historic stock development shows a switch from a system dominated by cod *Gadus morhua* in the early 1980s to a clupeid-dominated system beginning in the early 1990s (Möllmann et al. 2009; Fig. 1a). Up until the year 2006 no signs of cod recovery were observed.

In Management Scenario 1, we applied an unconstrained economic optimization of the multi-species fishery (Fig. 1b). This resulted in a fast rebuilding of the cod stock to ca. 700 000 t of SSB. The herring *Clupea harengus* stock increases in parallel to >1 million t of SSB, while the sprat *Sprattus sprattus* stock is reduced to 212 000 t of SSB, due to the strong predatory impact of the large cod stock. This level of sprat SSB is well below the recently defined precautionary biomass limit for impaired recruitment (570 000 t; ICES 2013).

In Scenario 2, the precautionary biomass level for sprat (sprat B_{PA}) was used as a constraint. Keeping a minimum of 570 000 t sprat SSB reduces the optimal cod SSB to 329 000 t. Optimal herring stock size reached almost 1.4 million t (Fig. 1c). Optimal fishing

Table 4. Parameters of stock-recruitment functions, obtained by either fitting a model to data from 1974 to 2011 as provided by ICES (2012) for herring, or functions by Quaas et al. (2012) for cod and Tahvonen et al.(2013) for sprat. Functions noted in parentheses

	φ_i	β_i
Cod (Ricker)	1.7	0.00182
Herring (Ricker)	30.33	0.000469
Sprat (Beverton-Holt)	104.2	0.5032

Table 5. *Gadus morhua*. Age-specific European reference prices for cod ((EC, 1999, 2009)

Age class	Price (€/kg)
1	0
2	0.35
3	0.35
4	0.35
5	0.477
6	0.477
7	0.636
8	0.731

mortality is considerably higher for all 3 species than in Scenario 1 (Table 6).

Setting a precautionary cod SSB (cod B_{PA}) of 88 000 t (ICES 2013) as a constraint, while otherwise optimizing profits from the clupeid fishery (Scenario 3), increasingly emphasizes the role of herring and sprat (Fig. 1d). The precautionary stock size of cod could be maintained at a fishing mortality as high as F = 0.9; the cod fishery would, however, be unprofitable (Table 6). SSB of herring and sprat peak at 1.5 million and 1.2 million t, respectively. This illustrates the broad range of strategic management options, while still accounting for ecological constraints.

The equilibrium stock sizes when simulating the cod management plan (Scenario 4) resemble the solution for the economically optimal solution (Fig. 1e). The cod stock increases slightly with corresponding smaller clupeid stock sizes as a result of intensified cod predation (Table 6). The target fishing mortality under the cod management plan (F = 0.3) is slightly lower than the steady-state fishing mortality under economically optimal management (F = 0.35).

The combined profits from all 3 fisheries are by far the highest if the cod stock is rebuilt (Table 6; Scenarios 1 and 4). The economically optimal management plan is only slightly better in terms of combined profits compared to the simulated cod management plan. (The difference in present-value terms is a bit larger because of the faster transition to an optimal steady state under optimal management.) According to our model, the cod fishery would not be profitable (zero profits) in Scenario 3, i.e. if the fishing mortality of

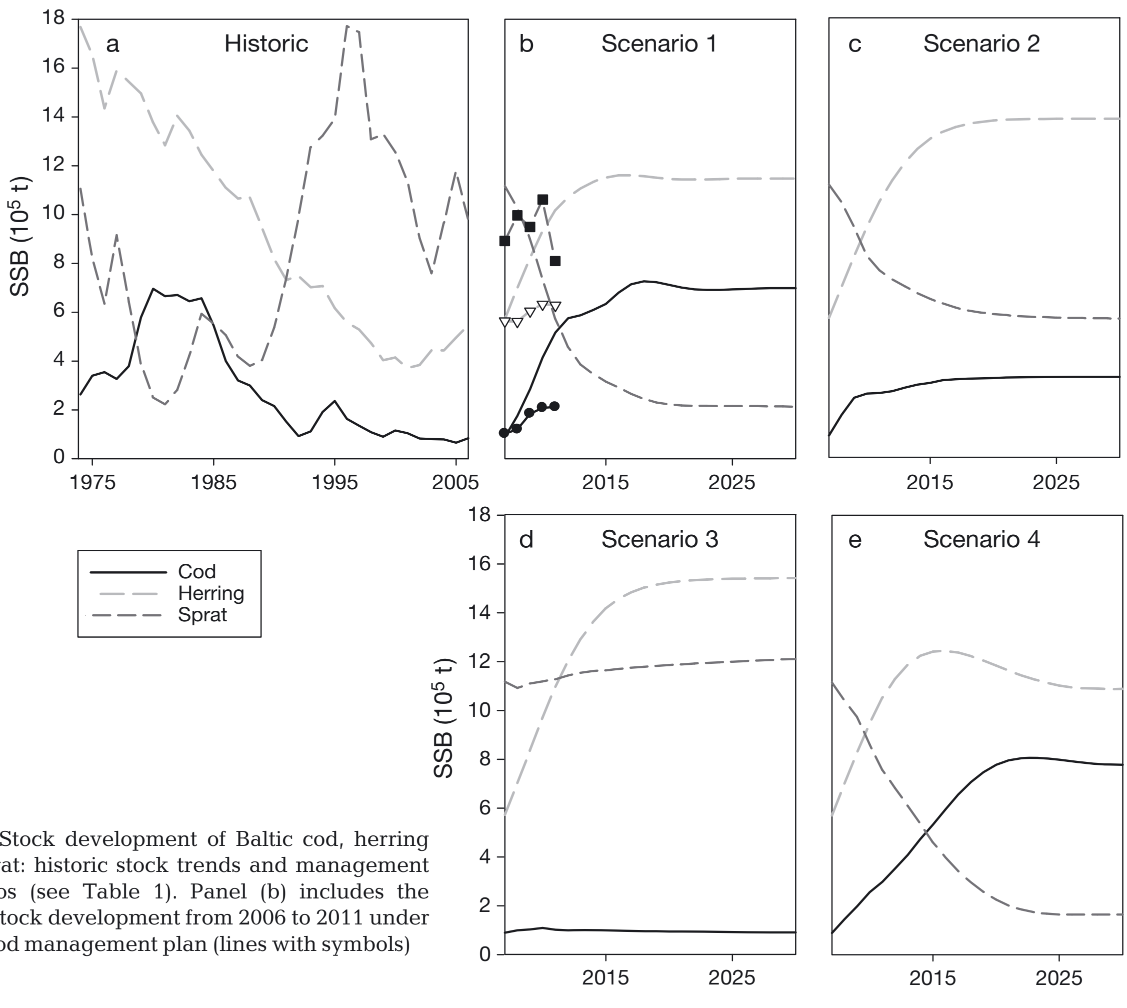

Fig. 1. Stock development of Baltic cod, herring and sprat: historic stock trends and management scenarios (see Table 1). Panel (b) includes the actual stock development from 2006 to 2011 under the cod management plan (lines with symbols)

Table 6. Projected profit (million € yr^{-1}), spawning-stock biomass (1000s of t) and fishing mortality for the year 2030, for the 4 selected long-term management goals (Scenarios 1 to 4, see Table 1). Values refer to 0% (5%) interest rate

Scenario	1	2	3	4
Profit				
Cod	97.5 (96.3)	43.8 (27)	0	99.8 (99.8)
Herring	17.7 (17.1)	26.1 (26.9)	32.4 (31.5)	15.7 (15.3)
Sprat	2.6 (2.9)	8.2 (9.6)	15.4 (15)	1.8 (1.8)
Sum	117.8 (116.3)	78.1 (63.5)	47.8 (46.5)	117.3 (116.9)
Spawning-stock biomass				
Cod	698 (689)	329 (264)	89 (89)	777 (777)
Herring	1146 (878)	1386 (1164)	1540 (1280)	1088 (805)
Sprat	212 (195)	568 (565)	1209 (965)	164 (130)
Fishing mortality				
Cod	0.35 (0.36)	0.67 (0.76)	0.9 (0.9)	0.3 (0.3)
Herring	0.18 (0.23)	0.23 (0.29)	0.26 (0.31)	0.17 (0.23)
Sprat	0.45 (0.58)	0.59 (0.74)	0.49 (0.66)	0.4 (0.51)

cod is not reduced considerably below the 2002 to 2006 level ($F_{2002\text{-}2006}$ = 0.93). A reduced cod stock would, however, result in higher profits for the herring and, especially, sprat fishery.

Setting the interest rate to 5% instead of 0%, as in our reference case, reduces steady-state biomasses as well as profits by maximal values of 24% (biomass) and 19% (profits). Fishing mortality is generally slightly higher. The results, however, are not qualitatively changed.

Transition dynamics

According to the economically optimal management plan (Scenario 1), the cod fishery would have been closed for almost 3 yr (F = 0.04 in third year). Afterwards, a gradual increase to the steady-state value of F = 0.35, with a concurrent strong increase in profits, would have been allowed (Fig. 2). Cod stock rebuilding under the cod management plan (Scenario 4) would take a few years longer, as fishing is kept up at the reduced rate of F = 0.3. Scenarios 2 and 3 allow for continued cod fishing at moderate (Scenario 2: sprat B_{PA}) or high levels (Scenario 3: cod B_{PA}). In Scenario 3 the cod fishery stays, however, unprofitable. Due to the bad stock status and the unprofitability of the cod fishery in 2006, all scenarios offer monotonically increasing total profits, even when the cod fishery is closed at the beginning.

Regional distribution of profits

In 2006, the regional distribution of profits mirrored the country-specific catch shares of herring and sprat, as the cod fishery was unprofitable, due to the severely overfished state of the cod stock in that year. The highest profits were gained in Poland and Sweden (Fig. 3a). Under the economically optimal management (Scenario 1), all countries would benefit (Fig. 3b, Table A1 in the Appendix). However, the amount of increase in benefits is very different regionally. Countries in the east, with a small share of the cod quota, e.g. Finland and Estonia, only gain a little, while countries in the west, e.g. Denmark and

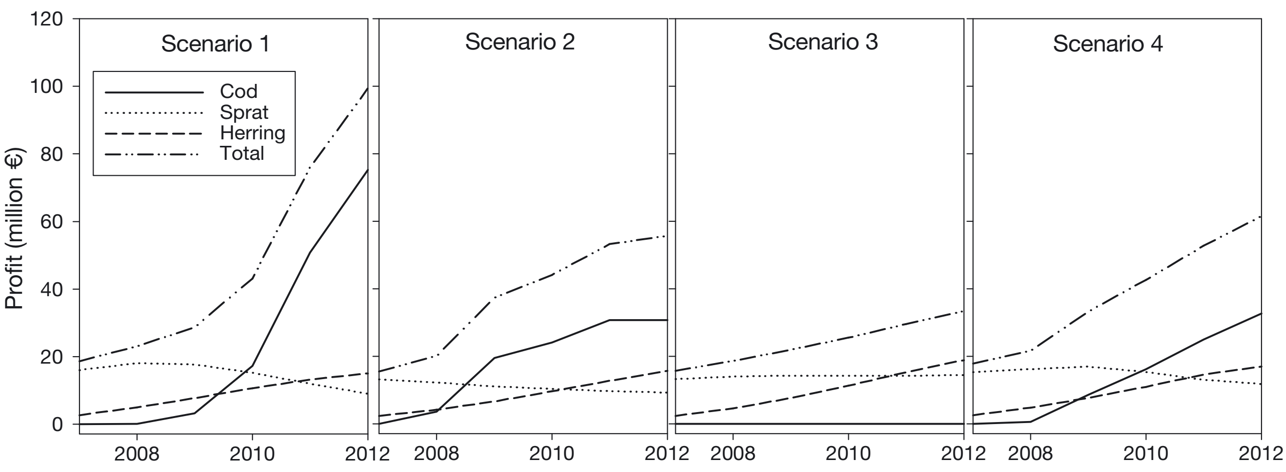

Fig. 2. Transition dynamics: path of fishery-specific, as well as total, profits (used as a performance indicator) from 2006 to 2012 for the 4 management scenarios: (1) unconstrained economic optimization, (2) optimization for profits of the cod fishery, while respecting a precautionary sprat spawning-stock biomass (SSB) (sprat B_{PA}) of 570 000 t, (3) optimization for profits of the sprat fishery, while respecting a precautionary cod SSB (cod B_{PA}) of 88 000 t, (4) simulation of the cod management plan

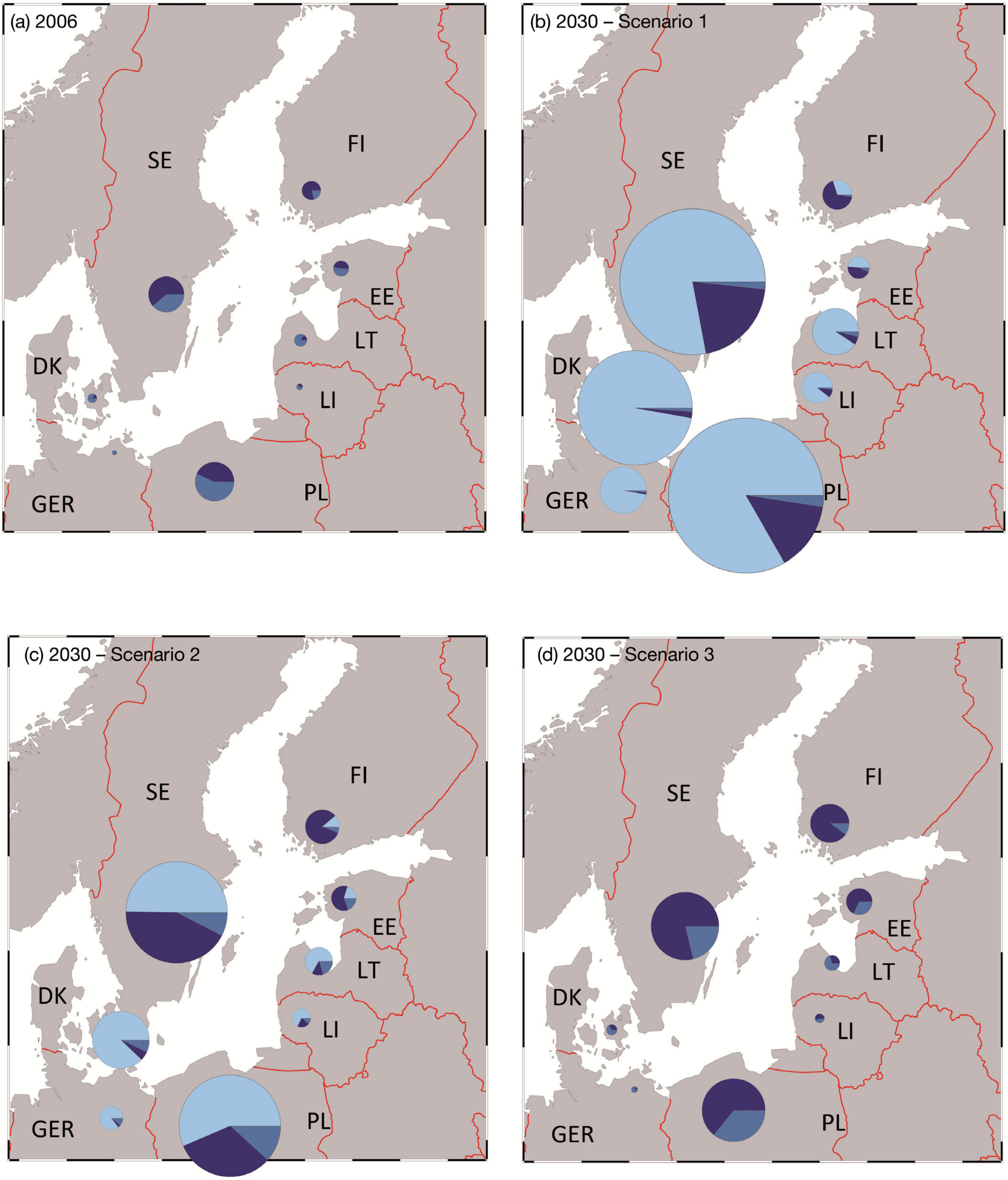

Fig. 3. Regional, country-specific distribution of profits from the cod (light shading), herring (dark shading) and sprat (medium shading) fishery; top left: situation at the beginning; remaining panels: distribution of the profits in the year 2030 according to the Scenarios. Actual values are given in Table A1 in the Appendix

Germany, realize the highest relative increases in profits. Currently quotas cannot be traded internationally. If such an international quota trade were introduced, these results would remain the same, as the market value of quotas corresponds to the (potential) profits that can be gained in the respective fishery.

Scenarios 2 and 3 respect ecological precautionary points and can be seen to set the boundaries for feasible management options (Fig. 3c,d). Within these constraints most countries realize the highest profits under the sprat B_{PA} scenario (Scenario 2); some other countries, however, would benefit more from a cod B_{PA} scenario (i.e. Estonia and Finland).

The different management options do not only cause an uneven regional profit increase, but they also imply social consequences within a certain country, as a re-distribution of profits between the different fisheries occurs (Table A1), e.g. Scenarios 1 (economically optimal management) and 4 (cod management plan) create high total profits; the sprat fishery, however, loses substantially.

DISCUSSION

We have shown that, in a multi-species set-up, different strategic management goals will result in regionally unequal distributions of future profits. Rebuilding a large predator (cod) stock will penalize countries holding the larger shares of forage-fish fishing rights. Unconstrained economic optimization would lead to a sprat stock size below commonly accepted ecological reference levels. Even when respecting precautionary stock size limits, there are many strategic management goals from which to choose. The inflexible system of distribution of catches according to the principle of relative stability in combination with species interactions might require new measures of compensation, to secure future acceptance and compliance by all states — no matter which strategic goal is chosen.

According to the relative stability principle, the Baltic countries hold fixed shares of cod, herring and sprat quota. Therefore, the absolute catch amounts may differ between years depending on the stock status, but not the percentage distribution of TACs to countries. All Baltic countries are involved in all 3 fisheries, however, with highly variable distributional shares to species. Poland holds the largest share of the cod (26.5 %) as well as the sprat quota (29.4 %). Sweden owns the largest share of the herring quota (33.4 %). The sum of allowable catches over all 3 species in 2006 (the start of our simulation) differed between ~164 000 t (Poland) and ~27 000 t (Lithuania). The composition of each country's catch portfolio should determine its interest in (or opposition to) the future multi-species management goals, in particular to change from a clupeid-dominated system back to a cod-dominated system (Möllmann et al. 2009).

Stock assessment in the Baltic Sea is regularly performed by the Baltic Fisheries Assessment Working Group. Its work is currently somewhere between single-species and multi-species assessment, as natural mortality of the clupeid stocks is calculated depending on the size of the cod stock, but the group does not explicitly provide multi-species advice (ICES 2012). In early 2013, a real multi-species assessment was provided for the first time by the Benchmark Workshop on Baltic Multi-Species Assessment (WKBALT) (ICES 2013), highlighting the ecological management trade-off. Species interaction, i.e. cod predation on clupeid species, is generally of high importance in stock forecast scenarios (Kellner et al. 2011). In such an environment, single-species projections (e.g. Froese & Proelss 2010) might be too optimistic, or even misleading. Even in the multi-species literature, economic aspects of management, especially regional distribution of profits, are rarely considered, and were not explicitly addressed in the work of the WKBALT as they were not part of the terms of reference. As shown here, such economic considerations might be as critical as ecological constraints, as they will ultimately have an influence on agreements and compliance with future management decisions.

Stock rebuilding plans can produce trade-offs due to species interactions (Gröger et al. 2007), which have to be communicated to stakeholders. Standard management trade-offs include trade-offs between harvest and spawner abundance (Collie et al. 2012), stock biomass and net financial returns (Little et al. 2011), or species conservation and size of marine protected areas (McClanahan 2011). From a more integrated point of view, trade-offs between restoration goals might be of interest (North et al. 2010), with the aim to predict benefits and quantify the associated costs. One of the few existing studies that take into account the economic impact of species interaction is on Pacific sardine *Sardinops sagax* (Hannesson et al. 2009). Contrary to our study, Hannesson et al. (2009) do not use an age-structured framework, and, therefore, their results cannot be directly translated into an ICES stock assessment. In the Baltic Sea, management that prioritizes profits will result in relative winners (cod fishery), but also in relative losers (sprat fishery). The system's dynamics are mainly driven by the cod stock: The range of optimal fishing mortality (F) for herring, sprat and cod is relatively narrow between scenarios (herring: 0.17 to 0.26; sprat: 0.4 to 0.59), but for cod optimal F ranges from 0.3 to 0.9 in steady state. In the model, the economically profitable cod stock is rapidly built up to a SSB of ~700 000 t. In reality stock rebuilding is largely dependent on recruitment success. Under unfavorable environmental conditions stock rebuilding might take longer. A high cod stock causes increased predation on sprat and herring, thus leaving less scope for improved catches from the clupeid fisheries. Cod stock recovery in the optimization model is faster than has been observed in reality after adoption of the management plan, i.e. from 2006 to 2011. This is due to a sharp reduction in catches from the cod fishery for 3 yr in the economically optimal solution, which was not conducted in reality. Instead, the management plan aims at smoothing variations by including maximum year-to-year variations of 15% (EC 2007). The steady-state biomass of cod is lower in the economically optimal solution than in the simulation of the cod management plan. Accordingly, a consequent realization of the long-term management plan for cod might lead to sprat stock sizes falling below ecological reference points, and might therefore need to be revised under multi-species- or even ecosystem-based (Pikitch et al. 2004) management.

Our model framework has room for improvements, in particular regarding environmental influences on recruitment (Köster et al. 2009), density-dependent growth (Casini et al. 2011, Gårdmark et al. 2013) and processes accounting for changes in the spatio-temporal overlap of cod and sprat (Eero et al. 2012). Nevertheless, we are confident in the range of simulated outcomes. Although future developments will most certainly increase the quantitative precision of simulations, the qualitative implications for management will likely remain robust. The Baltic Sea represents a suitable case study for demonstrating the principles of trade-off evaluation in multi-species fisheries. We are confident that our approach is readily transferrable to more complex systems, since reliable coupled ecological–economic models are increasingly becoming available.

Including ecosystem considerations other than commercially exploited species and quantifying the potential economic impacts might require valuing of, e.g., endangered species (Wallmo & Lew 2011). Taking all ecosystem and economic feedbacks into account might also reveal unforeseen trade-offs and externalities, like in the case of French Guiana, where the (economically sub-optimal) oversized trawl fishery for shrimps positively impacts endangered frigatebirds *Fregata* spp. (Martinet & Blanchard 2009). In the case of the Baltic Sea, we envisage a broadening of the scope of future work to an ecosystem level. This can be achieved by coupling to ecosystem models, in the sense of ensemble modeling as recently advocated by the Working Group on Integrated Assessments in the Baltic (Gårdmark et al. 2013).

Successful management in the future will require stakeholders to explicitly define commonly accepted fishery objectives against which trade-offs can be evaluated (Pilling et al. 2008). If quotas for the 3 species are not equally distributed among countries (like in the Baltic), new compensation schemes may be required to come to international agreements—ultimately the principle of relative stability may need to be abandoned. Additional pressure to revise the relative stability principle might also arise from the interaction of economic dynamics of fishing fleets and the European Community's legal framework, e.g. the right of establishment or free movement of workers (Morin 2000).

When performing model runs over periods of decades, it might become important to include climate change aspects, as reproductive success of all species has been shown to strongly depend on environmental conditions (cod: Köster et al. 2005; herring: Cardinale et al. 2009; sprat: Voss et al. 2006). Our model framework offers the possibility to simulate climate change scenarios for the most important environmental factors, and significantly different results may be obtained when comparing climate change to non-climate change scenarios (Voss et al. 2011). Besides the modeling of biological interaction and variability in physical forcing factors, assumptions on economic variables like cost functions, interest rates and prize elasticity also have a strong impact on the results (Voss et al. 2011, Tahvonen et al. 2013). Therefore, our data emphasize the need to proceed to inter-disciplinary multi-species management.

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Appendix

Table A1. *Gadus morhua*, *Clupea harengus*, *Sprattus sprattus*. Country- and fishery-specific profits (million € yr^{-1}) in the year 2006 (base year), as well as in year 2030, for the 4 selected long-term management goals (see Table 1). Values refer to 0% interest rate

	Finland	Sweden	Denmark	Germany	Poland	Estonia	Latvia	Lithuania
2006, base								
Cod	0	0	0	0	0	0	0	0
Herring	2.9	4.4	0.3	0.1	3.3	1.5	0.4	0.4
Sprat	0.8	2.8	1.5	0.9	4.3	1.7	2	0.7
Total	3.7	7.2	1.7	1	7.6	3.2	2.4	1.1
Scenario 1, econ. opt.								
Cod	1.7	22.7	22.4	8.9	25.8	2.2	8.3	5.5
Herring	3.9	5.9	0.4	0.1	4.4	2	0.5	0.5
Sprat	0.1	0.5	0.3	0.2	0.8	0.3	0.4	0.1
Total	5.7	29.1	23	9.2	31	4.5	9.2	6.1
Change	1.5	4	13.5	9.2	4.1	1.4	3.8	5.6
Scenario 2, sprat B_{PA}								
Cod	0.8	10.2	10.1	4	11.6	1	3.7	2.5
Herring	5.7	8.7	0.6	0.2	6.5	2.9	0.7	0.8
Sprat	0.4	1.6	0.8	0.5	2.4	0.9	1.1	0.4
Total	6.9	20.5	11.4	4.7	20.5	4.9	5.6	3.6
Change	1.9	2.8	6.7	4.7	2.7	1.5	2.3	3.3
Scenario 3, cod B_{PA}								
Cod	0	0	0	0	0	0	0	0
Herring	7.1	10.8	0.7	0.2	8.1	3.6	0.9	0.9
Sprat	0.8	2.9	1.5	1	4.5	1.8	2.1	0.8
Total	7.9	13.8	2.2	1.1	12.6	5.4	3	1.7
Change	2.1	1.9	1.3	1.1	1.7	1.7	1.3	1.5
Scenario 4, cod mgmt								
Cod	1.7	23.2	22.9	9.1	26.4	2.2	8.5	5.6
Herring	3.4	5.3	0.3	0.1	3.9	1.8	0.4	0.5
Sprat	0.1	0.4	0.2	0.1	0.5	0.2	0.3	0.1
Total	5.3	28.8	23.5	9.3	30.9	4.2	9.2	6.2
Change	1.4	4	13.8	9.3	4.1	1.3	3.8	5.6

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