THE ROLE OF HARVEST CONTROL LAWS, RISK AND UNCERTAINTY AND THE PRECAUTIONARY APPROACH IN ECOSYSTEM-BASED MANAGEMENT

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ABSTRACT

The traditional fisheries management approach involves scientists providing their best assessment of the status and productivity of a resource. They then use these results to recommend a control measure, such as a Total Allowable Catch (TAC), based upon some harvest control law, which is usually associated with a biological reference point (e.g. $F_{0.1}$). Superficially, the Operational Management Procedure (OMP), or equally the Management Strategy Evaluation (MSE), approach for providing TAC recommendations may appear identical, as this often also links the results from some form of assessment to a harvest control law. However, the key difference is that the OMP/MSE approach involves simulation testing of the whole process that gives rise to the TAC recommendation within an adaptive management framework. This testing includes checks that application of the control law adopted will not lead to major problems, even if key perceptions about the resource happen to be in error; in other words, explicit account is taken of scientific uncertainties, in the spirit of the precautionary approach. Furthermore, quantitative evaluations are provided of the levels of catch to be anticipated in the medium term, and how these trade off against levels of risk of unintended depletion of the resource, to provide managers with a readily interpretable basis to choose between different management options. However, the process involves some problems in defining risk, which have yet to be resolved.

Examples where ecosystem considerations have been taken into account in extending this OMP/MSE approach beyond the single-species level can be conveniently divided into two broad categories, depending on whether they concentrate primarily on operational (e.g. by-catch) or biological (e.g. predator-prey) interactions between species, and examples are given of each. To date, actual practical applications of this approach are more readily found for cases of operational interactions, particularly in the area of marine mammal by-catch. For practical applications involving biological interactions, the key limiting factor thus far is the paucity of data to estimate the form and magnitude of predation and competition interactions, which precludes confident computation of the trade-offs between harvest policy options that differ in the extents to which they concentrate upon different species. Nevertheless there are approximate approaches for dealing with this problem. We recommend the use of such approaches, while recognizing their limitations, until the data needed to develop more reliable models of biological interactions become available.

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INTRODUCTION

[1] Even aside of ecosystem considerations, our title covers as many as four topics. While the relationships between some of these are rather obvious – for example, greater uncertainties relate to increasing risk – those between others are perhaps not as self-evident.

[2] Our first task, therefore, is to make the argument that what has been called the "Operational Management Procedure" (OMP) (Butterworth and Bergh, 1993; Butterworth *et al.*, 1993, 1997; Cochrane *et al.*, 1998; De Oliveira *et al.*, 1998a, b; Butterworth and Punt, 1999) or, equivalently, the "Management Strategy Evaluation" (MSE) (Smith, 1994; Smith *et al.*, 1999; Punt *et al.*, in press) approach to fisheries management provides a framework within which these four topics are readily related and integrated.

[3] The customary process used by fishery scientists to provide advice on control measures for a resource involves two steps. First, the available data are integrated through a mathematical process called stock assessment to provide a "best" assessment of the status and productivity of the resource – for example its size, possibly expressed as a fraction of that before exploitation commenced, and the level of catch that is sustainable. Then a formula – a harvest control law – is used to translate this information into, say, a Total Allowable Catch (TAC) recommendation for the coming year. For example, the rule could be to recommend a TAC that is a fixed fraction of the estimated resource size, where that fraction is chosen to move the resource towards some level considered to provide optimal exploitation. Often the fraction harvested, or the target abundance level, are chosen from "biological reference points" in common use (e.g. $F_{0.1}$).

[4] Superficially, the end product of the OMP-MSE approach (referred to hereafter as an OMP) often appears near identical: some method of estimating current resource abundance (whether in tons or in terms of some relative index) and a harvest control law or formula to convert this to a TAC. Where the key difference lies is in the method used to select the law (or, more accurately, the combination of the data inputs, the basis for estimating abundance and the harvest control law that comprises the complete OMP). The OMP used to recommend TACs for the directed sardine fishery in South Africa (De Oliveira *et al.*, 1998b) provides a particularly simple example to illustrate this difference. This is because the data input specified – the annual hydro-acoustic spawning biomass survey of this resource – provides the requisite abundance index directly. The harvest control law is simply to set the TAC recommendation as 13.75% of this hydro-acoustic abundance estimate, subject to the constraints that the TAC should not change by more than 25% from that for the previous year. Note the adaptive nature of this approach: the TAC is raised or lowered in response to monitoring results that indicate respectively an increase or decrease in resource size. But on what basis were the 13.75% and 25% figures selected?

[5] The selection process involves projecting the resource status forward for a period of typically ten to twenty years in computer simulations, where the future catches are set by applying candidate OMPs with their harvest control laws. For example, in the sardine example above, the candidates reflect alternative values for the harvest percentage (the 13.75%). The choice between these alternatives is then made on the basis of anticipated performance as indicated by the statistics that summarize the results of the simulations. These "performance statistics" typically include measures relating to the size and variability of catches, and also to the risk of depleting the resource below a level at which future recruitment success might be seriously impaired.

[6] Conducting such simulations requires a basis (or "model") to predict how the resource will respond to different future levels of catch. If only the "best assessment" of the resource was used for this purpose, again there would be little difference from the traditional approach. The major novel feature of the OMP selection process is the formal recognition that this "best assessment" may well be wrong, so that simulations are conducted for several other plausible appraisals of the resource status and dynamics, which reflect the range of uncertainties in present knowledge of the

resource and the extent to which management actions can be implemented. The aim is to seek an OMP that is as "robust" as possible to these uncertainties.

[7] What this means is that the anticipated performance, as summarized by the performance statistics, should not change greatly over the range of uncertainties. In other words we are looking for an OMP that is adaptive in an even stronger sense than described above for the sardine example – we want it to self-correct over time even if some of the assumptions made in developing our "best assessment" were wrong. In the sardine case, for example, checks were made that the OMP (particularly here its harvest control law) would still perform appropriately even if assumptions for hydro-acoustic target strength made in deriving absolute abundance estimates from the surveys were in error by quite substantial amounts. Thus the requirements of the precautionary approach are being met: the OMP selection process provides a formal structure to take account of scientific uncertainties.

The uncertainties or potential errors that need to be considered in the OMP evaluation process fall into three categories. They are model errors (in assumptions about the dynamics of the real resource), data errors (regarding how what is measured and used for input to the TAC computations relates to the actual status of the resource), and implementation errors (relating to plausible differences between the management recommendations and what actually occurs in the fishery, such as between the TAC authorized and the actual subsequent catch). The process of selecting an OMP also requires that management objectives be operationalized, so that they can form the basis for the performance statistics used to compare candidate OMPs. Single-species management objectives generally fall into three categories: maximizing catch, minimizing risk to the resource and minimizing catch variability over time. These objectives cannot be satisfied simultaneously; for example, in the case of the sardine fishery mentioned above, increasing the proportion of the survey biomass harvested gives greater catches but increases the risk of depleting the resource more than desired. Reducing the maximum percentage change allowed in the TAC from year to year necessitates a lower level of utilization overall, so that catch reduction at times of downward fluctuations in resource size remain sufficient to control risks.

[8] The final selection between candidate OMPs involves decision-makers seeking their desired trade-off between these conflicting objectives. This process is facilitated by choosing measures of performance that are readily meaningful to non-scientists. This is straightforward as far as anticipated catch is concerned, and not too difficult for catch variability. The problem lies with risk. This is often expressed as the chance of depleting the resource below some reference level (often the criterion used is 20% of the abundance before exploitation commenced) over a simulation period of, say, 10 or 20 years (e.g. De Oliveira *et al.*, 1998a, b). The reason an extended period is needed is that, for all but very short-lived species, the risk associated with a single-year decision is often negligibly small, and becomes meaningful only when evaluated for the repeated application of a specific OMP, with its harvest control law, over a longer period. The key difficulty is how to interpret results for risks over simulation tests that span a wide range of uncertainties (Butterworth *et al.*, 1996). However good an OMP is at self-correcting for errors, it is always possible to envisage scenarios that result in a high probability of excessive resource depletion. In evaluating such results, it is critical that the relative plausibilities of such scenarios are also factored in – it is unnecessarily wasteful of the resource to manage in a manner that concentrates on securing against a situation considered very unlikely to apply in reality.

[9] In summary then, the OMP basis for fisheries management is a structured approach to take account of scientific uncertainties, in the spirit of the precautionary approach, when choosing a harvest control law that will reasonably contain the risk to the resource, i.e. it integrates over four of the five topics of our title. But this approach was developed in the context of single-species situations – can it be extended to ecosystem-based management?

EXTENSION OF THE OMP/MSE APPROACH TOWARDS AN ECOSYSTEM BASIS – SOME EXAMPLES

[10] The extension of the OMP approach towards an ecosystem basis involves no problems of principle, only of complexity and yet greater lack of knowledge (Sainsbury *et al.*, 2000). Essentially, it entails changing to a situation where more than one, rather than only a single, species is considered. This requires that the model used to project resource abundance forward under the impact of future catches includes all such species, and specifies how they interact with each other through predator-prey and other effects. Selection of OMPs also becomes more complicated, as separate performance statistics are needed for each species, and trade-off considerations now involve inter-species effects: increasing the anticipated catch of one species may require decreasing that of another.

[11] Because of this added complexity, applications to date of the OMP approach at the ecosystem level are unsurprisingly limited in number and extent. We summarize a few examples below. These are conveniently separated into cases of operational and biological interactions, i.e. whether the linkage between the species occurs because of their co-occurrence in catches, or because one eats the other or they compete for the same prey.

Operational interactions

South African pelagic fishery

[12] Two species dominate the South African pelagic fishery: the anchovy, which is used almost exclusively for the production of fish meal, and the more valuable (adult) sardine, which can be canned for human consumption. The operational interaction arises because the anchovy fishery concentrates on the recruits of the year during the winter months, as it is only over that period that these fish aggregate sufficiently and close enough to the coast to render catching economically viable. However, that is the very period when juvenile sardines are found mixed with the anchovy schools. The greater the anchovy catch, the greater the unavoidable associated catch of juvenile sardine, and hence the subsequent reduced directed catch of the more valuable adult sardine that can be permitted.

[13] This fishery is managed using a two-species OMP that sets limits each year on the directed adult sardine catch, the anchovy catch and the juvenile sardine by-catch, primarily on the basis of annual hydro-acoustic surveys of sardine and anchovy recruitments in May and spawning biomasses in November (De Oliveira *et al.*, 1998b). A key aspect in the selection of this OMP was the trade-off between anticipated directed sardine and anchovy catches, as illustrated in Figure 1. For high levels of directed sardine catch, the anchovy resource potential is wasted because of the need to keep the juvenile sardine by-catch low.

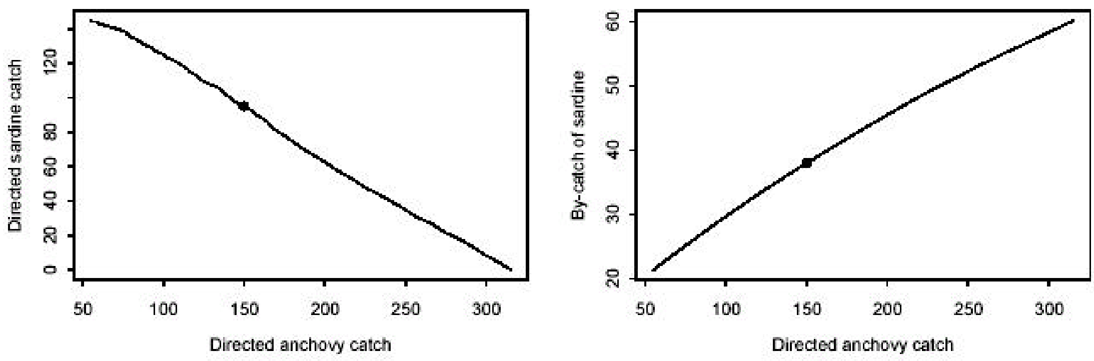

Figure 1. Trade-off between expected values of the directed catch of anchovy, the directed catch of sardine, and the level of sardine by-catch (units are '000 tons) (J. De Oliveira, pers. comm.). The black dots reflect the expectations for the selected OMP.

[14] A problem in selecting an OMP to give the desired trade-off in terms of the plot in Figure 1 is that different companies in the industry have different preferences because their existing processing capabilities are geared more towards fish meal or towards canned fish production. This renders consensus on a desired trade-off difficult to achieve. To cater for this, initiatives are now being pursued to allocate companies rights as proportions of the fishery as a whole, rather than of the TAC for each species separately. Each company then selects its own desired trade-off, and is managed under a company-specific OMP that yields that trade-off.

[15] Each company's quotas each year comprise the TACs indicated by "their" OMP, multiplied by that company's proportional right in the fishery as a whole. Thus rather than the conventional process of dividing a global TAC between rights holders, the annual TAC comprises the sum of allocations (under different OMPs) to the different companies. The OMP framework admits this industrially desirable enhanced flexibility, without compromising on the risk of unintended depletion of either resource.

Marine mammal by-catch

[16] Management advice on acceptable levels of by-catch of marine mammals in USA fisheries is based on the Potential Biological Removal (PBR) approach (Wade, 1998). The formula used to compute the PBR was selected to ensure, in particular, that such by-catches would not impair the recovery of previously depleted marine mammal populations. The parameters of the PBR were selected using the standard OMP simulation testing approach, considering all three categories of error or uncertainty (model, data and implementation) indicated above.

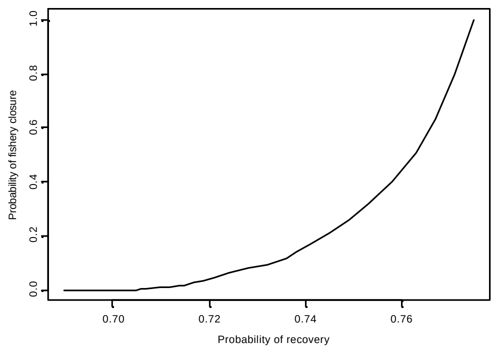

Figure 2. Relationship between the probability of the arrow squid fishery being closed and the probability that the Hooker's seal-lion population recovers to 90% of its pre-exploitation level (after Maunder *et al.*, 2000).

[17] In New Zealand, the fishery for arrow squid is closed if the estimated incidental kill of Hooker's sea lions exceeds an amount calculated by a method very similar to PBR. Though PBR and this approach were selected considering only containment of the risk to the marine mammal population, it is important to appreciate that there is, in reality, a trade-off issue involved. Varying the formula used to set the allowed incidental Hooker sea lion kill shows that the greater the probability of a recovery of the sea lion population, the greater also the probability of a squid fishery closure, with consequent economic loss to the squid industry (Figure 2).

Biological interactions

Fur seals and hake off South Africa's west coast

[18] The trawl fishery for hake off South Africa provides about half the landed value of all the country's fisheries combined. TACs for this fishery are set using an OMP, and approach 100 000 t/yr on the west coast. Early in the 1990s, concerns arose because the seal population at that time was estimated to consume a similar quantity of hake, and was forecast to double by the turn of the century (Butterworth and Harwood, 1991).

[19] An OMP approach was used to evaluate whether a seal cull would be beneficial to the hake fishery (Punt and Butterworth, 1995). The computations were based upon a model involving the two species of hake present (shallow- and deep-water), seals, other predatory fish lumped together, and the hake fishery with TACs set by the OMP. The perhaps surprising conclusion from the OMP evaluations was that a seal cull would lead to a slight reduction in realized hake catches. The underlying reason for this was that seals are thought to eat predominantly the younger shallow-water hake (Punt *et al.*, 1995), which when grown predate the young of the deep-water hake (Punt and Leslie, 1995). Fewer seals would mean more young and hence subsequently more adult shallow-water hake, which would eat more young deep-water hake and hence reduce the overall deep-water hake population (Punt and Butterworth, 1995). In combination, it turned out that the increase in shallow-water hake was more than offset by the decrease in the deep-water species.

Minke whales and the fisheries of the greater Barents Sea

[20] Structurally, this analysis (Schweder *et al.*, 1998), involving cod, capelin, herring and minke whales, was very similar to that for hake and fur seals described above. Whale catches in the simulations were set according to a variant of the Revised Management Procedure (RMP) developed by the International Whaling Commission (IWC, 1994). Cod and herring catches were computed so as to meet pre-specified fishing mortality targets.

[21] The results suggested that changing to a less risk-averse variant of the RMP would increase not only annual catches of minke whales by 300, but also annual cod catches by 100 000 t on average. The extra cod catch results from an increase in cod abundance as a result of decreased predation pressure from minke whales. However, the consequences for the herring fishery were not as clear. The reason is that both minke whales and cod eat herring, so that if the one decreases and the other increases in abundance, even the direction of the net effect on herring abundance, and hence catches, become difficult to predict.

Adaptive management of fisheries on Australia's northwest shelf

[22] After the introduction of fishing in this area, species composition showed a change, with an increase in species of lower value at the expense of higher-valued species. There were a number of different hypotheses to explain this change, including competitive interactions between species and alterations to the sea bed habitat. Each had different implications for how best to manage the fishery in the future (Sainsbury, 1991).

[23] The analysis was conducted on the OMP basis, but with a particular novel objective. This was to ascertain the relative potential of different forms of experimental management regimes, involving closed areas and uses of different gear types, to distinguish among the competing hypotheses and hence enhance the benefits from future management (Sainsbury, 1991; Sainsbury *et al.*, 1997). A key consideration was the length of time such an experimental regime should continue: longer periods provided better discrimination power, but reduced the overall value to be obtained from the series of catches to follow under future management.

Fur seals and krill off Antarctica

[24] The convention governing the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) became known as the first "ecosystem-orientated" convention because of the requirement of its Article II that levels of harvest allowed take account of the needs of dependent and related species. This arose from the particular concern that a developing fishery for Antarctic krill could harm populations of predators heavily dependent upon krill as a food source.

[25] The most developed analysis to date that attempts to assess such an impact models the biological interaction between krill and Antarctic fur seals (Thomson *et al.*, 2000). Precautionary annual catch limits for krill are set on the basis of a fraction of a survey estimate of krill biomass.

This analysis suggested that for a seal population of at least half the abundance expected in the absence of any krill fishing, the fraction of krill harvested should not exceed a value somewhere in the range of 4 to 23%.

SOME COMMON THEMES AND KEY LESSONS FROM THE EXAMPLES

[26] The leading question is: How many of the analyses listed above have been used in practice in formulating actual fisheries management decisions?

[27] For the operational interactions, the record is good. A joint sardine-anchovy OMP, incorporating the consequences of juvenile sardine by-catch, has been used to set TACs for the South African pelagic fishery over the last seven years. Marine mammal by-catch limits indicated by the PBR approach in the USA do not result in fishery closure if exceeded, but do at least lead to the formation of a "take reduction team" to work towards getting the by-catch back below the PBR limit. A harder line applies in New Zealand, where the squid fishery is closed if the Hooker sea-lion by-catch limit is reached. Moreover, computations of the trade-off between loss of squid catch and extent of recovery of the sea-lion population was not considered in specifying the sea-lion by-catch formula applied.

[28] For the biological interactions, the Australian northwest shelf analysis is a particular success story. The experimental regime commenced in 1985 and by 1991 had achieved successful discrimination among the competing hypotheses, and allowed a management system to be put in place with greater anticipated economic returns than would otherwise have been possible (Sainsbury *et al.*, 1997). However, none of the other examples quoted have directly affected management decisions. The reasons for this are numerous and varied. For example, in the South African fur seal example, the priority accorded the issue fell as a result of a lower seal population increase rate than had been predicted, as well as some other, political, factors.

[29] Nevertheless a common feature of most of these biological interaction analyses is the lack of certainty associated with the results they provide. The authors of the Barents Sea analysis term their quantitative results for the effects of increased whaling on the herring and cod fisheries "tentative." The forecast direction of the effect of a cull of fur seals on the South African hake fishery is reversed by assuming that the seals also consume the deep-water hake species to some extent – a possibility that existing dietary studies would not exclude. The attempt to estimate appropriate krill harvest levels by modelling the krill-Antarctic fur seal interaction directly yields a result that is much too imprecise for practical application. Instead, krill precautionary catch limits are set on the basis of keeping krill abundance, on average, at 75% of its level in the absence of harvesting, the 75% having been chosen as half way between the 100% as optimal for the predators in isolation, and 50% as (roughly) optimal for a krill fishery considered in a single-species context only. In other words, though "ecosystem considerations" are taken into account by reducing the krill harvest level from that which might otherwise have been chosen, the basis for the associated quantitative choice of this reduction is near arbitrary. Furthermore, this choice was made in the absence of any industrial pressure for higher krill harvest (existing harvests remain considerably below the precautionary catch limits set) – had such pressure been present, would a different trade-off choice have resulted?

[30] The reasons why these biological interaction analyses yield such "tentative" results are essentially two-fold. First, for many cases where the biological interactions involve predation, available dietary data are very limited. Second, even when a reasonable quantity of such data are available, they still are scarcely adequate to distinguish between alternative models of predator feeding behaviour in circumstances where these models can have very different implications for the outcomes of different management actions. What we need to know, but what current data are generally hardly able to tell us, is how do the predators change their diet composition and overall consumption in response to changes in the abundances of their various prey species. The situation

gets even more complicated when biological interactions are of the competitive type, because it is not at all easy to measure the extent of such competition.

THE BOTTOM LINE

[31] What are the immediate prospects for extending the OMP approach to take ecosystem considerations into account?

[32] As far as operational interactions, essentially by-catches in mixed species fisheries, are concerned, we see no reason why analyses with the potential to impact management decisions could not be usefully pursued immediately. Such approaches are already being applied in practice.

[33] Similar application of analyses that attempt to take account of biological interactions would still seem some time away. One requires the ability to confidentially predict how changing the management regulations (such as the TACs) for one species will affect likely catches or abundances of others, and the information needed to refine feeding and competition models sufficiently towards this end will probably still require considerable time to obtain.

[34] Nevertheless, one might anticipate that such predictive ability might come sooner for top predators than for species intermediate in the food chain, because of the lesser number of linkages that apply to the former (as shown, for example, by the relatively greater confidence in predictions for cod compared to those for herring in the Barents Sea analysis of Schweder *et al.*, 1998).

[35] Furthermore, until such time as species interaction models can be specified with more confidence, an interim approach adopted in simulations conducted by the IWC scientific Committee in developing its RMP merits attention (IWC, 1989). This is to mimic to effect of species interactions (essentially arising from changes in the abundances of other species) in the singles-species models used for the simulations by considering scenarios where the values of parameters (such as growth rate and carrying capacity) of those models change over time. While this, of course, cannot provide any information as to potential trade-offs in catches from interacting species, it does help to ensure that single-species management approaches are adequately self-corrective in the presence of changes to the environment in which those species exist.

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