# Comparative evaluation of a mixed-fisheries effort-management system based on the Faroe Islands example

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Total allowable catch (TAC) management has in many fisheries, especially mixed fisheries, failed to meet conservation objectives. For instance, for the Faroe Plateau mixed demersal fisheries, the TAC system failed to achieve the objective of an average annual fishing mortality of 0.45 for the three gadoid stocks cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), and saithe (*Pollachius virens*). Therefore, in 1996, an effort-regulation system with individual transferable effort quotas was introduced to manage the fisheries. Experience has shown that effort management without additional stock-specific measures may not be appropriate for such fisheries. A management strategy evaluation model was developed to compare an effort-management system based on the Faroese example with a TAC system as currently applied in EU fisheries. Results show that when stocks are considered in isolation, a total allowable effort system does not necessarily perform better than a TAC one. It depends on stock status and dynamics, the level of uncertainty, and the reactivity of the system to changes in scientific advice. When the stocks are considered together in mixed fisheries, effort management seems, however, to be appropriate, and interannual flexibility of the system appears to be the best compromise between short- and long-term objectives, as well as between biological sustainability and economic return.

**Keywords:** effort management, Faroe Islands, management strategy evaluation (MSE), mixed fisheries.

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## Introduction

Total allowable catch (TAC) management has been the traditional system used for managing fisheries in the western world. However, there is evidence that it has often not achieved its conservation objectives, especially in mixed fisheries where several species are caught simultaneously. This is particularly true in European fisheries, where a large proportion of stocks are overexploited (COM, 2001; FAO, 2007). Many studies have focused on the scientific and institutional caveats of the TAC system (see, e.g. COM, 2001; Kell *et al.*, 2005; Schwach *et al.*, 2007), emphasizing *inter alia*: (i) the intrinsic uncertainty in fisheries advice and the difficulty in providing precise point stock/fishery estimates and reliable TAC advice for most stocks; (ii) the political pressure when negotiating annual TACs, which often results in TACs not being consistent with scientific advice; (iii) the resistance of the fishing industry to TACs varying significantly between years; (iv) the difficulty in achieving single-species objectives in a mixed-fisheries context; and (v) the weak linkage between output control (TAC) and the levels of capacity and effort. All these were already acknowledged in the EU Common Fisheries Policy (CFP) Green Paper (COM, 2001). Indeed, a number of reforms of fisheries management have been initiated (Penas, 2007) and will likely be implemented fully in the future CFP expected to enter into force in 2012 (CEC, 2009).

In the meantime, there has been increasing awareness of the needs to manage the fisheries in a sustainable way, both as an ecosystem-based approach and in an integrated, multisectoral maritime policy recognizing the importance of the various components and usages of the ocean. This has led to a growing focus on alternative management strategies. Among others, some alternatives aim at improving the current TAC system through long-term management plans fixing long-term objectives and limiting the interannual variation (IAV) in yield (STECF, 2007a), and others address management through input control (Nielsen *et al.*, 2006). There is also a growing awareness that management must move from optimizing the catches to being robust to major sources of uncertainty (Degnbol and McCay, 2006).

This in turn has led to the development of scientific tools allowing evaluation of these alternative management strategies, largely based on scenario modelling through simulation. Management strategy evaluation (MSE) frameworks were first initiated by the International Whaling Commission (IWC, 1993; De La Mare, 1998) and are now widely recognized and used to test the performance of management scenarios under various plausible hypotheses on the dynamics of the fisheries system and fish stocks (Daan, 2007; ICES, 2007a; De Oliveira *et al.*, 2008).

Although most MSE applications have so far dealt with single-species and stock-based approaches, some studies have already addressed fleet-based mixed-fisheries interactions for flatfish (Ulrich *et al.*, 2002; Pastoors *et al.*, 2007; Kraak *et al.*, 2008; Andersen *et al.*, 2010) and roundfish (Hamon *et al.*, 2007) fisheries in the North Sea. All these studies focused on the issues arising from the current TAC system and simulated what alternative effort management would be. The present study was built on the reverse approach, focusing on the issues arising from a case study with the current effort-management system and simulating what an alternative TAC management would provide, so building on factual information and observations.

The Faroe Islands have received growing interest as a case study where relevant lessons could be learned (Nielsen *et al*., 2006; Jákupsstovu *et al*., 2007; Løkkegard *et al*., 2007; ICES, 2008b). In the mid-1990s, the TAC system in place was rejected by the fishing industry and the authorities because it did not lead to satisfactory management. It resulted in extensive discarding when single-species quotas were filled. Therefore, owing to the general dissatisfaction, the Faroese Parliament developed a new management system in close cooperation with the fishing industry for all vessel groups targeting demersal stocks on the Faroe plateau, and implemented it from 1996. This new system (hereafter referred to as one of total allowable effort, TAE) consists of individual transferable effort quotas (fishing days) for specific fleet categories (small trawlers, pairtrawlers, longliners, and coastal fishing vessels operating in "The Ring", in waters shallower than 200 m). Additional measures such as area closures during the spawning seasons, area restrictions for larger vessels, and minimum gear mesh sizes were implemented too.

In the first year of implementation, the initial allocation of fishing days was based on an estimated historical allocation from data on partial fishing mortalities. It was also estimated that sustainability of the fisheries could be achieved by a target fishing mortality ($F$) of 0.45 for each stock, corresponding to an average annual harvest of approximately one-third of the spawning stock (ICES, 2006; Jákupsstovu *et al*., 2007). Subsequently, the number of fishing days allocated has been regulated each year based on ICES advice and input from the fishing industry.

The Faroe Islands fisheries represent an innovative and unique system of a mixed groundfish fishery regulated by individual transferable fishing days. It is a relatively pure effort-regulation system, which also has been in force for a long time compared with other effort-regulation systems worldwide (Nielsen *et al*., 2006). After 10 years of implementation, it is possible to assess empirically the effects of this management system in practice (Jákupsstovu *et al*., 2007). These last authors described the key issues of the system and concluded that Faroese effort management had not achieved all its objectives. Management had failed to maintain average $F$ at 0.45 over the years. ICES (2006) did not consider this target to be consistent with its interpretation of the precautionary approach; ICES bases its advice on the precautionary approach that corresponds to a value of $F$ of 0.35. Since the introduction of the effort system, the total number of fishing days allocated has been reduced by some 15% in total, i.e. less than 2% per year since implementation, significantly less than that advised by ICES for the same period (ICES, 2008b). The allocated fishing days are still not fully utilized, however, which suggests that the initial effort allocation was too high to constrain $F$ to the target. In practice, effort management did not act as a restrictive and reactive management tool, but rather as a conservative *status quo*.

One of the main assumptions behind implementing effort management that fishers would switch their target automatically according to the relative availability of the stocks has not been verified (Jákupsstovu *et al*., 2007). Most fishers opportunistically target cod (*Gadus morhua*), which is the most valuable species. Changes to targeting behaviour towards stocks that are more abundant takes place progressively, so leading to ongoing high levels of mortality for the less abundant stocks, especially if their value is high or if prices increase while catches decline.

Effort regulation provides incentives for fishers to increase their catchability because they are limited by the time they are allowed to fish (Nielsen *et al*., 2006). In addition, catchability is likely to increase over time because of the so-called technological creep and increased knowledge of best fishing practice. However, it has proven difficult to demonstrate that changes in catchability were associated with the introduction of the effort system (Jákupsstovu *et al*., 2007) because of the influence of environmental conditions. There is considerable IAV in exchange rates between the warm, saline upper water layer and cold, less saline deeper water, leading to great variability in productivity between areas and years where primary production may vary by up to a factor of five. Environmental variability has a significant impact on fish stock dynamics and trends and may be considered as one of the main drivers of fluctuations in the stocks, with respect to both recruitment and growth (Steingrund and Gaard, 2005; ICES, 2006). Primary productivity seems to be negatively correlated with the catchability of longlines, suggesting that cod approach longline bait more often when natural food abundance is low (ICES, 2008a). Consequently, natural factors may impact catchability to a greater extent than technological ones.

The practical experience gained during the 10 years of effort management led to increased understanding of the efficiency of such a TAE system compared with a TAC system. The aim of this study was therefore to evaluate by simulation whether effort-based harvest control rules (HCRs) would be more appropriate and robust than catch-based HCRs, because they rely on other sources of information and are affected by different uncertainties. In particular, our evaluation focused on biological robustness, i.e. the ability to account for uncertainty and error in the biological dynamics and knowledge, so reducing the impact of uncertainties on the sustainability of the resources. Economic efficiency, the IAV in management decisions, and the management implementation in a mixed-fisheries context need also be considered. Finally, it is expected that the particular features of the Faroese fisheries may influence the perception of the TAE system. As a consequence, the model used for evaluation included generic aspects of both management systems, but was conditioned specifically to the Faroese characteristics.

The model was adapted from an MSE model developed in parallel for another EU fishery case study (Hamon *et al*., 2007). We formulated a model based on the Faroese example including the main stocks and fleets of the Faroe Plateau demersal fisheries, as well as some of the main sources of variability and uncertainty, to simulate a number of alternative management scenarios, first in a single-species approach where each stock is considered in isolation, then in mixed fisheries where catches of the various stocks are linked through technical interactions at a fleet level, assuming that they were being caught simultaneously in the fishing gears. The MSE model includes an operating model (OM), which simulates the "true world", and a management procedure (MP), which simulates the "perceived world" and the management actions based on it. The model was established using the FLR framework (Kell *et al*., 2007; www.flr-project.org; www.efimas.org), an open-source and flexible modelling toolbox running in the language R. The model covered two periods, one in the past (1998–2005), which was used to condition the model to observed data and to test its ability to reproduce the observed dynamics, and the other projected into the future (2006–2015), in which scenarios are run based on hypotheses and uncertainty about future parameter values.

Obviously, the system simulated here is only a rough simplification of the real Faroese fisheries and cannot encompass the full complexity of the actual fisheries dynamics and management in the Faroe Islands. Therefore, it is referred to as a "Faroese-type system". It must be emphasized that the work was undertaken as part of extensive and multidisciplinary analyses of a number of innovative and global fisheries-management systems put into a European context (Hauge and Wilson, 2009). Therefore, the work represents mainly the biological analysis of such a Faroese-type management system, and the associated comprehensive socio-economic analyses and more-detailed and elaborated economic and social aspects associated with it were addressed separately elsewhere (Buisman *et al.*, 2009; Christensen *et al.*, 2009; Hauge and Wilson, 2009) as part of a collaboration. As a consequence, the economic and social impacts of our scenarios were only addressed here in rather rough terms with simple proxies, whereas a more complete socio-economic investigation and evaluation of the system is covered in the findings of the companion studies and publications partly based on the modelling herein.

## Material and methods

### Input data

Stock data used in the MSE model are traditional VPA inputs from the ICES Northwestern Working Group (NWWG) at the time this work was initiated (ICES, 2006). These data are available from 1961 on for the three stocks evaluated and are referred to as "WG estimates". The precautionary approach and limit reference points in terms of $F$ and spawning-stock biomass (SSB) used are those defined by ICES for the three stocks. The value of $F$ associated with the precautionary approach ($F_{pa}$) is not used as a management objective, because the management target $F$ is defined. The stock assessment of saithe (*Pollachius virens*) is subject to great uncertainty and was only used as an indicative assessment (ICES, 2006), but in the evaluation here, the saithe data were used in a similar way as for other stocks.

Catches (in tonnage and value) and actual effort in fishing days by fleet were obtained directly from the Faroese Fisheries Laboratory data. However, only data for large longliners (>100 grt, fleet LG) and pairtrawlers (fleet PT) were available throughout. These two fleets account for more than half of the historical catches of the three stocks by the Faroese and were the only ones explicitly included in the model. The remaining catches were pooled into a single "other" fleet (fleet OTH). Effort data were available from the end of 1997 to 2005.

### The operating model

All equations used in the OM are given in the Appendix, and reference points and model parameters in Table 1. The biological OM was conditioned with all available data. Several sources of uncertainty were included when actual data and knowledge were available to condition them. For both past and projected years, natural mortality ($M$) and mortality before spawning were set to be constant and equal to ICES assumptions. Maturity in projected years was set to be constant and equal to the last year of available data. Stock numbers-at-age for the first year of the model were set equal to ICES estimates, and for all other years, both past and projected,

**Table 1.** Model parameter values.

| Parameter | Cod | Haddock | Saithe |
|---|---|---|---|
| BRP | | | |
| $B_{pa}$ | 40 000 | 55 000 | 60 000 |
| $B_{lim}$ | 21 000 | 40 000 | 85 000 |
| $\boldsymbol{F}_{pa}$ | 0.35 | 0.25 | 0.28 |
| $F_{lim}$ | 0.68 | 0.40 | 0.40 |
| $F_{min}$ | 5 | 3 | 4 |
| $F_{max}$ | 8 | 7 | 8 |
| SRR (Ricker) | | | |
| $\alpha$ | 0.776 | 2.558 | 2.393 |
| $\beta$ | 1.53e−05 | 2.71e−05 | 2.11e−05 |
| Acr | 0.461 | 0.569 | 0.292 |
| s.d. | 0.532 | 0.838 | 0.449 |
| Price (DKK kg$^{-1}$) | 16.23 | 11.28 | 4.90 |
| Selectivity smoother values | | | |
| Age | | | |
| 1 | 0 | 2.02e−04 | – |
| 2 | 0.100 | 0.032 | – |
| 3 | 0.259 | 0.234 | 0.0102 |
| 4 | 0.509 | 0.593 | 0.176 |
| 5 | 0.799 | 1.169 | 0.726 |
| 6 | 0.873 | 1.208 | 1.619 |
| 7 | 1.237 | 1.506 | 1.299 |
| 8 | 1.119 | 2.614 | 1.374 |
| 9 | 1.119 | 0.9219 | 1.280 |
| 10 | – | 0.9219 | 1.848 |
| 11 | – | – | 1.703 |
| 12 | – | – | 1.703 |
| Selectivity s.d. values | | | |
| Age | | | |
| 1 | – | – | – |
| 2 | 0.563 | 1.089 | – |
| 3 | 0.355 | 0.465 | 0.546 |
| 4 | 0.313 | 0.321 | 0.377 |
| 5 | 0.186 | 0.230 | 0.246 |
| 6 | 0.184 | 0.293 | 0.169 |
| 7 | 0.221 | 0.370 | 0.178 |
| 8 | 0.190 | 0.462 | 0.207 |
| 9 | 0.190 | 0.455 | 0.347 |
| 10 | – | 0.455 | 0.356 |
| 11 | – | – | 0.282 |
| 12 | – | – | 0.282 |
| Weight-at-age smoother values | | | |
| Age | | | |
| 1 | 0 | −0.005 | – |
| 2 | 0.986 | 0.559 | – |
| 3 | 1.338 | 0.703 | 1.094 |
| 4 | 1.870 | 0.859 | 1.273 |
| 5 | 2.609 | 1.055 | 1.448 |
| 6 | 3.712 | 1.554 | 1.687 |
| 7 | 5.008 | 1.854 | 2.301 |
| 8 | 6.717 | 2.182 | 3.099 |
| 9 | 7.624 | 2.323 | 3.959 |
| 10 | – | 2.505 | 5.474 |
| 11 | – | – | 5.975 |
| 12 | – | – | 6.382 |
| Weight-at-age autocorrelation | | | |
| Age | | | |
| 1 | – | −0.145 | – |
| 2 | 0.143 | −0.036 | – |
| 3 | 0.280 | 0.301 | 0.152 |
| 4 | 0.384 | 0.573 | 0.096 |
| 5 | 0.403 | 0.553 | 0.001 |

*Continued*

**Table 1.** *Continued*

| Parameter | Cod | Haddock | Saithe |
|---|---|---|---|
| 6 | 0.340 | 0.477 | 0.208 |
| 7 | −0.212 | 0.469 | 0.268 |
| 8 | 0.048 | 0.308 | 0.092 |
| 9 | 0.059 | 0.222 | 0.050 |
| 10 | – | 0.460 | −0.132 |
| 11 | – | – | −0.108 |
| 12 | – | – | −0.021 |
| Weight-at-age s.d. values | | | |
| Age | | | |
| 1 | – | – | – |
| 2 | 0.129 | 0.136 | – |
| 3 | 0.131 | 0.110 | 0.078 |
| 4 | 0.132 | 0.124 | 0.070 |
| 5 | 0.120 | 0.120 | 0.090 |
| 6 | 0.096 | 0.100 | 0.089 |
| 7 | 0.099 | 0.084 | 0.093 |
| 8 | 0.152 | 0.083 | 0.089 |
| 9 | 0.166 | 0.111 | 0.066 |
| 10 | – | 0.099 | 0.089 |
| 11 | – | – | 0.086 |
| 12 | – | – | 0.072 |
| Catchability mean values | | | |
| Fleets | | | |
| LG | 8.64e−05 | 5.02e−05 | 5.42e−07 |
| PT | 1.95e−05 | 7.91e−06 | 1.01e−04 |
| OTH | 6.87e−05 | 2.75e−05 | 2.61e−05 |
| Catchability s.d. | | | |
| Fleets | | | |
| LG | 4.09e−05 | 1.51e−05 | 2.38e−07 |
| PT | 6.47e−06 | 3.19e−06 | 3.05e−05 |
| OTH | 2.47e−05 | 6.66e−06 | 5.86e−06 |
| Catchability *CV* | | | |
| Fleets | | | |
| LG | 0.47 | 0.30 | 0.44 |
| PT | 0.33 | 0.40 | 0.30 |
| OTH | 0.36 | 0.24 | 0.23 |

stock numbers-at-age (except age 1) were calculated using Equations (A1) and (A2). For the past period, stock numbers-at-age 1 corresponded to the ICES estimates. For projected years, stock recruitment values at age 1 were estimated using a Ricker stock–recruitment relationship fitted to ICES data (ICES, 2006). Significant autocorrelation was observed for the three stocks in the past, reflecting the variations in environmental conditions that influence recruitment. Therefore, an error term including an autocorrelation parameter linking the residual of the year with the residual from the previous year, plus a lognormally distributed random error, was added to account for natural variability and uncertainty in the recruitment process. The same values of recruitment error by year and iteration were used in all scenarios.

OM weights-at-age in past years were equal to WG estimates, whereas in projected years they were simulated using a mean weight-at-age smoothed over the whole WG data range multiplied by an uncertainty term that also included autocorrelation reflecting the environmental conditions. For historical years, fishery selectivity-at-age was calculated by dividing $F$-at-age by the average $F$ over ages 5–8 ($F_{\mathrm{bar}}$). In projected years, selectivity-at-age and its variability were simulated in a similar way to weight-at-age.

We also included some simple elements of fleet dynamics to address the real situation of a mixed fishery with technical interactions. Fleets PT and LG were included explicitly in the model, and the OTH fleet was added as a fleet with constant effort and catchability. In historical years, effort, catch, and price data for the two fleets were those available from the Faroese Fisheries Laboratory. Historical values of catchability for each fleet were estimated by dividing the partial $F$ by the effort. For projected years, values of catchability were simulated by adding a lognormal random value on the average value of historical years [Equation (A10)]. In the absence of fleet-based catch-at-age data for certain periods, the overall fishery selectivity was applied to all fleets. Price values for the different species were kept constant over the projected years. No fishery costs data were directly available for these fleets, but some costs information was available in the Annual Economic Report for 2005, though were not fully consistent with the fleet data used here. Buisman *et al.* (2009) collected additional economic information directly from a Faroese auditing company. Therefore, only rough economic considerations dealing with revenue and value per unit effort (vpue) were included.

The OM value for $F$ is the key parameter linking the fleets, the stocks, and the management system. In the past period, no assumption was made regarding $F$, and it was calculated simply as the sum of partial $F$ by fleet. For the projected period, $F$ was calculated differently according to the management system simulated and the corresponding underlying assumptions and knowledge, as described below.

In a single-species TAC system without mixed-fisheries interactions, it is assumed that management influences only the fleet-specific catches and that all fleets catch their full quota. Estimated $F$ was allocated between fleets using a fixed allocation key, built using the historical average catches of the fleets, similar to the procedure for deploying effort using a fixed allocation key of days at sea.

In a TAE system, the effort of fleets is assumed to be directly influenced by management. The partial $F$-values induced by each fleet were calculated using effort, catchability, and selectivity estimates. In the present scenarios, the effort-allocation key was also considered constant and calculated based on historical data, which were stable over the 10 years of implementation. However, the model was also able to simulate scenarios dealing with changes in capacity and in the relative structure of the fleet in terms of the distribution over gears.

It was assumed that there was no discarding, and landings were set equal to catches and calculated for each fleet using Equation (A16). Although no estimates of discarding are available from the fisheries, the incentives to discard to highgrade the catches are considered to be low under present management arrangements.

## The management procedure

The MP model simulates, in a simplified manner, how the real stock represented in the OM is perceived and managed by scientific advisors and managers. Catches in the MP stock model were set equal to those in the OM stock model; sampling errors were not considered because no information was directly available on sampling accuracy. These catches were used as inputs to the assessment procedure, reflecting the fact that imperfect knowledge of the true stock abundance (use of an assessment model rather than a perfect monitoring of the stock) is a major source of bias and uncertainty in management decisions (Butterworth, 2007). The stock assessment was run using extended survivors analysis, XSA

(Shepherd, 1999), in FLR. The tuning procedure used an abundance index based on real OM stock numbers, with a lognormal error with a standard deviation equal to 0.3. This value was close to the standard error actually observed in both survey indices currently used in the assessment (ICES, 2006).

The outputs of the assessment procedure were used to derive future management actions, using a fixed HCR. These HCRs estimated the target $F$ to be applied in the following year ($F_{y+1}$) given the estimated SSB in the current year, $\text{SSB}_y$. The HCR implemented followed usual ICES procedures for the precautionary approach, except that the $F_{\text{target}}$ was set at 0.45 rather than $F_{\text{pa}}$:

$$F_{y+1} = \begin{cases} F_{\text{low}}(=0.2) & \text{if } \text{SSB}_y < B_{\text{lim}} \\ F_{\text{target}}(=0.45) & \text{if } \text{SSB}_y > B_{\text{pa}} \\ F_{\text{low}} + (F_{\text{target}} - F_{\text{low}})(\text{SSB}_y - B_{\text{lim}})/(B_{\text{pa}} - B_{\text{lim}}) & \\ & \text{if } B_{\text{lim}} < \text{SSB}_y < B_{\text{pa}}. \end{cases} \quad (1)$$

$F_{y+1}$ was accordingly translated into a $F_{\text{mult}}$ multiplier of perceived mean fishing effort in year $y$.

The key difference between the TAC and the TAE systems lies in the implementation of management rules in the following year, and how this links back to the OM. In a TAC system, the management rule is implemented using a short-term forecast predicting stock abundance in year $y+1$. This standard procedure in ICES requires assumptions (traditionally an average over the most recent years) concerning fish growth (i.e. weights-at-age), recruitment, and $F$ (fishing pattern) for the forecast years. The target $F_{y+1}$ was converted into a TAC using the catch equation

$$\text{TAC}_{y+1} = N_{y+1}\left[\frac{F_{y+1}}{M + F_{y+1}}\right](1 - e^{-(M+F_{y+1})})\text{Wt}. \quad (2)$$

In turn, this TAC was translated into the "true" $F_{y+1}$ in the OM, given the selectivity, weight-at-age, and recruitment parameters simulated for that year. Then the key source of uncertainty in management comprised the "true" values of these parameters in the OM in year $y+1$, which would differ from the projected values used in the MP at year $y$, so the TAC would in practice not correspond to the expected target mortality $F_{y+1}$. Therefore, the "true" $F$ in the OM would have to be recalculated. In a TAE system, no short-term forecast is necessary. The effort level as the total number of fishing days allowed in the fishery was directly estimated from the target $F$ assuming that catchability was constant [Equation (3) below]. In this case, the key source of uncertainty in management was the "true" value of catchability in the OM in year $y+1$, which would differ from the projected value used in the MP in year $y$. The difference can be large, because the fleets' catchability $CV$ is estimated to be 0.2–0.5 (Table 1). Therefore, the effort level ($E$) would in practice largely not correspond to the expected target mortality $F_{y+1}$.

$$E_{y+1} = \frac{F_{y+1}}{\text{catchability} \times \text{selectivity}} \quad (3)$$

Hence, both management systems rely on major assumptions and sources of uncertainty, but the sources differ between the two systems.

## Scenarios and evaluation criteria

A number of scenarios were run, each one based on 100 stochastic Monte Carlo iterations. For each of the three stocks, a single set of random errors by parameter, year, and iteration was generated during the initialization phase and was subsequently used across all scenarios, allowing comparison of the results.

The effect of the (lack of) flexibility of the system to adjusting management to scientific recommendations was tested under three different levels of maximum IAV in management decisions (both TAC and TAE), referred to as bounds (Kell *et al.*, 2006): 1% (as a contrasting proxy of a fully "rigid" system inspired by the historical situation in the Faroese fisheries, hereafter referred to as the "rigid scenario"), 15% [as a generic proxy for EU long-term management plans (Penas, 2007; STECF, 2007a), hereafter referred to as the "medium scenario"], and without bound (full flexibility, hereafter referred to as the "full-flexible scenario").

Although the Faroese-type system studied is by essence a mixed-fishery system, the comparison between TAE and TAC was first conducted on a single-species basis for the three stocks individually. This was motivated by the fact that EU fisheries are still regulated by single-species TACs, although they are in reality almost all mixed fisheries. In addition, this allowed obtaining more generic and theoretical understandings of the expected behaviour of the TAC and TAE systems, respectively. However, this single-species analysis is not intended to mask the main problem that TACs do not perform well in the Faroese Islands because of such mixed-fisheries interactions.

Comparing TAC and TAE systems in mixed fisheries where stocks are caught together is more difficult. For a TAE, one can assume, as we have done here, that management will be precautionary and that the final effort applied is set at the minimum across effort levels corresponding to the $F$ defined by the HCR for each individual species (Ulrich *et al.*, 2002; Kraak *et al.*, 2008). We also assumed perfect compliance with that level. For a TAC system, however, additional assumptions would be required for setting a particular combination of the three TACs as well as assumptions for the "true" effort levels under that combination of TACs (ICES, 2007b, 2008b, 2009). In the absence of underlying knowledge supporting these assumptions, we decided not to run the simulations with a full annual feedback loop between the fishery and the fish population. However, a central question to investigate is whether, if the effort management is perceived to be precautionary, there would be incentives, year after year, to switch back to a TAC in the short-term because it gives bigger catches and more value. Is there merit in the industry maintaining low levels of effort over a long period to allow the stocks to recover and hence to achieve high value output again?

To evaluate this question, we ran the simulation as with the TAE in mixed fisheries, but for each year $y$ of the simulation we also estimated the single-species TACs that would apply if a TAC system would be implemented in year $y+1$, using the same short-term forecasting procedures as in the single-stock approach. The difference in catches in year $y+1$ between this "potential TAC" and the full TAE model (i.e. calculated based on the stock assessment results up to year $y-1$) is calculated both in landings weight by stock and landings value by fleet, using the same allocation key as in the single-stock approach. As such, we measured how the incentives varied year after year, alongside the recovery of stocks.

Scenarios were compared using a set of usual evaluation criteria: (i) the probability of SSB being above $B_{\text{pa}}$ and $F$ being below 0.45 over the 10 projected years as a measurement of biological robustness and sustainability; (ii) the average revenue of catches for both fleets over years as a simple and rough proxy for economic efficiency, and vpue was also calculated for TAE scenarios; and (iii) the average IAV of the control variable (TAC or TAE) as a loose proxy for social robustness, assuming that the fishing industry would prefer stability and consistency in management decisions.

## Results

### Single-species approach

An MSE such as that conducted here usually produces extensive quantitative results and must be summarized through synthesis of the outputs. Scenarios are compared using time-series (Figure 1), and medians and coefficients of variation (*CVs*) over the projected period for the selected criteria (Tables 2 and 3).

The first finding from our analyses is that there is obviously no simple and generic answer as to which system performs best in a single-species-management scenario. There are extensive differences in the evaluation criteria, although the model was run with identical assumptions on stock dynamics, identical HCRs to estimate target $F_{y+1}$, and identical generated random errors,

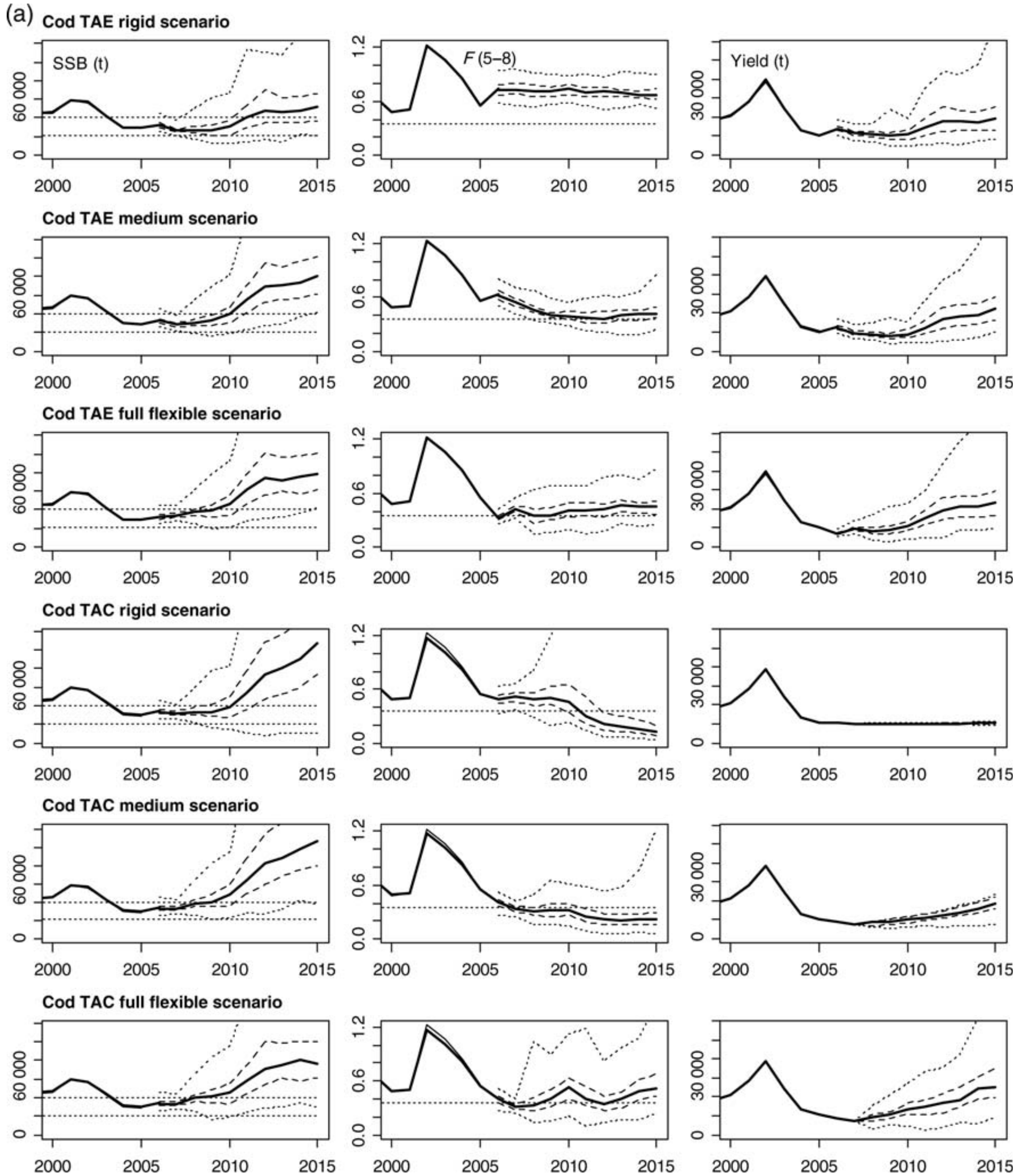


**Figure 1.** Time-series of SSB, fishing mortality ($F$), and yield and of the single-stock model for (a) cod, (b) haddock, and (c) saithe, with both TAE and TAC management. In the historical period (up to 2005), the thin line represents the estimates from the Working Group and the heavy line the deterministic calculations from the OM. In the projected period, the solid line represents the median values of the estimates from the OM. The quantiles 0.25/0.75 and 0.05/0.95 for the OM estimates are represented by the dashed and dotted lines, respectively. The straight dotted lines on the biomass graph represent $B_{pa}$ (upper) and $B_{lim}$ (lower). The one on the graph of $F$ represents $F_{pa}$.

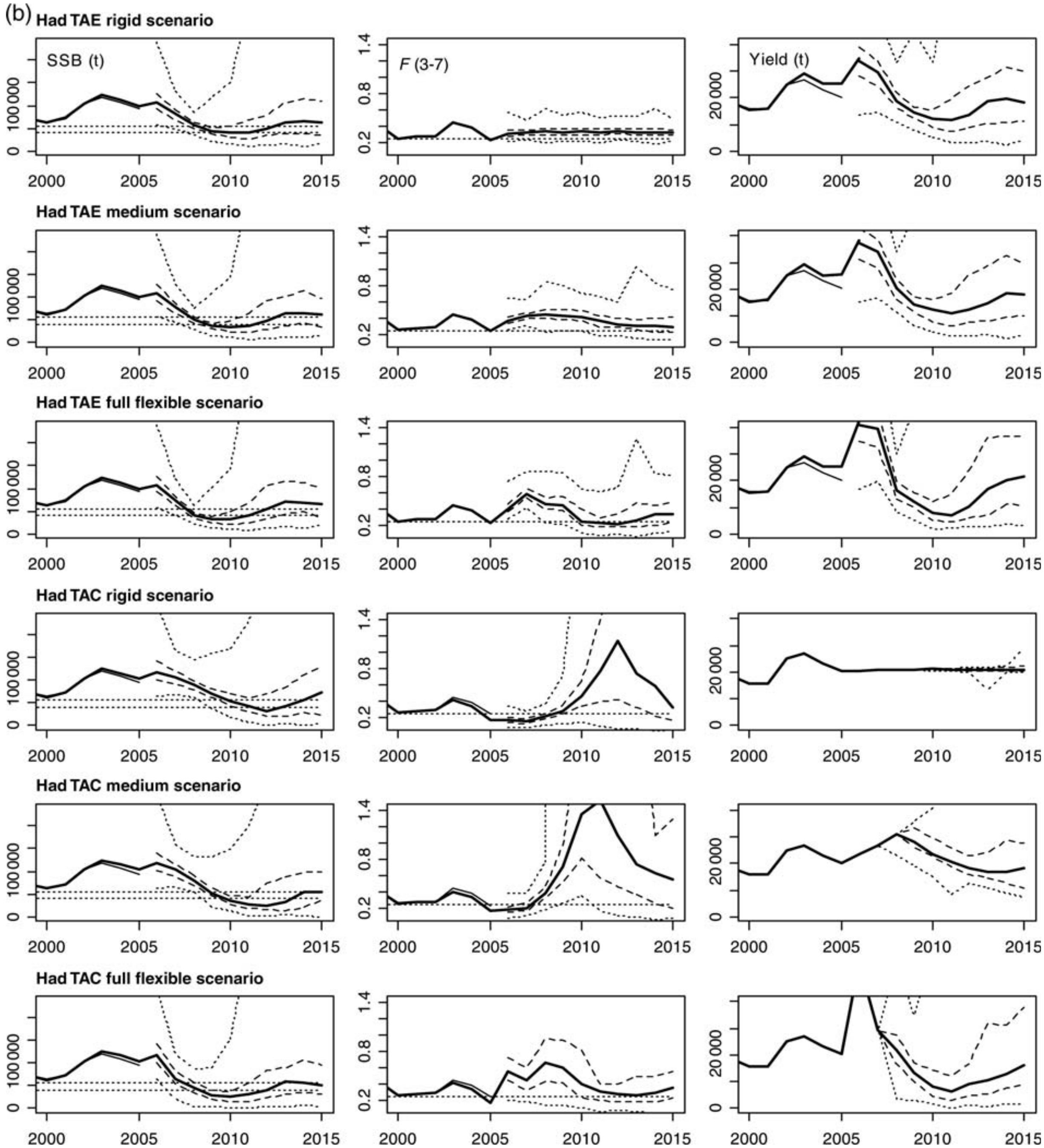


**Figure 1.** Continued.



dealing with three gadoid stocks. It is clear that the specific dynamics of each stock play a major role in the success of individual management strategies. In particular, the biomass starting conditions at the onset of the projected period seem to be a key determinant, because they influence the performance of the HCR.

The cod stock was assessed to be between $B_{pa}$ and $B_{lim}$ in 2006. The single-species TAC system appeared to be more sustainable than the TAE system at low (1%) and medium (15%) bounds on the IAV, with lower risks of SSB falling below $B_{pa}$ (Figure 1). The TAE system provided higher mean levels of catch and revenue, but the uncertainty was also much higher. Both systems performed equally well in relation to attaining sustainability levels in the "full-flexible" scenario. Within a TAE system, the same average catches and revenues were obtained with both the rigid (1% IAV) and full-flexible scenarios, but in the rigid scenario these were obtained with much higher effort (and hence lower catch per unit effort, cpue, and likely less economic efficiency) at the cost of much lower SSB (Table 2).

The haddock (*Melanogrammus aeglefinus*) stock was assessed to be within safe biological limits and largely above $B_{pa}$ in 2006. The stock fluctuated most in terms of recruitment, growth, and catchability, and simulations showed greater uncertainty in terms of future development than for cod and saithe (Figure 1). For haddock, the probability to stay within precautionary biomass

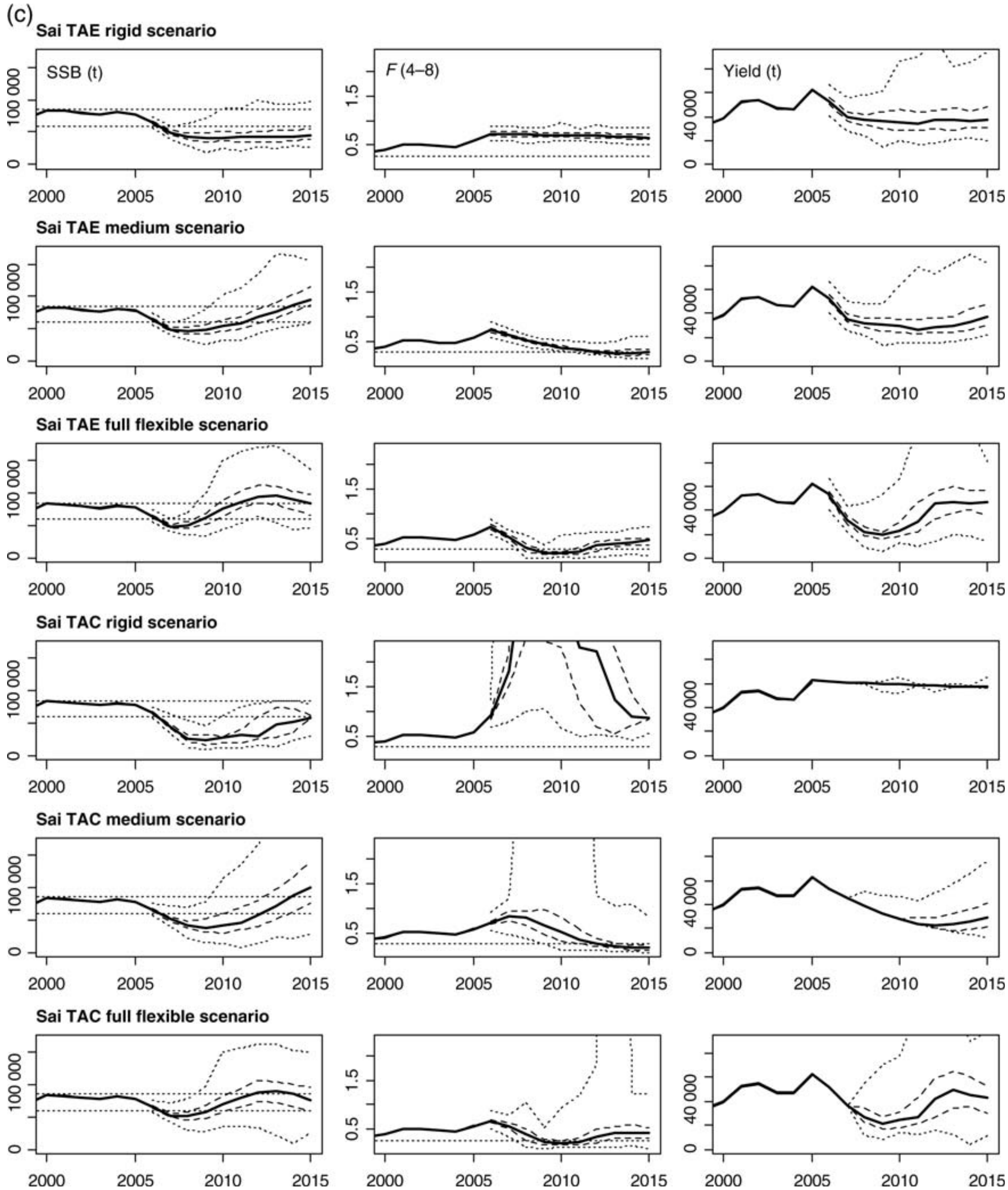


**Figure 1.** Continued.

limits was around or above 50% in all scenarios, especially with a rigid (1% IAV bound) TAC maintaining catches at low levels (Table 3). Haddock displayed successive years of low recruitment with strong autocorrelation in the stock–recruitment relationship, and a few strong year classes. There is a risk that biomass would fall below $B_{\mathrm{lim}}$ in future if recruitment fails. It is to be noted, though, that the XSA assessment procedure did not always perform well for this stock under a TAC system, with possible large assessment bias. The TAE system appeared to be fairly robust to the level of inter-annual flexibility, with comparable mean catches, SSB, and vpue across scenarios, although the rigid scenario seemed to reduce the range of uncertainty and the risk of large year-on-year fluctuations in management. In a single-species analysis, the current Faroese-type system seems to perform well compared with alternative management for that stock (Table 2).

The saithe stock was assessed to be around $B_{\mathrm{lim}}$ in 2006, but was the most depleted stock in the simulations. The level of flexibility seemed to be the main factor behind the success, rather than the management system itself (Figure 1). In a rigid scenario, both systems maintained the stock below $B_{\mathrm{pa}}$ in the projected period. Some signs of recovery were observed at the end of the period with the TAC system, but the uncertainty was also greater (Table 3). Stock recovery was achieved fairly quickly within both management systems under a full-flexible scenario, but this

**Table 2.** Result values of the single-stock model with TAE management for (top) cod, (middle) haddock, and (bottom) saithe over the projected period.

| Scenario | Catches | IAV in catches | SSB | $P(SSB > B_{pa})$ | $P$ $(F < F_{target})$ | Effort LG | Effort PT | Revenue LG | Revenue PT | IAV in revenue | Vpue LG | Vpue PT |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Rigid | 15 141 | 1.11 | 40 824 | 0.37 | 0 | 2 093 | 4 971 | 64 243 | 34 526 | 1.09 | 31 | 7 |
| | 0.47 | 0.31 | 0.49 | | | 0.03 | 0.03 | 0.47 | 0.47 | 0.32 | 0.49 | 0.49 |
| Medium | 14 248 | 1.13 | 55 119 | 0.56 | 0.23 | 1 307 | 3 105 | 60 457 | 32 491 | 1.11 | 48 | 11 |
| | 0.55 | 0.3 | 0.55 | | | 0.23 | 0.23 | 0.55 | 0.55 | 0.32 | 0.57 | 0.57 |
| Full | 15 128 | 1.17 | 59 411 | 0.66 | 0.33 | 1 194 | 2 836 | 64 188 | 34 496 | 1.22 | 53 | 12 |
| flexible | 0.6 | 0.42 | 0.51 | | | 0.24 | 0.24 | 0.6 | 0.6 | 0.4 | 0.5 | 0.5 |
| Rigid | 22 686 | 1.08 | 76 377 | 0.57 | 0.05 | 2 288 | 5 434 | 86 729 | 32 450 | 1.05 | 39 | 6 |
| | 0.76 | 0.5 | 0.76 | | | 0.02 | 0.02 | 0.76 | 0.76 | 0.52 | 0.75 | 0.75 |
| Medium | 24 107 | 1.09 | 73 437 | 0.52 | 0.1 | 2 617 | 6 216 | 92 161 | 34 482 | 1.05 | 35 | 5 |
| | 0.83 | 0.53 | 0.8 | | | 0.23 | 0.23 | 0.83 | 0.83 | 0.54 | 0.77 | 0.77 |
| Full | 24 972 | 1.21 | 74 570 | 0.5 | 0.27 | 2 612 | 6 202 | 95 467 | 35 719 | 1.16 | 35 | 6 |
| flexible | 0.94 | 0.71 | 0.82 | | | 0.39 | 0.39 | 0.94 | 0.94 | 0.75 | 0.82 | 0.82 |
| Rigid | 40 548 | 0.99 | 48 772 | 0.01 | 0 | 2 093 | 4 970 | 341 | 151 435 | 1 | 0.2 | 31 |
| | 0.29 | 0.26 | 0.27 | | | 0.03 | 0.03 | 0.29 | 0.29 | 0.27 | 0.3 | 0.3 |
| Medium | 35 683 | 0.98 | 69 196 | 0.22 | 0.2 | 1 255 | 2 981 | 300 | 133 267 | 1 | 0.3 | 49 |
| | 0.34 | 0.25 | 0.35 | | | 0.37 | 0.37 | 0.34 | 0.34 | 0.26 | 0.4 | 0.4 |
| Full | 37 948 | 1.05 | 79 225 | 0.38 | 0.36 | 1 150 | 2 731 | 319 | 141 728 | 1.08 | 0.3 | 55 |
| flexible | 0.46 | 0.44 | 0.33 | | | 0.45 | 0.45 | 0.46 | 0.46 | 0.45 | 0.36 | 0.36 |

Upper values are medians, and lower values are *CV*s.

**Table 3.** Results of the single-stock model with TAC management for (top) cod, (middle) haddock, and (bottom) saithe over the projected period.

| Scenario | Catches | IAV in catches | SSB | $P$ $(SSB > B_{pa})$ | $P$ $(F < F_{target})$ | Revenue LG | Revenue PT | IAV in revenue | TAC |
|---|---|---|---|---|---|---|---|---|---|
| Rigid | 10 278 | 1 | 62 001 | 0.52 | 0.43 | 41 702 | 25 021 | 1 | 10 278 |
| | 0.03 | 0.01 | 0.72 | | | 0.03 | 0.03 | 0.01 | 0.03 |
| Medium | 11 386 | 1.06 | 70 680 | 0.67 | 0.72 | 46 198 | 27 719 | 1.09 | 11 386 |
| | 0.37 | 0.13 | 0.65 | | | 0.37 | 0.37 | 0.11 | 0.37 |
| Full | 15 385 | 1.15 | 59 954 | 0.66 | 0.36 | 62 425 | 37 455 | 1.18 | 15 385 |
| flexible | 0.59 | 0.32 | 0.52 | | | 0.59 | 0.59 | 0.32 | 0.59 |
| Rigid | 20 992 | 1 | 84 582 | 0.69 | 0.44 | 80 452 | 30 761 | 1.01 | 20 967 |
| | 0.03 | 0.01 | 0.72 | | | 0.07 | 0.07 | 0.09 | 0.03 |
| Medium | 24 698 | 1.01 | 75 518 | 0.58 | 0.27 | 94 168 | 36 005 | 1 | 24 110 |
| | 0.34 | 0.25 | 0.82 | | | 0.34 | 0.34 | 0.23 | 0.34 |
| Full | 23 654 | 1.25 | 66 009 | 0.44 | 0.25 | 90 691 | 34 676 | 1.1 | 23 771 |
| flexible | 0.99 | 0.71 | 0.91 | | | 0.99 | 0.99 | 0.73 | 1.01 |
| Rigid | 59 302 | 0.99 | 45 968 | 0.1 | 0.05 | 5 723 | 217 454 | 1 | 54 112 |
| | 0.03 | 0.01 | 0.46 | | | 0.04 | 0.04 | 0.04 | 0.28 |
| Medium | 33 923 | 0.93 | 66 442 | 0.2 | 0.33 | 3 270 | 124 248 | 0.94 | 33 323 |
| | 0.33 | 0.14 | 0.57 | | | 0.33 | 0.33 | 0.14 | 0.33 |
| Full | 38 813 | 1.04 | 75 533 | 0.3 | 0.32 | 3 741 | 142 165 | 1.07 | 38 127 |
| flexible | 0.47 | 0.44 | 0.34 | | | 0.47 | 0.47 | 0.45 | 0.47 |

Upper values are medians, and lower values *CV*s.

flexibility hampered maintenance of sustainability in the long term, because $F$ increased as soon as the stock reached $B_{pa}$, so jeopardizing the effect of the stock recovery.

## Mixed-fisheries approach

In the mixed-fisheries approach, the three stocks were not considered in isolation but assumed to be caught simultaneously, and the catches for the three stocks depended on the total effort level by fleet. Management was assumed to be precautionary, i.e. to adapt to the level of the most depleted stock, which was saithe at the beginning of the period (Table 4). This meant that the results obtained were globally similar to those in the single-stock approach for that stock.

Similarly, the effort levels estimated for cod in the single-species approach were close to those for saithe, and as such the mixed-fisheries results were close to those described under the single-stock approach. However, results differed for haddock, whose HCR would suggest that larger catches were sustainable at the beginning of the period. This would lead to underexploitation of the stock, compared with whether the stock was under a single-species quota. This had economic consequences for the fishery, especially for the PT fleet, which experienced systematic positive incentives to revert to a TAC system, in contrast to the longliners for which these were negative or close to zero (Figure 2). However, the simulations for haddock showed also a high probability of poor recruitment in future, i.e. a high probability of it becoming the most depleted stock after 2010, so driving the HCR in the simulations. In that case, precautionary mixed-fisheries management, which would shift from adapting to saithe at the beginning of the period to adapting to haddock later, would help to maintain the stock above $B_{pa}$ throughout the period, even for successive poor recruitments.

Overall, the mixed-fisheries approach showed that the medium scenario (15% IAV bound) appeared to be the most sustainable. The rigid scenario provided the best catches and revenues, but resulted in an unsustainable fishery, illustrated by the lower probabilities of biomass being above $B_{pa}$ for cod and haddock. On the other hand, a full-flexible system maintained $F$ below $F_{target}$ but did not allow the most depleted stock to recover fully above $B_{pa}$, because catches and effort would be allowed to increase too quickly after the first signs of recovery. Moreover, as observed in single-species simulations, uncertainty (i.e. *CV* values) generally increased with the level of interannual flexibility (Table 4). Such observations confirmed the need for a balanced trade-off between flexibility and uncertainty.

However, although this medium scenario may appear to be a reasonable compromise between stock sustainability, stability, revenue, and vpue, it also created the largest incentives to revert to single-stock TACs (Figure 2). By maintaining the effort at a conservative level with medium variability, the scenario showed high probabilities of stock recovery, so increasing the potential for high TACs in the short term, especially for the LG fleet. In comparison, the rigid scenario showed strong negative incentives for the PT fleet, mainly because the limited decrease in effort maintained high catches of saithe compared with those of a precautionary TAC. The incentives were closer to zero in the full-flexible scenario, because the effort adapted automatically to the most depleted stock, also when stocks recovered, and followed more closely the potential effect of the single-species TACs.

## Discussion

We could not ascertain whether TAEs led systematically to more biological robustness than TACs. In a single-species approach over a 10-year projected period, this was only true for the most depleted stock, where mean biomass was higher and uncertainty lower than with the TAC scenarios. For the other two stocks, and without accounting for mixed-fishery interactions, single-species TACs performed equally well or sometimes better than a TAE system. Revenues were significantly higher under a TAE system for the cod stock only. For the other stocks, revenues of both fleets were comparable or higher with the TAC system, and the uncertainty was generally less when dealing with rigid (1% bound on IAV) or medium (15% bound on IAV) scenarios.

Effort-based HCRs were expected to be more biologically robust than catch-based HCRs because they are less dependent on uncertainty in growth, recruitment, and the results of stock assessments. There were a number of simulations where the TAC system induced large fluctuations in $F$, along with poor performance of the assessment method. However, it was not clear from the results that the TAE was more biologically robust when looking at each stock in isolation. For a highly variable stock such as haddock, effort management stabilized the harvest level around the target, but led to highly variable catches, whereas TAC influenced the outcome in the opposite direction, stabilizing the catches but leading to variable harvest rates. However, their performances in terms of maintaining the biomass within safe biological limits were comparable. One key issue in a TAE system is

**Table 4.** Results of the mixed-fishery model with TAE management for cod, haddock, and saithe over the projected period, in (top) rigid (1% bound on IAV), (middle) medium (15% bound on IAV), and (bottom) full-flexible scenarios (no bound on IAV).

| Stock | Catches | IAV in catches | SSB | $P$ (SSB > $B_{pa}$) | $P$ ($F < F_{target}$) | Effort LG | Effort PT | Revenue LG | Revenue PT | IAV in revenue LG | IAV in revenue PT | Vpue LG | Vpue PT |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cod | 15 150 | 1.12 | 41 049 | 0.37 | 0 | – | – | – | – | – | – | – | – |
| | 0.47 | 0.31 | 0.49 | | | – | – | – | – | – | – | – | – |
| Haddock | 21 016 | 1.08 | 75 314 | 0.57 | 0.13 | 2 093 | 4 970 | 144 966 | 217 620 | 1.04 | 1.0 | 69.4 | 43.8 |
| | 0.7 | 0.51 | 0.72 | | | 0.03 | 0.03 | 0.46 | 0.25 | 0.34 | 0.23 | 0.47 | 0.25 |
| Saithe | 40 970 | 0.99 | 49 127 | 0.01 | 0 | – | – | – | – | – | – | – | – |
| | 0.29 | 0.27 | 0.27 | | | – | – | – | – | – | – | – | – |
| | | | | | | | | | | | | | |
| Cod | 12 886 | 1.11 | 58 164 | 0.55 | 0.42 | – | – | – | – | – | – | – | – |
| | 0.49 | 0.29 | 0.6 | | | – | – | – | – | – | – | – | – |
| Haddock | 15 901 | 1.03 | 92 183 | 0.74 | 0.87 | 1 168 | 2 773 | 115 764 | 186 294 | 1.03 | 1.0 | 106 | 71.5 |
| | 0.71 | 0.48 | 0.74 | | | 0.31 | 0.31 | 0.45 | 0.29 | 0.32 | 0.22 | 0.56 | 0.35 |
| Saithe | 35 924 | 0.99 | 72 723 | 0.24 | 0.18 | – | – | – | – | – | – | – | – |
| | 0.32 | 0.25 | 0.32 | | | – | – | – | – | – | – | – | – |
| | | | | | | | | | | | | | |
| Cod | 14 222 | 1.17 | 62 564 | 0.67 | 0.47 | – | – | – | – | – | – | – | – |
| | 0.61 | 0.45 | 0.55 | | | – | – | – | – | – | – | – | – |
| Haddock | 15 076 | 1.09 | 93 868 | 0.8 | 0.94 | 1 068 | 2 536 | 118 286 | 190 426 | 1.15 | 1.1 | 112 | 76 |
| | 0.68 | 0.63 | 0.63 | | | 0.27 | 0.27 | 0.49 | 0.34 | 0.46 | 0.37 | 0.44 | 0.24 |
| Saithe | 36 530 | 1.03 | 77 948 | 0.27 | 0.23 | – | – | – | – | – | – | – | – |
| | 0.36 | 0.42 | 0.23 | | | – | – | – | – | – | – | – | – |

Upper values are medians, and lower values *CV*s.

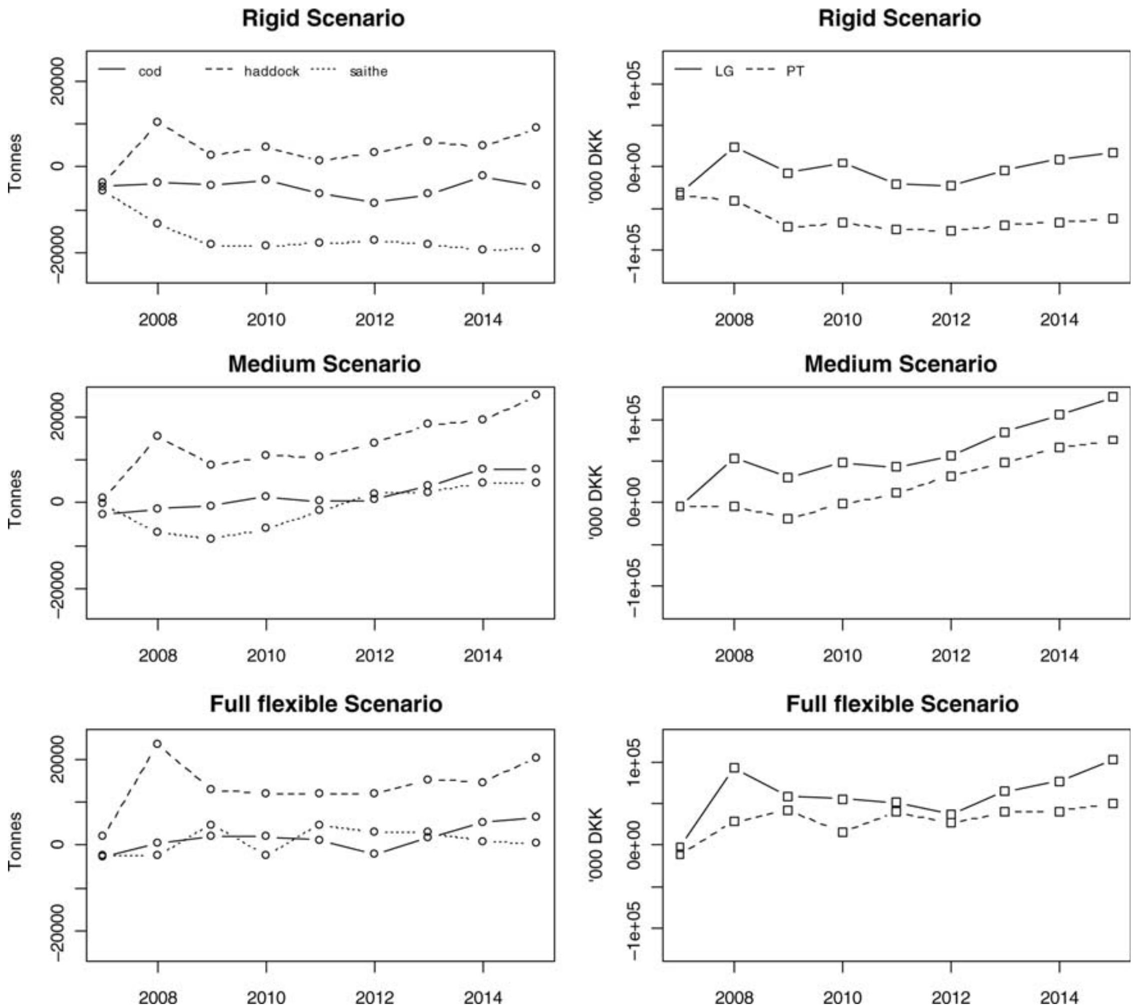


**Figure 2.** Difference between the potential TAC and the actual catches from the TAE mixed-fisheries scenarios, in weight by stock (left) and value by fleet (right).

the great uncertainty in catchability estimates, which blurs the relationship between *E* and *F*. The uncertainty in this parameter is comparatively higher than the uncertainty in biological forecast parameters, owing to the generally poor relationship between *E* and *F*, and the potentially great impact of environmental variables (Jákupsstovu *et al.*, 2007). This fact undermines the ability of a TAE to control *F* effectively. This issue is a generic feature in any effort-management system, but in the particular context of the Faroese-type fisheries studied here, where nominal effort is high and lacking in flexibility, this high variability in catchability contributes substantially to the risks of non-sustainability of the system.

Our mixed-fisheries approach, based simply on the combined catches of the main fleets, showed that single-species objectives cannot be met simultaneously because of differences in the dynamics and initial states of the various stocks. One single precautionary effort applied to all fleets would ensure biological sustainability and high revenues, but also lead to underexploitation of some stocks in the short-term and the greatest interannual variability in the catches, which may not be desirable for the fishing industry. On the contrary, one single effort set at a high level, as happens in the rigid scenario, would jeopardize the sustainability of the most-depleted stocks. The results showed little probability of stocks being within safe biological limits, especially saithe, which is assessed here as being the most depleted stock and the one least likely to produce strong recruitment soon. The TAE system maintained *F* well above $F_{pa}$.

However, the TAE system would more often be sustainable if it was more flexible in following the scientific recommendations. Even with extensive fluctuations in the stock dynamics, and great uncertainty in the catchability of the fleets, there is high probability of maintaining cod and haddock stocks above $B_{pa}$ and recovering the saithe stock, leading to higher average values of vpue over the whole period. As a consequence, medium flexibility in the system would appear to be the most attractive compromise, but such an intermediate effort-management scenario creates incentives to revert to single-stock TACs in the medium term as stocks recover.

Similar patterns were obtained by Buisman *et al.* (2009), who observed that an effort control system does not seem to improve the economic performance of the Faroese fleet. Setting the effort level at an intermediate level, but with additional measures to protect the depleted stock, would appear to be an acceptable compromise between sustainability and optimal yield, if there is some spatio-temporal separation of stocks on some fishing grounds. In reality, the Faroe Islands also have an advanced system of technical measures ensuring clear spatial separation between gears, but these need also be designed carefully to ascertain the best exploitation scenarios for the various stocks.

The model we have presented here is a mixed-fisheries extension of a now-established standard for single-species MSE based on assessment data and using FLR (Kell *et al.*, 2007; www.efimas.org; Hamon *et al.*, 2007; ICES, 2007a; STECF, 2007a, 2008). Major sources of uncertainties and variability were included to cover a wide range of plausible developments of the stocks. Although the model failed to reflect the full complexity of Faroese fisheries, particular attention was paid to an acknowledged causality, such as when sinusoidal fluctuations in primary production possibly may have had a strong influence on the recruitment and growth of the stocks (ICES, 2008a). This was accounted for by including the significant autocorrelation parameter observed in the assessment data in the projection. As such, the scenarios are likely to produce a number of worst case iterations, with consecutive years of poor recruitment and slow growth. New stock assessment data were made available after the work presented here was performed (ICES, 2008a), evidencing medium-to-low recent recruitments for the three stocks. Although this may slightly affect the results of our simulations (given the importance of starting conditions on medium-term simulation results), it is not expected that the main findings and comparisons of scenarios would differ substantially, because most iterations involved reduced recruitment in the first years. However, several other potential sources of uncertainty were disregarded, in the absence of available data to condition upon, perhaps impacting the results and conclusions of this study. Theoretical sensitivity analyses could be conducted, but they would likely not be fully informative in the absence of reference levels about the actual situation. In particular, no sampling error on the catches was included in the MP. It should be emphasized that sampling error in an effort-regulation system may likely be proportionally lower than in a TAC system in terms of bias, because of the different incentives to misreport catches.

Here, we used experience from the only European full implementation of effort management in a mixed groundfish fishery to formulate our model and to evaluate the potential outcomes of effort management vs. TAC management. Our results were similar to those of some previous comparative studies for other stocks under TAC regulation and considered in isolation (e.g. Kell *et al.*, 2006; STECF, 2007a, 2008). However, the key issue is the mixed-fisheries interactions, and our results clearly show that single-species TACs would not be consistent with each other when the stocks are caught by the same fleets. Other studies have also compared TAC and potential TAE in mixed fisheries, but based on sometimes simplistic assumptions on linkages between a set of TACs and the resulting effort (Ulrich *et al.*, 2002; ICES, 2007b, 2008b, 2009; Kraak *et al.*, 2008). However, the truth is obviously more complex, especially when more than two species are being considered. These apparently weak direct linkages can be analysed in many different ways, often on a trip-by-trip basis (Marchal *et al.*, 2006; Gillis *et al.*, 2008). The absence of a clear relationship is actually one of the main concerns with TAC management. Decreasing TACs have not contributed to reducing capacity and fishing activity in mixed-fisheries management, unless being supplemented with major effort reduction programmes (days at sea, and decommissioning schemes; STECF, 2007b; Reeves *et al.*, 2008; ICES, 2009). Beddington *et al.* (2007) even suggested that overcapacity could be intrinsically induced by a TAC system. Van Oostenbrugge *et al.* (2008) demonstrated non-linear optimizing behaviour in a restrictive effort-management system, and accounting for this may have drastic consequences on the results from a MSE framework (Kraak *et al.*, 2008). Their findings could certainly be applied to Faroese fisheries, but individual trip data were not available for our work.

The case of management of the Faroese fisheries is particularly interesting. These fisheries have been experiencing the same problems of managing mixed fisheries with TACs as in other parts of the western world. However, their smaller scale, their closure to foreign fleets, their uni-jurisdictional management, the importance of fisheries to society, and some co-management schemes between industry, scientists, and management bodies have made it possible to establish new governance rules and new innovative management systems. However, even under these favourable conditions, 10 years of experience have proved that the system has not achieved all its conservation objectives (Jákupsstovu *et al.*, 2007), partly because the system has long not been effectively restrictive (Løkkegard *et al.*, 2007). The initial effort agreed in 1996 was set at a high level, which only prevented the effort exerted increasing but did not actually limit it. The fishery was nevertheless maintained at high productivity during the period of analysis, because recruitment to the stocks has been relatively high. Since then, there has been resistance from industry to decreasing the amount of effort authorized, but the current decrease in the cod stock has led to recent proposals for drastic reductions which are impacting the whole of Faroese society (Anon., 2008). Our results suggest that effort management seems to be appropriate, but that some interannual flexibility in the system would appear to be the best compromise between short- and long-term objectives, as well as between biological sustainability and economic return. This would allow adapting management to natural fluctuations in stock abundance and uncertainty in the catchability parameter.

MSE is increasingly being used in a management context. However, although an MSE is designed theoretically to evaluate the robustness of management strategies to uncertainty, and although generic guidelines are being proposed (ICES, 2007a, 2008c; Rademeyer *et al.*, 2007), experience gained recently underlines the difficulty in building consistent and scientifically validated MSEs. MSE is only a projected simplification of reality and includes few processes. For instance, the known fishing processes of targeting cod regardless of price in a mixed fishery is difficult to implement and have not been included in the evaluation here. Similarly, we used an assessment method (XSA) for saithe, although its use for that stock is considered by ICES (2006) to be too uncertain to yield robust scientific advice. We have stressed, though, the sensitivity of the model to the starting conditions.

Model uncertainty, i.e. the uncertainty arising from the design of the model itself, is a key source of uncertainty. From an academic perspective, this can be addressed through additional testing of alternative underlying hypotheses. Ultimately, however, it must always be borne in mind that MSEs are exploratory tools, not providers of solutions. Their principal purpose is to serve as quantitative support to decision-making, by simulating and comparing a range of plausible futures, so helping to reach management consensus agreement, as advocated by Rochet and Rice (2009), rather than providing absolute predictions. In this respect, the FLR toolbox has proven to be a very useful approach, providing a flexible, transparent, and consistent approach (Schnute *et al.*, 2007).

## Conclusion and perspectives

Despite its many and necessary simplifications, this simulation has led to a number of results and conclusions about potential benefits and drawbacks of two alternative management systems in various contexts. A TAE system is not likely to be implemented in a European single-species fishery, so the single-species analysis was mainly useful in deriving generic understanding of the behaviour of TAC and TAE systems under selected hypotheses of simulations, rather than being applicable to the Faroese context.

Our results demonstrated also the importance of the initial design of a management system. We showed that the main issue was not effort management itself, but rather its inability to adjust to scientific recommendations and to variability and trends in catchability. This in turn is linked to the fact that the initial effort was set by Faroese authorities too high, and it could not be reduced easily thereafter. A sustainable TAE system is accommodated if the initial effort level is set sustainably. Only then, and allowing for adequate year-on-year flexibility, the TAE would appear to be a more sustainable and economically robust management strategy than TAC-based management, considering the fluctuations in the single-species HCR and the extensive discarding this could create.

Finally, the model and range of scenarios presented here were deliberately reduced to key features and few fleets. Should this approach be used in future for quantitative support to management decision-making and consensus between the fishing industry, scientists, and management bodies, the model could be developed further to address additional specific issues and scenarios.

## Acknowledgements

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## References

Anon. 2008. Fiskerikrise på Færøerne. Fiskeri Tidende, Week 34 (in Danish).

Andersen, B. S., Vermard, Y., Ulrich, C., Hutton, T., and Poos, J. J. 2010. Challenges in integrating short-term behaviour in a mixed-fishery Management Strategies Evaluation frame: a case study of the North Sea flatfish fishery. Fisheries Research, 102: 26–40.

Beddington, J. R., Agnew, D. J., and Clark, C. W. 2007. Current problems in the management of marine fisheries. Science, 316: 1713–1716.

Buisman, E., Frost, H., Hoff, A., Murillas, A., and Powell, J. P. 2009. Evaluating economic efficiency of innovative management regimes. *In* Comparative Evaluations of Innovative Fisheries Management: Global Experiences and European Prospects, pp. 143–162. Ed. by K. H. Hauge, and D. C. Wilson. Springer, Dordrecht.

Butterworth, D. S. 2007. Why a management procedure approach? Some positives and negatives. ICES Journal of Marine Science, 64: 613–617.

CEC. 2009. Green Paper. Reform of the Common Fishery Policy. Commission for the European Communities, COM (2009)163. 28 pp.

Christensen, A-S., Aranda, M., McCay, B., McLay, H. A., Rova, C., Leme da Silva, A., and Wolff, F. (2009) Understanding social robustness in selected European fisheries management systems. *In* Comparative Evaluations of Innovative Fisheries Management: Global Experiences and European Prospects, pp. 163–190. Ed. by K. H. Hauge, and D. C. Wilson. Springer, Dordrecht.

COM. 2001. Green Paper on the Review of the Common Fisheries Policy. European Commission, Fisheries Directorate.

Daan, N. (Ed.) 2007. Fisheries Management Strategies. ICES Journal of Marine Science, 64: 577–862.

Degnbol, P., and McCay, B. J. 2006. Unintended and perverse consequences of ignoring linkages in fisheries systems. ICES Journal of Marine Science, 64: 793–797.

De La Mare, W. K. 1998. Tidier fisheries management requires a new mop (management-oriented paradigm). Reviews in Fish Biology and Fisheries, 8: 349–356.

De Oliveira, J. A. A., Kell, L. T., Punt, A. E., Roel, B. A., and Butterworth, D. S. 2008. Managing without best predictions: the management strategy evaluation framework. *In* Advances in Fisheries Science. 50 Years on from Beverton and Holt, pp. 104–134. Ed. by A. Payne, T. Potter, and J. Cotter. Blackwell Publishing, Oxford, UK.

FAO. 2007. The state of world fisheries and aquaculture 2006. FAO Fisheries and Aquaculture Department, Rome.

Gillis, D. M., Rijnsdorp, A. D., and Poos, J. J. 2008. Behavioral inferences from the statistical distribution of commercial catch: patterns of targeting in the landings of the Dutch beam trawler fleet. Canadian Journal of Fisheries and Aquatic Sciences, 65: 27–37.

Hamon, K., Ulrich, C., Hoff, A., and Kell, L. T. 2007. Evaluation of management strategies for the mixed North Sea roundfish fisheries with the FLR framework. *In* MODSIM 2007 International Congress on Modelling and Simulation, pp. 74–80. Ed. by L. Oxley, and D. Kulasiri. Modelling and Simulation Society of Australia and New Zealand, December 2007. ISBN: 978-0-9758400-4-7. http:/www.mssanz.au/modsim07/Papers/51_s26/EvaluationOfmanagements26_Hamon_.pdf.

Hauge, K. H., and Wilson, D. C. 2009. Comparative Evaluations of Innovative Fisheries Management: Global Experiences and European Prospects. Springer, Dordrecht.

ICES. 2006. Report of the Northwestern Working Group, 25 April–4 May 2006. ICES Document CM 2006/ACFM: 26.

ICES. 2007a. Report of the Study Group on Management Strategies (SGMAS), 22–26 January 2007, ICES Headquarters, Copenhagen. ICES Document CM 2007/ACFM: 04. 59 pp.

ICES. 2007b. Report of the Study Group on Mixed Fisheries Management (SGMixMan), 15–19 January, ICES Headquarters, Copenhagen. ICES Document CM 2007/ACFM: 02. 56 pp.

ICES. 2008a. Report of the Northwestern Working Group, 21–29 April 2009. ICES Document CM 2008/ACOM: 03.

ICES. 2008b. Report of the Study Group on Mixed Fisheries Management (SGMixMan), 14–18 January, ICES Headquarters, Copenhagen. ICES Document CM 2007/ACOM: 23. 65 pp.

ICES. 2008c. Report of the Study Group on Management Strategies (SGMAS). ICES Document CM 2008/ACOM: 24. 74 pp.

ICES. 2009. Report of the Workshop on Mixed-Fisheries Advice (WKMIXFISH). ICES Document CM 2009/ACOM: 47. 62 pp.

IWC. 1993. Report of the Scientific Committee. Report of the International Whaling Commission, 43: 57–64.

Jákupsstovu, S. H. í, Cruz, L. R., Maguire, J-J., and Reinert, J. 2007. Effort regulation of the demersal fisheries at the Faroe Islands: a 10-year appraisal. ICES Journal of Marine Science, 64: 730–737.

Kell, L. T., Mosqueira, I., Grosjean, P., Fromentin, J-M., Garcia, D., Hillary, R., Jardim, E., *et al.* 2007. FLR: an open-source framework for the evaluation and development of management strategies. ICES Journal of Marine Science, 64: 640–646.

Kell, L. T., Pilling, G., Kirkwood, G., Pastoors, M., Mesnil, B., Korsbrekke, K., Abaunza, P., *et al.* 2005. An evaluation of the implicit management procedure for some ICES roundfish stocks. ICES Journal of Marine Science, 62: 750–759.

Kell, L. T., Pilling, G., Kirkwood, G., Pastoors, M., Mesnil, B., Korsbrekke, K., Abaunza, P., *et al.* 2006. An evaluation of multi-annual management strategies for ICES roundfish stocks. ICES Journal of Marine Science, 63: 12–24.

Kraak, S. B. M., Buisman, F. C., Dickey-Collas, M., Poos, J. J., Pastoors, M. A., Smit, J. G. P., van Oostenbrugge, J. A. E., *et al.* 2008. The effect of management choices on the sustainability and economic performance of a mixed fishery: a simulation study. ICES Journal of Marine Science, 65: 697–712.

Løkkegard, J., Andersen, J., Boje, J., Frost, H., and Hovgård, H. 2007. Report on the Faroese Fisheries regulation—the Faroe Model. Report of the Institute of Food and Resource Economics. University of Copenhagen, 193.

Marchal, P., Andersen, B., Bromley, D., Iriondo, A., Mahévas, S., Quirijns, F., Rackham, B., *et al.* 2006. Improving the definition of fishing effort for important European fleets by accounting for the skipper effect. Canadian Journal of Fisheries and Aquatic Sciences, 63: 510–533.

Nielsen, J. R., Sparre, P. J., Hovgård, H., Frost, H., and Tserpes, G. 2006. Effort and capacity-based fisheries management. *In* The Knowledge Base for Fisheries Management, pp. 163–212. Ed. by L. Motos, and D. C. Wilson. Developments in Aquaculture and Fisheries Science, 36. Elsevier, Amsterdam.

Pastoors, M. A., Poos, J. J., Kraak, S. B. M., and Machiels, M. A. M. 2007. Validating management simulation models and implications for communicating results to stakeholders. ICES Journal of Marine Science, 64: 818–824.

Penas, E. 2007. The fishery conservation policy of the European Union after 2002: towards long-term sustainability. ICES Journal of Marine Science, 64: 588–595.

Rademeyer, R. A., Plagányi, É. E., and Butterworth, D. S. 2007. Tips and tricks in designing management procedures. ICES Journal of Marine Science, 64: 618–625.

Reeves, S. A., Marchal, P., Mardle, S., Pascoe, S., Prellezo, R., Thébaud, O., and Travers, M. 2008. From fish to fisheries: the changing focus of management advice. *In* Advances in Fisheries Science. 50 Years on from Beverton and Holt, pp. 135–154. Ed. by A. Payne, T. Potter, and J. Cotter. Blackwell Publishing, Oxford, UK.

Rochet, M-J., and Rice, J. C. 2009. Simulation-based management strategy evaluation: ignorance disguised as mathematics? ICES Journal of Marine Science, 66: 754–762.

Schnute, J. T., Maunder, M. N., and Ianelli, J. N. 2007. Designing tools to evaluate fishery management strategies: can the scientific community deliver? ICES Journal of Marine Science, 64: 1077–1084.

Schwach, V., Bailly, D., Christensen, A-S., Delaney, A. E., Degnbol, P., van Densen, W. L. T., Holm, P., *et al.* 2007. Policy and knowledge in fisheries management: a policy brief. ICES Journal of Marine Science, 64: 798–803.

Shepherd, J. G. 1999. Extended survivors analysis: an improved method for the analysis of catch-at-age data and abundance indices. ICES Journal of Marine Science, 56: 584–591.

STECF. 2007a. Working Group Report on Evaluation of "Policy Statement" Harvest Rules Report, Copenhagen, 10–14 September 2007. STECF/SGMOS-07-07.

STECF. 2007b. Working Group Report on Assessment of Fishing Effort Regime, Lisbon, 21–25 June 2007, Ispra, 24–28 September 2007. STECF/SGRST 07-02 and 07-04. 282 pp.

STECF. 2008. Working Group report on Harvest Control Rules, Lowestoft, 9–13 June 2008. STECF/SGRST-08-02. 104 pp.

Steingrund, P., and Gaard, E. 2005. Relationship between phytoplankton production and cod production on the Faroe Shelf. ICES Journal of Marine Science, 62: 163–176.

Ulrich, C., Pascoe, S., Sparre, P. J., De Wilde, J. W., and Marchal, P. 2002. Influence of technical development on bio-economics in the North Sea flatfish fishery regulated by catches or by effort quotas. Canadian Journal of Fisheries and Aquatic Sciences, 59: 829–843.

Van Oostenbrugge, H. J. A. E., Powell, J. P., Smit, J. P. G., Poos, J. J., Kraak, S. B. M., and Buisman, E. F. C. 2008. Linking catchability and fisher behaviour under effort management. Aquatic Living Resources, 21: 265–273.

# Appendix

## Equations used

*Population numbers*

$$N_{a+1,y+1} = N_{a,y}\exp(Z_{a,y}), \tag{A1}$$

$$N_{+\mathrm{gp},y+1} = N_{+\mathrm{gp}-1,y}\exp(Z_{+\mathrm{gp}-1,y}) + N_{+\mathrm{gp},y}\exp(Z_{+\mathrm{gp},y}). \tag{A2}$$

*Weight-at-age*

$$\mathrm{Wt}_{a,y} = w_a e^{\varepsilon} \quad \varepsilon \sim N(0, \varphi_a). \tag{A3}$$

*Stock–recruitment relationship*

Ricker

$$R = \alpha S e^{-\beta S}. \tag{A4}$$

Recruitment residuals

$$N_{r,y} = f\,(\mathrm{SSB}_{y-r})\exp(\varepsilon_y - \sigma^2/2), \tag{A5}$$

$$\varepsilon_{y+1} = \gamma\varepsilon_y + \eta_{y+1}, \tag{A6}$$

$$\eta_y \sim N(0, \sigma^2_\eta), \tag{A7}$$

$$\sigma^2 = \ln(CV^2 + 1), \tag{A8}$$

$$\sigma^2_\eta = (1 - \gamma^2)\sigma^2. \tag{A9}$$

*Fleet dynamics*

Catchability

$$q_{\mathrm{fl},y} = \bar{q}_{\mathrm{fl}} \times \varepsilon \quad \varepsilon \sim \mathrm{LogN}(1, 0, \sigma_q). \tag{A10}$$

Selectivity

$$S = \delta * \varepsilon \quad \varepsilon \sim \mathrm{LogN}(1, 0, \sigma_{S)}. \tag{A11}$$

Effort

$$E_y = E_{\mathrm{mult}} \times E_{y-1}. \tag{A12}$$

*Mortality rates*

$$Z_{a,y} = M_{a,y} + F_{a,y}, \tag{A13}$$

$$F_{a,y} = \sum_{\mathrm{fl}} q_{\mathrm{fl},y} S_{a,\mathrm{fl},y} E_{\mathrm{fl},y}. \tag{A14}$$

Catch equation

$$C_{a,\mathrm{fl},y} = \frac{N_{a,y} F_{a,\mathrm{fl},y}(1 - \exp(-Z_{a,y}))}{Z_{a,y}}. \tag{A15}$$

## Symbols used in equations

| | |
|---|---|
| $a$ | Age index |
| $y$ | Year index |
| fl | Fleet index |
| $N_{a,y}$ | Numbers of fish by age and year |
| $Z_{a,y}$ | Total mortality by age and year |
| $+$gp | Age of the plus group |
| $\mathrm{Wt}_{a,y}$ | Mean weights-at-age by year |
| $w_a$ | Expected weight-at-age |
| $\varphi_a$ | Variance of weight-at-age |
| $\mathrm{SSB}_y$ | Spawning-stock biomass by year |
| $\alpha$, $\beta$ | Stock–recruitment model parameters |
| $r$ | Age at recruitment |
| $\varepsilon_y$ | Recruitment residual by year |
| $\sigma^2$ | Variance in recruitment residuals |
| $\gamma$ | Autocorrelation in recruitment residuals |
| $\eta_y$ | Innovation in recruitment residuals |
| $\sigma^2_\eta$ | Variance in recruitment residual innovation |
| $CV$ | Coefficient of variation in recruitment residuals |
| $q_{\mathrm{fl},y}$ | Catchability by fleet and year |
| $E_{\mathrm{fl},y}$ | Effort by fleet and year |
| $F_{\mathrm{fl},y}$ | Fishing mortality by fleet and year |
| $\sigma_q$ | Standard deviation of the catchability |
| $C_{a,\mathrm{fl},y}$ | Catch by age, fleet, and year |
| $S_{a,\mathrm{fl},y}$ | Selectivity by age, fleet, and year |
| $\delta$ | Selectivity smooth value of the last historical year |
| $\sigma_S$ | Standard deviation of the selectivity |
| $F_{a,y}$ | Fishing mortality by age and year |
| $M_{a,y}$ | Natural mortality by age and year |

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