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Original Article

Integrating stochastic age-structured population dynamics into complex fisheries economic models for management evaluations: the North Sea saithe fishery as a case study

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There is growing interest in bioeconomic models as tools for understanding pathways of fishery behaviour in order to assess the impact of alternative policies on natural resources. A model system is presented that combines stochastic age-structured population dynamics with complex fisheries economics. Explicitly, the economic response of fleet segments to changes in stock development is analysed by applying observed values and stochastic recruitment. The optimization of net profits determines the fishing effort and the investment and disinvestment behaviour of fleet segments, which, in turn, affect the level of catch rates and discards. This tool was applied to the North Sea saithe fishery, where ICES re-evaluated the existing EU–Norway management plan, focusing on biological reference points only. Two scenarios were tested with alternative harvest control rules and then contrasted with one unregulated scenario with no quotas and driven by optimizing the net profit of the whole fleet. The model showed the success of both harvest control rules in rebuilding the stock and the associated costs to the fleets in terms of maximal 21% reduction in net profits, 21% reduction in crew wages and 11% reduction in fleet size in the midterm (2007–2015). In the long term (2022), successful stock recovery coincided with net profits almost equalling that of the unrestricted fishery. The model is highly sensitive to the parameter values but can be used strategically, providing a qualitative understanding of the anticipated relative changes.

Keywords: bioeconomic modelling, fishery management, gadoid species, impact assessment.

Introduction

Population dynamics of fish stocks in the North Sea are estimated based on short- and long-term prediction models that are often parameterized with data derived from surveys and commercial catch data (ICES, 2012a). This analysis is conducted by working groups of the International Council for the Exploration of the Sea (ICES) and the Scientific, Technical and Economic Committee for Fisheries (STECF). The model outputs form the basis for scientific management advice in the framework of the European Common Fisheries Policy (CFP). Although understanding and anticipating fisher response to changes in biological, economic and regulatory conditions in fisheries is critical in designing management plans that will sustain resources and fishing activities (Béné *et al.*, 2001), the advice given by ICES is mainly based on operational assessment models that do not account for fisheries economics (ICES, 2012a). Ideally these models would include short- and long-term fleet dynamics, such as effort distribution and entry–exit behaviours, as they influence the fishing mortality. Fleet dynamics are driven by revenue that depends on fish prices and variable costs (such as fuel cost) and greatly influence short-term effort distribution between fisheries. In addition, the profit of a fleet segment will affect the investment or disinvestment behaviour and thus the long-term development of the targeted fish stocks. The fishery system comprises a dynamic interplay between the biological and economic parts of the system. When trying to understand a fishery, it is essential to take account of biological and economic pressures.

Unfortunately, existing bioeconomic models often focus on either the biological component or the economics and hence capture only some of the relevant feedbacks. For instance, the model developed by Da Rocha *et al.* (2010), which was applied to evaluating recovery plans, assumed costs to be a source of uncertainty and fleet size (the number of vessels participating in the fishery) to be constant. Similarly, Pelletier *et al.* (2009) developed the "ISIS-Fish Model" to evaluate the bioeconomic sustainability of multispecies, multifleet fisheries under a range of policy options but did not include the age-structure of the populations and the entry or exit of vessels. In another model developed by Poos *et al.* (2010), marketable fish was represented as a homogeneous group, whereas, in real fisheries, the marketable catch consists of several size classes that may differ in value and directly affect economic performance and the related fleet dynamic consequences. Moreover, Naqib and Stollery (1982) developed a bioeconomic model of multicohort fisheries but included neither the distribution of effort over years nor any stock–recruitment relationship. The latter is crucial, as recruitment is likely to decline, or fail, if stock size is reduced too far (Shepherd, 1982). They did not consider any type of investment or disinvestment behaviour, which is important as this can indirectly influence fishing mortality. To ensure the sustainability of fisheries, maintain incomes, and preserve regional communities that depend on fishing, it is crucial to understand both biological and economic mechanisms linking fish stocks and fisheries as comprehensively as possible (Stephenson and Lane, 1995).

This study combines stochastic age-structured population dynamics of the stock with a detailed representation of the economy of fleets into one model. Specifically, the model integrates essential aspects of those two components and includes both the economics of fleet segments in terms of fish and fuel prices, fixed and variable costs, and fleet adjustments, as well as the biology of the fish stock in terms of individual growth, maturity, variable recruitment, spawning stock biomass (SSB) and instantaneous mortality rates. The approach is based on a bioeconomic optimization and simulation model called "FishRent" (originally developed during the EU-funded project "Renumeration of Spawning Stock Biomass" by Salz *et al.*, 2011). Compared with the models mentioned previously, the basic version of FishRent is an advanced model from the economic point of view because it includes prices, costs, and fisher behaviour, in terms of investment, disinvestment and fishing effort distribution between fleet segments for a long period of time (Salz *et al.*, 2011). This basic version of FishRent was extended by replacing the Schaefer model (Schaefer, 1957), which was a simple deterministic stock growth production function, with a dynamic age-structured population model that accounts for stochasticity in the stock–recruitment relationship. The model was applied to the North Sea saithe fishery, where ICES re-evaluated the current management plan in 2012.

Material and methods

The North Sea saithe fishery

Saithe (*Pollachius virens*) is of major economic importance for North Sea fisheries, with annual landings values of around 15 million Euros (Anderson and Guillen, 2009). It is targeted by Norwegian, French, German, British, Danish and, to a small extent, Swedish trawlers (ICES, 2012a). There is an EU–Norway long-term management plan for North Sea saithe. This plan involves a Harvest Control Rule (HCR) based on annual Total Allowable Catches (TACs), and reference points. B_{lim} is a reference point for SSB, below which there is a high probability that recruitment is impaired (Lassen and Medley, 2001; ICES, 2010). B_{pa} is the precautionary reference point for SSB, below which the stock would be regarded as potentially overfished (Lassen and Medley, 2001; ICES, 2010). F_{tar} is the target fishing mortality for age class 3–6 (Lassen and Medley, 2001; ICES, 2010). In the long-term management plan for North Sea saithe F_{tar} is set to 0.1 ($F_{tar-low}$) when SSB is estimated to be below the minimum level of 106 000 t (B_{lim}) (ICES, 2013). Usually the fishing mortality is ~0.4, therefore an F_{tar} of 0.1 is a large reduction to allow SSB to recover. Where SSB is above 200 000 t (B_{pa}), the parties have agreed to restrict fishing on the basis of a TAC consistent with a target fishing mortality of 0.3 (F_{tar-up}) (ICES, 2013). In the case where SSB is estimated to be between B_{pa} and B_{lim} the target fishing mortality rate ($F_{tar-mid}$) is calculated as:

$$F_{tar-mid} = F_{tar-up} - (F_{tar-up} - F_{tar-low}) \times \frac{(B_{pa} - SSB)}{(B_{pa} - B_{lim})} \quad (1)$$

Another element of the plan is that the annual TAC should not vary by more than 15% (ICES, 2013). Although there exists a long-term management plan, SSB of saithe has declined in the last few years and is currently close to $B_{trigger}$ (ICES, 2012a), which is inside the Maximum Sustainable Yield (MSY) framework of ICES. $B_{trigger}$ is the value of SSB that triggers specific management actions in order to avoid a further decline of the stock in regions with increased probability of a stock collapse (ICES, 2012a). Besides the declining SSB values, saithe has exhibited lower growth rates and recruitment has been below average since 2006 (ICES, 2012a). These factors, when taken together, indicate a decline in stock productivity. This questions the sustainability of the HCR with its target fishing mortalities and constrained change of TACs (the restriction on the maximum interannual change of TACs allowed) used in the management plan for North Sea saithe (ICES, 2012b). At the moment, the annual TAC is not allowed to vary by more than 15%. Only if it is considered to be appropriate can the parties agree to abolish the constraint on TAC change (ICES, 2012b). At the moment, B_{pa} is used as the reference point that triggers a decrease in the annual TAC by more than 15%. ICES re-evaluated the management plan in 2012 using a standard Management Strategy Evaluation (MSE) approach (ICES, 2012b). However, there has never been an impact assessment for the plan that takes into account fisher behaviour or the economic performance of the fleet under different management plan options and scenarios. The model presented here was developed for such a purpose, as it integrates the biological component, fisher behaviour and the economics of fleet segments. The model was successfully applied to the North Sea saithe fishery to evaluate whether B_{lim} or B_{pa} is a more appropriate reference point for an annual TAC adjustment by more than 15%.

Scenarios

Simulations of an unregulated case, which represents a fishery without quotas only driven by optimizing the net profits of the whole fleet, were contrasted with two alternative HCRs (see Table 1), where the current HCR of North Sea saithe was modelled with the two reference points (B_{lim} and B_{pa}) referred to as HCR_{Blim} and HCR_{Bpa}. The unregulated case facilitated the assessment of potential economic costs and benefits that may occur due to the

Table 1. Scenario description.

Scenario	Description
Unrestricted fishery	Neither a TAC nor a target fishing mortality rate was applied. Exclusively driven by maximizing net profits of the fishery
HCR B_{lim}	Considering the target fishing mortality rate, and a 15% constraint for annual TAC adjustments if SSB is at or above B_{lim}
HCR B_{pa}	Considering the target fishing mortality rate, and a 15% constraint for annual TAC adjustments if SSB is at or above B_{pa}

implementation of one of the HCR options. The first HCR option includes a 15% constraint on the annual TAC change if SSB is at or above B_{lim}. Such an option might stabilize catches, as the TAC cannot vary by more than 15%, unless SSB drops below B_{lim} (106 000 tons). The other HCR option includes a 15% constraint on the annual TAC change if SSB is at or above B_{pa} in order to account for uncertainties and to ensure that the probability of a stock collapse is low (ICES, 2010). This option might facilitate a more immediate response to stock development, especially when SSB is declining below B_{pa} (200 000 tons), but could lead to less interannual stability in fishing opportunities. Target fishing mortality rates of both HCRs were modelled according to the long-term management plan for North Sea saithe (Lassen and Medley, 2001; ICES, 2010).

Settings

The model was run for a period of 16 years (2007–2022). For the years 2007–2009, low recruitment values as observed in the official assessment from 2012 (ICES, 2012a) were used. For the following years, recruitment was predicted based on stochastic simulations applying a Beverton and Holt stock–recruitment relationship (Figure 1). This kind of stochasticity was added to the originally deterministic model, because recruitment failure is an important driver of the North Sea saithe fishery right now. The model accounted for six fleet segments covering vessels from Denmark, England, France and Germany that fished North Sea saithe either as the main target species or an important bycatch species. According to the Data Collection Framework (DCF), fleet segments were classified by vessel length and predominant gear type (European Commission, 2010). The calibration of the model was based on average biological and economic data for the period 2005–2007 (Anderson and Guillen, 2009; ICES, 2012a).

Model description

The presented modelling approach is based on a bioeconomic optimization and simulation model called "FishRent" (Salz *et al.*, 2011). It is a dynamic feedback model with annual time-steps, including independent procedures for the stock development (e.g. growth, recruitment and mortality), the catch, the effort distribution, and the investment behaviour. The economic performance of individual fleet segments can be compared with each other over a long period of time (e.g. 50 years). The model is composed of six submodules (Figure 2). It is a model of a fishery system that focuses on the economic drivers, among which the profit earned by the fleet segments is the main driver. Profit depends on the amount of landed fish, prices for the landed fish, the costs of fishing, and on the interest rate for capital invested in the fleet. It is presumed that effort in realistic settings responds to economic incentives. In particular, it is assumed that fleet segments seek to maximize their profits by setting an optimal level of fishing effort, which in turn affects the commercial fish stock. Each year, the applied CONOPT solver [for the detailed description of the CONOPT algorithm see Drud (1991)] finds the optimum levels of fishing effort for each fleet segment (within the historical minimum and maximum values of fishing effort) that maximize the total net profit of the fleet. Based on the calculated profits from the two years prior to a particular year, the model determines the level of investment or disinvestment in the fleet [for details see *Behaviour submodule* or (Salz *et al.*, 2011)]. Given that free access in the fisheries is allowed, any fleet segment that is highly profitable will become bigger, and hence the profit of the individual vessels would dissipate in the long term. The idea of fishers responding to economic incentives with effort allocation is supported by several studies (Bockstael and Opaluch, 1983; Robinson and Pascoe, 1997; Dorn, 1998). For instance, Bockstael and Opaluch (1983) provide empirical documentation showing that fishers adjust their effort in response to changes in expected returns. In the model, management constraint activities affect the stock and control the fishery. Simulations of changes in stock biology (e.g. changes in stock productivity), fisheries economics (e.g. changing fuel costs) and/or policy (e.g. alternative management strategies) can be conducted using the model. A full description of the basic version of the model can be found in Salz *et al.* (2011). The list of parameters and their estimation can be found in the Supplementary data.

Biological submodule

The Biological Submodule calculates the annual population dynamics of the stock. Individual fish grow according to the von Bertalanffy weight-at-age function (von Bertalanffy, 1938). For the case study, the parameters used in this function were estimated directly from weight-at-age data of the North Sea saithe stock (ICES, 2010). Once a year, stochastic recruitment (the number of age class 3 fish at the beginning of the year) was calculated via a Beverton and Holt stock–recruitment function (Beverton and Holt, 1957), which showed the best fit to stock recruitment data from 1967–2012 (ICES, 2013).

$$R_t = \frac{a \times SSB_t}{c + SSB_t} \times e^{(D \times CV - 0.5 \times CV^2)}, \quad (2)$$

with SSB as the overall SSB for saithe at the peak of the spawning period. The parameters a ($a = 190.9$) and c ($c = 76.3$) are species-specific and were estimated via the non-linear least-squares approach with data of the North Sea saithe stock (ICES, 2010, 2013). D is a standard normal deviation and CV is the coefficient of variation (CV = standard deviation/mean), estimated based on historical stock sizes at age 3 from 1967–2012 (ICES, 2013). Each time the stochastic recruitment model is employed, 1000 stochastic iterations are run and median recruitment and SSB values are taken for further calculations. This means that for each time-step, i.e. year, 1000 random iterations from the probability distribution in the stock–recruitment function are run. At the end of each year, all fish of *i*th age are moved to the next age class. All fish older than the maximum age are accumulated in the last age class (plus group at age 10). The catch calculated via a standard Cobb–Douglas production function (see *Interface submodule*) is used for Pope's approximate solution to the Baranov equation (Pope,

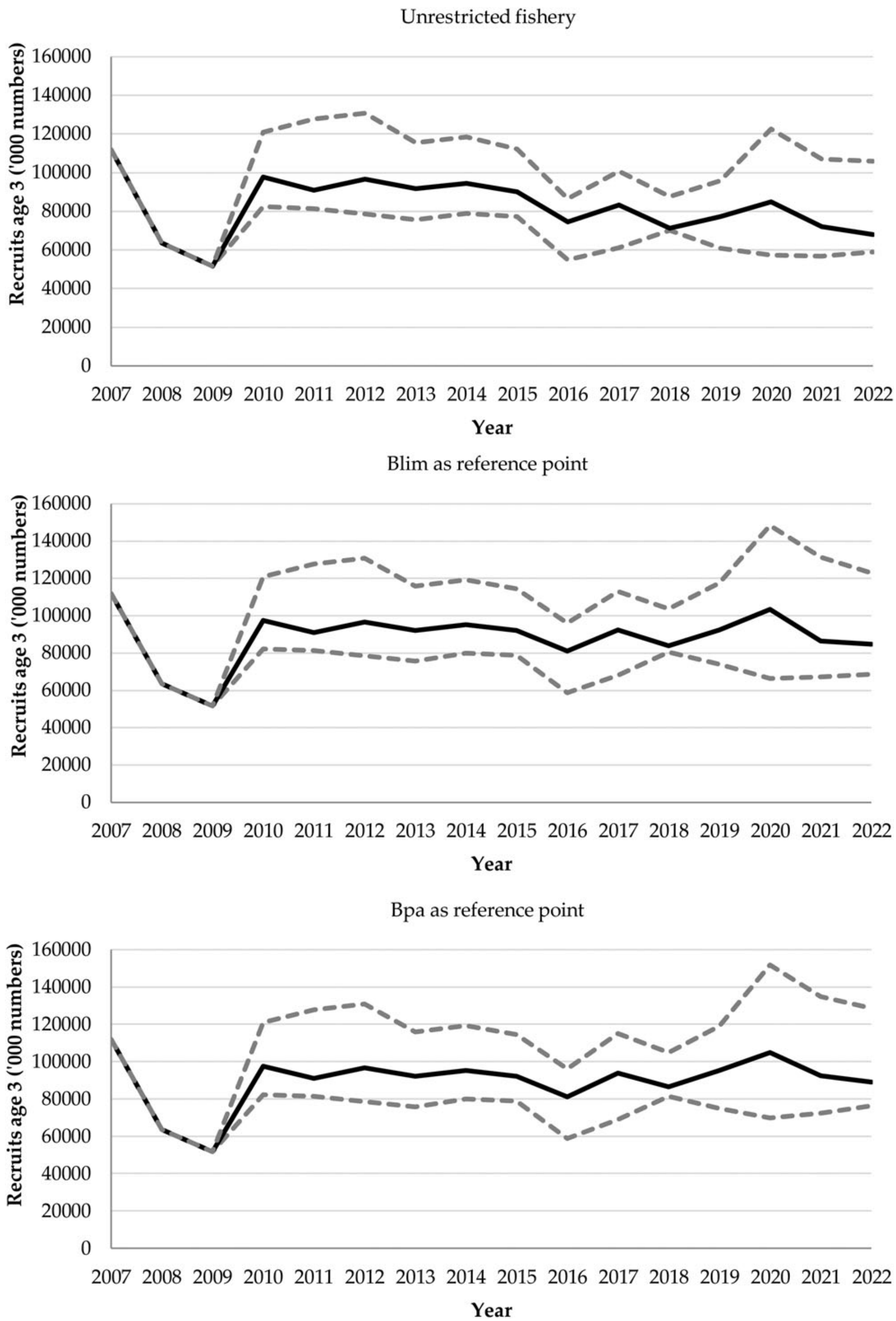

Figure 1. Predicted number of recruits of age 3 for the modelling period. From 2007 – 2009, observed recruitment values were used. From 2010 onwards, median recruitment values (solid lines) with 5 and 95% intervals (dotted lines) based on 1000 iterations are shown. Upper graph: unrestricted fishery; middle graph: HCR_{Blim}; lower graph: HCR_{Bpa}.

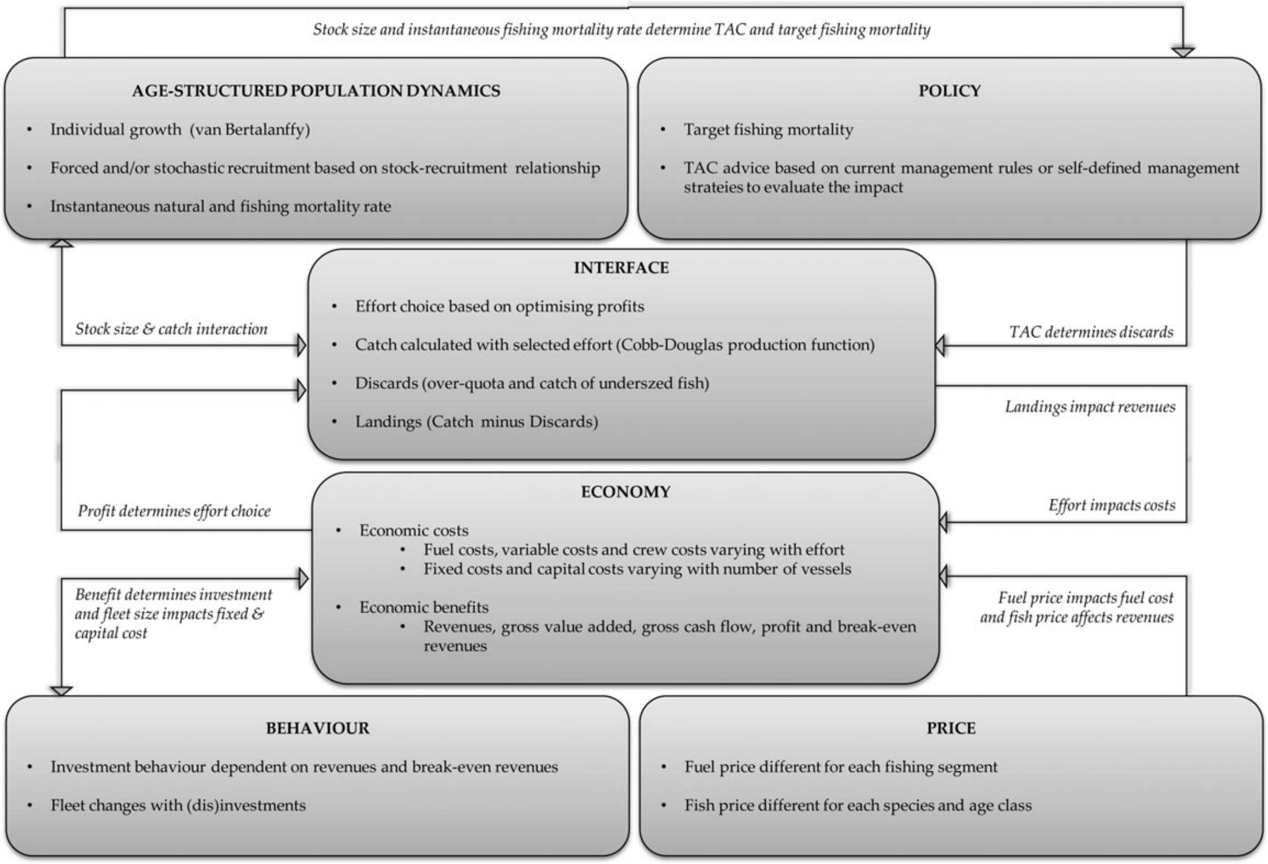

Figure 2. Conceptual model design with arrows that explain the interaction between the six submodules (age-structured population dynamics, policy, interface, economy, behaviour and price submodules).

1972) to calculate the number of individuals of ith age at time t,:

$$N_{t,i} = N_{t-1,i-1}e^{-M_i} - \sum_j \left(\frac{C_{t-1,i-1,j}}{s_{i,j}}\right) e^{-\frac{M_i}{2}}, \quad (3)$$

where $N_{t,i}$ is the number of fish of ith age at time t, $C_{t,i,j}$ is the catch in numbers of ith age and jth fleet segment at time$t$ and $S_{i,j}$ is the catch share for ith age and jth fleet segment (constant over time). The catch share serves to estimate the total catch of a species considering the catches of non-modelled fleet segments. M_i is the instantaneous natural mortality rate for ith age. The estimated number of individuals is then used in Equation 4 to calculate the age-specific instantaneous fishing mortality:

$$F_{t,i} = -\ln\left(\frac{N_{t,i}}{N_{t-1,i-1}}\right) - M_i. \quad (4)$$

Pope's approximation can be used in the Virtual Population Analysis (VPA) to avoid numerical estimation procedures. Moreover, as long as total mortality is below 1, it has been proven that Pope's approximation works well, leading to very small relative errors (MacCall, 1986; Lassen and Medley, 2001).

Policy submodule

The Baranov function (Baranov, 1918), including the target fishing mortality rate, is used in the Policy Submodule to determine the TAC for the next year. In particular, the Baranov model is referred to as a catch model, as it provides a catch estimate that is compared with a certain percentage (ts_t) of the TAC from the previous year (e.g. 85% of the TAC, if the TAC change constraint was 15%) (Equation 5). If this catch estimate is below or above the certain TAC level, the TAC for the following year is decreased (Equation 5a) or increased (Equation 5b), respectively, within the maximum allowed annual change. This maximum allowed annual change of the TAC is 15%, unless SSB drops below the reference point of HCR_{Blim} or HCR_{Bpa}, in which case the TAC can be changed by more than 15%. If none of the two options is true the TAC for the following year is calculated based on the Baranov catch model alone (Equation 5c). The TAC was calculated for the saithe fishery in the North Sea and Skagerrak if:

$$\sum_i \left[TSB_{t,i} \times \frac{\left(\frac{F_{t,i}}{Ftar_{t-1}}\right) \times Ftar_t}{Z_{t,i}} \times (1 - e^{-Z_{t,i}}) \right] < (1 - ts_t) \times TAC_{t-1} \quad (5)$$

$$TAC_t = (1 - ts_t) \times TAC_{t-1}, \quad (5a)$$

or if:

$$\sum_i \left[TSB_{t,i} \times \frac{\left(\frac{F_{t,i}}{Ftar_{t-1}}\right) \times Ftar_t}{Z_{t,i}} \times (1 - e^{-Z_{t,i}}) \right] > (1 - ts_t) \times TAC_{t-1}$$

$$TAC_t = (1 + ts_t) \times TAC_{t-1} \quad (5b)$$

or:

$$TAC_t = \sum_i \left[TSB_{t,i} \times \frac{\left(\frac{F_{t,i}}{Ftar_{t-1}}\right) \times Ftar_t}{Z_{t,i}} \times (1 - e^{-Z_{t,i}}) \right], \quad (5c)$$

where $TSB_{t,i}$ is the total stock biomass of ith age at time t calculated as the product of number of individuals and mean weight-at-age. $Z_{t,i}$ is the instantaneous total mortality rate calculated as the sum of instantaneous fishing and natural mortality rate.

Interface submodule

The Interface Submodule links the submodules together. In this submodule the levels of catch and effort are determined and enter into the Economic Submodule and into the Biological Submodule. The effort level in terms of fishing days is based on maximizing the sum of the net profits of the modelled fleet segments with a fixed quota allocation to each fleet segment mimicking the relative stability. The level of fishing effort that maximizes the overall profit under the given quota is used in a standard Cobb–Douglas production function to calculate the catch. The Cobb–Douglas production function is chosen to calculate the catch because it assumes a bi-non-linear relationship between the two inputs (fishing effort and total stock biomass) and the produced catch. In particular, two exponents (alpha and beta) are used as scaling factors for fishing effort and total stock biomass (Table A1). This is in contrast to the common assumption that fishing mortality is directly proportional to effort and that yield is proportional to stock size (Eide *et al.*, 2003). It is assumed that modelled fleet segments have a perfect knowledge of potential catch rates. As the catch in the model is estimated from the effort applied in the Cobb–Douglas production function, it is not necessarily equal to the quota. As long as the total catch of a species is less than the quota, the whole catch can be landed. When the total catch exceeds the quota, only the quota is landed and the catch above the quota is discarded.

Price submodule

Technically fish prices per age are included in the model but no further investigation was performed, as saithe fish prices do not significantly vary between age classes (Table A2). Fuel prices are fixed over time and were set at 60 Euro cents per litre.

Economic submodule

In the Economic Submodule, gross revenues for each fleet segment are calculated considering the landings value of the modelled species and also the landings value that comes from catches of other not explicitly modelled species (Equation A1, Table A1–A3 in the Supplementary data). Landings are the difference between catch and discard, whereas discard consists of over-quota catch and catch of undersized species (defined as a fixed proportion of the total catch). Net profit of a fleet segment is calculated as the gross revenue minus all economic costs (fuel costs, variable costs, crew costs, capital costs and fixed costs). The total net profit of all fleet segments is then maximized as described in *Model description*. In the model there is a differentiation between fixed and variable costs. Fixed costs include vessels costs (such as administrative costs, insurance and maintenance costs) and are directly proportional to the number of vessels, while variable costs are dynamic and are associated with variations in fishing effort. In the North Sea saithe fishery, crew costs are determined as a percentage of the difference between revenues and fuel costs. In the model, crew costs are calculated in the same way. Independently from the modelling approach, crew costs were used to estimate the average wage of a crew member. Hereby, the predicted crew costs were divided by the predicted number of vessels in a fleet segment. Consistent with the real saithe fishery, the calculated skipper wage is 8% of the predicted crew costs per vessel, the steersman's wage is 5%, and the wages for the rest of the crew (on average three members) is 4% per person. The estimated wages of the three crew members is particularly relevant, as it provides insights into the social effects of both HCR options. It is, in general, difficult to compare fishers' wages with wages in other sectors. So crew wages here were compared with two values: (i) the German gross unemployment benefit, serving as an indicator for the minimum wage that a skipper has to pay to a crew member, (ii) the average wage of a German crew member on a fishing vessel, assuming that a crew member would switch to another fishery/vessel if his wage is below that average wage (Federal Statistical Office, 2012/2013). Fuel costs vary directly with effort (Equation A2, Tables A1 and A3). They represent the most relevant cost item in fishing activities for most European fleets especially since a recent significant increase in the price of fuel has been one of the most critical factors for the profitability of fishing activities (Prellezo *et al.*, 2012). Capital costs involving depreciation and interest payments are defined as a fixed share of the number of vessels.

Behaviour submodule

The economic response of the fleet is modelled through a dynamic investment and disinvestment function (number of vessels), which evaluates the change in the fleet capacity given the economic outcome of the fishery two years ago (Equation 6). In reality, the investment/disinvestment function is based on future expectation, but because of the lack of information, past evidence (in terms of profitability) is used in the model. Thereby, the break-even-revenue ($BER_{t,j}$) is an important variable (Equation A3). It considers revenues and costs (with salary to the skipper/owner of the vessel included in the crew costs), and provides the value of gross revenue, where net profit is zero. It is assumed that the fleet changes, i.e. investment and disinvestment take place, proportionately to the relation between the break-even-revenues and the realized revenues. In particular, at the end of each year the number of vessels ($FLE_{t,j}$) in jth fleet segment is adjusted in terms of exit (Equation 6a) or entry (Equation 6b) of vessels depending on whether gross revenues ($R_{t,j}$) fall short of (unprofitable fishery) or exceed (profitable fishery) break-even-revenues of two years before.

This leads in some years to quite substantial changes in the number of vessels in a fleet segment, as vessels from other fleet segments may enter the fishery. However, it is recognized that the inertia of the system (e.g. licensing, knowledge of skippers) does not allow such full flexibility. Consequently, parameters have been introduced to limit the fluctuation in investment and disinvestment (change in the number of vessels). In particular, a maximum percentage of 10% in disinvestment ($d_j^{\max}$) and a maximum change of 5% in investment ($i_j^{\max}$) is applied (Equation 6). As these two limits are different, it creates asymmetric investment and disinvestment behaviour. To avoid a continuous growth of fleet size while vessels in the fleet segments have a low activity, the days-at-sea of a fleet segment ($DAS_{t,j}$) have to achieve a certain minimum level of days-at-sea per vessel ($das_j^{\min}$) before the fleet size can be expanded (Equation 6b, Table A3). This minimum level is based on the historical average level of days-at-sea for the modelled fleet segments.

If $BER_{t-2,j} > R_{t-2,j}$,

$$Inv_{t,j} = MAX\left[\begin{array}{c} d_j^{max} \times FLE_{t-1,j}, \\ \dfrac{R_{t-1,j} - BER_{t-1,j}}{R_{t-1,j}} \times FLE_{t-1,j} \end{array}\right]. \qquad (6a)$$

If $BER_{t-2,j} \leq R_{t-2,j}$ and $DAS_{t-1,j} < das_j^{min}$,

$$Inv_{t,j} = MIN\left[\begin{array}{c} i_j^{max} \times FLE_{t-1,j}, \\ \dfrac{R_{t-1,j} - BER_{t-1,j}}{R_{t-1,j}} \times FLE_{t-1,j} \end{array}\right], \qquad (6b)$$

where $Inv_{t,j}$ is the number of vessels that are entering (Equation 6a) or leaving (Equation 6b) the fleet/fishery.

Results

Simulation results

Stock development

Both HCRs, with either a TAC change constraint of 15% if SSB was at or above B_{lim} or a 15% TAC change constraint if SSB was at or above B_{pa}, enabled a successful recovery of the stock (Figure 3). In 2015, the probability of SSB being above B_{pa} was 30% for HCR_{Blim} and 60% for HCR_{Bpa}. The probability was higher for HCR_{Bpa} because the TAC was reduced by more than 15% when SSB dropped below 200 000 tons, allowing for a faster response to the declining stock. Since SSB dropped below B_{pa} in 2012, but never below B_{lim}, the TAC was reduced by more than 15% only for HCR_{Bpa} (Figure 3). For the unregulated case, SSB reached B_{lim} in 2020, implying a high probability of a stock collapse (Figure 3). In long-term simulations (2022) the probability of SSB being above B_{pa} was 80% for HCR_{Blim}, and 50% for HCR_{Bpa}. The probability was higher for the HCR_{Blim} scenario where the TAC was increased gradually during the rebuilding phase of the stock. For HCR_{Bpa} the TAC was increased by more than 15% while the stock was still recovering (Figures 3 and 4). As the model accounts for overquota catch, discarding behaviour of the modelled fleet segments could be investigated. In particular, catches often exceeded the quota when the TAC was reduced or when the stock size was high (Figures 3 and 5). However, both SSB and catches decreased sharply over 2007–2013, and a slight stabilization and increase occurred after 2013 (Figures 3 and 5). Catches of the unrestricted fishery decreased during the low recruitment period (Figure 5). After 2009 they were more or less stable among years (Figure 5).

Costs and benefits

Integrating the economic component into the model made it possible to estimate the associated costs of implementing the alternative HCRs. In midterm (2007–2015) these costs included 17–21% lower net profits for the fleet but also 12–21% lower crew costs, and 9–11% smaller fleet sizes than for an unrestricted fishery (Figure 5). Benefits included 42–46% higher SSB values (Figure 5). More importantly, in the long-term simulations (2022) the successful stock recovery coincided with the net profits being 5% lower for HCR_{Blim} and 1% higher for HCR_{Bpa} than the net profits of an unrestricted fishery (Figure 6). Although net profits of the alternative HCRs were almost equal to those of an unrestricted fishery, the stock was likely to collapse under an unrestricted fishery in the long term (2022) (Figure 3). Crew wages were increased from mid to long-term, but were still 6–16% lower than that of an unrestricted fishery (Figure 6). Fleet size was still 9–11% lower than that of an unrestricted fishery (Figure 6).

Crew wages

Crew costs were determined as a certain percentage of the difference between revenues and fuel costs (corresponding to the real calculation used in the saithe fishery), and were used independently of the modelling approach to estimate the average wage of a crew member per month. Estimated individual monthly crew wages were 780 Euro (HCR_{Blim}) and 848 Euro (HCR_{Bpa}) above the German gross unemployment benefit, which served as an indicator of the minimum wage that an owner has to pay to a crew member each month (Table 2). Over the long term, the estimated monthly crew wages were 814 Euro for HCR_{Blim} and 882 Euro for HCR_{Bpa} above the German gross unemployment benefit (Table 2). Even though estimated monthly crew wages were reduced in midterm due to the alternative HCRs, they were still 300 Euro for HCR_{Blim} and 500 Euro for HCR_{Bpa} above the mean wage of a German crew member on a fishing vessel (Table 2). In the long term, the estimated monthly crew wages were 400 Euro (HCR_{Blim}) and 600 Euro (HCR_{Bpa}) above the mean wage of a crew member working in the fishing sector (Table 2).

Sensitivity of the model

The percentage deviation from base case values (values of the scenarios discussed above, Table 1), both of profits and SSB by varying parameter values, was evaluated. Even if the standard variation of recruitment was halved or doubled or the fuel cost halved or doubled, both HCRs were successful in rebuilding the stock and SBB did not drop below B_{lim} (high risk of a stock collapse) in any of the conducted iterations. These results indicate that not only was the model robust, but the tested HCRs were also robust (Table 3). However, when the standard variation of recruitment was set to five times the base case values, it overwhelmed the density-dependence portion of the model, and both HCRs had little effect on stock rebuilding. Doubling of the fuel costs was actually beneficial for stock recovery because this led to a significantly stronger reduction in effort and fleet size. The model was highly sensitive towards the effort and total stock biomass scaling factors (the exponents of fishing effort and total stock biomass in the Cobb–Douglas production function), especially when those were set to 0.1—then profits were around 100% lower than the base case values and SSB estimates were more than 100% higher (Table 3).

Discussion

The model presented here is an extension of the bioeconomic model called FishRent (Salz *et al.*, 2011). The initial FishRent model represents a complex economic model with simplified biology, where the fish population is described by a single variable, interpreted as the biomass of the population. The deterministic stock growth production function is mostly used in economic models

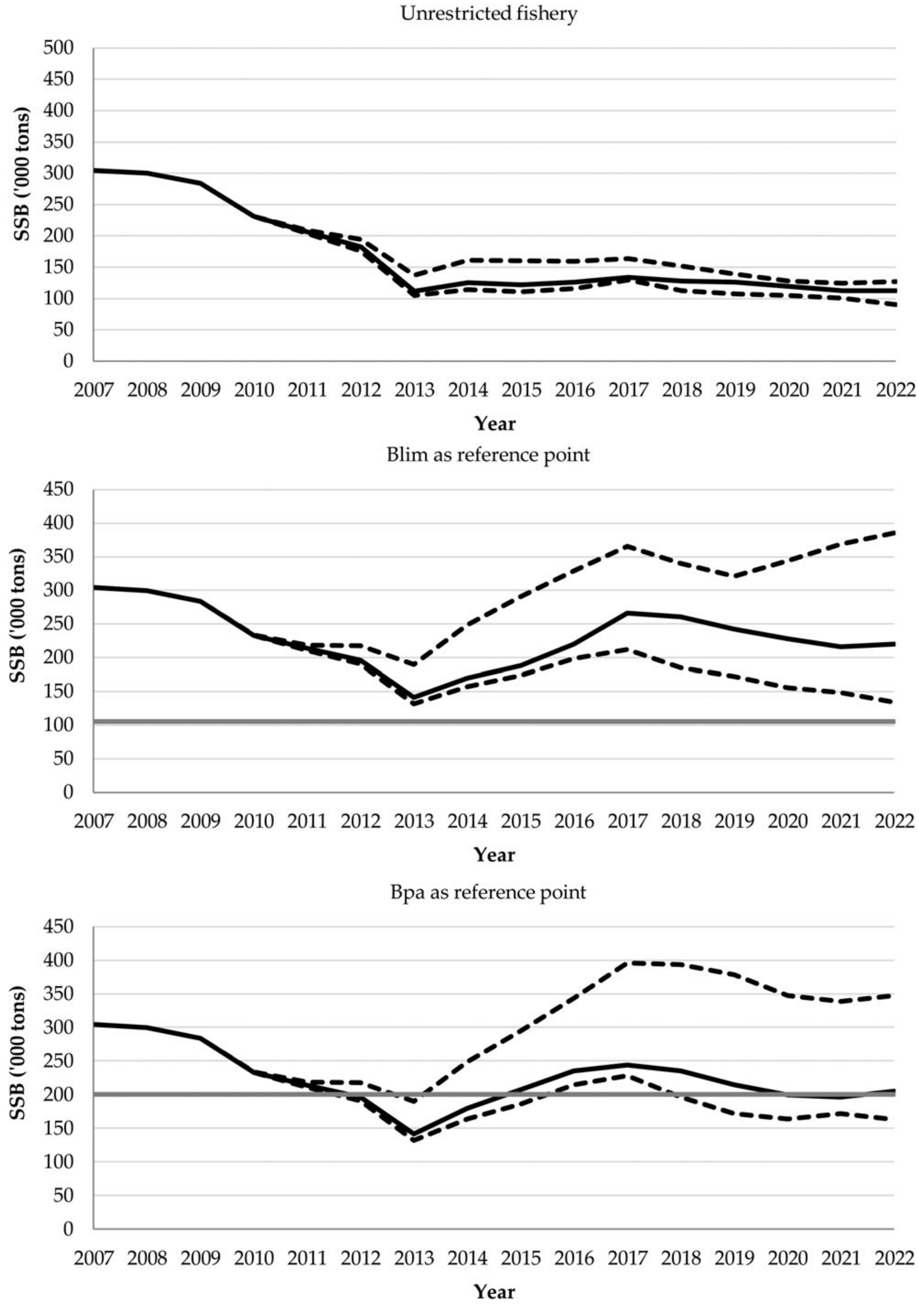

Figure 3. Median SSB values (black solid line) with 5 and 95% intervals (dotted lines) based on 1000 iterations. The two reference points B_{pa} and B_{lim} (grey lines) are shown. Upper graph: unrestricted fishery, middle graph: HCR_{Blim}, lower graph: HCR_{Bpa}.

(Clark, 1976; Clark and Kirkwood, 1979; Goh, 1979; Charles, 1983; McKelvey, 1985; Cohen, 1987; SEC, 2004). In these models, overfishing occurs by definition if the biomass falls below the MSY. However, these models are not able to disentangle the effect of whether fishing may reduce the production of young fish (recruitment overfishing) or may remove most of the older fish, reducing

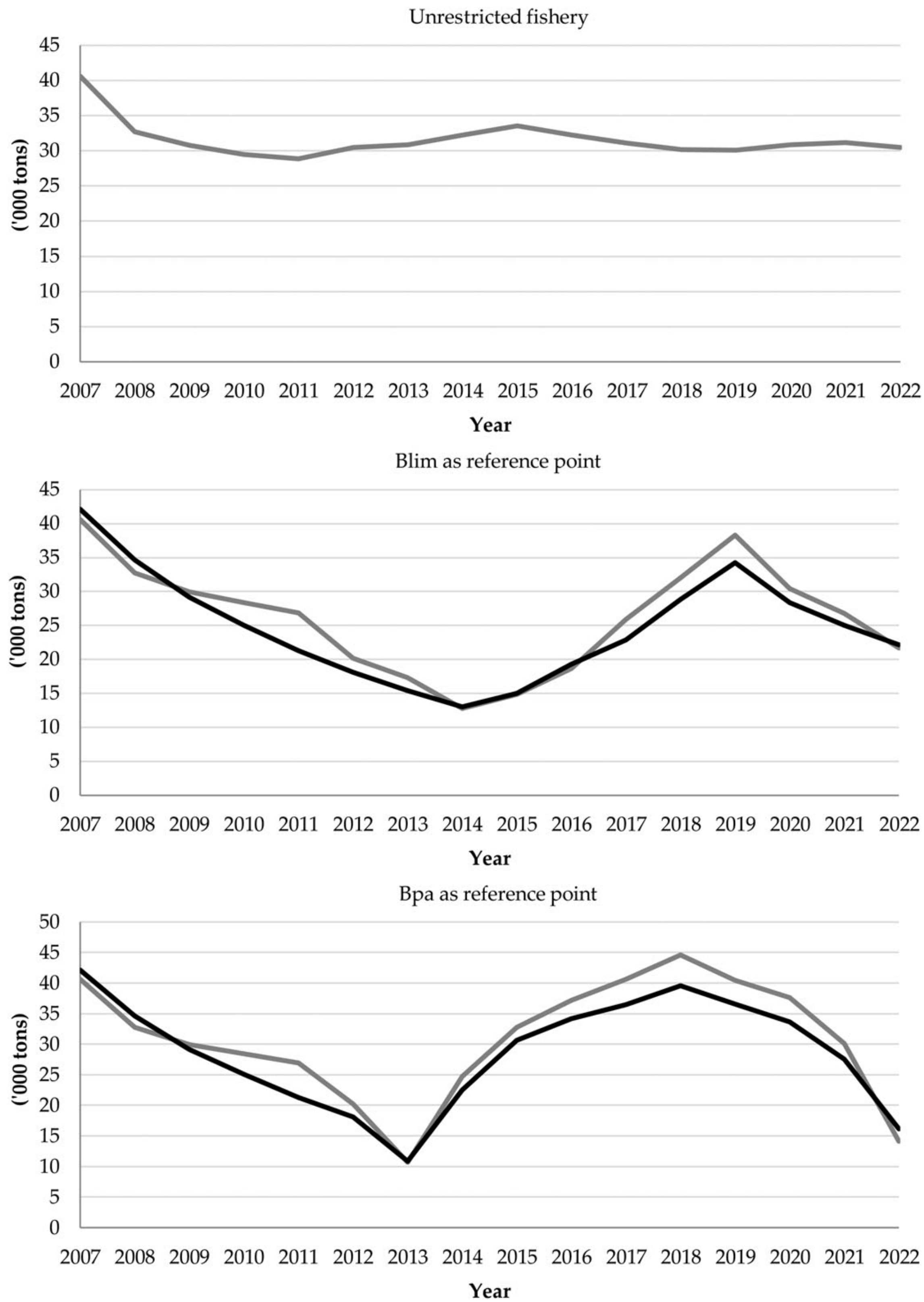

Figure 4. Catches (grey lines) and quotas (black lines) for the modelled segments. Results are shown for simulations of the unrestricted fishery, HCR_{Blim} and HCR_{Bpa}.

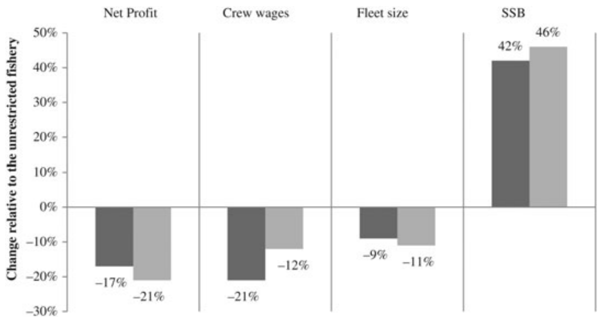

Figure 5. Changes (%) in median SSB values and for the whole fleet in net profit, crew wages and fleet size (number of vessels) relative to the unrestricted fishery. Diagrams show midterm (2015) changes for simulations of HCR_{Blim} (black) and HCR_{Bpa} (grey).

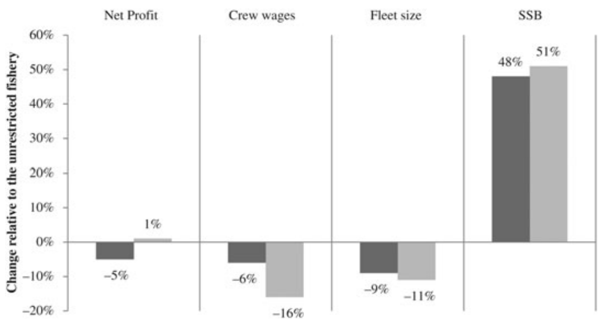

Figure 6. Changes (%) in median SSB values and for the whole fleet in net profit, crew wages and fleet size (number of vessels) relative to the unrestricted fishery scenario. Diagrams show long-term (2022) changes for simulations of HCR_{Blim} (black) and HCR_{Bpa} (grey).

Table 2. Mean gross crew wages (Euro/month) determined by the model, assuming five crew members per vessel.

Scenario	Time line	Mean crew wage	Unemployment benefit	German mean crew wage
HCR_{Blim}	midterm	2300	1520	2000
	long term	2400	1586	2000
HCR_{Bpa}	midterm	2500	1652	2000
	long term	2600	1718	2000

Gross unemployment benefit wages (Euro/month) were based on crew wages of the previous year and estimated by applying the calculation process from the German Federal Employment Agency. The mean gross wages (Euro/month) of a German crew member on a fishing vessel was derived from Statistisches Bundesamt (2012/2013).

the average age and size of remaining fish (growth overfishing). In this study, the catch age composition, the growth of individuals and the estimation of recruitment are fully integrated with the dynamics of multiple fleet segments. These extensions allow the new model system presented here now to indicate whether recruitment or growth is a case of biological overfishing. The importance of such a feature is recognized by researchers worldwide, and hence more and more cohort models are being applied in bioeconomic studies (e.g. Sumaila, 1998; Tahvonen, 2010; Skonhoft *et al.*, 2012; Quaas *et al.*, 2013).

Biological models are useful for short-term predictions (e.g. one year ahead, single species TAC) (Beverton and Holt, 1957; Hilborn and Walters, 1992), but when trying to understand long-term impacts of management regulations, biological, economic and social processes need to be considered (Brander, 2003). As opposed to other models (Ganguly and Chaudhuri, 1995; Marchal, 1997; Garza-Gil, 1998; Pelletier *et al.*, 2009; Da Rocha et al., 2010; Poos et al., 2010), this model combines the economic processes such as price setting, fleet, effort and cost dynamics with a detailed age-structured population model that includes recruitment dynamics. This model can investigate fishers' response to alternative management regulations along with the stock development and market conditions (e.g. fuel prices and fish market prices). Being able to perform such an impact assessment is crucial for the performance of fisheries management, especially in the EU when a new management measure or regulation is proposed (SEC, 2009).

Recruitment

An important feature of the extended model is that it is possible to account for the risks of a management failure through a stochastic stock–recruitment relationship. In particular, the developed model provides estimates of the risk of SSB falling below B_{lim}, the probability of SSB being above B_{pa}, and the time needed to reach the target fishing mortality rate. These indicators are the same as those used by ICES (2012b) when re-evaluating the HCR options. However, the modelling approach presented here differs from the one used in the evaluation process of ICES (2012b), as it includes the fisher behaviour and economic performance of multiple fleet segments. In the presented model, recruitment was forced for certain years, using observed values for certain years to investigate the response of the fleet. Results showed that SSB started to decline and even dropped below B_{pa} when the given low number of recruits of age 3 attained older age classes in subsequent years. The simulated HCR_{Bpa} was favourable in the midterm, as it allowed a fast reaction to that stock decline. However the simulated HCR_{Bpa} led to only a 50% probability of SSB being above the precautionary reference point in the long term. This was due to the fact that TAC was increased by more than 15% when SSB was still below B_{pa}. On the other hand, simulations with the HCR_{Blim} did not allow such a strong catch reduction due to its 15% constraint at a lower level of SSB (B_{lim}). However, the HCR_{Blim} did allow the stock to recover to a significantly higher level above the precautionary reference point (B_{pa}), because the TAC increase was constrained within 15%. This in turn resulted in an 80% probability of SSB being above B_{pa} in the long term. A combination where the TAC is allowed to be reduced by more than 15% if SSB is falling below B_{pa}, but allowing then only a 15% increase in the TAC when the stock is recovering, would probably be the best solution in terms of stock conservation. This option may be difficult to accept by fishers due to its high economic costs in terms (20% lower net profits) and a potential further reduction of fleet size. In contrast to the present study, ICES (2012b) provided only biological indicators. They used stochastic recruitment for the whole modelling period, but the parameter values of the stock–recruitment relationship are not documented. As SSB for both HCRs remained above B_{pa} for the whole modelling period, it is likely that ICES used higher mean recruitment levels. In ICES simulations, SSB remained above B_{pa}, and hence the annual TAC adjustment was always kept within the 15%. Consequently, the performance of both HCRs was identical in ICES (2012b), and therefore in contrast to this study.

Table 3. Results of the sensitivity analysis, shown as deviations (%) from base case values of profit and SSB for the midterm (2015) and long term (2022).

			Profit		SSB	
Parameter	**Values**	**Scenario**	**2015**	**2022**	**2015**	**2022**
Recruitment	0.5 × std. variation	unrestricted fishery	−8	−9	6	10
		HCR_{Blim}	−9	−8	12	14
		HCR_{Bpa}	−8	−7	15	12
	2 × std. variation	unrestricted fishery	−5	−19	−4	30
		HCR_{Blim}	−2	−15	2	15
		HCR_{Bpa}	−4	−15	9	19
Fuel costs (FuC)	0.5 × FuC	unrestricted fishery	−4	−5	−8	−13
		HCR_{Blim}	−68	−15	4	5
		HCR_{Bpa}	−50	−10	5	2
	2 × FuC	unrestricted fishery	−17	−15	7	15
		HCR_{Blim}	−38	−14	12	37
		HCR_{Bpa}	−50	−10	15	25
Effort scaling factor alpha	0.1	unrestricted fishery	−120	−198	140	194
	0.1	HCR_{Blim}	−160	−195	100	300
	0.1	HCR_{Bpa}	−190	−190	150	200
	1	unrestricted fishery	−25	−54	−23	−6
	1	HCR_{Blim}	10	−2	−10	15
	1	HCR_{Bpa}	16	−5	−20	20
Biomass scaling factor beta	0.1	unrestricted fishery	−120	−160	−17	−7
	0.1	HCR_{Blim}	−101	−135	70	230
	0.1	HCR_{Bpa}	−115	−190	100	200
	1	unrestricted fishery	23	17	−23	−6
	1	HCR_{Blim}	12	15	16	34
	1	HCR_{Bpa}	15	13	25	17

Positive percentages mean that the value when varying the parameter is higher than the base case value, and vice versa.

Fleet dynamics

In the presented model the catch of the species in each year depends on the dynamics of the fleet segments. For the unrestricted fishery scenario, catches declined during the low recruitment period, but remained more or less stable afterwards. First, the relative stable catch after the low recruitment period was due to the lack of any regulation that would have forced a reduction in catches. Second, the initial condition of fishing effort is of importance. For example, initially the modelled fleet segments exhibited low effort, and hence there was neither an investment nor a disinvestment, because fishing was still profitable (although profits were decreasing among the modelled years). As a result, the fleet size remained exactly the initial size in the unrestricted fishery scenario. For HCR simulations, fleet size and effort in terms of days at sea were reduced when profit decreased. This in turn resulted in a lower fishing pressure on the stock and was beneficial for the stock recovery. In the long term (2022), the successful stock recovery for both HCR options coincided with increased net profits. For an unrestricted fishery the risk of a stock collapse was high. In contrast, ICES (2012b) ignored fleet dynamics completely. ICES (2012b) estimated the yields by assuming a constant fishing mortality, and hence predicted higher yields and a higher risk of SSB falling below B_{lim} in the long term. This overestimation of fishing mortality indicates the need of a bioeconomic assessment approach, where the feedback works in both directions, i.e. takes into account the fact that the stock development influences the fleet economy, but also that the fleet economy will influence stock development.

Discards

In general not all vessels in the European fishing fleets are fishing at their maximum performance, as they are limited by the amount of allocated days-at-sea, and quotas often *do* limit the quantity of fish being caught (Hoff and Frost, 2008). Moreover, the use of TACs represents a way to control the outputs (yield) of a fishery but does not allow direct regulation of the level of input (e.g. fishing effort) (Holden, 1994). A reduction in TAC without equivalent reduction in inputs results in an imbalance between catches of the fleet and the TAC. This problem was reflected by the model when the TAC estimate was reduced, and overquota catch predicted as inputs in terms of fishing effort and fleet size were still too high. These outcomes are supported by other examples where fishers continued to fish and discarded marketable fish (overquota discarding) (Daan, 1997; Pascoe, 1997; Rijnsdorp *et al.*, 2007; Hamon *et al.*, 2007). Furthermore, until the implementation of the new CFP, it is legal and mandatory to discard undersized fish and overquota catches within the EU (Holden, 1994), which leads to catches above the TAC as fishers are incentivized to maximize the value of their catch. This incentive was incorporated in the model where fleet segments were assumed to maximize the total net profit (revenues minus costs). In particular, discards were predicted when they resulted in a higher total net profit due the age-specific fish prices. Thus, in other fisheries than the saithe fishery, where fish prices differ considerably between age classes this discarding behaviour might be stronger. Moreover, most European fleet segments harvest several species, each equipped with individual quotas that are often exhausted at different rates (Jákupsstovu *et al.*, 2007). This is also happening in the North Sea saithe fishery. In particular, the EU fleets targeting North Sea saithe have fallen under the effort regime of the cod recovery plan since 2009 (ICES, 2013). In particular, if their cod catch exceeds 1.5% of the total catch the days-at-sea will be restricted. Thus, when reducing the quota of North Sea cod, discarding of North Sea cod might occur, as the quota of saithe has still not been filled. As in general this model is able to mimic this discarding behaviour when including the dynamics of the North Sea cod stock and the fleets targeting that stock, it might be an interesting aspect for future investigations.

Economic costs

In reality, crew members will compare their income to the next-best alternative. In the present study the average monthly wage of a crew member on a German fishing vessel (Federal Statistical Office, 2012/2013) was used for comparison. In particular, it was assumed that crew members in the modelled saithe fishery stay in that fishery if their income is still above that average wage. Estimated monthly crew wages were reduced in the mid and long term, but for both tested HCR options they were still higher than the average monthly salary of a crew member on a German fishing vessel. Therefore, it is likely that under the given circumstances crew members stay in that fishery. However, simulations may represent the best case, and in reality crew wages may sometimes be below the average wage of the fishing sector, because fishers may not always act in a way that maximizes their total net profits due to a lack of knowledge about potential catch rates, traditions, weather condition or accessibility of a region (Hilborn and Kennedy, 1992; Prince and Hilborn, 1998; Swain and Wade, 2003; Salas and Gaertner, 2004). Moreover, in the German saithe fishery a crew member's income is around 4% of the difference between revenues and fuel costs. However, the skipper has the right to change that percentage depending on how efficient and hardworking he thinks each crew member has been. Thus crew wages can vary according to the individual and create competition between crew members. In most fisheries, the crew is rewarded by such a lay system (the crew is paid with a share of revenue less costs), rather than a fixed wage. Those uncertain wages, and the fact that fishing is a hazardous occupation, make it difficult to compare the wages from fisheries with wages in other sectors. For these reasons the average wage of a German crew member might be the best way to represent the opportunity cost of labour.

Another reason why the average wage of a crew member working on a German fishing vessel might be more suitable than a reference wage from another sector is that crew members may switch to another fishery/vessel rather than to another sector, as they tend to have skills that are specific to the fishing sector. In Germany especially, it is difficult and time-consuming to acquire the qualification required to work on a fishing trawler. There are no studies that could be used for comparative purpose or as reference in terms of modelled crew salaries. At present, social indicators such as employment are mostly ignored in simulation models that investigate the effects of alternative management strategies (Pelletier and Laurec, 1992; Marchal, 1997). Other factors, such as fuel cost, significantly influenced the effort level. For instance, doubling the fuel costs (see Table 3) reduced the effort and was beneficial for stock recovery. This highlights the importance of fuel costs and the potential risks of an unsustainable fishery with fuel subsidies, because it encourages the maintenance of fishing effort even when stock levels decline (Sumaila *et al.*, 2006; Tidd *et al.*, 2011).

Investment behaviour

Unlike in many other models, where capacity is kept constant (e.g. Ulrich *et al.*, 2002), the capacity in the model presented here is subject to change by the use of an investment/disinvestment function. For instance, compared with the unrestricted fishery where fleet size remained constant, it was reduced by 9% for HCR_{Blim} and by 11% for HCR_{Bpa} when the quota was reduced in response to the decline of SSB below B_{pa}. In reality, the number of vessels of the considered fleet segments declined in recent years (Anderson *et al.*, 2012). However, this decline was considerably lower than the predicted reduction. The maximization process of the model predicted stronger reduction in fleet size, indicating that the real saithe fishery could be more profitable by further reducing the number of vessels. Thereby, the model presented here might be more suitable to explain a situation where switching of vessels from one fishery to another can occur, instead of investments in new vessels or scrapping of existing vessels. One underlying assumption of the investment function is that capital costs are constant per vessel, which may be unrealistic, as a new vessel will have higher capital costs. In reality, when a vessel is taken out of the fishery it can be sold (providing money), it can be scrapped (which may even cost money), or it might switch to another fishery, which is the case that is mimicked by the model. Tidd *et al.* (2011) demonstrated that for the English North Sea beam trawl fleet, vessel age, vessel length, stock status, fuel cost, the availability of decommissioning grants, fleet size, and the revenues from target species are significant factors in determining fisher decision-making. Factors such as the vessel age, vessel length, and the availability of decommissioning grants are not included in the model, which may explain why the predicted fleet size was lower than the observed number of vessels. Moreover, in the presented model, the behaviour is modelled on fleet segment level, but most likely there is a considerable individual variability in fishing success among vessels (Hilborn and Ledbetter, 1985; Smith and McKelvey, 1986; Thorlindsson, 1988). Thus, the investment function used in the presented model could be further elaborated if this individual variability could be taken into account, e.g. by classifying vessels by certain characteristics (such as catch success derived from logbooks). However, for the saithe fishery there is no detailed information about the behaviour of fishers with regard to investment or switching between fisheries. This lack of economic detail is quite general, and apart from Bjørndal and Conrad (1987), there is generally little empirical data on investments within fisheries. Because of the lack of detailed information, fisher behaviour is often modelled as either simplistic (Horwood *et al.*, 1998; Apostolaki *et al.*, 2002) or based on case-specific assumptions (Holland, 2000). Nevertheless, including the investment and disinvestment behaviour in the present model clearly demonstrated its importance with regard to the stock development. In particular, when the associated costs of the introduced HCR options made the fishery less profitable, fleet size was reduced, lowering the fishing pressure on the stock and allowing the stock to recover.

The link between biology and economy

In the presented model, the link between biology and economy is implemented via the Cobb–Douglas production function. For this function a bi-non-linear relationship is assumed between the two inputs, fishing effort and total stock biomass, and the produced catch. This is different to Ulrich *et al.* (2002), where total fishing mortality is a linear function of effort in terms of sea days. However, using a non-linear relationship between effort and fishing mortality might be more realistic because it takes into account the possibility of crowding. In turn, crowding makes extra trips less efficient, resulting in a flattened fishing mortality rate. Gillis (2003) describes crowding as a direct interference between vessels that reduces their efficiency, e.g. when a trawler and its gear are blocking the way of another trawler. The effect of crowding was also described by Rijnsdorp *et al.* (2000), who showed that the catch rates of Dutch beam trawlers increased by 10% when the vessel density decreased by 25%. Moreover, in Ulrich *et al.* (2002) the model tended to overestimate the level of effort employed, as it does not account for the economic costs. As a result the quota was fully fished. In reality, fishers stop fishing once the marginal cost of fishing exceeds the marginal value of the catch, even if the TAC remains unfilled. The presented model considers the costs of fishing, and therefore the catches of the modelled fleet segments can be below the quota. However, this was predicted only for the first two modelling years, when the quota started to be reduced (Figure 4).

The non-linear relationship between stock and catch might also be appropriate, as mobile species such as saithe can concentrate in restricted areas due to food availability or spawning events. The assumed value for the stock exponent was set at 0.4, and hence well below 1. This implies that the density of the stock at the fishing ground does not increase proportionally when total stock biomass increases (Eide *et al.*, 2003). This may apply to North Sea saithe, as Casini *et al.* (2005) found that the population of North Sea saithe appeared to aggregate at low levels of total stock biomass and disperse at high levels of total stock biomass. The sensitivity analysis has shown that the model is highly sensitive to the actual values used as exponents for effort and total stock biomass. Since profit and SSB were highly varying when changing the values of those parameters, the values used as exponents must be determined carefully. In particular, catches and total fishing mortality were around 10–13% higher when both exponents were set to one, compared with the values used here of 0.6 and 0.4 for effort and the total stock biomass, respectively. Thus, depending on the values used for those exponents, the produced management advice may differ. Frost *et al.* (2009) highlighted the importance of the values for the two exponents in the Cobb–Douglas function and found that they significantly influence the catch and the fishing costs (being a function of effort). However, in the simulations presented here the values for the exponents were identical between the unrestricted fishery scenario and the two HCR scenarios, and hence it was possible to compare the performance of different strategies/scenarios investigated with one another.

Conclusion

The modelling approach presented here is a step forward in the development of models for fisheries management towards including the effect of fleet behaviour on the stock, and the effect of stock abundance on the overall performance of the fleet. The main contribution is that the model has a level of detail in both the economic and biological component. On the one hand, modelled economic variables (e.g. fish and fuel prices, variable and fixed costs, effort distribution and capital investment) affect fishing mortality through modifications in the fishing behaviour, which in turn affects stock size. On the other hand, modelled biological variables such as variable recruitment and its effect on stock size also influence fishing mortality, fishing effort and hence fishing behaviour. This is different from most other models that first evaluate the stock development independent from the fleet dynamics, and second evaluate various economic indicators at the end of each year as a function of the biological catches, but without these economic factors feeding back into the biological development in the next year (SEC, 2004). In our case study it was possible to show that both B_{lim} and B_{pa} are suitable reference points to provide guidance in deciding whether an annual TAC should be increased or decreased by more than 15%. Moreover, it was demonstrated that both reference points are important and should be used in combination when decreasing (B_{pa}) and increasing (B_{lim}) the annual TAC by more than 15%. Thus, the modelling approach is not only a step forward, but the results are relevant to policy in the current re-evaluation process of the HCR for North Sea saithe.

Supplementary data

Supplementary data is available at *ICES Journal of Marine Science* online. It includes a list of all parameters used in the model, their values and estimation methods and/or literature source.

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