International Council for the Exploration of the Seas ICES C.M. 1999/Z:07
Ecosystem Management – Can We Make It Operational

HOW COMPLEX SHOULD OPERATIONAL ECOSYSTEM OBJECTIVES BE?

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ABSTRACT

A necessary (but not sufficient) component of successful resource management is clear, operational objectives. It has proven difficult to specify operational objectives even for comparatively simple single-species fisheries – at least a complementary set of compatible objectives for the biological, economic, and social aspects of fisheries. This talk will consider the problem of what is the lowest level of complexity necessary if operational objectives are to address more than the status of the a single target species.

I review part of the Report of the last meeting of the Working Group on Ecosystem Effects of Fishing, summarizing both the conclusions of the Working Group and the lines of reasoning which led to them. WGECO concluded that objectives were needed in addition to those assuring conservation of the target species of fisheries. However, it also concluded that based on present knowledge the additional objectives required for conservation of "the ecosystem" were simply a longer list of individually single-species objectives. At the time of that meeting, it had not been possible to demonstrate the need for additional objectives based on emergent properties of the community or ecosystem.

This final conclusion was offered as a tentative one, possibly determined largely by limitations on the available ecosystem models and the data to test them. The majority of this paper will revisit this conclusion, and consider the possible role of ecosystem-level objectives in conservation and management of living marine resources. I will argue that the need for such "ecosystem" objectives remains unproven, and the basis for setting such objectives in an operational framework is far more problematic than the basis for setting current objectives. The serious problems are reinforced by results of an review of metrics of ecosystem status, which was prepared for the Symposium on Ecosystem Effects of Fishing. Major results of that review are also summarized.

Evaluating compliance with complex ecosystem objectives will also be far more difficult than the tasks presently proving a challenge to the fisheries science and management communities. Finally, I will argue that developing and implementing management plans necessary to increase the probability of achieving objectives based on emergent ecosystem properties require a belief in our ability to engineer living systems into

selected configurations and keep them there. Such a belief has little foundation in either ecological theory or historic system dynamics.

INTRODUCTION

The case for the importance of explicit management objectives in operational fisheries management has already been made (Gulland 1977, Smith et al. 1993, FAO 1996, Pitcher et al. 1998). Not all proposed objectives are operationally useful, however. General objectives about abstract attributes of species or ecosystems can serve many useful social functions, such as building a sense of camaraderie, public support, and shared vision for conservation. However, the task of providing scientific advice to managers and decision makers requires that the advisory community identify specific management actions needed for achieving the objectives, or evaluating the compatibility of each of several alternative actions with the overall biological, social and economic objectives. In these advisory contexts, objectives must refer to attributes of stocks or species which can be quantified, and whose likelihoods can be estimated (Rosenburg and Restrepo 1994, Kirkwood 1998, de la Mare 1998). These quantifications and estimations may have substantial uncertainty (Hilborn et al. 1993, Caddy and Mahon 1996), but tools to address that uncertainty are available (Evans and Rice 1988, Charles 1992, Frederick and Peterman 1993). In this context, operational management objectives are closely tied to biological reference points. As is clear in recent ICES advice (ICES 1999), operational single-species management objectives are readily translatable into quantitative reference points. Vague, general objectives leave science advisors having to use their own judgment in guessing what quantitative population properties are necessary and sufficient to have "achieved" the objectives.

The generalizations above are made for advice in a single-species management context. However, there are no reasons to think that objectives would be any less important in ecosystem management contexts, nor that there would be less need to quantify the state of the ecosystem relative to the objectives or to estimate the consequences of management actions relative to attainment of the objectives. Hence, in truly operational ecosystem management, explicit properties of operational objectives would be retained in the advisory context. This would include objectives that are themselves quantitative, and which would be directly translatable into reference points that would be consistently quantifiable and have estimable uncertainty. It is also desirable to have some biological (and logical) linkage from operational objectives to target and limit reference points, whether the reference points were in exactly the same biological currency as the objective or in some other property of the stock, community, or ecosystem.

The reasoning associated with having clear linkages between objectives and reference points, when they are expressed in very different biological currencies, is not complex, although it is often confusing. Consider, for example, an operational single-species biological objective that the probability of a recruitment poorer than x-million fish at age 3 not fall below y, where x and y are chosen after thorough analyses of historic data and simulation modeling, to ensure conservation of the resource and reasonably high yield.

(A biological objective to "keep recruitment as high as possible – or practical" is not an operational objective, although all would agree it is a laudable goal.) The objective technically meets the criteria that one can quantify year-class size (with associated uncertainty) and estimate the probability of the year-class exceeding some value, given appropriate co-variates. Nonetheless, the core biological objective might prompt adoption of ancillary objectives, in different ecological currencies. The ancillary objectives might address size and age composition of the spawning biomass and aspects of habitat quality, because there was scientific evidence that probabilities of good year-classes were higher (the original objective) when experienced spawners were abundant and habitats were unpolluted. In the context of ecosystem objectives, it is also probable that ancillary objectives in a variety of currencies will proliferate from initial operational ones, as knowledge expands about what perturbations are incompatible with larger goals.

MULTISPECIES AND COMMUNITY INDICES AS REFERENCE POINTS

The first place one might look for operational ecosystem objectives is among the metrics of community and systems ecology. If our sole criterion for choosing operational objectives was that they were attributes which could be quantified or estimated from field data, then the only problem with identifying ecosystem or community objectives would be choosing which ecosystem metric(s) to use. Large numbers of metrics of ecosystem or community status have been proposed. However, a recent comprehensive review of metrics of ecosystem status, prepared for the ICES/SCOR Symposium on Ecosystem Effects of Fishing, concluded that most of them presented serious problems when applied to systems exploited by fisheries (Rice, in press).

Single value indices of diversity, similarity, etc have a variety of problems. For example, they may be indeterminate - many different communities produce the same index value, so maintaining a specific index value may not be conserving valued ecosystem components. They may be insensitive - over wide range of exploitation rates diversity indices emphasizing richness may show little change until species begin to be lost from the community, at which point conservation has already failed. They may be sensitive to the wrong things - recruitment of a strong year-class to one stock may depress diversity indices emphasising evenness, and fishing out abundant species will elevate such indices.

Despite these problems the number of indices proposed for evaluating ecosystem or community status continues to proliferate, and methods have been proposed for estimating their uncertainty and using the uncertainty in statistical contrasts (Rice, in press). Perhaps their most serious drawback for use as quantitative ecosystem objectives is that they often cannot be interpreted without falling back on the constituent data themselves. For example, although impacts of environmental stress from toxicants, thermal pollution, etc., are often evaluated by the degree to which diversity index values of a site have declined (ex. Hosmani 1987, McIntyre 1993, Srivastava et al. 1993), modest over-fishing can make diversity indices increase through affecting evenness much more than richness. These antagonistic effects of different ecosystem stressors would make interpreting achievement – or failure to achieve – a diversity objective problematic to interpret, even if its probability could be estimated.

Multivariate community representations, commonly through ordinations, can avoid some of these problems, but ordination results are far too complex to be taken as an empirical biological objective. It is possible that specific attributes of a community ordination could be adopted as an operational objective; for example, that the first principal component or correspondence axis should explain at least z percent of the total variance in the community abundance matrix. Certainly communities have been contrasted with regard to how much variance is captured by the first few ordination axes, and major perturbations in communities have been detected that way (Gomes 1993). Nonetheless, it is a very different use to specify that as an objective community ordinations should taken on particular configurations. Theory would have to be pushed much further to know what configurations should be specified. Management actions needed to keep an ordination in a particular configuration are also unlikely to be obvious, just because ordination structure reflects the community matrix as a whole (Gauch 1982, Jongman et al. 1987). Management actions which have even moderate consequences on several species may reconfigure the entire ordination solution, as may actions which have a large effect on even one species.

The review cited earlier concluded that metrics of community or ecosystem status which summarize data sets may have more promise as indicators of impacts of fisheries on ecosystems. These include ABC- and k-dominance curves (Beukema et al. 1988, Clarke 1990) and slopes of biomass or abundance size spectra (Rice and Gislason 1996, Gislason and Rice 1998), which are treated in another paper in this session. Part of the reason why these summary metrics were considered more useful than alternative metrics is that substantial progress has been made on developing theory to link analytically exploitation to the metrics (Gislason and Rice 1998, Gislason and Lassen 1998). For such metrics to be used as operational ecosystem objectives, however, it is also necessary to identify values of the metrics which can be used are ecological benchmarks and management reference points. Theory has not progressed to the point where such designations are possible.

The same problem is encountered when one considers using model-based metrics of ecosystem status as operational ecosystem objectives. Notwithstanding the need to accept quite strong assumptions about ecological processes in order to accept model-based metrics (Rice, in press), one must also have an external biological justification for managing resource uses to maintain the metric-based value within a specified range. Models, by their nature, can require that input parameters be kept within a specific range, because outside that range model behaviour is considered pathological. Similarly, output parameters of the model also may remain within a specific range when the model is performing "satisfactorily". Such performance attributes of models invite users to define positions within those ranges of "acceptable" input and output parameters as reference points. However, very often parameterization data are very limited, so boundaries of "acceptable" performance have to be determined by exploring parameter space in simulation mode. Following such an approach in setting target and limit reference points is not just accepting the validity of the ecological assumptions underlying the model and the model's representation of them, but is also accepting the quantitative performance of the model outside the range of parameterization data.

I am not attempting to argue that it is impossible to use ecosystem and community metrics and models directly as operational management objectives. I am arguing that the task will not be straightforward. Some types of community metrics have been used successfully as indicators of community perturbations due to pollution (summary in Spellerberg 1991) so it could be argued that the success should transfer directly to other ecosystem stresses, in particular exploitation through fisheries. Fisheries differ in some important ways from toxic chemicals, though, although both can cause significant mortality of species exposed to the stressor.

Itemizing the differences between effects of fishing and effects of pollutants on ecosystems would be another talk, but for these purposes two are important. First, fisheries management actions and fishers' strategies both should make the rate of mortality due to fishing sensitive to abundance of the stocks being harvested, at least at low abundances; managers try to reduce exploitation of depressed stocks and fishers quit or switch target species when catch rates are poor enough. Contrastingly, mortality rate due to pollutants is unlikely to be similarly sensitive to abundance. Second, there are often a small number of taxa which thrive under polluted conditions (see discussion in Warwick and Clarke 1995) while populations of most other taxa decline or are extirpated. Constrastingly, there are usually no characteristic taxa which only thrive under over-exploited conditions, although when whole trophic levels are depleted there may be cascading effects (Carpenter et al. 1985). Both of those differences mean that index values may change in complex ways as communities are perturbed by fishing, either by itself or in conjunction with other environmental stressors.

This complexity would carry over into using any of the community or assemblage metrics as operational ecosystem management objectives. One would have to do a great deal of comparative analyses among ecosystems to develop guidelines for what values of which metrics should be management reference points. With data sets on nearly every system incomplete in very different ways (ICES 1998; Sherman et al. in press) there are no assurances that scientifically sound ecosystem reference points would emerge from such comparative analyses.

Theoretical guidance would be invaluable in selecting reference points of course. However, precedents suggest development of the theory linking measurable features of resource exploitation to metrics of community and ecosystem structure and function will not be easy, although it may be possible. Progress from single species fish population dynamics modeling to multi-species assessment models has been possible (Gislason and Helgason 1985, Daan and Sissenwine 1991, Magnusson 1995) but those models work in the currencies of the same populations, the same population parameters, and for the same (exploited) part of the system. Taking theory to the step of linking dynamics of exploited – or secondarily impacted - populations to community and ecosystem theory is a much harder challenge. If we look to the development of theory within the discipline of community ecology, it has proven very difficult to link descriptive community properties such as diversity and connectance to dynamic community properties such as stability or invasability (Pimm 1991), despite decades of work by theorists. As noted earlier, theory linking resource exploitation to community metrics is in its infancy, and only for a few metrics, such as the slope of size spectra. For most community and ecosystem metrics researchers are still struggling with conception of the linkage, or still seeking for how to form the union directly at all.

Perhaps we are setting too high a goal, and it is at least premature, if not fundamentally in error, to consider setting specific values of an ecosystem metric as a reference point, or using particular metrics as operational objectives. It may not be necessary manage marine resource usage in ways that keep Hill's N1 no lower than 2.7, say; the first three principle components explaining between 65 and 80% of the variance in the community abundance matrix; or a particular suite of species co-located on an MDS axis. It may be sufficient to simply quantify the present value of a metric, and set as a quantitative objective not allowing that metric value to change by more than some set percentage. This strategy is still not simple, and may not achieve general conservation objectives. Much data analyses and theory development is necessary to determine whether that tolerable percentage change should be small (say 5-10%) or should be "double-or-halve". Moreover, because of the differential sensitivities of indices noted earlier, tolerable changes may be very different for indices which emphasise richness and those which emphasise evenness. Also, some very large changes in index values may be due to things managers would not want to reduce in likelihood, such the recruitment of a strong year-class to a herring stock. Finally, perhaps the most serious problem with using some fixed value of a community metric as an operational ecosystem reference point, it that it ignores the intrinsic dynamic nature of the community (Hallowed, et al. in press). Then fisheries or ecosystem managers who fail to prevent change from happening will be judged to have failed, whereas in reality, perhaps the most artificial perturbation that could be inflicted on any ecosystem would be to freeze it in one state over time.

ALTERNATIVE OPERATIONAL OBJECTIVES

At its 1997 meeting the ICES Working Group on Ecosystem Effects of Fishing (WGECO) considered the question of ecosystem management objectives (ICES 1999). WGECO concluded that even if fisheries were suddenly managed in ways which ensured, with high likelihood, compliance with sound target and limit reference points of fishing mortality and biomass of target species, ecosystem conservation could not assured. The concern was not that other stressors such as toxic chemicals and climate change could lead to excessive mortality or depressed biomass of target species, and place some of the fished stocks at risk despite the management reference points. Such concerns are supposed to be addressed fully in estimating the risk of violating reference points. Rather, even if all the target stocks were being conserved, the fishery could be having impacts on other ecosystem components which would be incompatible with conservation objectives. WGECO went on to identify four classes of such ecosystem components:

- Non-target species killed directly as bycatch or by mechanical damage *in situ.*

- Ecologically dependent species, whose food supplies may be reduced enough by a fishery to be inadequate for reproduction of the dependent species, even if conservation of the target stock (the food supply) was not jeopardized.
- Genetic diversity of the target species, which could be reduced by disproportionate harvesting of stock components, even if overall the species or total stock unit was not at risk.
- Competitors or prey of species which scavenge fish wastes (discards and offal), where fisheries may produce an abundance of food for scavengers, allowing their populations to build up sufficiently that they do serious ancillary damage to their competitors or other prey taxa.

The WGECO report provides examples and full explanations of each class.

Importantly, each of these additional concerns can be addressed by adding more species, (or in the case of genetic diversity, a feature in addition to biomass and fishing mortality) to the list of species for which there are operational objectives, and corresponding target and limit reference points. WGECO recognized that point, and went further to pose the question that if there were this expanded group of single-species reference points, and fisheries were managed to comply with all of them, would ecosystem conservation be achieved with high probability. The conclusion two years ago was that "We don't know", and one could argue that we still don't. Some of the reasons why WGECO didn't know if ecosystem conservation would be achieved in that hypothetical situation were unanswered questions about complex processes such as nutrient transfer from sediments. However, WGECO also noted that, at the time of their meeting, although several very different modeling approaches were investigated or reviewed, no modeling approach had identified an ecosystem property which would be at risk, if the component stocks or species were individually NOT at risk. Earlier this year, at the Symposium on Ecosystem Effects of Fishing that question was posed again. To this point it has still not been possible to identify attributes of ecosystem models which would be at risk, if the constituent pieces were not at risk. This is a particularly relevant consideration for choosing operational objectives for ecosystem management.

As argued in the previous section, it may be possible to develop reference points for operational management out of complex metrics of ecosystem status, but our discipline is not ready to do so now, and I believe we will not be ready for some time to come. The same arguments apply to developing operational ecosystem objectives based on "emergent properties" of ecosystems. It may be possible to find some higher-order properties of interest, but based on several years of discussion in ICES Working Groups (WGECO and MAWG), we are not there yet. In the general field of community and ecosystem theory, for that matter, there is no consensus on whether "emergent properties" exist, let alone consensus on what they are. These are research lines worth exploring, but they are not solutions to today's problems – or probably next year's problems either.

At the operational scale, I believe the WGECO deliberations do give us an avenue which at least will be useful in the near-term, and may prove sufficient in the long term. The WGECO conclusions have been mis-interpreted or mis-represented as calling for the establishment of target and limit reference points of every species in the ocean, and for evaluation of the status of every species relative to them on a regular basis. Such a charge would be no more operational than trying to evaluate the status of the ecosystem relative to conservation of some abstract property such as "integral pathways". For good reasons ICES Working Groups have not endorsed the concept of globally applicable "indicator species". However for specified threats to specified systems there are many examples of ranking species according to their sensitivities to the threat, and evaluating status of the most vulnerable.

For example the Study Group on Effects of Sandeel Fisheries (ICES 1999b) reviewed substantial information and concluded that kittiwakes (*Rissa tridactyla*) were highly dependent on sandeels during the breeding season, and would be very sensitive to local abundance of sandeel. Monitoring kittiwake breeding success would be an effective surrogate for monitoring sandeel availability to all sandeel predators in the same coastal areas. If actions were taken to ensure conservation of kittiwake populations, through maintaining breeding success at least at the replacement level, it was likely (although not guaranteed) that populations of other dependent predators would also be conserved. Appropriate cautions were stressed about the limitations on the use of such surrogates, for example kittiwake breeding success says little about the possible impacts of sandeel fisheries outside the foraging range of breeding kittiwakes. However, all types of reference points have limits on the degree which site-specific findings can be generalized. Similar approaches could be used to seek comparable operational surrogates in other areas as well.

As another example, when WGECO was collecting information for the first Environmental Quality of the North Sea Report, they were asked to provide some evaluation of the contribution of damage by fishing gear to mortality of the benthos. Evaluating mortality of all the benthos is not a tractable task at present, even if it is of great interest to all marine scientists and managers. However, based on life-history characteristics, distribution in substrates, and other properties which were clearly identified, the WG as able to rank a large number of benthic species according to their potential vulnerability. (I do not mean to argue that WGECO was the first to do this. There was a literature on the general topic already available, of which WGECO made use.). The contribution of mortality due to damage by fishing gear was evaluated for several species whose abundances had been quantified, in order to test the validity of the ranking. Then the most vulnerable species of those whose abundances could be estimated with reasonable confidence were selected for more intensive study and possibly, eventual monitoring (ICES 1995).

This is not the only case where species have been ranked by their perceived or measured vulnerability to population declines, range reductions, or other indicators of failure to achieve conservation objectives. Prompted by changes to criteria used and expansion of species considered by groups such as the IUCN, which focus on protection of threatened or endangered species, biological (and environmental) characteristics which are related to vulnerability have received a great deal of attention (Powles et al in press, Casey and Myers 1998). The specific features associated with high vulnerability are not crucial to our present consideration of operational objectives for ecosystem management. What does matter is that there appears to be consensus that such features do exist, and at least some of them, for example life history attributes such as fecundity and age of maturity, are biological characteristics which can be measured for many species. If conservation is being achieved for the most vulnerable species, it is likely to be being achieved for less vulnerable species as well, at least in the context to some threat of particular interest to management.

The core message is that we do not need complex ecosystem objectives in order to implement ecosystem management. Without question we need *more* objectives than we did when fisheries were being managed as if fishing mortality was the only factor of importance to dynamics of the target species, and fisheries did nothing but create employment and wealth by harvesting only target species. However, those additional objectives do not have to be hugely different in kind from the objectives and reference points we have been developing in recent years. The necessary tasks can be listed readily:

- We need to identify the major threats to conservation of ecosystem components. We need to do this task whatever objectives and reference points one chooses to adopt.
- We need to evaluate stocks or taxa with regard to their vulnerability to the major threats. Using complex ecosystem metrics might mean one would not have to undertake such a population-scale evaluation, but I question if this is a benefit or a drawback. Surely both managers and conservationists want to know what ecosystem components are at greatest risk from the various major threats.
- We need to identify population sizes, mortality rates, or other features (genetic diversity) of the vulnerable stocks, species, or ecosystem components which are associated with a high likelihood of population viability (target reference points) and unacceptable likelihoods of continued decline (limit reference points). This is proving tractable, if highly contentious in detail, for target fish stocks (ICES 1998). As principles and practices emerge for stocks which are targets of fisheries, however, there are no conceptual reasons why the same principles and practice should not apply to other populations as well.
- We need to monitor the statuses of at least a rationally chosen sub-sample of the vulnerable populations, and evaluate their status relative to the reference points. It could be argued that it is too costly, or beyond the capabilities of current technology to evaluate the status of non-target species with sufficient accuracy and precision to evaluate their statuses relative to reference points. This may be true, but if it is, it is illogical to argue that we do better with more abstract properties of ecosystems, such as their richness and diversity, properties of their ordinations, or properties of models of them. Any textbook on using these multivariate and modeling methods includes admonitions that one must start with reliably representative sampling of the system whose properties are being summarized with the index or ordination or are being modeled. Many texts even include analyses which are recommended to be performed to ensure that the sampling is adequate before community metrics are calculated, or to investigate whether the "best" model parameterization is really any good at all. Using an aggregate metric to cover up inadequate sampling is certainly possible, but could be considered scientifically dishonest, and is at best banking the future on the

robustness of the central limit theorem to overcome poor sampling tools or insufficient sampling effort.

- We need to have management take appropriate action when our evaluations suggest conservation measures are needed.

This latter point has not been addressed in critiquing alternative ecosystem management objectives. In terms of identifying necessary management actions, and motivating their implementation, I also think that we come out ahead using simple objectives, and simple reference points. If our monitoring and evaluation shows that a few populations, known *a priori* to be particularly vulnerable to a specific threat, show alarming trends, we are directed to take action first to reduce that threat. We could be wrong in diagnosing the problem, but at least in the short term we are still likely to be aiding the conservation of the populations at risk, and the larger ecosystem. It is unclear, at least to me, how one would do better at diagnosing the problem in need of management action if one were monitoring a diversity index or an ordination structure, and judging conservation of a very complex property of the ecosystem. However, that is an opinion without systematic testing, and I could be shown to be wrong. The necessary work to systematically test whether trends in community indices are more effective guides to remedial action than trends in stocks selected because of their vulnerability to particular threats and their reliability of measurement would be very interesting science. It might be complex and time-consuming, however.

There is another consideration in ensuring that the necessary management actions are taken when our ecosystem indicators say actions are needed. That is the strength of the indicators to motivate action by resource managers and compliance by resource users. Again, in my personal experience I would prefer to go to a manager or a fisherman with graphs of trends in perhaps population sizes of skates and rays or breeding success in a kittiwake population, and show how things were going bad, rather than use trends in a diversity index, an ordination score, or an abstract model property, to achieve the same goal. There is no guarantee that they would be willing to take management action or change fishing practices to reverse a trend in a skate population, but if they were unwilling, I judge it less likely still that they would take action to change the trend in a more abstract index. Again, this is a personal opinion, and highlights again the need for some good research on what sorts of science advice is most useful for managers.

In summary my view of operational ecosystem management objectives, for the present time, is that they should look much like current species management objectives. There should just be more of them, covering more components of the ecosystem. They should be simple, but wisely chosen, using what is known of major potential threats to ecosystem, species vulnerabilities, and what we believe we can monitor with existing tools. We should generally not look to abstract or highly integrative indices of ecosystem status as the basis of our ecosystem objectives, nor of our tools for trying to achieve those objectives. Such metrics have both analytical problems and interpretational problems. Furthermore, we presently lack the knowledge both to diagnose from their behaviour the very specific actions which need to taken (i.e., a reduction of fishing effort by x percent over the next y years) to enhance conservation of the ecosystem, and to use them to convince those whose behaviours must change that their sacrifices are necessary.

These problematic features are not fundamental attributes of the more complex metrics. They are all tractable to research, analysis, and modeling. But they are all hard problems; work on community metrics has been on-going for several decades, with no resolution to some of the analytical and interpretational problems in sight. We should not expect solutions to the problems of implementation to be easier to find. However, we should be greatly encouraged. With what we know now, we can be sure we can do a lot towards conservation of the ecosystem if we just work for conservation of its component pieces. Perhaps someday we may learn what the more complex ecosystem metrics truly are telling us (and not telling us), and how to use them effectively in advice to managers. At that time I think we may conclude that we have done all that was needed, in just conserving all the individual pieces at ecologically effective population sizes.

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