Original Article

Maximum sustainable yield from interacting fish stocks in an uncertain world: two policy choices and underlying trade-offs

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The case of fisheries management illustrates how the inherent structural instability of ecosystems can have deep-running policy implications. We contrast 10 types of management plans to achieve maximum sustainable yield (MSY) from multiple stocks and compare their effectiveness based on a management strategy evaluation (MSE) that uses complex food webs in its operating model. Plans targeting specific stock sizes (B_MSY) consistently led to higher yields than plans targeting specific fishing pressures (F_MSY). A new self-optimizing control rule, introduced here for its robustness to structural instability, led to intermediate yields. Most plans outperformed single-species management plans with pressure targets that were set without considering multispecies interactions. However, more refined plans to “maximize the yield from each stock separately”, in the sense of a Nash equilibrium, produced total yields comparable to plans aiming to maximize total harvested biomass, and were more robust to structural instability. Our analyses highlight trade-offs between yields, amenability to negotiations, pressures on biodiversity and continuity with current approaches in the European context. Based on these results, we recommend directions for developments of EU fisheries policy.

Keywords: Common Fisheries Policy, food webs, harvest control rule, management strategy evaluation, maximum sustainable yield, structural instability, theoretical ecology.

Introduction

Four questions about multispecies maximum sustainable yield

Fisheries management aiming to attain maximum sustainable yield (MSY) from multiple interacting stocks is considerably more complicated than single-stock management (Pope, 1976; Bulgakova and Kizner, 1986; Collie et al., 2003; Walters et al., 2005; Matsuda and Abrams, 2006; Geczek and Legović, 2012; Houle et al., 2013; Voss et al., 2014; Thorpe et al., 2016). A priori it is not even clear what the best translation of the MSY objective for a single, isolated stock is to cases with multispecies interactions, i.e. with feeding and competitive interactions among species (for simplicity, we use “stock” interchangeable with “fish species” in this paper). Even when a management objective considering multispecies interactions is defined, attaining it can be difficult because these interactions are generally not well known. It is therefore not surprising if legislation aiming at MSY acknowledges the role of multispecies interactions in principle, but tends to play it down. Examples are §§301.a.3, 303.b.12 of the Magnuson-Stevens Act (U.S. Department of Commerce, 2007) or Article 9.3.b of the Common Fisheries Policy (EU, 2013), but see, Section 13(2) of New Zealand’s Fisheries Act (Parliamentary Council Office, 2014). In the current practice of fisheries management, multispecies interactions among managed stocks play only a minor role. Noteworthy exceptions are cases where stock assessments take account of the dependence of predation mortality on the abundances of other species (Gjøsæter et al., 2015; ICES, 2014b).
To contribute to the development of management practices mindful of multispecies interactions we ask here:

Q1: How can the single-species MSY objective be translated into the multispecies case?
Q2: Which strategies are suited to achieve such objectives?
Q3: How do these options compare with regards to the yields they achieve, the degree of collaboration among players required to reach the objectives, pressures on biodiversity, and their political acceptability?
Q4: How much can be gained from multispecies management in comparison with management disregarding ecological interactions?

Three conceivable answers to each Q1 and Q2 are given in the following two sections. Hence there are two high-level policy choices to be made. The first choice is of the management objective, the second of the type of strategy employed to achieve it. Each of the resulting nine variants we call a management plan.

To answer Q3, we performed formal management strategy evaluations (MSEs, Walters and Hilborn, 1976; Hilborn and Walters, 1992) of these plans. That is, the outcomes of the plans were evaluated by applying them to fisheries in hypothetical ecosystems, described by an operating model, that are not fully known to the manager. To answer Q4, evaluation scores were compared with those for management that aims for MSY for the same set of stocks while disregarding multispecies interactions, modelled after current EU practice.

**First policy choice: the management objective**

There is no unique way of translating the single-species MSY objective to the multispecies case. Maximization of yield from one stock will generally require different strategies than maximization of yield from another. In a simple predator-prey system, for example, the maximization of yield from the prey requires culling the predator, while by not exploiting the prey yield from the predator can be maximized (May et al., 1979; Clark, 1990).

The three types of high-level objective we consider here are:

(i) **Nash Pressure**: To fish all exploited stocks at such rates that changes in the exploitation rate of any single stock cannot increase the long-term yield from that stock.

(ii) **Nash State**: To keep all exploited stocks at such sizes (e.g. in terms of spawning stock biomass) that changes in the size of any single stock cannot increase the long-term yield from that stock.

(iii) **Total Yield**: Maximization of the summed long-term yield from all exploited stocks.

Formal mathematical definitions of these objectives are given in Supplementary Material S1. In general, they are equivalent only in absence of multispecies interactions.

The objectives Nash Pressure and Nash State are two ways of implementing the idea of “maximization of yield from each stock separately”. They correspond to the Nash equilibrium outcome in game theory (Osborne and Rubinstein, 1994), where no player of a game could improve its gains by changing its moves. Nash equilibria are traditionally understood as arising naturally when players are not collaborating. For the Nash Pressure objective the players are hypothetical fleets, each targeting one specific stock, and the permitted moves change in their exploitation rates. For the Nash State objective the players are hypothetical managers of individual stocks and their moves are the stock sizes they aim at. The Total Yield objective corresponds a situation where players chose their moves such that the total gain by all players is maximized. Attaining this objective generally requires collaboration or enforcement through governing institutions.

Variants of Total Yield (Pope, 1976; Bulgakova and Kizner, 1986; Matsuda and Abrams, 2006; Geçek and Legović, 2012; Houle et al., 2013; Voss et al., 2014) and Nash Pressure objective (Walters et al., 2005; Collie et al., 2003; Moffit et al., 10.1016/j.dsr.2015.08.002, in press) have been considered in the literature. We are unaware of studies addressing the Nash State objective.

**Second policy choice: the strategy**

The management strategies that we consider differ by the structure of the corresponding harvest control rules (Deroba and Bence, 2008). All rules ultimately prescribe the total catch from each stock for a given time period, e.g. a year. This corresponds to total allowable catches if one assumes that allowances are fully used. Combined with estimates of stock sizes, rules for catches can be formulated in terms of rules for fishing mortality rates. Depending on how these rates are determined based on managers’ knowledge of stock sizes and interactions, we consider:

(i) **Pressure Target Control**, where fishing mortality rates are kept fixed at target values deemed to be consistent with the selected objective.

(ii) **State Target Control**, where fishing mortalities are continuously modified such as to adjust stock sizes to target values deemed to be consistent with the selected objective.

(iii) **Self-Optimizing Control**, where neither targets for fishing mortalities nor stock sizes are fixed, and instead target fishing mortalities are varied according to homogeneous linear functions of stock sizes. These functions are chosen such that the expected resulting equilibrium state is consistent with the selected objective.

**Simplifications, complications, and negotiations**

To simplify analysis and discussion, and to isolate the particular implications of multispecies interactions, a number of complications relevant in the practice of management are not covered in our analysis. First, plans do not generally include constraints resulting from conservation objectives. There are several conceivable implementations of such constraints, among others those associated with $F_{PA}$, $B_{PA}$, or $B_{rigid}$ reference points (ICES, 2014a; Task Force on Multiausal Plans, 2014), and we would not want to confound the side effects of specific implementations with the implications of the policy choices studied here. Instead, we determine biodiversity impacts of the unconstrained management plants. Second, neither technical interactions among fleets, that is, complications due to limitations to stock-selectivity of fishing, nor the resulting issues surrounding by-catch and discards are considered. Third, we disregard differences in market value of species, and other factors differentiating between biomass and economic yields. Forth, environmental stochasticity, including recruitment variability, is not considered, despite its importance for stock assessments and decision making. Real-world management
must take all these and many other complications into account and adapt objectives and strategies accordingly. The plans we consider could be modified to this end, at moderate cost in complexity and presumably with little effect on the trends we find.

However, managing each of these complications involves value judgements, and presumably requires negotiations among stakeholders. Rather than prejudicing such negotiations, one can ask how easy the negotiations would be when based on the different objectives we investigate. By definition, long-term yield is maximal for each player under the Nash Pressure objective, and so will not decline much when the player fishes at slightly different rates—and similarly for Nash State (impacts on yields from other stocks, however, can be larger!). Stakeholders will therefore be open to negotiations of their targets. This intuition is captured by proposals to define MSY as a “range” of targets rather than a particular “point” (Task Force on Multiannual Plans, 2014), and to use this flexibility to take technical constraints into account. Achievement of the Total Yield objective generally involves trading the yield from one stock off against that from another, generating winners and losers among fleets (Voss et al., 2014). Negotiations aiming for Total Yield can therefore be more difficult. The question arises what the loss in yields would be when opting for a Nash equilibrium to avoid these difficulties. This is addressed below.

As a technical simplification, required for computational reasons, our models do not structure populations by size or age. For food-web interactions, corresponding population dynamics can be derived from size-structured models using projection methods (Rossberg and Farnsworth, 2011; Rossberg 2012b). However, this means we disregard size-selectivity effects. We consider this a legitimate simplification, because, in a first approximation, optimization of size selectivity and of fishing mortality for given size selectivity are separate problems (Law and Grey, 1988; Getz, 2012; Scott and Sampson, 2011).

**Structural instability of natural and modelled ecological communities**

A particular concern of this study is the management implications of structural instability of ecological communities (Rossberg, 2013, Chapter 18). By structural instability we mean that, in a community of interacting populations, small changes in environmental conditions or external pressures can lead to large changes in population sizes (Yodzis, 1988). Structural instability manifests itself, among others, through the difficulties fisheries modellers experience in parameterizing models of interacting species such that these reproduce observed community structure and dynamics. Small deviations of model structure or parameters from reality can lead to very different model states. While the dynamics of aquatic food webs with species resolution. Species in the PDMM have different maturation body masses, which determine, through allometric scaling laws (Yodzis and Innes, 1992), maximum growth rates and consumption-independent loss rates (metabolic losses and non-predation mortality). The consumption of a resource species by a consumer species is modelled through a Type II functional response, modified to model prey-switching following van Leeuwen et al., (2013). For any two species in the PDMM, their trophic interaction strength depends on their relative body masses and the match between two kinds of abstract traits that characterize them as consumers and resources. Size preference is parameterized through a population-level predator-prey size ratio window (Rossberg, 2012b). For a full description of the model, see Rossberg (2013, Chapter 22) and Supplementary Material S2.

In the parameterization used here, the model spans approximately five trophic levels and resolves species and their interactions over 17 orders of magnitude in maturation body size (median range $10^{15}$–$10^{1.8}$ kg). Following Fung et al. (2013), species with maturation body sizes larger than 1 g are interpreted as fish. When numbering the S species in the model from the largest state relations might not be correctly represented, leading to incorrect projections for MSY. If, on the other hand, the model’s structural instability is as large as in nature, the model will be difficult to parameterize. Besides, comparing a model’s structural instability to reality is difficult. In our MSE, these problems are avoided by abandoning the idea of using an operating model representative of a specific natural community—for our general analysis, this might not even be desirable. Instead, model communities are constructed through a random process that mimics natural assembly and turnover of aquatic food webs. This leads to the kind of community saturation thought to be responsible for structural instability in nature (Borelli et al., 2015). Since, in addition, the resulting communities have macroecological properties similar to marine communities (Fung et al., 2013), we expect them to exhibit structural instability of a degree similar to that of marine systems.

We introduce below a method (“conservatism”) capable of mitigating the adverse effects of structural instability on management outcomes. In essence, it consists in avoiding management targets that are too different from the current community state to be reliably described by inherently inaccurate management models. Instead, management gradually adapts targets to meet the objective.

It turns out that the answers to above questions Q3 and Q4 are essentially determined by structural instability. Key results of our MSE can be anticipated mathematically from structural instability and general principles of community dynamics alone, as we highlight below by referring to corresponding mathematical considerations developed in Supplementary Material. The good agreement between general theoretical expectations and simulations using a complex model suggests that these expectations are sufficiently robust to be fulfilled in the real world as well. This justifies us in deriving recommendations for practical fisheries management from this study.

**Methods**

**Operating model**

The operating model representing “reality” in our MSE is the Population Dynamical Matching Model (PDMM; Rossberg et al., 2008; Fung et al., 2013; Rossberg 2013). It describes structure and dynamics of aquatic food webs with species resolution. Species in the PDMM have different maturation body masses, which determine, through allometric scaling laws (Yodzis and Innes, 1992), maximum growth rates and consumption-independent loss rates (metabolic losses and non-predation mortality). The consumption of a resource species by a consumer species is modelled through a Type II functional response, modified to model prey-switching following van Leeuwen et al., (2013). For any two species in the PDMM, their trophic interaction strength depends on their relative body masses and the match between two kinds of abstract traits that characterize them as consumers and resources. Size preference is parameterized through a population-level predator-prey size ratio window (Rossberg, 2012b). For a full description of the model, see Rossberg (2013, Chapter 22) and Supplementary Material S2.

In the parameterization used here, the model spans approximately five trophic levels and resolves species and their interactions over 17 orders of magnitude in maturation body size (median range $10^{15}$–$10^{1.8}$ kg). Following Fung et al. (2013), species with maturation body sizes larger than 1 g are interpreted as fish. When numbering the S species in the model from the largest
to the smallest, so the first \( S_0 \) species are the fish, this leads to population dynamics of the general form

\[
\frac{dB_i}{dt} = g_i(B_1, \ldots, B_S)B_i - F_iB_i \quad (1 \leq i \leq S_0),
\]

(1a)

\[
\frac{dB_i}{dt} = g_i(B_1, \ldots, B_S)B_i \quad (S_0 < i \leq S),
\]

(1b)

where \( t \) is time. The momentary growth rate \( g_i(\ldots) \) of the population biomass \( B_i \) of each species \( i \) depends on its direct biological interactions with other species. The parameters \( F_i \) represent exploitation rates, and for each fish species \( i \) the product \( Y_i = F_iB_i \) is the yield from that species per unit time. Exploitation rates are proportional to adult fishing mortalities and typically attain numerically similar values (Shephard et al., 2012), hence our use of the symbol \( F \).

PDMM model communities are generated by simulating the natural processes of community assembly and turnover by iteratively adding species to the model and removing those that go extinct, until \( S \) fluctuates around some equilibrium. This is done with all exploitation rates \( (F_i) \) set to zero. Each MSE was performed using 37 different samples from this community, taken after every 10,000 species additions and therefore composed of largely independent sets of species (of 40 communities so sampled, the first 3 were discarded as burn-in). In our parameterization (Supplementary Material S2), each sample contained around 2000 species, of which between 9 and 38 were fish (mean 22.4).

Management model

As the management model, we used the multispecies extension of Schaefer’s (1954) surplus production (or multispecies Lotka-Volterra) model (Pope, 1976, 1979)

\[
\frac{dB_i}{dt} = \left[ r_i - \sum_{j=1}^{S_0} G_{ij}B_j \right] B_i - F_iB_i,
\]

(2)

where the surplus productivities \( r_i \) and interaction coefficients \( G_{ij} \) \( (1 \leq i, j \leq S_0) \) are constants. This model has frequently been studied for management applications (Pope, 1979; Bulgakova and Kizner, 1986; ICES, 1989; Collie et al., 2003; Gaichas et al., 2012).

It has the advantages of compatibility with the abstractions employed in the operating model, Equation (1), formal simplicity, and a low number of fitting parameters compared with processed-based models. Management models used in practice are often more complex.

Supplementary Material S3 describes a method to calibrate the parameters of the management model \( (r_i, G_{ij}, 1 \leq i, j \leq S_0) \) such that it approximates the dynamics of fish populations in the operating model, Equation (1), for states similar to the current. The method is based on the assumption that the responses of non-fish species to changes in fish populations are fast compared with these changes. It should therefore work best for fish populations close to equilibrium, where they change slowly.

A mathematical analysis of the calibration method reveals that structural instability of the operating model can lead to inaccurate approximations of the operating model by the management model and to structural instability of the management model itself (Supplementary Material S4). Our MSE is designed to capture the implications for management outcomes of these effects and of incomplete representation of reality by the management model, because similar issues can arise for management models used in practice. For simplicity, the MSE does not consider measurement errors of population-dynamical parameters or implementation errors, and it assumes perfect knowledge of all \( B_i \) and \( F_i \).

Management plans are applied in cycles simulating adaptive management. At the beginning of each cycle, the management model is calibrated to approximate the dynamics of the operating model for states similar to the current state. The multispecies harvest control rule (mHCR, see below) corresponding to the plan is then (re)parameterized using the management model and used throughout the management cycle. A rather long period (50 years) is chosen for these cycles to allow the operating model to reach an equilibrium, because, with our simple calibration algorithm, this results in better fit of the management model to the operating model in the next cycle. Conceivable effects of shorter management periods were not investigated.

Multispecies harvest control rules

The three management strategies we consider differ by the formulae used to determine exploitation rates (mHCR). The free parameters in these formulae \( (F_{\text{MSY},i}, B_{\text{MSY},i}, G_i) \) are chosen such that, if the management model was the correct model of reality, the corresponding equilibrium state of the system would meet the objective of the plan. Corresponding analytic expressions are derived in Supplementary Material S5 following Pope (1979).

The simplest case is Pressure Target Control, where exploitation rates are kept constant at

\[
F_i = F_{\text{MSY},i} \quad \text{(Pressure Target Control)} \quad (3)
\]

for each fish species \( i \).

State Target Control is implemented by a rule

\[
F_i = \max \left[ 0, F_{0,i} + \frac{1}{T} \left( 1 - \frac{B_{\text{MSY},i}}{B_i} \right) \right] \quad \text{(State Target Control)},
\]

(4)

where the \( F_{0,i} \) are defined in the following paragraph, \( B_{\text{MSY},i} \) are the targeted stock sizes, and the parameter \( T \) (dimension Time) depends on how fast management attempts to reach the target. Here we chose \( T = 1 \) year rather small, to model “fixed escapement” management as recommend by bio-economic analyses with low discount rate. The operation \( \max[0, \ldots] \) replaces any negative exploitation rate by zero.

As the (time dependent) neutralising exploitation rate \( F_{0,i} \) we define the exploitation rate that would keep stock \( i \) at its current size, provided all other stocks and long-term environmental conditions remain in their current state as well. The values \( F_{0,i} \) can be computed independently of the management model, Equation (2), in practice during yearly stock-assessments using standard methods or their multispecies extensions. In our MSE we set \( F_{0,i} = g_i(B_1, \ldots, B_S) \), with \( g_i(B_1, \ldots, B_S) \) as in Equation (1a). As a result, Equation (1a) becomes effective

\[
\frac{dB_i}{dt} = \frac{1}{T} \left( B_{\text{MSY},i} - B_i \right) \quad \text{(effective dynamics under State Target Control)}
\]

(5)

whenever \( F_i \), as given by Equation (4), is positive. That is, the State Target Control rule is designed to achieve an exponential
approach of all stocks to their target sizes $B_{\text{MSY}}$ at a rate $1/T$. By modifying Equation (4) and/or the parameter $T$, other dynamics for approaching $B_{\text{MSY}}$ can be obtained. For the type of plan we recommend in Conclusions below, one such variant will briefly be discussed.

For the Nash Pressure and Nash State objectives, the Self-Optimizing Control rule has the form

$$F_i = \max \{0, \hat{G}_i B_i\} \quad \text{(Self-Optimizing Control I),}$$

where the constants $\hat{G}_i$ depend on the interaction coefficients $G_{ij}$ of the fitted management model, Equation (2). On average, approximately 4% of the calculated $\hat{G}_i$ were negative for both Nash Pressure and Nash State and therefore $F_i$ set to zero.

Self-Optimizing Control for the Total Yield objective requires

$$F_i = \max \{0, \sum_{j-1}^{N} G_{ij} B_j\} \quad \text{(Self-Optimizing Control II).}$$

The coefficients $G_{ij}$ entering Equation (7) are directly given by the interaction strengths from the fitted surplus production model, Equation (2). Because the matrix $(G_{ij})$ entering Equation (7) is the transpose of the interaction matrix $G = (G_{ij})$ from Equation (2), we call the scheme in Equation (7) Transposed Interaction Control, alluding to the “Transposed Jacobian” scheme of control theory (Craig, 1989).

### Single species harvest control rule

To model Single Species Control as currently being implemented in the EU (Task Force on Multianual Plans, 2014), we fixed target exploitation rates as in Equation (3), but now computed the targets $F_{\text{MSY}} = F_{\text{SSC}}$ using single-species surplus production models, fitted to the operating model’s apparent dynamics for that species when disregarding all others. See Supplementary Material S6.

### Addressing structural instability through conservative management plans

Mathematical analyses suggest that, due to structural instability, the parameters of mHCRs can sensitively respond to small errors in the population-dynamical parameters of management models (Supplementary Material S7) and that inaccuracies in mHCR parameters can lead to large reductions in yields (Supplementary Material S8).

It might therefore be advantageous to pursue management strategies that are conservative in the sense of preferring targets that differ less from current states than other, non-conservative strategies, even if the latter are predicted to be optimal by a naive evaluation of the management model. We achieve such conservatism through modifications of the formulae for mHCR parameters. Technically, we implement a variant of Lavrentiev regularization of matrix inversion (Engl et al., 2000), which improves the numerical robustness of the calculation, see Supplementary Material S9. A regularization parameter controls the degree of conservatism. Where applicable, we performed the MSE both with that parameter set to zero (no conservatism) and to a fixed reasonable non-zero value, given in Supplementary Material S11.2 (standard conservatism). To obtain upper bounds on conceivable further improvements of outcomes through conservatism, we evaluated in addition plans for the case where the parameter was chosen for each plan and sample community such that the actual total long-term yield attained was maximized (optimal conservatism).

### Theoretical maximum sustainable total yield

As a yardstick to compare the outcomes of management plans against, we computed the theoretical maximum long-term total yield ($\text{MSY}_{\text{theo}}$) achievable from each sampled model community. This was determined using an evolutionary optimization algorithm (Stafford, 2008). Specifically, we applied the method of Runarsson and Yao (2005) as implemented in NLopt (http://abinitio.mit.edu/wiki/index.php/NLopt), modified to improve convergence (Supplementary Material S2.2).

### Comparison of management plans

We compared all possible management plans, i.e. combinations of objectives and strategies, based on the total yield they achieved and the resulting impacts on biodiversity and community size structure. The long-term yield resulting from applying a plan to a sample community was computed as the time-averaged yield over the duration of 8 adaptive management cycles, after a 24 cycle transition phase. Because $\text{MSY}_{\text{theo}}$ varied substantially among sample communities ($\text{CV} = 0.40$), we quantified the yield from each plan as the average over all sample communities of the percentage of $\text{MSY}_{\text{theo}}$ attained.

As secondary criteria for comparison, we computed the proportion of fish species conserved (100% – proportion extirpated) and the mean logarithmic maturation body mass of species at the end of each MSE run. The latter addresses the specific concern that maximization of yields might require culling large piscivorous species to release pressure on smaller and more productive planktivores. As for yields, both indices were averaged over all sample communities. For comparison, we calculated these indices also for unfished communities after simulating them over a period corresponding to 32 management cycles, because slow residual dynamics of stocks (Fung et al., 2013) can result in natural extirpations.

In the policy context, other criteria will also play a role, e.g. the similarity of plans to those currently in place or, based on game theoretical considerations, the likelihood with which stakeholders will agree to plans or small modifications of them. Rather than attempting to score plans based on such criteria, key considerations are suggested in Discussion.

### Results

Table 1 displays for each type of plan the average proportion of $\text{MSY}_{\text{theo}}$ achieved in the MSE, together with the proportion of surviving species and changes in size structure. Typical standard errors are around 3% for yield scores, 2% for survival rates, and 0.12 for mean log size. More detailed statistical results, including error estimates and statistical significance of differences, are documented in Supplementary Material S11. There we also report yields for a second, statistically independent MSE based on model communities of half the size used here. Remarkably, differences in yield scores between the two MSE are small, suggesting that our findings depend little on the size of communities.

Table 1 shows that exploitation leading to $\text{MSY}_{\text{theo}}$ would come at the cost of conserving only about 72% of fish species and a change in species size structure corresponding a drop of 52% ($= 1 - 10^{0.85-1.97}$) in geometric mean maturation body mass. Such impacts are plausible in view of a corresponding survival rate of 51%
obtained by Matsuda and Abrams (2006). Yields from Single Species Control were only about half of MSYtheo and resulted in survival of 63% of species and a decline in geometric mean size by 45%.

Among the full range of plans considered, a number of general trends can be observed:

– Independent of objective, State Target Control gave higher long-term yields than Pressure Target Control. This is expected from structural instability (Rossberg, 2013, Chapter 24), as explained in Supplementary Material S8. Remarkably, these yield increases were paralleled by higher species survival rates.

– For the Total Yield objective, conservatism did substantially increase yields with both Pressure Target Control and State Target Control. The same holds, to a lesser extent, for the Nash Pressure objective. As theoretically expected from structural instability (Supplementary Materials S7 and S9), this was not the case for the Nash State objective. Conservatism increased species survival without exception.

Table 1 Illustration of trade-offs among multispecies management plans to achieve MSY, as derived from our MSE. Intensity of shading increases with score attained.

| Strategy                     | Conservation (standard/optimal) | Objective other | Nash Pressure | Nash State | Total Yield |
|------------------------------|---------------------------------|-----------------|---------------|------------|-------------|
| Pressure (‘F’) Target Control| no                               | 50.6            | 47.4          | 33.9       |
|                              | standard                        | 56.7            | 38.2          | 56.1       |
|                              | optimal                         | 72.9            | 46.5          | 72.8       |
| State (‘B’) Target Control   | no                               | 70.6            | 62.2          | 57.0       |
|                              | standard                        | 73.3            | 65.9          | 85.0       |
|                              | optimal                         | 83.4            | 73.2          | 93.0       |
| Self-Optimising Control      |                                 | 71.7            | 62.6          | 76.4       |
| Single Species Control       |                                 | 51.7            |               |            |

| Species survival (% initial number of fish species) |
|----------------------------------------------------|
| Pressure (‘F’) Target Control                      |
| no                                                 |
| standard                                           |
| optimal                                            |
| 62.0                                               |
| 68.9                                               |
| 77.0                                               |
| 70.4                                               |
| 77.6                                               |
| 80.7                                               |
| 88.9                                               |
| 62.7                                               |
| MSYtheo                                            |
| Unfinished                                         |

| Fish community mean \(\log_{10}\) maturation body mass (grams) at the end of MSE |
|-----------------------------------------------------------------------------|
| Pressure (‘F’) Target Control                                              |
| no                                                                         |
| standard                                                                   |
| optimal                                                                    |
| 1.70                                                                       |
| 1.84                                                                       |
| 1.90                                                                       |
| 1.68                                                                       |
| 1.77                                                                       |
| 1.80                                                                       |
| 1.84                                                                       |
| 1.71                                                                       |
| MSYtheo                                                                    |
| Unfinished                                                                 |
| 1.65                                                                       |
| 1.97                                                                       |
With standard conservatism, Self-Optimizing Control resulted in yields intermediate between State Target Control and Pressure Target Control, independent of objective.

Optimal conservatism led to improvements over standard conservatism by 8.3–16.9% of MSY_{theo}. The pattern of improvements was similar to that for the difference between standard and no conservatism.

Independent of objective, yields from Pressure Target Control using standard conservatism improved only slightly over Single Species Control. However, yields from Pressure Target Control with optimal conservatism were notably higher, which indicates potential for some improvements through advances in mathematical methods. Without conservatism, yield score, survival rate and impact on community size structure for Pressure Target Control with Nash Pressure objective were almost identical to Single Species Control.

Independent of strategy, total yields obtained under Nash State objective were consistently lower than those under Nash Pressure objective.

Remarkably, total yields attained aiming for Nash Pressure objective tended to be similar to or higher than those attained aiming for Total Yield objective. The expected reversal occurred only with the conservative variants of State Target Control. With optimal conservatism, State Target Control yield came (on average) within 7% of MSY_{theo}, while survival rate and impact on community size structure were almost identical as for MSY_{theo}.

Impacts on species survival and community size structure were consistently lowest with Nash State, highest with Total Yield and intermediate with Nash Pressure objective.

**Discussion**

**Biodiversity impacts**

In our MSE, strategies that yield more are also beneficial to species survival and community size structure, while for management objectives this relation is reversed. Although noteworthy, we caution not to over-interpret these relations, because the management plans could be modified to explicitly incorporate conservation constraints (Matsuda and Abrams, 2006). To illustrate possible implications of conservation constraints, Table 2 compares yield, survival rate and community size structure with and without explicit inclusion of such constraints in State Target Control management plans for Nash Pressure and Total Yield objectives. The constraint in this case is to not purposely deplete fish populations to less than 10% of their virgin biomass, see Supplementary Material S5.2. Interestingly, this modification reduces extirpations and erosion of community size structure substantially, without significantly affecting yields. In view of this robustness of yields, we disregard conservation in the following, acknowledging that conservation while aiming at multispecies MSY requires further study.

**Structural instability**

Our MSE demonstrates how structural instability of ecosystems, and so of ecosystem models, can strongly impact management outcomes. Results from our MSE based on a complex food-web model closely follow expectations derived from general mathematical arguments (Rossberg, 2013, Chapter 24; Supplementary Materials). The mathematical arguments also appear to be robust to many real-world complications not considered in the MSE, as already demonstrated for conservation constraints. Structural instability issues persist when transitioning from stock- to fleet-based management (Rossberg, 2012a) or when multiplying yields by market values. Recruitment variability and imperfect control of applied fishing pressures lead to additional uncertainty, which structural instability can enhance (Supplementary Material S8).

The MSE also highlighted options to mitigate these impacts. The most important may be adoption of state-rather than pressure targets in management plans. Another mitigation option is conservatism, i.e. regularization of the inverse problems underlying the computation of multispecies MSY reference points. The mathematical scheme for this proposed in Supplementary Material S9 can be extended to more detailed management models if required. To determine good degrees of conservatism when direct comparison of long-term management outcomes is not practical, one can compare simulated outcomes using members of a plausible model ensemble (Ianelli et al., 2015), applying conservative reference points derived using other members.

The EU’s current approach to implementing its Common Fisheries Policy (CFP, EU, 2013), here modelled as Single Species Control, is mathematically similar to aiming at the Nash Pressure objective using Pressure Target Control without systematic conservatism (Supplementary Material S6). Our analysis shows that this type of management plan is particularly vulnerable to structural instability. Expected yields are therefore comparatively low. These difficulties are illustrated in simulations by Lynam and

| Scoring criterion | Management plan | Species conservation | paired t-test p-values |
|-------------------|-----------------|----------------------|----------------------|
|                   | State Target Control for Nash Pressure objective | No | Yes | 0.744 |
| Yield             | State Target Control for Total Yield objective | 73.3 | 72.6 | 0.744 |
|                   | State Target Control for Nash Pressure objective | 85.0 | 85.5 | 0.713 |
| Survival rate     | State Target Control for Nash Pressure objective | 77.6 | 87.0 | 0 |
|                   | State Target Control for Total Yield objective | 70.4 | 91.9 | 0 |
| Mean log_{10} size| State Target Control for Nash Pressure objective | 1.77 | 1.85 | 0.038 |
|                   | State Target Control for Total Yield objective | 1.71 | 1.90 | 0.004 |
Mackinson (2015, Fig. 6c) predicting that a transition to the MSY exploitation rates recommended by the International Council for the Exploration of the Sea (ICES) will lower rather than raise total long-term yields.

Choice of objective

The MSE showed that the consequences of the two major policy choices we considered, i.e. of objective and of strategy to achieve it, are qualitatively independent. They are therefore discussed separately hereafter. In doing so, we shall assume that management models do take structural instability sufficiently into account such that the “standard conservatism” case in Tab. 1 is representative.

Among the three objectives analysed, sensitivity to structural instability increases with increasing expected total yield, i.e. in the expected order Nash State → Nash Pressure → Total Yield objective (theoretical expectations are summarized in Supplementary Material S10). In view of the high empirical uncertainty in the strengths of multispecies interactions, the Nash State objective might therefore be favoured by stakeholders, despite the comparatively low yields this implies. When combined with State Target Control, it has the added advantages of transparency and simplicity; stakeholders can focus on simply reaching agreement on the targeted ecosystem state. Exploitation rates and corresponding yields required to achieve this state would play a role in negotiations, but not explicitly become part of agreements. Our results suggest this approach would yield about 20–30% higher than Single Species Control.

Despite Nash Pressure and Nash State objective having similar definitions, corresponding expected long-term yields clearly differ. The higher yields predicted for Nash Pressure could be reason for stakeholders to favour this objectives over Nash State. We therefore caution that we could not find a reason why Nash Pressure should generally yield more. Indeed, Supplementary Material S10 constructs counter-examples. This pattern should therefore be verified using MSeS that vary our setup.

Independent of the higher expected yields, there are two considerations that favour Nash Pressure over Nash State objective. First, Nash Pressure could be more acceptable to stakeholders familiar with Single Species Control because it is conceptually similar. Secondly, the fictitious game underlying the Nash Pressure objective comes closer to the situation of real multi-stakeholder fisheries than that underlying Nash State. Agreeing on this objective or variations of it might therefore be easier.

Improvements in yields when going over from Nash Pressure to Total Yield objective were surprisingly small, if present at all. Even with State Target Control, they were just 16% (standard conservatism). This might be insufficient reason to adopt the Total Yield objective, considering that this might lead to difficult negotiations, as explained in the Introduction. Nevertheless we anticipate that, when either Nash Pressure or Nash State objective is taken as the starting point for negotiations, they will develop towards the Total Yield objective by agreements on modifications of targets that give higher overall yields or incomes.

Choice of strategy

Considering the choice of strategy, out MSE strongly favours State Target Control over Pressure Target Control. This result is expected (Supplementary Material S10), because structural instability leads to large changes in community structure in response to small changes (or errors) in applied pressures. It is worth noting that, for the same reasons, community structure is expected to be highly sensitive to changes in the physical environment. Within limits, State Target Control counteracts this sensitivity. We warn that MSE that do not use a structurally unstable operating model, e.g. an operating model with non-interacting species, will be unable to reproduce these phenomena.

Although Pressure Target Control is currently favoured by the EU for most stocks, State Target Control is either legal or de-facto policy, e.g., in the United States, Canada, and New Zealand. Even the EU does not appear to be legally bound to Pressure Target Control. The CFP in its current form requires, according to Article 2 that “management […] restores and maintains populations of harvested species above levels which can produce the maximum sustainable yield.” It does then set deadlines for adjusting exploitation rates accordingly, but these are not the principal objective. Further, the CFP requires formulation of multiannual management plans with “objectives that are consistent with the objectives set out in Article 2 […]”. The plans shall include “quantifiable targets such as fishing mortality rates and/or spawning stock biomass”. However, considering the low expected yields from Pressure Target Control, we suggest that multiannual plans setting targets for fishing mortality or exploitation rates may not actually be consistent with the objectives set out in Article 2.

Self-Optimizing Control strategies have their own strengths. They are largely insensitive to structural instability, do not require conservatism, and yet achieve long-term yields nearly as large as those from State Target Control. Further, the corresponding mHCRs are independent of productivity rates $r_i$ in Equation (2)], which lets these strategies automatically adjusts stock sizes to corrected MSY levels if productivities change because of short- or long-term environmental change.

A specific disadvantage of Self-Optimizing Control with Total Yield objective, Transposed Interactions Control, is that the prescribed exploitation rate of a stock depends on the abundances of all stocks on which it has a direct ecological effect: prey, predators, and competitors. This can increase measurement and planning uncertainty. (For State Target Control such dependencies enter through neutralising exploitation rates, but there only as corrections to past observations, so uncertainty is lower.) Despite its elegance, Transposed Interactions Control might therefore be preferred only in exceptional cases, e.g. for data-rich single-stakeholder fisheries that have already approached the Total Yield objective and require a scheme to effectively adapt to uncertain environmental change.

Implementation

Most management plans we discussed require for their implementation two models with different skills (Dickey-Collas et al., 2014). The first is what we called the management model. It serves to compute parameters of mHCRs consistent with objectives. Emphasis is on the skill to make long-term MSY projections.

In practice, a second type of model is needed to determine the stock sizes $B_i$ required to compute exploitation rates $F_i$ from mHCRs and to convert these into catch allowances. Conventional age or size-structured stock-assessment models can be use for this. They have the desired skill to accurately determine current states of stocks based on current and historical data. State Target Control requires in addition that these models can estimate neutralizing exploitation rates. To achieve this, they must be capable
of forecasting states for given exploitation rates a few years into the future. The key skill is therefore short-term projection.

Strategies to implement these two model types will be different. For long-term projections, interactions with the ecosystem at large and general principles such as conservation and dissipation of energy as it flows through food chains will be important. For short-term projections, optimal use of available data will be key, as with current stock-assessment models.

Conclusions

Because in fact fish stocks do interact, it is not obvious how the frequently invoked MSY policy objective should be translated to real-world fisheries. Answering questions Q1 raised in Introduction, we listed a range of management objectives that would for an isolated stock all reduce to classical MSY. Policy needs to clarify which one is meant! Addressing Q2, we considered a range of harvest control rules and implementation details, and found that management outcomes depend sensitively on their choice.

Answering Q3, the preceding Discussion highlighted trade-offs among management options: the plans expected to generate highest yields require large changes in management practice, accurate data, and/or good collaboration among stakeholders. The EU’s current scheme of Single Species Control yields substantially less than the target value, than most other options (Table 1), which answers Q4.

While there are tradeoffs among all options considered, we propose as a conceivable step towards increased effectiveness, sophistication, and predictability of EU fisheries management a transition to State Target Control with a Nash Pressure objective. In this study, we evaluated the scheme \( F = F_0 + (1 - B_{SSC}/B) / T \), where \( F_0 \) is the neutralising exploitation rate, \( B \) is the current stock size, and \( T \) is a relaxation time parameter, arbitrarily chosen as \( T = 1 \) year. If the resulting value for \( F \) is negative, it is replaced by zero. When applied to an isolated species, the graph of \( F \) vs. \( B \) for the corresponding HCR is similar, e.g., to that of the Fishery Decision-Making Framework used by the Canadian government (http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/fish-ren-peche/sf-cpd/precaution-eng.htm), except that \( F \) is allowed to exceed the predicted value for \( F_{MSY} \). Increases of \( F \) beyond \( F_{MSY} \) can be necessary because \( F_{SSC} \) is uncertain and an uncontrolled stock size larger than the target value \( B_{SSC} \) could lead, through multispecies interactions, to unintended and uncontrolled effects on other stocks, a situation that State Target Control aims to avoid.

Variations of this mHCR are conceivable, and many may be similarly effective. For example, following discussion within the Working Group on Multispecies Assessment Methods of ICES (ICES, 2014b), we evaluated the modified rule \( F = F_0 - |F_{SSC}| (1 - B/B_{SSC}) \), where \( F_{SSC} \) is the target exploitation rate of the Single Species Control scheme. While with this modified rule target stock size is approached at the potentially slower rate \( |F_{SSC}| \), the rule has the intriguing property that it would reduce to \( F = F_{SSC} \), i.e. to conventional Single Species Control, if the single-species Schaefer model was accurate. Reality is more complicated and the HCR will deviate from Single Species Control, but these deviations may be small. Applying our MSE to this rule with a Nash Pressure objective, we obtain on average long-term total yield of 75.4% MSY\(_{dueo} \) (standard conservatism), not significantly different from the result using our original State Target Control rule \((P = 0.15, \text{paired } t\text{-test})\). Such results open up the possibility of swiftly transitioning from the current scheme to State Target Control, enabling higher future yields and more predictable ecological outcomes, with minimal changes to currently recommended exploitation rates.

As discussed earlier, the formulation of the EU’s CFP is open to both Pressure Target Control and State Target Control. We recommend using of this flexibility to achieve best possible outcomes for society and the marine environment.

Supplementary data

Supplementary material is available at the ICES JMS online version of the article.

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