

On scientists' discomfort in fisheries advisory science: the example of simulation-based fisheries management-strategy evaluations

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Abstract

Scientists feel discomfort when they are asked to create certainty, where none exists, for use as an alibi in policy-making. Recently, the scientific literature has drawn attention to some pitfalls of simulation-based fisheries management-strategy evaluation (MSE). For example, while estimates concerning central tendencies of distributions of simulation outcomes are usually fairly robust because they are conditioned on ample data, estimates concerning the tails of distributions (such as the probability of falling below a critical biomass) are usually conditional on few data and thus often rely on assumptions that have no strong knowledge base. The clients of scientific advice, such as the European Commission, are embracing the mechanization of the evaluation of proposed Harvest Control Rules against the precautionary principle and management objectives. Where the fisheries management institutions aim for simple answers from the scientists, giving 'green/red light' to a proposed management strategy, the scientists are forced into a split position between satisfying the demands of their advisory role and living up to the standards of scientific rigour. We argue against the mechanization of scientific advice that aims to incorporate all relevant processes into one big model algorithm that, after construction, can be run without circumspection. We rather encourage that fisheries advice should be a dynamic process of expert judgement, incorporating separate parallel concurrent, lines of scientific evidence, from quantitative and qualitative modelling exercises and factual knowledge of the biology and the fishery dynamics. This process can be formalized to a certain degree and can easily accommodate stakeholder viewpoints.

Keywords EU fisheries-management advice, expert judgement, management-strategy evaluations, scientific credibility, simulations, uncertainty

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Introduction

Recently, attention has been drawn to the pitfalls of Monte Carlo simulation-based fisheries management-strategy evaluation (MSE) (Rochet and Rice 2009). In such MSE simulated fish populations are tracked in time, while the removals of fish are simulated under a rule specified by the management strategy to be evaluated. MSE simulations are commonly requested by fisheries managers, for example to provide quantitative statements of the probability that an undesirable event would occur under the management regime in question if it were implemented. Critics of this approach argue that, because of the lack of data relating to uncertainty and variability, probability estimates concerning the tails of distributions are usually based on assumptions that have no strong knowledge base. Hilborn and Liermann (1998) stated that 'we underestimate uncertainty because the assessments are contingent upon a long list of assumptions that we know are often not true' and 'more cynically, if we were to admit this type of uncertainty to decision makers and stakeholders, they would likely conclude that there was little benefit in funding stock assessment because it would tell you so little about how to manage the stock, and we would all be out of work'. We are less cynical, and believe that scientists sincerely try to inform on the likely consequences of a range of alternative management actions, but are held hostage by the dilemma that the type of simple quantitative or 'traffic light' advice managers ask for can often not be scientifically justified. In fisheries science, the kind of certainty that decision makers would like is rarely available. As a result, scientists are being drawn into making poorly founded statements while at the same time knowing better. Wilson (2009), in a sociological study of fisheries science, reports that scientists feel they are 'asked to create certainty that is not really there'.

In this study, we discuss the resulting discomfort among scientists. While we are aware that similar situations may exist in a broad range of policy-

directed science, such as the science of climate change, we choose to focus our explorations on the case study of European MSE. We review a set of MSE recently conducted in response to requests for advice and compare these with recent, peer-reviewed papers on MSE that appeared in scientific journals. Given that there is often overlap in authorship between the advisory reports and the peer-reviewed articles, we argue that, although the scientific community is well aware of the many caveats of MSE, scientists, when working through the institutions of fisheries advice, express poorly founded statements, upon which fisheries management is legitimized. We discuss the reasons for this discrepancy between what is called 'regulatory science' and 'academic science' (Anonymous 2009b) and discuss its (un)desirability. Concluding that the role of science should not be eroded, we discuss ways forward. These will include a dynamic process of expert judgement (Schnute and Richards 2001), incorporating separate parallel, concurrent, lines of scientific evidence, including insights gained from quantitative and qualitative modelling exercises, as well as scientific insights and factual knowledge of the biology and the fishery dynamics of the system. Such a process can be formalized to a certain degree and can easily accommodate stakeholder viewpoints (Funtowicz *et al.* 1999).

Current fisheries management and its science

The European Commission (EC) has been managing the commercially exploited fish stocks in the European waters under the Common Fisheries Policy (CFP), the main instruments of which are the annual total allowable catches (TAC), supplemented by various technical measures, including closed areas/seasons, effort regulations and mesh size regulations. Partly as a response to the current poor state of many stocks, and partly to avoid the annual political battles over setting the TAC, multi-annual management plans (MP) have been and are currently being adopted. These prescribe, usually through a Harvest Control Rule (HCR), how the annual TAC should be set and whether they should

be accompanied by fishing effort restrictions [the North Sea flatfish MP, Council Regulation (EC) No 676/2007; the long-term plan for depleted European cod stocks, Council Regulation (EC) No 1342/2008]. An impact assessment or an evaluation of the consequences of the implementation of the MP is often required, by the stakeholders or by the managing authorities. The EC has asked scientific bodies such as the Scientific, Technical, and Economic Committee for Fisheries (STECF) and the International Council for the Exploration of the Sea (ICES) to carry out such impact assessments and evaluations [see for example the list of MSE in ICES (2008b)].

The evaluation of an MP should in principle address the following elements: (i) the ecological impact of the plan; (ii) the socio-economic impact of the plan; (iii) the extent of stakeholder support that the implementation of the plan can build on to earn a certain degree of legitimacy (resulting in a level of compliance); and (iv) the extent to which the performance metrics in the plan are realistically measurable (Kelly *et al.* 2006). Unfortunately, recent practice has been that MSE have been carried out with a very limited scope (ICES 2008b), namely where the MP is equated to a HCR and where the MSE is limited to investigating the performance of the HCR in terms of future yields (a limited aspect of element 2 above) and the risk of the spawning-stock biomass (SSB) falling below the level at which recruitment is likely impaired (a limited aspect of element 1 above). Elements 3 and 4 are usually ignored by these small-scope MSE; an exception is the case of stakeholder participation through the Pelagic Regional Advisory Council (Pelagic RAC) in the MSE modelling of the MP for Western horse mackerel (*Trachurus trachurus*, Carangidae) where element 3 was considered (Hegland and Wilson 2009). The MSE are carried out by simulations, conditioned on available data of the relevant stock and projecting into the future under various assumptions and scenarios. The simulations usually incorporate stochasticity in one or more processes (see below), which gives them the potential to produce quantitative statements of probabilities and risk. Managers request these quantitative MSE aiming to evaluate HCR objectively.

While the requests for evaluation often explicitly asked for a statement on whether the MP is 'precautionary', the community of fisheries scientists and managers in Europe became aware that an operational definition of 'precautionary' was lacking

(ICES 2009a). This situation has given rise to a wish for standardization of the practice of MSE and for an operational definition of 'precautionary'. In an attempt to formulate such standardization and definition, a Workshop of ICES and STECF (ICES 2009b) agreed that 'The HCR (MP) is considered to be precautionary if the probability of $SSB < B_{lim}$ is less than 5%', where B_{lim} is the estimated SSB level below which recruitment is impaired or below which stock dynamics are unknown (ICES 2003). The Workshop explicitly defined the probability in terms of the number of Monte Carlo simulation runs (see table 5.1 in ICES 2009b). Clearly, this is an operational simplification of the concept of precaution. We refer to other publications on this issue (Hauge *et al.* 2007). For the current discussion, it suffices to note that managers implicitly or explicitly request estimates of probabilities at the tails of distributions. However, while estimates concerning central tendencies of distributions of simulation outcomes (such as expected future mean yield) are usually fairly robust because they are conditioned on ample data, the estimates of probabilities concerning the tails of distributions are usually based on assumptions that have no strong knowledge base (Rochet and Rice 2009). In many cases of stock assessment, the degree of uncertainty coming from the various sources is not accurately determined (Hilborn and Liermann 1998), although tools are becoming available (Patterson *et al.* 2001). If MSE simulation models quantitatively specify uncertainty, partly based on unsupported (tacit) assumptions, the resulting probability statements, such as the probability of falling below a particular limit reference point when following a particular HCR, may be inaccurate while giving managers as well as the public an illusion of scientific rigour (Hilborn and Liermann 1998; Rochet and Rice 2009). To make matters worse, the uncertainty is quite likely to be seriously underestimated (Hilborn and Liermann 1998).

Sources of variability and uncertainty

Population and fishery dynamics as well as stock assessment and management are characterized by variability and uncertainty, not all of which is or can be incorporated into the simulations of the system. We distinguish between (i) stochastic variability, (ii) temporal factors and (iii) unknown or unaccounted for processes. Examples of (i) would be the variability in natural mortality or gear

selectivity; owing to stochastic processes (which are really deterministic processes at a smaller scale) natural mortality and gear selectivity will never be constant. Also data collection is subject to stochastic variability, known as sampling error. Temporal factors (ii) could, for example, be changes in underlying processes that influence natural mortality (such as mean temperature, predator or prey abundances, prevalence of diseases), gear changes, changes in effort allocation, or changes in assessment settings. Temporal changes could be abrupt or gradual. Finally, processes (iii) that are not accounted for could be, for example, depensation or migration; here the true population dynamics may be misrepresented in the simulation if the process is ignored. An example of an unaccounted for process affecting fishing dynamics is high-grading, and an example of a process that is unaccounted for affecting data collection is biased age reading. Bias and error may systematically vary across the range of dynamics being investigated. We also note that 'full-feedback' MSE models incorporate simulation of the stock assessment process, which may lead to emergent discrepancies between the modelled stock and the simulated perception of it. 'Non-feedback' MSE models assume prefixed levels of error and bias of the assessment and the implementation of the HCR, thereby ignoring any possible systematic variation of this error or bias. We do not argue that all sources of uncertainty must be incorporated into the MSE simulation models. To the contrary, we argue that no model will ever be able to fully capture the complexity of reality nor should it. Nevertheless, while this neglect will usually not greatly affect the robustness of simulation results that represent central tendencies of distributions, it casts doubt on the credibility of statements of probability concerning the tails of distributions (Rochet and Rice 2009).

Because of the many sources of uncertainty and variability that are not taken into account, the total amount of uncertainty assumed in MSE, will generally be an underestimation of the actual uncertainty. For example, in a study on multi-model inference on fish growth by model-averaging based on Akaike weights (Katsanevakis 2006), it was shown that ignoring model-selection uncertainty could not only cause biased point estimation but also lead to false evaluation of precision. In this particular study, model-selection uncertainty caused a fourfold increase of the standard error of the asymptotic length as estimated by the best

model. It becomes very obvious then, that quantitative estimates of risk derived from MSE will also typically be too low.

The choice of sources of uncertainty to be incorporated in MSE simulations often seems quite arbitrary and does not necessarily reflect a lack of scientific knowledge of the relevant process. Let us look in some more detail at some sources of variability and uncertainty. Most stock assessments and MSE consider SSB as the index of stock reproductive potential (SRP) instead of direct measures of the egg production of the stock. This assumes that egg production per unit of biomass is constant. However, if fecundity varies in relation to parental age and quality, resource availability, environmental and other factors, the relationship between SSB and egg production would not be constant (Lambert 2008; Marshall 2009). The relationship would also change if, for example, through differential mortality on males and females, the population sex ratio changes. Demography can have a significant effect on SRP as well; in many species older fish spawn earlier within a season and may produce more egg batches over a longer period (Wright and Trippel 2009). If, by whatever mechanism, younger females have lower fecundity or produce less viable larvae, then any decrease in SSB accompanied by truncation of the age-structure will result in a more than proportional decrease in SRP, and this is usually not taken into account in MSE. In a study on perceptions of population productivity of eight fish populations across the North Atlantic (Morgan *et al.* 2009), it was found that the inclusion of biological complexity such as variation in maturation, sex ratio, and fecundity, greatly affected the perceived status of the stock relative to the given reference points. Recently, efforts are being made to integrate knowledge of reproduction into fisheries-management advice [Morgan 2008; Marshall 2009, see also the Fish Reproduction and Fisheries (FRESH) network; <http://www.fresh-cost.org>].

Another example of a process that is usually ignored in stock assessments and MSE comes from a recent study on spatial variation in fishing mortality (Ralston and O'Farrell 2008). In this model study it was shown that spatial heterogeneity in fishing mortality may affect yield differentially, depending on the life stage in which density-dependent compensation operates. Simulations demonstrated that if fishing mortality is excessive, spatial variation in fishing intensity often improves yield when com-

compensation occurs after dispersal, where juvenile mortality of newly settled fish is a function of the local density of adults (as opposed to before dispersal, where egg production is a function of the local density of spawners). The yield premium generated here was explained as being caused by a low level of compensatory mortality in heavily fished areas coupled with dispersal of propagules into these areas from lightly fished adjacent regions.

Furthermore, MSE often ignore the possibility that the fishers' behaviour may alter in response to new management measures. A well-known case is the question of whether the closing of areas to fishing leads to effort reallocation and whether this reallocation is random or according to some pattern caused by fishers optimizing their profits which will in turn affect total catchability (Bastardie *et al.* 2010).

Management-strategy evaluations (MSE) in advisory reports vs. MSE in peer-reviewed scientific journals

Table 1 lists a set of recent MSE studies that were carried out in response to requests for advice from fisheries managers or stakeholders (taken from ICES 2008b). These studies are published as reports (of ICES, STECF or research institutes) and not usually peer-reviewed. All but one of the requests asked for a risk statement explicitly or implicitly (if the request was to evaluate whether the MP is precautionary this was interpreted to mean whether the risk of SSB falling below B_{lim} is $<5\%$). Only the North Western Waters RAC (NWWRAC) requesting an evaluation of the Northern hake (*Merluccius merluccius*, Merlucciidae) long-term MP did not ask for a statement on risk (D. García, personal communication); accordingly, the report on that MSE (García *et al.* 2008b) does not include such a statement. The other reports provide quantitative statements on risk (Table 1), which we believe are poorly founded. Evidently, managers ask for scientists to provide statements on risk. In an attempt to address the requirement that an MP be precautionary, they use the simplified operational definition (mentioned above; ICES 2009b) referring to the proportion of simulation runs where SSB falls below a reference point whose value may itself be poorly defined and poorly estimated.

Table 2 lists examples of recent MSE studies that appeared in the peer-reviewed literature. Most of these do not provide statements of absolute risk but

present results in terms of relative levels of risk, of one management strategy relative to another or under differing assumptions of underlying dynamics. The studies in Table 2 are examples of the kind of research questions that can be approached by the MSE framework, without committing to unjustified quantitative risk statements. In the first example (Kell *et al.* 2005), robustness of both short-term stock-biomass recovery and longer-term sustainable management strategies to different plausible climatic change scenarios were evaluated for North Sea cod (*Gadus morhua*, Gadidae), where climate was assumed to impact growth and recruitment. It was found that the implications of climate change for biological reference points depend upon the mechanism through which temperature acts on recruitment. It is uncertain whether this is through an effect of temperature on juvenile survival or on the carrying capacity. The study also indicated that reference points based on fishing mortality appear more robust to uncertainty than those based on biomass. While the study does not provide managers with unsound quantitative risk statements, it informs managers in two ways, namely with respect to the best choice of reference points and suggesting that it may be worthwhile to invest in studies on temperature effects on recruitment.

In the second example, on the Irish Sea cod recovery plan (Kelly *et al.* 2006), it was shown that the metrics defined in the MP were not measurable given the assessment uncertainty, and that elements of the implementation of the plan lacked foresight and the application of best practice. Simulations were used to compare the effect of simple management strategies in a relative sense only.

Another example is the study on the mixed roundfish fisheries in the North Sea (Hamon *et al.* 2007). This study makes explicit to the managers that single-stock management objectives for cod and haddock (*Melanogrammus aeglefinus*, Gadidae) cannot be achieved simultaneously using stock-based annual TAC because of conflicting incentives to the fleets. The two opposite hypotheses of fishermen behaviour (stop fishing when cod quota is exhausted or fish until haddock quota is exhausted while discarding overquota catches of cod), representing the two extremes bounding the range in which the actual behaviour probably lies, lead to dramatic differences in the probability of reaching cod recovery in the short-term. Also here, the merit of this study is not to provide absolute probabilities

Table 1 Management-strategy evaluations (MSE) carried out on request from fisheries managers or stakeholders (ICES 2008b); whether they make explicit statements about probabilities in the tails of distributions; which uncertainty was explicitly taken into account.

MSE	Probability statements on tails of distributions	Uncertainty taken into account
<p>North Sea flatfish management plan [Council Regulation (EC) No 676/2007] evaluation</p> <ul style="list-style-type: none"> Commissioned by Dutch ministry (Machiels <i>et al.</i> 2008) Reviewed by ICES (unpublished document: review of flatfish plan_combined.doc) An earlier but similar MSE commissioned by the Dutch ministry (Machiels <i>et al.</i> 2007) of an earlier draft of the management plan (5403/06 PECHE 14) was used by the STECF impact assessment (STECF 2006, 2007) 	<p>Explicit statements on probability of $SSB < B_{lim}$</p>	<ul style="list-style-type: none"> Two scenarios of S–R relationship Noise in R based on observed residuals Two scenarios of fisher behaviour, implicitly influencing catchability through spatio-temporal effort allocation Landing, discards, and survey catches with fixed error coefficient of variation (CV) of 10% Assessment error and bias emergent through full feedback Three scenarios for intermediate year assumption in short term forecast
<p>NEA mackerel long-term management scientific evaluations</p> <ul style="list-style-type: none"> Evaluated by ICES (ICES 2008c) on request from EC 	<p>Explicit statements on probability (called 'risk') of $SSB < B_{lim}$; explicit statements on probability (called 'risk') of depletion (i.e. TAC cannot be taken)</p>	<ul style="list-style-type: none"> Probabilistic hybrid S-R model based on Bayesian analysis of observations; model output comparable with observations Uncertainty in initial stock size: log-normally distributed age-specific CV with log-normally distributed year error scaled to CV of 29% on SSB Stochastic selection-at-age Assessment error handled in 2 ways: <ul style="list-style-type: none"> Emergent through full feedback Log-normally distributed random multipliers for age and year applied to stock numbers-at-age (latter included bias); assessment error with CV of 29% on SSB; autocorrelation Effects of different levels of implementation error and bias tested Model validation through use of different frameworks and comparison of outcomes, e.g. 'risk' levels
<p>Evaluations of management plan of North Sea herring and development of management plan for western Baltic spring spawning herring</p> <ul style="list-style-type: none"> Evaluated by ICES (2009c) 	<p>Explicit statements on probability of $SSB < B_{lim}$</p>	<ul style="list-style-type: none"> Stochastic R; two levels of mean R tested; two levels of variability tested Stochastic weight- and maturity-at-age, and initial stock numbers Stochastic assessment error; two levels of error tested Stochastic implementation error; two levels of bias tested Two levels of TAC in 2008 tested
<p>Cod recovery management plans, i.e. the 2008 Commission proposal and the 2008 Norwegian lawful Authorities proposal</p> <ul style="list-style-type: none"> Evaluated by ICES (2008e, 2009a) on request from EC 	<p>Explicit statements on probabilities of $SSB > B_{lim}$ and of $SSB > B_{pa}$</p>	<ul style="list-style-type: none"> Two sources of bias were tested: unreported catch and changes in natural mortality (m) Three assessment settings were tested: no correction for bias, correction for bias because of unreported catch, and correction for bias because of change in m Two levels of stochastic R tested ('standard' and 'low') Assessment error and bias emergent through full feedback

Table 1 (Continued).

MSE	Probability statements on tails of distributions	Uncertainty taken into account
Blue whiting management plan proposal • Evaluated by ICES (ICES 2008d) requested by the coastal states (EU, Norway, Iceland and Faroe Islands)	Explicit statements on probability of $SSB < B_{lim}$	<ul style="list-style-type: none"> • Observation error of stock numbers-at-age: year factor from log-normal distribution with fixed CV; random noise from log-normal distribution with range of CVs tested • Stochastic R, log-normal error with fixed CV • Simulation results checked with other software that included¹ <ul style="list-style-type: none"> • Variable selection-at-age • Assessment error with correlated parameters
Northern hake long-term management • Evaluated by STECF requested by EC and NWWRAC (García <i>et al.</i> 2008a,b)	Explicit statements on probability of $SSB < B_{pa}$ only in the first of the two reports (García <i>et al.</i> 2008a). In the second (García <i>et al.</i> 2008b), there is no absolute quantitative statement on risk; focus is on central tendencies, widths of distributions are mentioned only comparatively	<ul style="list-style-type: none"> • Stochastic R, log-normal error with fixed CV • Stochastic initial population numbers, generated by repeating XSA on data with log-normal error added to the abundance indices
Management plan for Western horse mackerel (prepared by the Pelagic RAC in July 2007) • Evaluated by ICES requested by EC (Clarke <i>et al.</i> 2007; ICES 2007a,b; Roel and De Oliveira 2007)	Explicit statements on probability of $SSB < B_{lim}$	<ul style="list-style-type: none"> • Stochastic R; using a combination of the Ricker function and a process that allows very large recruitment with a frequency of roughly one in 20 year • Stochastic initial population numbers • Implementation error, bias and random variation • Egg abundance modelled with 'process' error and 'observation' error
Management plan for saithe in the North Sea and West of Scotland • Evaluated by ICES requested by EC and Norway (ICES 2008a)	Explicit statements on probability of $SSB < B_{lim}$ and on probability of $F > F_{lim}$	<ul style="list-style-type: none"> • Stochastic R (hockey-stick) • Stochastic initial population numbers with age-specific log-normal error • Fixed weights-at-age, maturity, natural mortality and no measurement or implementation error (authors note that this may lead to underestimation of risk)

SSB, spawning-stock biomass; NWWRAC, North Western Waters Regional Advisory Council.

¹The authors note that the risk of falling below B_{lim} indeed increased when this extra variability was included.

of cod recovery, but to advise the managers on the fact that in such a situation single-species management is illogical and untenable.

In the fourth example, Kraak *et al.* (2008) investigated the comparative biological and economic performance of different management strategies. Using a full-feedback model, it was found that

the high 'shrinkage' setting (Darby and Flatman 1994; Shepherd 1999) that had commonly been used for the assessment of, among others, North Sea plaice (*Pleuronectes platessa*, Pleuronectidae) and sole (*Solea solea*, Soleidae), led to a cyclical alternation of underestimation and overestimation of the stock parameters. The high shrinkage of recent

Table 2 Peer-reviewed management-strategy evaluation (MSE) studies and how they use probability statements.

MSE	Probability statements on tails of distributions	Aim of the study
Implications of climate change for the management of North Sea cod (<i>Gadus morhua</i>) (Kell <i>et al.</i> 2005)	Explicit statements on probability of $SSB > B_{pa}$, but only for comparative purposes	Testing of robustness of both short-term stock-biomass recovery and longer-term sustainable management strategies to different plausible climatic change scenarios
The Irish Sea cod recovery plan: some lessons learned (Kelly <i>et al.</i> 2006)	Explicit statements on probability of $SSB < B_{lim}$, but only for comparative purposes	<ul style="list-style-type: none"> • Testing of sensitivity of model outcomes to different levels of assessment error and bias and of implementation error and bias • Comparison of different management strategies (with or without 15% bound year-on-year changes in TAC) • Comparison of different management targets of fishing mortality
The potential for new <i>Nephrops</i> trawl designs to positively effect North Sea stocks of cod, haddock and whiting (Catchpole <i>et al.</i> 2007)	No	Comparison of potential impact on whitefish stocks of implementation of different fishing gear designs and also of scenarios of cessation of discarding
Evaluation of management strategies for the mixed North Sea roundfish fisheries with the FLR framework (Hamon <i>et al.</i> 2007)	Explicit statements on probability of $SSB < B_{lim}$ (which happen not to concern the tails of distributions), but only for comparative purposes	Testing of sensitivity of model outcomes to different hypotheses of fishermen behaviour (stop fishing when cod quota is exhausted or fish until haddock quota is exhausted while discarding overquota catches of cod)
Evaluation of the robustness of maximum sustainable yield based management strategies to variations in carrying capacity or migration pattern of Atlantic bluefin tuna (<i>Thunnus thynnus</i>) (Kell and Fromentin 2007)	No	<ul style="list-style-type: none"> • Comparison of different management strategies • Testing of robustness of management strategies to different hypotheses of stock dynamics (fluctuations come from changes in the carrying capacity or migration)
The effect of management choices on the sustainability and economic performance of a mixed fishery: a simulation study (Kraak <i>et al.</i> 2008)	Explicit statements on probability of $SSB < B_{lim}$ but only for comparative purposes and with health warning not to take the absolute values serious	<ul style="list-style-type: none"> • Comparison of different management strategies (TAC vs. effort restrictions) • Comparison of different assessment settings (shrinkage)
Linking catchability and fisher behaviour under effort management (van Oostenbrugge <i>et al.</i> 2008)	No	Testing of sensitivity of model outcomes to the assumption of a (non-)linear relationship between fishing effort and fishing mortality
Insurance mechanisms to mediate economic risks in marine fisheries (Mumford <i>et al.</i> 2009)	No	Exploration of how insurance funds could protect revenue and encourage increased sustainability of fisheries and improve compliance with and enforcement of fisheries regulation

TAC, total allowable catches; SS, spawning-stock biomass.

fishing mortality to values in the more distant past caused the simulated perception of stock development to lag several years behind the simulated 'true' development of the stock. During periods with a trend in fishing pressure, shrinkage will therefore

lead to inappropriate management measures resulting in undershooting and overshooting of the targets aimed for. Although the effect of shrinkage was known in the assessment community, it had not yet been shown so explicitly.

Finally, the example of the study by Mumford *et al.* (2009) illustrates that an entirely novel proposal for a new instrument for fisheries management, namely insurance, can be explored in terms of the implications of different sources of uncertainty for insurance costs and the effectiveness of particular insurance schemes to mitigate risks. The insights gained from such studies can be the starting point for further discussions.

The role of science: regulatory and academic science

The authors of the MSE in Table 1 partly overlap with those of the studies in Table 2, and in any case these authors are often involved in close collaborations in ICES Working Groups and international EU-funded projects. In the science realm where definitive statements of risk cannot be made without supporting evidence, there are very few statements where 'risk' is enumerated, whereas in the operational world of fisheries, where managers would like a definitive basis for decisions, there appears to be a pressure on scientists to enumerate figures to support what is in essence 'expert opinion' (and thus not always quantifiable). Various reasons can be given for the occurrence of this discrepancy between scientists' statements in advisory reports and peer-reviewed papers.

1. The clients request simplistic advice, such as absolute probability statements;
2. Time constraints exist for advisory reports (Hegland and Wilson 2009); advice often has to be finalized on time scales of weeks;
3. Budgetary constraints exist for advisory reports (Hegland and Wilson 2009); the scientists' time has to be paid for;
4. Advisory reports are often written by large groups aiming for consensus;
5. Political aims such as conservation issues on the one hand and national fisheries interest on the other are often mixed in with impartial scientific advice.

The Center for Science and Technology Policy Research, University of Colorado (Anonymous 2009b) contrasts regulatory science and academic science. The first is expected to generate information needed to meet regulatory requirements and to provide reliable information for decision makers, while research questions are framed by legislators and regulators and often have social and economic

implications, and the ultimate goal is conflict resolution via public debate over competing interests and values. Academic science is original research framed by scientists and driven by rational analysis and expert judgment. Its goal is to expand understanding and knowledge of the natural world through an ongoing process of questioning, hypothesizing, validation, and refutation. The role of uncertainty is also different: predictive certainty of regulatory science is required by the political process and by legal requirements, while uncertainty is expected and 'embraced' by academic science.

At another level the issues are being experienced because there is a failure to adapt to the paradigm that fisheries management is as much about resolving the trade-offs in the human interaction with the natural world, as it is about setting TAC. In the past, scientists 'predicted' stock levels, and once a harvest rate was agreed, it was just a matter of applying the latter to the former to arrive at an advice on how to harvest the stock (i.e. an advised TAC). Currently it is acknowledged that there is uncertainty in the system, and we have to find ways to deal with this uncertainty in the face of the management objectives. The MP is supposed to encompass a range of objectives including biological, societal and economic, to set out a harvest strategy which forms the agreed compromise between these. In reality this concept of an MP is being reduced to an HCR used as a sophisticated way to set the TAC. In the advisory documentation, the fisheries scientists see themselves as impartial observers to the system, and their 'part' only consists in computational aspects of fish stocks. The managers react the same way to the information coming from the HCR evaluations as they have in the past (i.e. that this is a prediction specifying how to harvest the stock).

The question as to whether scientists 'buy in' to their own advice is not often asked (but see Wilson 2009). However, the recurrent appearance of papers critical of the misuse of quantitative modelling (Schnute and Richards 2001; Rochet and Rice 2009), and the difference documented here between the use of MSE in advisory reports and that in scientific papers, as well as the existence of various scientist-driven initiatives such as the FRESH network (<http://www.fresh-cost.org>, aiming to improve stock assessment through implementation of knowledge of SRP), all attest to the discomfort felt among some scientists about the current format in which the advice is to be given. Wilson (2009) found that

scientists working in National Fisheries Institutes experience a lower level of job satisfaction than scientists working elsewhere (Wilson 2009; table 5.2, p. 138); dissatisfaction was related to the experience of being asked to 'create certainty that is not really there' (Wilson 2009, table 5.10, p. 158).

At this point we want to make clear that we do not argue against the use of HCR; but we consider that they are only a component of an MP. We believe (for all the reasons described previously) that quantitative statements on risk based on a simulation of the application of an HCR do not provide an adequate representation of the full risk profile of an MP. The probability estimates from MSE simulations, are being treated as if they were synonymous with 'real risk', and are being presented to managers as an evaluation of the HCR, as if it were an evaluation of the whole MP (including stakeholder buy-in and societal aspects). Although the clients of the advice, such as the EC, seem to embrace the mechanization of the evaluation of MP against the precautionary principle and management objectives (ICES 2008b, 2009b), we contend that the scientists' discontentment should be transformed into giving more appropriate advice that is credible to the scientists themselves. We acknowledge the different goals and resources of regulatory vs. academic science, but we believe that nobody benefits where the quality of science is compromised. Wilson (2009) notes that the importance of science, in the applied context, derives from its transparency. In addition, we stress that science loses its *raison d'être* if it gives up the one thing that distinguishes it from informed opinion: its value-free and open-minded search for improved and falsifiable understanding of reality. Scientists should not (be asked to) provide scientifically indefensible statements. The challenge we face is: what are ways forward?

Ways forward

We have already pointed to the valid information content relevant to fisheries management gained from the type of comparative MSE studies published in the peer-reviewed literature. There is certainly a role for such studies in the advisory context. These simulation studies view the outcomes as relative, either comparing different management strategies or testing the sensitivity to alternative assumptions about the underlying world. MSE studies can be valuable if interpreted in an appropriate context:

namely for the investigation of small-scope specific issues to which relative or qualitative answers are expected. MSE as simple models can be used as tools to give insight into processes. They can help us achieve a better understanding of the complexities of fisheries systems and their responses to management measures. The continuing challenge remains to design and carry out studies aimed at obtaining ever more insight.

Furthermore, we encourage scientists to take up the suggestions given by Rochet and Rice (2009), such as retrospective examination of management-strategy performance and qualitative modelling. The latter provides an attractive way of modelling vague knowledge to provide decision support. Rather than models that need parameterization of quantitative formulations of processes, this approach allows a formal analysis of the qualitative consequences, such as the direction of change, under qualitative assumptions. For example, Dambacher *et al.* (2009) have examined the utility of qualitative modelling for predicting change in populations belonging to complex exploited communities under various pressures, and investigated the main feedback controls in commercial fisheries. An approach that is similar in principle is the idea of using an empirical indicator in a qualitative or relative sense to monitor stock status and subsequently make management decisions. The idea is that basic statistical process control techniques can be used to objectively determine the direction and rate of change in key stock indicators (for example mean catch weight of fish) that act as proxies to the overall stock status (Froese 2004; Scandol 2005; Kelly and Codling 2006). This can either be completed at a single-species level (Froese 2004; Scandol 2005), or community-level indicators can be used to manage stocks at a multi-species or ecosystem level (Trenkel and Rochet 2003).

Our suggestion for the integration of quantitative and qualitative modelling with other knowledge follows a proposal by Schnute and Richards (2001). These authors view models more as tools for thought, than for rigorous analyses of system dynamics. Similar to our own suggestions above, they propose to consider a suite of models to pose a suite of single-issue what-if questions such as 'what would happen if the environment played a major role in regulating abundance?' or 'what if fish movement and spatial structure govern the interaction between fish and fishermen?' or 'could

species interactions produce effects not noticed when investigating species individually?'. They further argue that not only insights gained from modelling (whether quantitative or qualitative) should be considered, but also scientific knowledge of facts and scientific expert judgment. Released from the obligation to be formalized into a unified modelling structure, and released from the obligation to be quantitative, such knowledge can enter the advisory debate on equal footing to quantitative model outcomes. This may well apply to the mission of the FRESH network of bringing knowledge on SRP into fisheries-management advice. Rather than trying to force such knowledge into a modelling framework, it should effectively sit side by side to a model and be treated as another piece of evidence. This will lead to a dynamic process where separate, concurrent lines of scientific evidence are considered. Schnute and Richards (2001) propose that by exploring lines of evidence concurrently, the focus remains on the basic thought processes, rather than on the details, and the final product can be more readily understood, interpreted, and explained to others. We believe therefore that this process can easily include stakeholder viewpoints. Funtowicz *et al.* (1999) state that in issue-driven science relating to environmental debates, typically facts are uncertain, values in dispute, stakes high, and decisions urgent. And in these cases, they argue (Funtowicz *et al.* 1999), 'Post-Normal Science' is needed; this is a process taking place in extended peer communities where a range of viewpoints and perspectives is to be considered.

The pitfalls of this process are that the non-formalized contribution of several strands of evidence may lead to chaos and indecisiveness. The questions then arise: What to do with opposing strands of evidence and opposing views? How will the final decision be determined, if there is no fixed procedure? Will the process take too long to reach consensus and will it be too time-intensive? However, the process is already starting to take place: the CFP reform in 2002 had already started a process of strengthened participation in fisheries governance; and the recently released Green Paper on the CFP (Anonymous 2009a) further explores avenues to increase participatory decision-making. According to this Green Paper, the general framework for fisheries policy would be set on the basis of a Commission proposal, but detailed implementing decisions, for example, on types of gear or on which

area should be closed to fishing and when, could then be taken at a regional level where scientists would need to interact with stakeholders and governments. It also foresees that the industry is asked to develop its own fisheries plans, for which they would need scientific advice – especially if this is to form part of a results-based management system. We have already mentioned the participatory modelling for the Western horse mackerel MP, which was partly successful (Hegland and Wilson 2009).

A certain level of formalization of the process could be considered and methods to formalize the contribution of different views to the decision-making process are being developed. Examples are: a survey-based tool that allows for trade-off decision-making and joint preference articulation (Dorow *et al.* 2009); interview-based quantitative probability models reflecting differing viewpoints, which can be used for communication (Haapasaari *et al.* 2009) and which can be merged through Bayesian model-averaging and conditioned on stock assessment data (Mäntyniemi *et al.* 2009); the bridging of biological and socio-economic models to aid policy makers in examining trade-offs and consequences of different management actions (Dankel *et al.* 2009b); a quantitative technique defining fishery management objectives using stakeholder-specific utility functions to illustrate a 'zone of consensus' (where a certain proportion, for example 90%, of each group's maximum achievable utility is reached) (Dankel *et al.* 2009a). Each of these methods focuses on stakeholder-views, but they can potentially also be used to formalize the contribution of different types of scientific evidence to the decision-making process. It is well-recognized that fisheries-management decisions often suffer from a lack of stakeholder support. This deficit not only undermines the legitimacy of management policy, but it may be responsible for many of the policy's failures to achieve its objectives (Backstrand 2004; Daw and Gray 2005).

However, we should be careful not to fall into the automation-trap again, which is indeed the process we are arguing against: not everything can be or should be quantified, formalized, and automated. Moreover, it is conceivable that stakeholders do not feel their view to be represented if it is abstracted to a technical model, even if it was based on interviews with them. Therefore, if formalization is not desirable, the process could just consist of scenario discussion in a multi-stage iterative process as

proposed by Walker *et al.* (2003): (i) problem identification and framing by policy makers, stakeholders, and analysts; (ii) decision support activities; (iii) peer reviews; (iv) evaluation of outcomes by policy makers and stakeholders – with feedback to (i); (v) policy decision; (vi) implementation and communication; and (vii) monitoring. It seems that such a process was largely followed in case of the Western horse mackerel (Hegland and Wilson 2009). In this context, we see great promise in the picture that Schnute and Richards (2001) sketched [corresponding to the concept of Post-Normal Science (Funtowicz *et al.* 1999)], of a board room at some fishery institute. This may well be a board room at ICES headquarters, or a room at a RAC meeting, or an STECF meeting that is open to observers. In this imagined room, one of the modellers may note an observed outcome of their model, saying that they are not sure whether it reflects the real stock development or rather represents an artefact of the data owing to recent sampling difficulties. An oceanographer, also present in the room, may bring in some information that sheds light on the situation. An experienced biologist, who has just returned from a research vessel survey trip, may relate this information to observed developments in the abundance of the predator of the stock under consideration. In our own, slightly modified picture of that scene (Schnute and Richards 2001), a fisherman or their representative may propose to reduce the quota of the stock but allow increased by-catch of the predatory fish to compensate for the lost catches of the target species. We envisage that iterative rounds of such consultation between various scientists, managers, and other stakeholders may lead to effective development of MP. This may apply to *a priori* evaluation of MP to be developed, as well as to *a posteriori* evaluation of MP that have been in place for some years. And in the course of time we will learn how to do it better.

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