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An evaluation of multi-annual management strategies for ICES roundfish stocks

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Abstract:

Current scientific management objectives for ICES roundfish stocks are to ensure conservation of the biological resource and do not explicitly consider economic or social objectives. For example, there are currently no objectives to maximize the sustainable yield or to reduce variability in total allowable catches (TACs). This is despite the fact that the current system can result in wide annual fluctuations in TAC, limiting the ability of the fishing industry to plan for the future. Therefore, this study evaluated management strategies that stabilized catches by setting bounds on the interannual variability in TACs. An integrated modelling framework was used, which simulated both the real and observed systems and the interactions between system components. This allowed the evaluation of candidate management strategies with respect to the intrinsic properties of the systems, as well as our ability to observe, monitor, assess, and control them. Strategies were evaluated in terms of risk (measured as the probability of spawning-stock biomass falling below a biomass threshold for the stock) and cumulative yield. In general, bounds on interannual TAC change of 10% and 20% affected the ability to achieve management targets, although the outcome of applying TAC bounds could not have been pre-judged because results were highly dependent on the specific biology of the stock, current status, and the interaction with assessment and management. For example, for North Sea haddock, management became less responsive to fluctuations resulting from large recruitment events. Simulated target fishing mortality levels were rarely achieved, regardless of the TAC bound applied, and actual fishing mortality rates oscillated around the target. In the longer term, more restrictive bounds resulted in oscillations of greater amplitude and wavelength in yield and SSB. Bounds had less effect when a stock was close to the biomass corresponding to the target F. Risk for stocks that are declining or currently at low abundance may be greater, because if bounds restrict the extent to which TACs can be reduced each year, they could lead to collapse of the stock and the loss of all future revenue. However, for a recovered stock or one at high abundance, the loss of yield as a result of bounds would be smaller than that caused by the total collapse of the fishery. At low stock size or if the stock was declining, catches should be changed more rapidly than when the stock was increasing or at a high level, especially high stock sizes acting as an insurance against uncertainty. Therefore, rebuilding strategies, and strategies aimed at maintaining the stock above prescribed limits, should be considered separately.

Keywords: bounds, cod, evaluation, haddock, hake, harvest strategies, limiting variations, management, North Sea, population modelling, saithe, simulation, TAC, whiting

Introduction

Scientific advice on the management of roundfish stocks in the Northeast Atlantic is provided by the Advisory Committee on Fisheries Management (ACFM) of the International Council for the Exploration of the Sea (ICES). The main management objective for these stocks is to ensure that spawning-stock biomass (SSB) remains above a threshold at which recruitment may be impaired, and that fishing mortality remains below a threshold level that would drive the stock below the biomass threshold. The thresholds are referred to as limit values (i.e. F_{lim} and B_{lim} for the fishing mortality and biomass limits, respectively). In recognition of uncertainty in stock estimates and in an attempt to apply the precautionary approach (Doulman, 1995), precautionary reference points (i.e. pa-values, F_{pa} and B_{pa}) have also been defined, to trigger management action before the thresholds are reached. Scientific advice is based upon recommending an Allowable Biological Catch (ABC) corresponding to a level of fishing mortality that will ensure that SSB remains above or recovers to the precautionary biomass level. Total allowable catches (TACs), limits on fishing effort, or a range of technical conservation measures are then implemented by managers (i.e. the European Commission) in order to ensure that fishing mortality and SSB remain within the precautionary range, or, for stocks subject to a recovery plan, to achieve a defined rate of recovery of SSB. There is no management objective based upon a target such as the fishing mortality that would achieve the maximum sustainable yield, although in practice, ICES has used F_{pa} as a target to advise on TACs.

The current system can result in wide annual fluctuations in TAC, limiting the ability of the fishing industry to plan for the future. This has led to calls for longer term management strategies, including those that attempt to achieve stability in catch (see Gauthiez, 2000). However, natural fluctuations in stocks attributable to recruitment variability are assumed to make the achievement of interannual stability in TACs impossible (Shepherd, 1990, 1993). Therefore, attempts to maintain TACs at a constant level could threaten the sustainability of fishery resources unless TACs are set at a very conservative level, necessary to achieve the requirements of the precautionary approach. Furthermore, even if it was possible to reduce annual fluctuations in TAC, reducing fluctuations in both catches and exploitation levels (and thus fishing effort) at the same time would be impossible, so compromises would be necessary (COM, 2000).

This study evaluated modifications to the current management strategy to constrain interannual variation in TACs. This work was performed on behalf of the European Commission. The inevitable trade-off between risk to the stock and yield was examined. For each strategy, the catch was calculated for a specific target fishing mortality and the TAC was then calculated by bounding the annual fluctuations in the TAC (i.e. TAC bounds).

A management procedure approach, as pioneered by the IWC (Hammond and Donovan, in press), was used. A management procedure is defined as a set of rules used to determine management actions in which the data, assessment methods, or rules used for decision-making, and the harvest control rules for implementing management action are pre-specified. In the IWC approach, management procedures are rigorously tested before implementation through computer simulation to ensure robustness to a wide range of uncertainty, over a wide variety of plausible operating conditions and hypotheses. This is done through the development of an operating model that represents the underlying reality against which candidate management procedures are tested with respect to explicitly stated and prioritized objectives (e.g. low risk of depletion, high and stable catches). Fournier and Warburton (1989) first recommended the evaluation of management procedures as part of an integrated management system, and the approach has now been used to evaluate management options in a diverse range of situations in South Africa (Butterworth and Bergh, 1993; Cochrane et al., 1998) and Australia (e.g. Smith et al., 1999; Sainsbury et al., 2000; Punt et al., 2002). One of the main features of the approach is that the success of an implemented or operational management procedure depends on the interactions of the monitoring, assessment, and management components within it, rather than each in isolation (Kirkwood, 1997; De Oliveira and Butterworth, 2004).

Kell et al. (2005a, b) previously showed for ICES roundfish and flatfish stocks that it is important to include the management procedure within management strategy evaluations. When the management procedure was included, the dynamic behaviour of the stocks and fisheries in terms of yield and SSB could not be predicted from the biological assumptions alone, nor from stock projections based upon a target fishing mortality (i.e. without feedback from the management procedure to the operating model).

The current study used a pre-agreed methodology, consistent with the fact that management strategies should commence at a state equivalent to the "current" situation. This was because the study was conducted on behalf of the European Commission, who wanted advice on both the short- and long-term consequences of modification to the current advisory framework. Evaluation was performed using performance statistics corresponding to the probability of the stock falling below B_{lim} and the mean yield in the short, medium, and long terms. The stocks evaluated were some of the main ICES roundfish stocks: North Sea cod, haddock, saithe, and whiting, northern and southern hake, and eastern and western Baltic cod. Stocks can be defined on either an ecological, evolutionary, or operational basis (Waples and Gaggiotti, in press). In this paper, the definition of stocks is based upon the latter, i.e. that used by ICES to perform stock assessment and to provide management advice. It was therefore assumed that there is no immigration or emigration, and that stocks are homogeneous. All data and population estimates were taken from the relevant ICES Working Group report (ICES, 2001a, b, c, d).

1. Material and methods

The simulation framework used to investigate the response of fishery systems to management was comparable to that described by Kell et al. (2005a, b). In summary, the approach models both the "real" system and the "perceived" system (observed data, assessment of current status, and reference points used to guide management), allowing testing of alternative management strategies before implementation with respect to both the intrinsic properties of natural systems and our ability to understand and monitor them. The "real" stock and fishery dynamics are represented as the operating model, from which simulated data are sampled. These data are used within a management procedure to assess the status of the stock and, depending on the perception of the stock, management controls are applied to the fishery and fed back into the real system (Figure 1). Performance statistics based upon the operating model are then used to evaluate the behaviour of the management procedure. The framework explicitly models a variety of sources of uncertainty, as categorized by Rosenberg and Restrepo (1994). In particular, process, measurement, estimation, and model error were incorporated, as described below.

The operating model

The operating model consisted of a simulated population comprising historical and future components. In the past, the system corresponded to assumptions made in, and population estimates derived by, the most recent ICES assessment at the time of simulation (i.e. stock status up to and including the year 2000 corresponded to the ICES values estimated in 2001). The starting state of the system therefore corresponded to the 2001 ICES perception. All parameters were deterministic, apart from numbers-at-age in 2000, which were lognormal random variables with expected values and CVs estimated by the Working Group. This ensured consistency of simulated biomass, reference points, and stock-recruitment relationship with current perceptions. In the future part of the simulation, the population was simulated from the initial state in 2000 for a period of 30 years. This allowed the examination of short-, medium-, and long-term impacts of management strategies. In the future component, recruitment was modelled by a stochastic stock-recruitment relationship, and selectivity, mass-, and catchability-at-age as random variables. Constant values of natural mortality and maturity-at-age were used.

For each stock, a single human consumption fishery was modelled, and in addition, in the case of North Sea haddock and whiting, discards and an industrial (i.e. non-human consumption) fishery were also modelled. The historical fishing mortality was as estimated by ICES, and for the future, selectivity-at-age was modelled as a random variable, where expected selectivity-at-age was equal to the expected value in the last year (2000), as given by a lowess smoother (span 0.75), and variability was modelled by bootstrapping the residuals to the smoothed fit.

In the operating model, historical mass-at-age corresponds to values used by ICES. For the future component, masses-at-age were modelled as random variables, with expected values equal to the smoothed values in the last year (2000). Variability was modelled by bootstrapping residuals to the smoothed fits. No trends in growth were modelled for any of the stocks, for consistency with current ICES advice.

For stocks other than North Sea haddock and whiting (for which discard data were available and catch masses were modelled explicitly), if mass-at-age in the catch differed from that in the stock, then the ratio between the two was calculated. Values were then smoothed within an age, and the expected ratios in the last year were used to model the future ratios. In the case of the stock masses-at-age, year-class effects were included by also modelling autocorrelation within a cohort. Historical catch-at-age was taken from the appropriate Working Group report.

Yield taken by the fishery corresponded to the total allowable catch (TAC), as set by the management procedure. However, to prevent unrealistic fishing mortalities being generated, fishing mortality was constrained, so that, in any year, the average fishing mortality was never more than 2.5. If fishing mortality was constrained, the TAC was not reached. Historical catches were as reported to ICES, even though these may not have been accurate, i.e. no implementation error was assumed, so that the reported catch was equal to the actual catch.

To evaluate the robustness of the management procedures to the assumed stock dynamics, three alternative Ricker stock-recruitment relationships (SRR) were included in the operating model for each stock: (i) Ricker with lognormal errors; (ii) Ricker with autocorrelation and lognormal errors; and (iii) Ricker with a "pessimistic" value of the slope-at-the-origin, set equal to the 25th percentile of the Ricker with lognormal errors. A Ricker functional form (i.e. recruitment declines at bigger population sizes) was chosen for all stocks because it is commonly assumed to be most appropriate for gadoids (Garrod and Jones, 1974; Jakobsen, 1996).

The management procedure

A management procedure is defined as a set of rules used to determine management actions in which the data, assessment methods, or rules used for decision-making, and the harvest control rules for implementing management action are pre-specified. The management procedure used in this study corresponded to the *de facto* assessment methodology and advice criteria used by ICES (Kell et al., 2005b). Stock assessment was undertaken using eXtended Survivors Analysis (XSA; Darby and Flatman, 1994; Shepherd, 1999), an implementation of virtual population analysis (VPA).

Given the XSA result in each year, a "short-term projection" was performed, using the same methodology as the relevant ICES Working Group, to estimate the ABC. Exploitation pattern and masses-at-age for the forecast were assumed to be equal to the mean of the estimates from the last 3 years. Natural mortality and maturity-at-age were the same as the values assumed in the assessment. Numbers-at-age were projected through the year of the assessment (for which total catch data were not yet available), assuming that the average fishing mortality-at-age was equal to the value in the previous year. A projection based on a fixed fishing mortality corresponding to F_{pa} was then made in the following year to estimate the ABC.

The TAC was set on an annual basis equal to the ABC, unless the ABC differed from the previous year's TAC by an amount greater than the TAC bounds, i.e.

$$\begin{array}{ll} \text{If } \mathrm{ABC}_{t+1} > \mathrm{TAC}_t(1+\alpha), & \text{then } \mathrm{TAC}_{t+1} = \mathrm{TAC}_t(1+\alpha) \\ \text{If } \mathrm{ABC}_{t+1} < \mathrm{TAC}_t(1-\alpha), & \text{then } \mathrm{TAC}_{t+1} = \mathrm{TAC}_t(1-\alpha) \\ \text{Otherwise,} & \mathrm{TAC}_{t+1} = \mathrm{ABC}_{t+1}, \end{array}$$

where α is the constraint on the annual change in TAC (i.e. 10%, 20%, 30%, or 40%).

If at the start of the simulations the current fishing mortality was greater than twice the target fishing mortality, an initial transition period was implemented, where fishing mortality was reduced first by 50%, and then to the target value. There was no transition period if the target mortality was greater than current fishing mortality.

Data for use in the management procedure, catch per unit effort (cpue) and catch- and mass-at-age, were generated via an "Observation Error Model". Cpue was used to calibrate the XSA in the assessment procedure and to generate "best estimates" of terminal populations. A single cpue fleet that covered all the age classes in the population was constructed, assuming Equations (10)–(13) in the Appendix, and a CV of 30% (an average value for the fleets studied). The results of limited simulations showed that the performance of multiple- and single-fleet assessments was broadly comparable. Natural mortality and maturity-at-age varied with age, but were held constant over years, and corresponded to values used by the 2001 Working Group.

Experimental treatments

The effect of applying interannual bounds on catches was investigated for a management strategy based upon a target of F_{pa} for comparison, although F_{pa} may not be an appropriate target. Table 1 presents the F_{pa} values for the study species, and the average F value (F_{bar}) in 2000. In the case of southern and northern hake and eastern and western Baltic cod, F_{pa} also corresponded to fishing mortality levels that are close to the maximum equilibrium or sustainable yield (MSY), although this is not the case for the North Sea stocks.

To examine the bounds on the annual fluctuations in TACs, symmetric bounds of 10%, 20%, 30%, and 40% were investigated. The results of applying bounds were compared with a control, where no bounds were applied to the interannual TAC change.

Performance statistics

For each treatment, Monte Carlo simulations were performed 100 times, and performance statistics were calculated using the results from the operating model. Evaluation was undertaken by comparing the performance of the alternative management procedures with respect to the management objectives using these statistics (i.e. risk to stock and yield), and the inevitable trade-off between them. The performance statistics were:
S1 the cumulative yield achieved over the 30-year simulation period;

S2 the probability of SSB being maintained above the corresponding biomass limit reference point B_{lim} over time; and
S3 the probability of SSB being greater than the target biomass reference point B_{pa}, over time.

2. Results

All results were obtained from the operating model.

Simulation trajectories of median yield and SSB from 2000 to 2030 are presented for the WG Ricker model as an example of the dynamic behaviour of the system in Figure 2, where the strategy with no interannual TAC bounds is compared with strategies with bounds of 40%, 30%, 20%, and 10%. Trajectories are superimposed upon the equilibrium yield–SSB curve for each species, and are expected to converge on a point (yellow disk) corresponding to the equilibrium point for F_{pa} in an anti-clockwise direction. The further the point is from the equilibrium curve, the greater will be the annual change in SSB and yield. As noted in Kell et al. (2005b), although the equilibrium SSB–yield curves are all similar in shape, there is considerable variation in the positions of the biomass reference points (the vertical lines), the relative distance between them (i.e. the precautionary level), and the expected SSB and yield for Fpa (the yellow disk). This shows that both limit and precautionary reference points, as chosen by ICES, are not consistent within or between stocks, and are often inappropriate. The rates at which trajectories converge on the expected point on the curve depends on the productivity of the stock and also importantly on lags between being able to detect changes, implement a management measure, and obtain a response from a stock, because the management procedure is explicitly included in the simulations (Kell et al., 2005a, b). Results are therefore different from those that would be expected from the type of projection used by ICES to set TACs or often used to define the precautionary and limit values.

Examination of the first column in Figure 2 (i.e. the control where no interannual TAC bounds were imposed) shows that, for many stocks, the simulated trajectories do not reach the implied equilibrium level during the simulated period. Examples are for North Sea cod, where the stock collapses despite fishing mortality being equal to F_{pa}, western Baltic cod, where the stock trajectory oscillates in the vicinity of the expected point on the curve, and for eastern Baltic cod, where the stock is underestimated by the management procedure and the trajectory converges on a point far to the right of the equilibrium point. Only in the case of North Sea haddock and saithe do the trajectories converge to the expected point.

When considering the effect of bounding the interannual variability of TACs, the factors of interest are the level of bound at which an effect is seen, and the magnitude of changes in yield and SSB. For North Sea cod and haddock, bounds of 20% and 10% result in the actual range of SSB and yield over the whole time-series being much greater than in the unbounded case. For North Sea saithe only a bound of 10% has an effect, leading to the target being over-run and eventually to collapse of the stock. Only a small effect is seen for North Sea whiting; the time taken to achieve equilibrium is increased where 20% or in particular 10% bounds are applied. For western Baltic cod, variations are large with and without bounds, while for eastern Baltic cod, bounds of 10% reduce the yield from the fishery during recovery, although the general trend and end-point are unaffected. Interannual TAC bounds do not affect the trajectory of southern and northern hake stocks, because the bounds seldom come into play; even without bounds, interannual variation in TACs is low.

The effect of interannual TAC bounds, averaged over all three stock-recruitment relationships as results were similar for all functional forms, are summarized in Figure 3. The probability of SSB being greater than B_{lim}, the implicit target SSB, and the expected cumulative yield over the 30-year period are presented.

It would be expected that a fishing mortality of F_{pa} by definition should maintain the stock at or rebuild it to a level above B_{lim}. Three stocks (North Sea cod, eastern Baltic cod, and northern hake) start the simulation below B_{lim}. North Sea cod recovers but then declines in the medium term, and in the long term the probability of being above B_{lim} declines to 20% regardless of the bound. For the other stocks below B_{lim}, the eastern Baltic cod stock recovers and bounds have little effect, whereas for northern hake, SSB recovers in the short term and bounds hasten recovery slightly because TACs are reduced. Stocks that start the simulation above B_{lim} with a high probability are North Sea haddock, saithe, whiting, western Baltic cod, and southern hake. For southern hake, bounds on TACs have a minimal effect. For North Sea whiting and western Baltic cod, there is a slight effect, but the probability of being above B_{lim} remains about 80% for all TAC bounds. North Sea haddock show no effect of TAC bounds in the short term, but in the medium term, the stricter the bound, the greater the probability of being above B_{lim}, although there is some evidence of cyclical behaviour. Similarly in the case of North Sea saithe, bounds have no effect in the short term, but do show an effect in the medium to long term; the stricter the bound, the greater the probability of falling below B_{lim} and hence the risk to the stock. The stricter the TAC bound applied for this species, the longer the stock remains above B_{lim} with a high probability. This is because the TAC is originally low and, as the stock increases with strict bounds, TACs remain lower than those implied by the target fishing mortality. In the long term, TACs increase, even with strict bounds, and then the bounds act to prevent TACs being reduced at the lower stock sizes that occur as a consequence of random variation in recruitment.

The second column of Figure 3 shows the probability of exceeding the implicit biomass target. Only North Sea haddock started the simulation near the target biomass (it has a 50% probability of being at or above the target level); the remaining stocks were below the target biomass. It would be expected that, in the long term, all stocks should have a 50% probability of being above the target biomass. However, because of estimation error in the management procedure, eastern Baltic cod and both southern and northern hake in the long term have a high probability of exceeding the expected equilibrium value. While the North Sea cod stock collapses at a fishing mortality of $F_{\mathrm{pa},}$ bounds just delay this collapse. The biggest difference between bounds is seen in the short term, and a 10% symmetric bound in nearly all cases causes biomass to increase more rapidly (and hence the target biomass to be reached sooner) than cases with less strict bounds. This is because recent yields are relatively low. For North Sea cod and haddock and western Baltic cod, bounds cause SSB to fluctuate over longer time periods.

The third column of Figure 3 shows the cumulative yield over the 30-year period. For the hake stocks and whiting the bounds have no effect. For saithe, a 10% bound has an effect in the long term, whereas for North Sea cod and haddock, a 10% bound has an effect in the short or medium term. For Baltic cod stocks, the stricter the bound the lower the yield in the short, medium, and long terms.

The biology of the stock also has significant impacts on the performance of management incorporating TAC bounds (although not presented, results were robust to the assumed stock-recruitment relationship, at least within the range of the data seen historically). For example, hake stocks responded slowly to the imposed management measures, and the TAC bound applied had little effect on either the yield or the risk to the fishery. In contrast, the biology of the other gadoid stocks (e.g. high variability in growth and recruitment and earlier recruitment to the fishery) meant that it was possible to obtain large increases or decreases in SSB in a relatively short time in response to management (Kell et al., 2005a, b). However, recent tagging studies (de Pontual et al., 2003) suggest that growth of hake may be faster than assumed by ICES. This means that the age composition on which ICES bases its stock assessment advice may be wrong and would have important implications for the scientific advice on hake currently provided by ICES. Such uncertainty about biological processes should be included in future evaluations of management strategies, to ensure their robustness.

3. Discussion

The study evaluated, at the request of managers, the trade-off between yield and risk to the stock of limiting the interannual variation in TACs set by the implicit ICES roundfish management procedure. This modification limited the change in TAC allowed between years. The study was not intended to provide tactical advice (i.e. what the actual target fishing mortality and quota should be) and therefore methodology differed from the ICES approach, which is based upon a limited consideration of uncertainty. Instead a simulation evaluation approach was used in which both the "true" and "perceived" systems were modelled, and the uncertainty attributable to measurement, estimation, model, and process error, and the interactions between the various system components were explicitly considered. The intention was not to implement a management procedure for any of these stocks, however, as this would require a much more comprehensive set of simulations trials to test robustness and to decide upon actual implementation details.

In the short term, performance is highly dependent on current stock status. Many stocks were initially below B_{pa} (and all were below B_{MSY}), which was also the biomass corresponding to the target F (i.e. F_{pa}). This meant that if initially fishing mortality was greater than F_{pa} then bounds could result in TACs not being decreased as rapidly as required to meet the target F. This increased the risk of the stock falling below B_{lim}. There is therefore a difference in the acceptable risks for stocks that are below B_{lim} and/or B_{pa}, and those above those levels. For example, if the stock is depleted, then a bound that restricted the extent to which TACs can be reduced (and hence F) each year could lead to collapse of the stock and the loss of all future revenue. However, a bound that prevents TACs (and hence F) increasing to the target level would result in more biomass being "banked" and may act as an insurance against uncertainty. Once a stock has recovered, however, the loss of yield due to TAC bounds would be smaller than that caused by the total collapse of the fishery. Therefore in the short term for depleted stocks, bounds that restrict decreases in TACs may be inappropriate and asymmetric bounds may be more appropriate. Asymmetric bounds would need to be evaluated separately before any recommendations can be made.

In the medium to long term when a stock has recovered from a depleted state, bounds of both 10% and 20% generally had an effect on the ability to achieve management targets. Examination of the historical variations in yields (Figure 4) indicates that variations in reported yields have generally been small, however, in the region of the lowest level of interannual variations explored in the simulations. The largest changes have generally been seen in recent years when TACs were being reduced. This suggests that the current system is not performing as implied by the implicit management procedure (where there are no bounds on TACs), perhaps because institutional factors are taking precedence over scientific advice.

It was also shown that a 10% bound can result in large changes in stock size and hence TACs over time, for example in the case of North Sea cod. A previous flatfish study also showed that TAC bounds of 10% could affect the ability to achieve management targets and result in low-frequency cycling in the stock (Kell et al., 2005a). Over the longer term, therefore, the fishing industry would need to adjust to considerable variations in yield, but would have a longer response time. Restricting interannual variation in TACs also means that management was less responsive to fluctuations resulting from large recruitment events, such as those of haddock in the North Sea. If good recruitments followed a period of low stock size, yield would be forgone because TACs could not be increased sufficiently fast enough to benefit from the increased abundance resulting from good recruitment.

The current study, although valuable, only looked at the consequences of making minor modifications to the current system and did not compare its performance (i.e. the ability to meet management objectives) with that of alternative management procedures. An example is the harvest control rule proposed following the implementation and subsequent review of an earlier harvest control rule for Icelandic cod (Baldursson et al., 1996; G. Stefánsson, pers. comm.):

$$\mathrm{TAC}_{t+1} = \alpha\ \mathrm{Biomass}_t(1-\beta) + \beta\ \mathrm{TAC}_t,$$

where α is the harvested proportion of the biomass and ß is a catch stabilizer. Selection of appropriate values of α and ß would "tune" the harvest control rule.

A harvest control rule of this form has an important difference from the ICES advice framework in that it has no explicit or implicit limit or threshold reference points, and there is no short-term population projection. Its properties are therefore likely to be very different from the management procedures evaluated in this study.

In practice management procedures are also subject to an "Implementation Review" after a set period (e.g. 3–5 years in the case of South African management procedures, and 5 years or less for the IWC), when the management procedure would be reviewed and modified as necessary in the light of any changes in the understanding of the resource or fishery gained in the interim.

Setting appropriate catch levels will also require an economic analysis, so that the trade-off between stabilizing yield, risk to the stock, and the loss of potential yield can be fully evaluated. Alternatively, a multi-annual fishery rather than a stock approach could be explored, based on bio-economic considerations, so that rather than trying to control the interannual variability in a single stock, the objective is to stabilize revenue from a portfolio of stocks as a hedge against biological variability (Edwards et al., 2004). This might be of particular value in mixed fisheries, where current single-species management objectives are often conflicting and encourage discarding, highgrading, and/or black landing. The current study did not evaluate implementation error that, particularly in the case of under-reported catch, may mean that the perception of the stock is biased. The actual bias is difficult to predict, because it is a function of management action; its magnitude depends upon whether quotas are increasing or decreasing. By explicitly considering the needs of the industry, compliance with management measures may be improved. This would have a positive effect on the ability to monitor and control stocks.

The results of the current study illustrate that the outcome of applying TAC bounds cannot be pre-judged; their influence is highly dependent on the specific biology of the stock, their status, and their interaction with assessment and management. For example, the cycling around equilibrium points results from the lag in the stock assessment process, a consequence of the requirement for ICES to provide catch options for TACs to be applied 2 years ahead of the data on which they are based. When combined with the assessment process, this results in a time-lag of up to 5 years within the management procedure for both detecting and attempting to react to population changes (Kell et al., 2005b). The lag interacts with the management strategy, and is not considered in current ICES procedures. This demonstrates the utility of the management procedure simulation approach, which can incorporate uncertainty attributable to stock biology, assessment, and management, and offers the ability to examine trade-offs in terms of yield and risk inherent within any management strategy (Cooke, 1999). Moreover, while interannual TAC bounds may confer stability on fisheries, their use should be closely monitored to ensure that sufficient management flexibility remains to take action under the changing conditions encountered by fish stocks.

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Table 1 Fishing mortality corresponding to F_{pa} and F_{bar} (i.e. mean fishing mortality) in 2000.

F value	North Sea cod	North Sea haddock	North Sea whiting	North Sea saithe	Southern hake	Northern hake	Eastern Baltic cod	Western Baltic cod
F_{pa}	0.65	0.64	0.61	0.25	0.27	0.20	0.60	0.16
F_{bar}	0.83	0.92	0.45	0.29	0.27	0.31	1.15	1.00

Appendix

Population dynamics

$$N_{a+1,y+1} = N_{a,y}\,\mathrm{e}^{-Z_{a,y}} \quad (1)$$

$$N_{p,y} = N_{p-1,y-1}\,\mathrm{e}^{-Z_{p-1,y-1}} + N_{p,y}\,\mathrm{e}^{-Z_{p,y}} \quad (2)$$

$$N_{r,y} = f(B_{y-r}) \quad (3)$$

Mortality rates

$$Z_{a,y} = F_{a,y} + D_{a,y} + M_{a,y} \quad (4)$$

$$F_{a,y} = \sum_{i=1}^{f} P_{i,a,y} S_{i,a,y} E_{i,y} \quad (5)$$

$$D_{a,y} = \sum_{i=1}^{f} (1 - P_{i,a,y}) S_{i,a,y} E_{i,y} \quad (5a)$$

Catch equation

$$C_{f,a,y} = N_{a,y} \frac{F_{f,a,y}}{Z_{f,a,y}} \left(1 - e^{-Z_{a,y}}\right) \tag{6}$$

Stock-recruitment relationships

Ricker

$$N_{r,y} = \alpha B_{y-r} e^{-\beta B_{y-r}} \tag{7}$$

Recruitment residuals

$$\begin{aligned} N_{r,y} &= f\left(B_{y-r}\right) e^{\varepsilon_y - \sigma^2/2} \\ \varepsilon_{y+1} &= \rho \varepsilon_y + \eta_{y+1} \\ \eta_y &\sim N\left(0, \sigma_\eta^2\right) \\ \sigma^2 &= \ln\left(\mathrm{CV}^2 + 1\right) \\ \sigma_\eta^2 &= \left(1 - \rho^2\right) \sigma^2 \end{aligned} \tag{8}$$

Derivation of effort

$$\sum^{age} C_{f,a,y} W_{f,a,y} - Y_{f,y} = 0 \tag{9}$$

Cpue models

$$U'_{f,a,y} = q_{f,a} N_{a,y} \tag{10}$$

$$U'_{f,a,y} = \frac{U_{f,a,y}}{A_{f,a,y}} \tag{11}$$

$$A_{f,a,y} = \frac{\left(e^{-\alpha_f Z_{a,y}} - e^{-\beta_f Z_{a,y}}\right)}{\left(\beta_f - \alpha_f\right) Z_{a,y}} \tag{12}$$

$$U'_{f,a,y} = q_{f,a} N_{a,y} \gamma \, e^{N\left(0,\sigma^2\right) - \sigma^2/2} \tag{13}$$

Selectivity

$$U_{f,y} = \mathrm{MVN}\left(\mu_f, \sum_f\right) \tag{14}$$

Mass-at-age

$$W_{f,y} = \mathrm{MVN}\left(\nu_f, \Omega_f\right) \tag{15}$$

Yield

$$Y_{f,y} = \sum_{i=r}^{p} C_{f,i,y} W_{f,i,y} \qquad (16)$$

SSB

$$B_y = \sum_{i=r}^{p} N_{i,y} W_{i,y} O_{i,y} \qquad (17)$$

Symbols used in equations

Parameter	Definition
$N_{a,y}$	Numbers of fish of age *a* at the start of year *y*
$M_{a,y}$	Natural mortality-at-age *a* in year *y*
$F_{a,y}$	Fishing mortality-at-age *a* in year *y*
$F_{f,a,y}$	Partial fishing mortality of fleet *f* at age *a* in year *y*
$D_{a,y}$	Discard mortality-at-age *a* in year *y*
$Z_{a,y}$	Total mortality-at-age *a* in year *y*
$S_{f,a,y}$	Selection pattern for fleet *f* at age *a* in year *y*
$P_{f,a,y}$	Proportion of catch retained for fleet *f* at age *a* in year *y*
$C_{f,a,y}$	Catch in numbers of fleet *f* at age *a* in year *y*
R	Age at first recruitment to the fishery
P	Age of the plus group
B_y	Spawning-stock biomass in year *y*
α, β	Stock-recruitment model parameters
$W_{a,y}$	Mass-at-age *a* in year *y*
$W_{f,a,y}$	Mass-at-age *a* in year *y* in catch of fleet *f*
$O_{a,y}$	Proportion mature-at-age
$Y_{f,y}$	Total catch mass of all ages of fish in year *y* by fleet *f*
$U_{a,y}$	Cpue of age *a* in year *y*
$U'_{a,y}$	Cpue of age *a* adjusted to start of year *y*
$q_{f,a}$	Catchability, relationship between cpue and numbers-at-age *a* for tuning index *f*
γ	Relationship between catchability and abundance
α_f	Start of the period of fishing in cpue series *f*

$ß_f$	End of the period of fishing in cpue series *f*
ε_y	Recruitment residual in year *y*
σ	Standard error of recruitment residuals
ρ	Autocorrelation of recruitment residuals
η_y	Recruitment innovation in year *y*
$\sigma\eta$	Standard error of recruitment residual innovations η_{year}
μ_f	Expected selectivity vector
σ_f	Covariance matrix used in selectivity modelling
ν_f	Expected mass-at-age in the stock
Ψ_f	Covariance between the ratio of stock to catch mass-at-ages
Ω_f	Covariance between mass-at-ages in the stock
ϕ	Standard error of cpue residuals
MVN	Multivariate normal

Equations and symbols used in the framework.

Figures

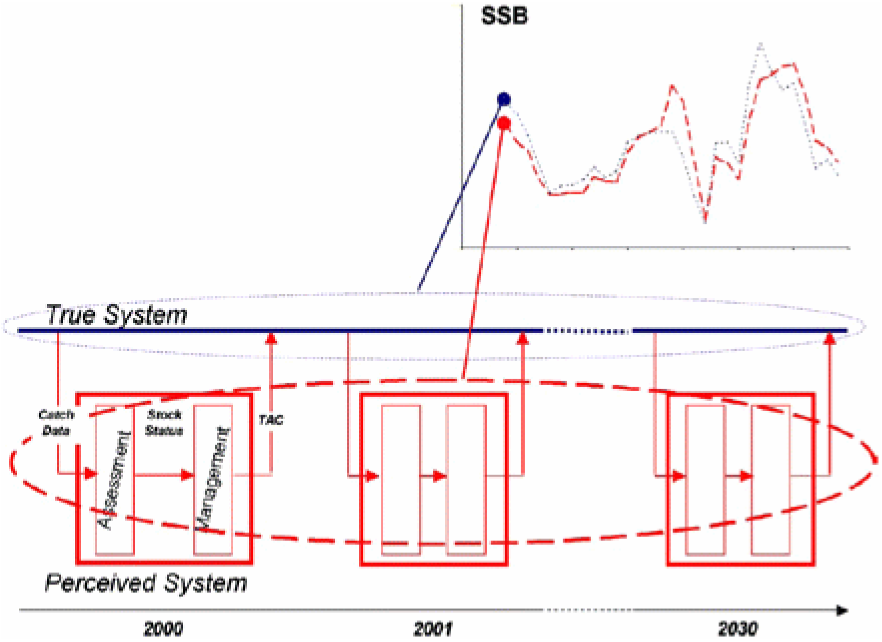

Figure 1 The simulation framework.

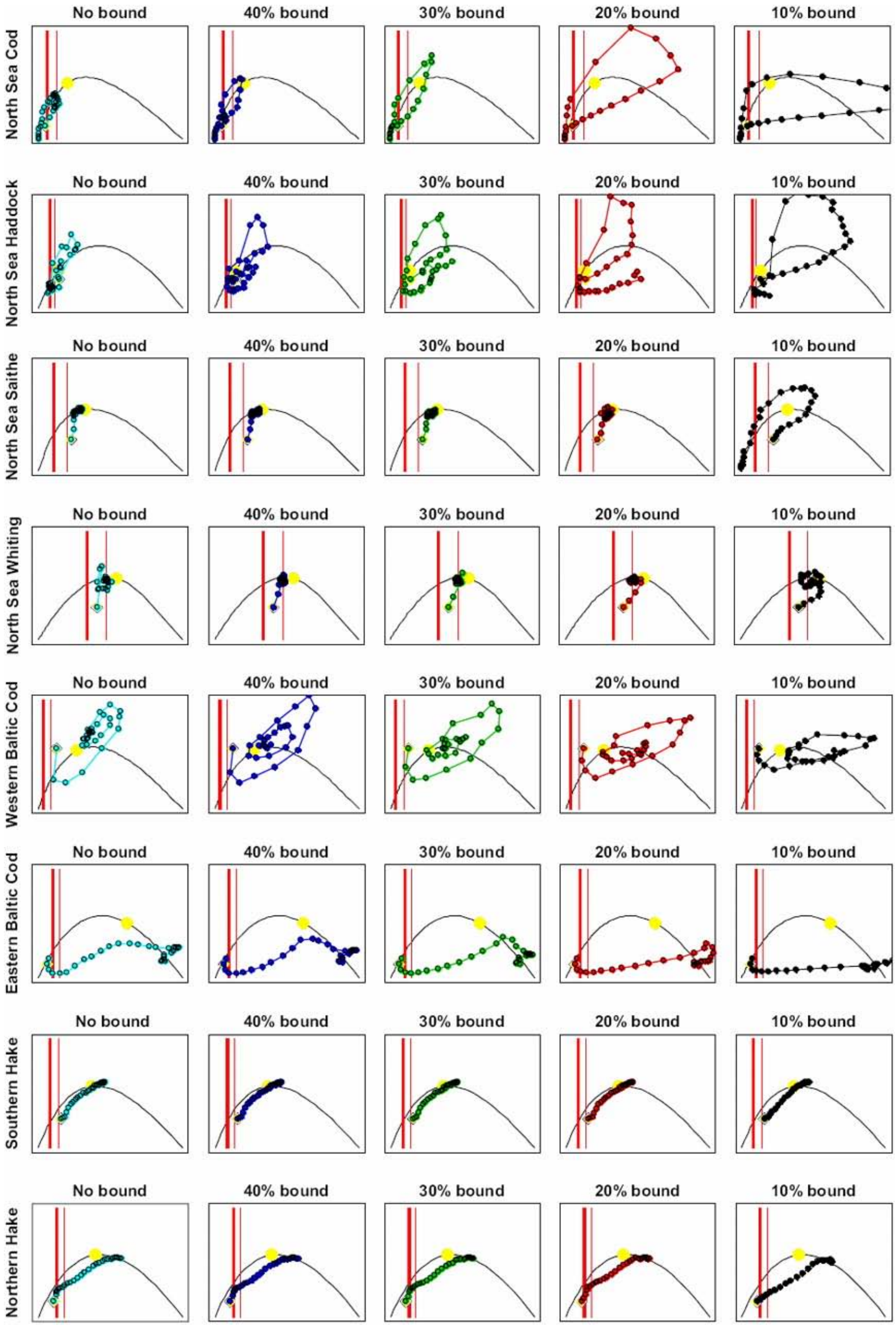

Figure 2 Curves of equilibrium yield against SSB (*x* and *y* axes correspond to SSB and yield, respectively), with simulated trajectories for 30 years for no (turquoise), 40% (blue), 30% (green), 20% (red), and 10% (black) bounds on interannual TACs. The vertical red lines represent B_{lim} (left) and B_{pa} (right), the yellow diamond shows the starting position, and the yellow circle is the implied target at F_{pa}.

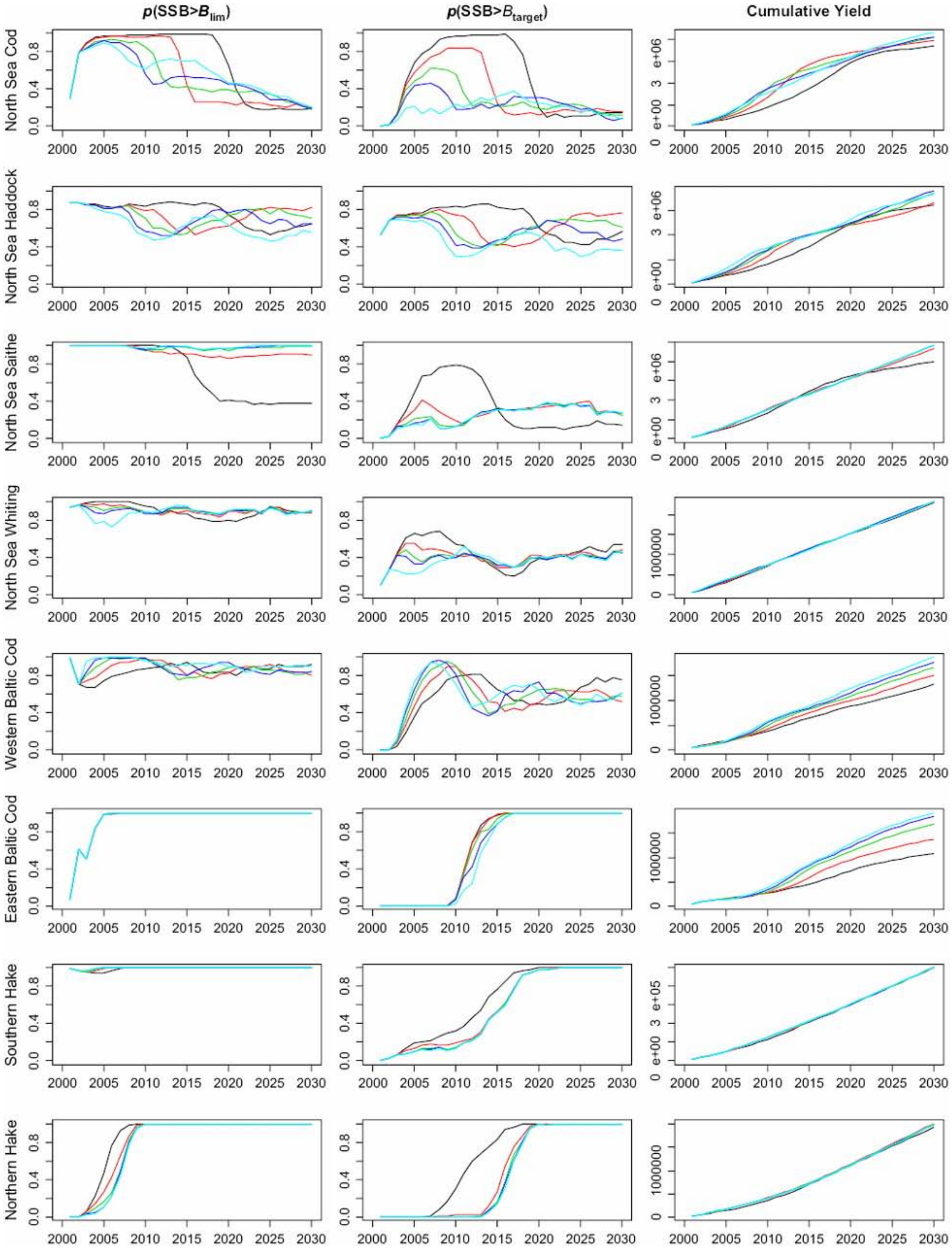

Figure 3 Comparison of the probabilities of SSB being greater than B_{lim} and B_{target} (the point on the equilibrium curve corresponding to the given fishing mortality level), and the cumulative yield for the different interannual bound levels. The different interannual TAC bound levels are shown, 10% (black), 20% (red), 30% (green), 40% (blue), and none (turquoise).

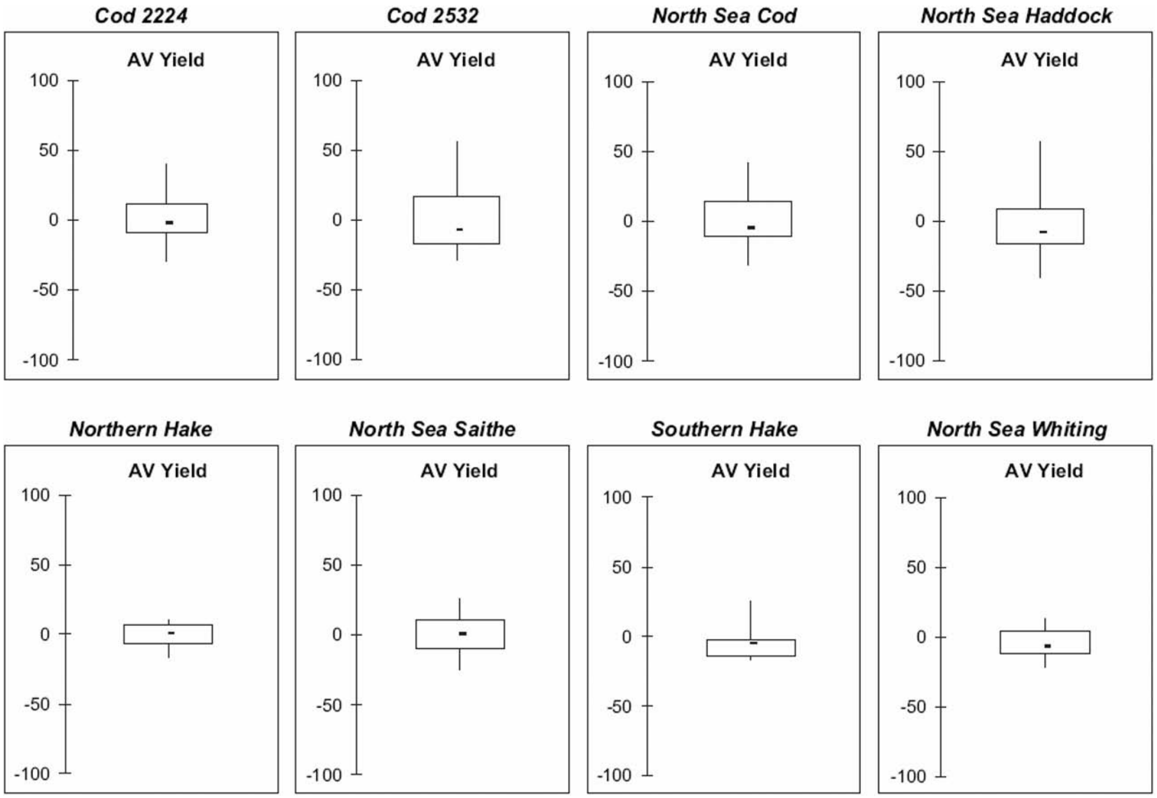

Figure 4 Comparison of historical variability in TACs, boxes representing the interquartile range and whiskers the 5–95 percentiles.