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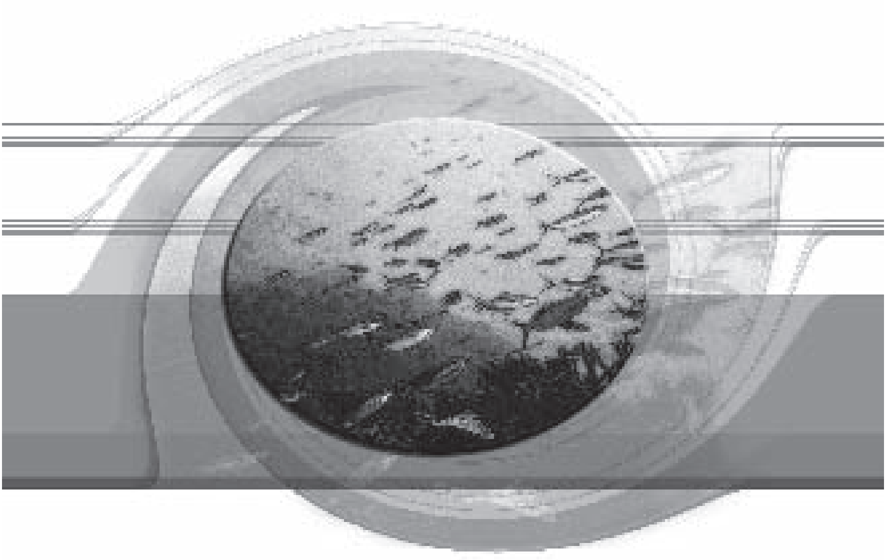
FAO TECHNICAL GUIDELINES FOR RESPONSIBLE FISHERIES

4

Suppl. 2 Add.1

FISHERIES MANAGEMENT

2. The ecosystem approach to fisheries
2.1 Best practices in ecosystem modelling for informing an ecosystem approach to fisheries



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FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
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ABSTRACT

Awareness of the limitations of a single-species approach to fisheries management has led to global acceptance of the need to adopt a wider ecosystem approach to fisheries (EAF) assessment and management. Applying EAF in management requires the application of scientific methods and tools that also go beyond the single-species approaches which used to be, to a large extent, the exclusive sources of scientific advice. Managers and decision-makers must now find management solutions that take into account the wider range of societal objectives that must be explicitly considered under EAF and the interactions in the ecosystem. Ecosystem models, i.e. models that represent a wider range of technological and ecological processes affecting the species in the ecosystem (including multispecies and whole ecosystem models), are potentially important tools for providing this wider scientific information.

There are many different types of ecosystem models and they can vary enormously in terms of complexity. They can be used in different ways, ranging from contributing to conceptual understanding, providing information for strategic decisions through to making tactical decisions, although they are rarely used as yet for the last purpose. These guidelines were developed by a group of leading practioners in aquatic ecosystem modelling as a tool for provision of management advice. They are intended to assist users in the construction and application of ecosystem models for EAF. The guidelines address all steps of the modelling process, encompassing scoping and specifying the model, implementation, evaluation and advice on how to present and use the outputs. The overall goal of the guidelines is to assist in ensuring that the best possible information and advice is generated from ecosystem models and used wisely in management.

The considerable uncertainties in the predictions provided by ecosystem/multispecies models notwithstanding, decisions have to be made and actions implemented to ensure sustainable and optimal utilization of marine living resources. These decisions must be informed by the best available scientific advice and, in the context of EAF, this scientific advice must include ecosystem considerations. Ecosystem models, adhering as far as possible to the best practices described here, will frequently be the best sources of such information and can lead to advice that rests on explicit and principled arguments. In their absence, managers and decision-makers will have no choice but to fall back on their own mental models which may frequently be subjective, untested and incomplete, a situation which clearly needs to be avoided.

Ecosystem models are not at the stage where a single such model could be selected as a "management" model and reliably used at the tactical level to provide management recommendations in a particular case. However, the use for this purpose of simple models with an ecosystem foundation could become more widespread in the near future. Such a foundation would be provided by evaluating these simpler "management" models using Management Strategy Evaluation (MSE), where the operating models reflecting alternative possible underlying dynamics that are used in this evaluation process would include a range of ecosystem models.

CONTENTS

BACKGROUND

1. From ancient times, fishing has been a major source of food for humanity and a provider of employment and economic benefits to those engaged in this activity. However, with increased knowledge and the dynamic development of fisheries, it was realized that living aquatic resources, although renewable, are not infinite and need to be properly managed, if their contribution to the nutritional, economic and social well-being of the growing world's population was to be sustained.
2. The adoption in 1982 of the United Nations Convention on the Law of the Sea provided a new framework for the better management of marine resources. The new legal regime of the oceans gave coastal States rights and responsibilities for the management and use of fishery resources within the areas of their national jurisdiction, which embrace some 90 percent of the world's marine fisheries.
3. In recent years, world fisheries have become a dynamically developing sector of the food industry, and many States have striven to take advantage of their new opportunities by investing in modern fishing fleets and processing factories in response to growing international demand for fish and fishery products. It became clear, however, that many fisheries resources could not sustain an often uncontrolled increase of exploitation.
4. Clear signs of overexploitation of important fish stocks, modifications of ecosystems, significant economic losses, and international conflicts on management and fish trade threatened the long-term sustainability of fisheries and the contribution of fisheries to food supply. Therefore, the nineteenth session of the FAO Committee on Fisheries (COFI), held in March 1991, recommended that new approaches to fisheries management embracing conservation and environmental, as well as social and economic, considerations were urgently needed. FAO was asked to develop the concept of responsible fisheries and elaborate a Code of Conduct to foster its application.
5. Subsequently, the Government of Mexico, in collaboration with FAO, organized an International Conference on Responsible Fishing in Cancún in May 1992. The Declaration of Cancún endorsed at that Conference was brought to the attention of the UNCED Summit in Rio de Janeiro, Brazil, in June 1992, which supported the preparation of a Code of Conduct for Responsible Fisheries. The FAO Technical Consultation on High Seas Fishing, held in September 1992, further

recommended the elaboration of a Code to address the issues regarding high seas fisheries.

6. The one hundred and second session of the FAO Council, held in November 1992, discussed the elaboration of the Code, recommending that priority be given to high seas issues and requested that proposals for the Code be presented to the 1993 session of the Committee on Fisheries.
7. The twentieth session of COFI, held in March 1993, examined in general the proposed framework and content for such a Code, including the elaboration of guidelines, and endorsed a time frame for the further elaboration of the Code. It also requested FAO to prepare, on a "fast track" basis, as part of the Code, proposals to prevent reflagging of fishing vessels which affect conservation and management measures on the high seas. This resulted in the FAO Conference, at its twenty-seventh session in November 1993, adopting the Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas, which, according to FAO Conference Resolution 15/93, forms an integral part of the Code.
8. The Code was formulated so as to be interpreted and applied in conformity with the relevant rules of international law, as reflected in the United Nations Convention on the Law of the Sea, 1982, as well as with the Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, 1995, and in the light of, *inter alia*, the 1992 Declaration of Cancún and the 1992 Rio Declaration on Environment and Development, in particular Chapter 17 of Agenda 21.
9. The development of the Code was carried out by FAO in consultation and collaboration with relevant United Nations Agencies and other international organizations, including non-governmental organizations.
10. The Code of Conduct consists of five introductory articles: Nature and Scope; Objectives; Relationship with Other International Instruments; Implementation, Monitoring and Updating and Special Requirements of Developing Countries. These introductory articles are followed by an article on General Principles, which precedes the six thematic

articles on Fisheries Management, Fishing Operations, Aquaculture Development, Integration of Fisheries into Coastal Area Management, Post-Harvest Practices and Trade, and Fisheries Research. As already mentioned, the Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas forms an integral part of the Code.

11. The Code is voluntary. However, certain parts of it are based on relevant rules of international law, as reflected in the United Nations Convention on the Law of the Sea of 10 December 1982. The Code also contains provisions that may be or have already been given binding effect by means of other obligatory legal instruments amongst the Parties, such as the Agreement to Promote Compliance with Conservation and Management Measures by Fishing Vessels on the High Seas, 1993.
12. The twenty-eighth session of the Conference in Resolution 4/95 adopted the Code of Conduct for Responsible Fisheries on 31 October 1995. The same Resolution requested FAO *inter alia* to elaborate appropriate technical guidelines in support of the implementation of the Code in collaboration with members and interested relevant organizations.

1. INTRODUCTION

1.1 THE ECOSYSTEM APPROACH TO FISHERIES

Individual fisheries normally target from one to several species depending on the fishing methods and the ecological community being fished. As a result, until towards the end of the last century, fisheries management tended to focus only on regulating fishing activities in order to achieve sustainable utilization of those target species. However, fishing usually affects other components of the ecosystem in which it occurs. For example, there is often bycatch of non-targeted species, physical damage to habitats, food-chain effects and others, and in recent years there has been a growing realization of (FAO 2003):

- the importance of interactions among fishery resources, and between fishery resources and the ecosystems within which they exist;
- the wide range of goods and services provided by fishery resources and marine ecosystems, and the need to sustain those;
- the poor performance of fisheries management in many cases, leading to the poor state of many the world's fisheries; and
- increased knowledge of the functional value of ecosystems to humans, and awareness of the many uncertainties about ecosystem function and dynamics.

This awareness has led to recognition of the need for fisheries management to consider the broader impact of fisheries on the ecosystem as a whole and also the impact of the ecosystem, and other users of the ecosystem, on fisheries. The overall goal must be the sustainable use of the whole system, not just of the targeted species. Achieving this goal requires the implementation of an ecosystem approach to fisheries (EAF) which can be defined as (FAO 2003):

> *"....an ecosystem approach to fisheries (EAF) strives to balance diverse societal objectives, by taking account of the knowledge and uncertainties of biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries."*

Collectively, the nations of the world, through the Plan of Implementation of the World Summit on Sustainable Development (Johannesburg, 2002) have committed to "Encourage the application by 2010 of the ecosystem approach, noting the Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem and decision V/6 of the Conference of Parties to the Convention on Biological Diversity".

1.2 WHERE DO ECOSYSTEM MODELS FIT INTO MANAGEMENT ADVICE AND WHAT ARE THE BENEFITS?

Single-species stock assessment methods were developed as a tool to predict how a fish stock would respond over time to one or more management measures (e.g. an annual TAC, changing the mesh size) and what effect this would have on the status of the stock and the yield to the fishery. Stock assessment models can feed into the management process during the scoping phase of their development, and should inform the process of setting objectives and the formulation of "rules" or appropriate management measures. Single-species assessment methods remain an important tool for implementation of EAF but, with the need to "balance diverse societal objectives" and to take into account the interactions in the ecosystem, fisheries managers and policy makers now also need scientific information that allows them to consider the impacts of the fishery on other ecosystem components and to take into account changes in the ecosystem other than those caused by fishing, whether natural or anthropogenic in origin, that may be impacting the fishery. Ecosystem models, i.e. models that represent a wider range of technological and ecological processes affecting the species in the ecosystem (including multispecies and whole ecosystem models), are potentially important tools for providing this wider scientific information.

The need to consider multiple-users of the ecosystem means that a wide range of objectives, frequently ignored in the past, must be considered in selecting optimal fisheries management measures and strategies. This inevitably highlights a number, sometimes a large number, of conflicts between different stakeholder groups that need to be reconciled and resolved if management is to be successful and the overall societal goals achieved (Table 1). Such conflicts have always been there but in the past were largely not directly addressed by fisheries management or management of the other relevant sectors. Specifically designed ecosystem models which incorporate the relevant variables and processes (which can include biological, ecological, social and economic factors) can be used to simulate the implications and trade offs of alternative management actions and trade-offs for the different, conflicting stakeholders or objectives. In this way, they can provide valuable information to managers in the search for optimal management measures and approaches.

TABLE 1

Hypothetical multiple objectives for a fishery and potential consistency and conflicts between them. "+" indicates that management measures aimed at achieving the objective in that row will probably also favour achieving the objective shown in the column while "–" indicates that such management measures will probably hinder achieving the objective shown in the column

Objective	1	2	3	4
Reduce effort to ensure that F does not exceed target F (which should be below FMSY)	0	+	–	+ (because CPUE should increase)
Reduce impacts of fishery on species of conservation concern (e.g. turtles, sharks)		0	–	–
Maintain employment opportunities in the fishery			0	–
Maximize economic efficiency of the fishery to ensure competitive access to markets				0

1.3 EXAMPLES OF USES FOR MANAGEMENT ADVICE

Ecosystem models can be used for a variety of purposes which can be broadly classified as: improving conceptual understanding of a system; providing information and advice to inform strategic planning and decision-making; and providing information and advice to facilitate tactical planning and decision-making. In reality, there is no clear distinction between each of these three categories and they can be seen as a continuum running from conceptual understanding at one extreme to tactical support at the other. However, for the purposes of this report, the three broad areas within the continuum are loosely defined as follows.

- Conceptual understanding: a broad understanding of the structure, functioning and interactions of the ecosystem, or sub-system, under consideration. This understanding may not be used explicitly in decision-making or scientific advice but forms the underlying context for any detailed management planning and decision-making. An example of such an application is the experimental approach followed to distinguish different hypotheses to explain multispecies trends on the NW Australia shelf (Sainsbury, 1991; Sainsbury *et al*., 1997). Another is the krill surplus hypothesis of Laws (1977) to provide a qualitative explanation of increases of minke whales and crabeater seals in the Antarctic as competitive release in response to an increase of krill following the severe depletion of blue, fin and other large baleen whales through overharvesting.

- Strategic decisions are linked to policy goals and are generally long-range, broadly-based and inherently adaptable. An example of a strategic decision, based on ecosystem considerations and advice from models, can be found in the Walleye Pollock assessment and quota-setting process in the Gulf of Alaska in 2005 and 2006. In this case, a long-term decline in productivity was linked to the rise of a predator (arrow tooth flounder) which may have increased natural mortality on prey. This led to the development of a strategic decision to develop management strategy analyses (still in progress) to explore the results of conditioning M used in stock assessment, and therefore future quotas and reference points, on arrow tooth flounder biomass levels.
- A tactical decision is typically aimed at the short-term (e.g. next 3–5 years), linked to an operational objective and in the form of a rigid set of instructions. An example involving a technical (in contrast to an ecological) interaction is provided by the pelagic fishery for sardine and anchovy off South Africa, where the management procedure adopted to provide TAC recommendations for the directed fishery on adult sardine takes quantitative account of the inevitable bycatch of juvenile sardine with the anchovy fishery, so that large directed sardine catches necessitate lesser anchovy catches and vice versa (De Oliveira and Butterworth, 2004)

Sometimes management advice will be based on a combination of the above categories of model uses. The fishery for the Walleye Pollock in the Bering Sea provides an example of such combined use of models. In 2006, Bering Sea Walleye Pollock had experienced 5 years of low recruitment. Ecosystem indices and fitted ecosystem models showed that plankton production had been at unprecedented lows during this time, and potential predatory species had been increasing. Based in part on these modelling results (taken qualitatively), the North Pacific Fishery Management Council took the ad hoc tactical decision to reduce quota to approximately 7 percent below the maximum permitted from the results of the single species assessment model, in order to take a precautionary approach to the spawning stock during this period of climate/food web uncertainty (Dorn *et al.,* 2005; Boldt, 2006; North Pacific Fisheries Management Council SSC Minutes, December 2006).

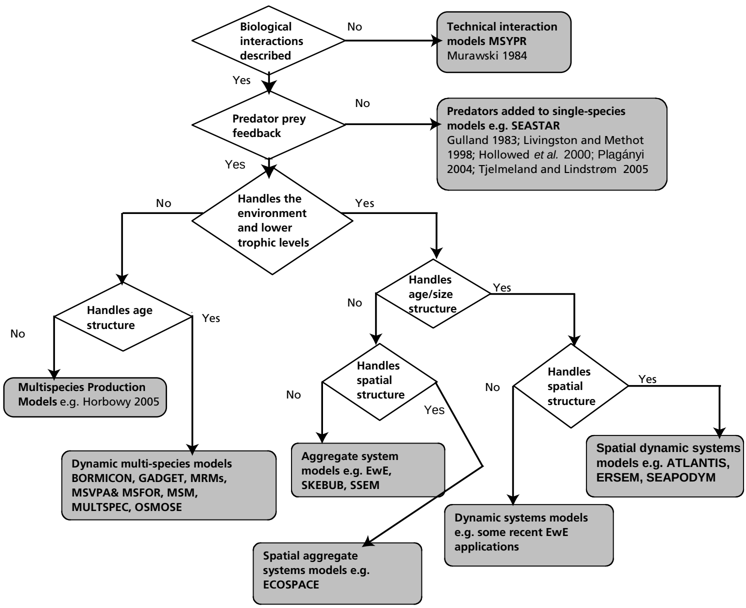
1.4 THE SCOPE OF ECOSYSTEM MODELS

There are many different types of ecosystem models and they can vary enormously in terms of complexity. The simplest models may consider, for example, how to harvest a target species appropriately while simultaneously accounting for the needs of a predator dependent on the target species as prey. More complex models may attempt, for example, to take into account the direct and indirect effects of predation and competition or other, non-trophic impacts, on a target population, the direct impacts of the fishery on the target species, as well as the direct and indirect impacts of the fishery on the rest of the ecosystem. In general, increasingly complex models attempt to increase the ecological realism of the model but this also has a cost as it may also lead to greater scientific uncertainty because of imperfect knowledge of both the functional relationships and the parameters that are incorporated in the model.

The range of different types of ecosystem model currently available can be classified as shown in Figure 1. All these model types are considered to be "ecosystem models" for the purposes of this publication.

Ecosystem models can have an important role to play in Management Strategy Evaluation (MSE) or the analogous Management Procedure (MP) approach (e.g. Butterworth *et al.*, 1997; Smith *et al.,* 1999; Rademeyer, Plaganyi and Butterworth, 2007). MSE or MP frameworks are used to identify and model uncertainties and to balance different resource dynamics representations. As such they provide key examples of formal methods for addressing uncertainty issues. The approach involves an evaluation of the implications of alternative combinations of monitoring data, analytical procedures, and decision rules to provide advice on management measures that are reasonably robust to inherent uncertainties in all inputs and assumptions used. The MSE framework typically involves both harvest rules and "operating models" (also termed "testing models"). Operating models (OMs) simulate alternative plausible scenarios for the "true" dynamics of the resource and generate "data" that are used by the MP modules. They may seek a high degree of realism, and hence may be quite complex. Thus models such as Ecopath with Ecosim (EwE) and ATLANTIS may be used as OMs. Operating models provide the basis for

FIGURE 1
Flowchart (from Plagányi, 2007) summarizing the classification of the various existing ecosystem model types. The flowchart has been modified and updated from that presented in Hollowed *et al.* (2000)



simulation testing to assess how well alternative candidate harvest rules achieve the objectives sought by the management authority.

Ecosystem models are also important for testing potential indicators and identifying reference points. An ecosystem model being used to test a management measure or strategy should allow for simulation of the indicators to be used in management and for their trial application. The role and application of ecosystem indicators is not discussed in these guidelines and the reader is referred to the ICES Journal of Marine Science vol. 62, 2005 for a recent review of this topic.

1.5 ROBUSTNESS AND THE PRECAUTIONARY APPROACH: ADDRESSING UNCERTAINTY IN MANAGEMENT

The complexity of the ecosystem in which fisheries operate means that science cannot possibly hope to deliver on all the information required. Appropriate research to reduce some of the critical uncertainties will be required and should improve understanding in the future, but in the meantime management decisions have to be made on the best information available at the time. It is essential that these management decisions and the resulting actions are robust to uncertainties. Within this context, appropriate application of the precautionary approach is very important in implementation of the ecosystem approach to fisheries and in the use of ecosystem models for informing management. The precautionary approach requires that "where there are threats of serious irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (FAO, 1996). In practice this requires that scientists and managers need to evaluate in a systematic way whether any key uncertainties in their knowledge could lead to a management action not producing the results that were expected from it. If there is an unacceptably high risk of something going wrong, because an assumption used in deciding on the management action is subsequently found to be incorrect, then a different management action, either more conservative or robust to the uncertainty in some other way, should be used instead. Ecosystem models can be used to test the robustness of management actions to such uncertainties, either through the formal testing of an MSE process or, if that is not possible, through using the model for a thorough and rigorous evaluation of the management action and potential problems that could be encountered.

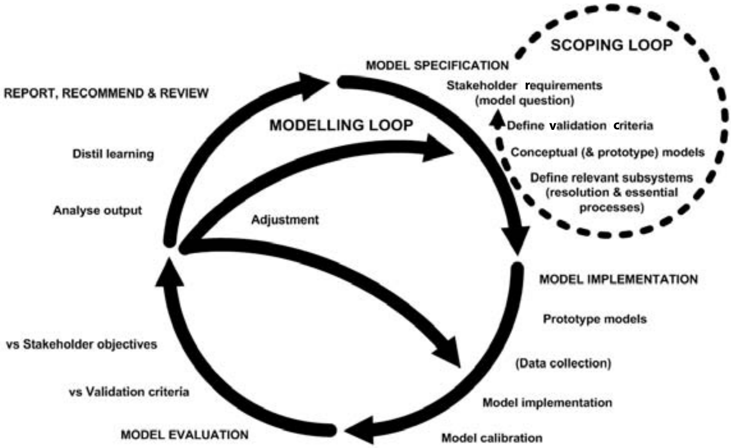
2. MODELLING

2.1 MODELLING APPROACH

When making the transition from models focused on single species to multispecies or whole ecosystem models the very basic steps of best practice model development should be followed (specification, implementation, evaluation, report and review – see Figure 2). With the expansion in model scope and the questions to be addressed, a broader focus is needed through the whole procedure (model development and use), and conceptual and prototype models take on a much greater role. It may seem a natural transition to assume that best practice for ecosystem models is simply single species best practice extended to the entire system, but this is not really the case. While the core principles remain the same, a simple expansion of that form is not typically feasible, as it would overwhelm existing resources and run the risk of omitting extra considerations that present themselves only at the multispecies and ecosystem levels. One aspect of best practice that is common to the two is that processes such as MSE (see section 1.4) are of great value in both settings. Use of those methods does not by definition demand fully quantitative models of large complexity; it is the approach and thought processes involved, rather than the model, which are the key. A brief discussion of the modelling process is presented here, but a more detailed and technical discussion is presented in the Appendix.

Key components of this modelling loop are discussed briefly below or expanded at length in Chapters 4 and 5. Perhaps the key consideration however is that the reason for modelling (the question to be addressed) must be kept in mind at all times. Another key concept is that **there is no one single correct model**; rather there will be a range of models that can address the question and that overlap in resolution or form and complement each other. A tension between prediction and understanding does exist, but experience has shown, given the uncertainty associated with ecosystem-level questions, that the greatest leverage is gained via considering combinations of models (referred to as ensembles) that may be of quite different forms. There is a continuum of model types from qualitative to simple empirical functions through to fully specified models that represent specific processes. These different models may be coupled together to capture different parts of a greater system, but they have a

FIGURE 2
Diagram of the steps used in modelling (modified from Figure 1 in Dambacher et al., 2007). This loop can be used for any model type in any role (understanding, strategic or tactical), but in the context of strategic models the scoping loop can produce a model for understanding, the main loop deals with the strategic model and the end result of the review process may be recommendations on the form of a tactical model



deeper role whereby they can inform each other and resolve different aspects of reality. They can be used as separate stages of a larger strategic implementation, but also be taken pragmatically and used to deliver useful insights under practical constraints, such as time and money available. In many cases conceptual understanding represents an important advance, and in those situations qualitative, statistical and simple quantitative methods can deliver very useful results. These methods can also be valuable for some strategic and tactical modelling questions. Strategic and tactical questions may require more quantitative approaches however.

2.1.1 Model scoping

The first (and most critical) step in model development is model specification. This not only addresses the specific question to be answered and the validation criteria to be used to check if the model is performing well, but also covers all the other steps of the model scoping loop (as this can be an iterative process). Specifically it includes the development of conceptual models which help in turn to identify relevant subsystems, appropriate resolutions and essential processes for in corporation in the final model. Without this step there is the danger of the development of a model that fails to address the purpose intended. It is also a very useful means of ensuring that excessive detail is avoided. This is a very important consideration in itself, as there are often considerable computation, uncertainty and performance issues associated with the inclusion of details beyond those absolutely required to address the specific issue in question. Ecosystem models need not be huge and all encompassing; in fact such large models should be the exception rather than the rule. Models are sufficiently detailed if they capture the critical processes, drivers and resolution of the components under scrutiny.

Conceptual models capture understanding of the system structure, interactions and drivers and are basically descriptive (often box and arrow) models of "how the system works". Development of conceptual models should be conducted in consultation with stakeholders so that their knowledge is appropriately captured. Additional information or hypotheses can be proposed, but these should be presented to stakeholders for comment in an iterative process so that a complete understanding is achieved. The full form of the resultant model need not then be taken further into prototype or final models, but it should be used to define relevant subsystems. This development of conceptual models and the definition of relevant subsystems links directly to two other key aspects of the scoping stage: that it is a very effective way of increasing stakeholder involvement and understanding; and that it is when evaluation criteria (by which model performance will be judged) are articulated. It should be recognized at this time what data are available for model validation, which in combination with conceptual models and the definition of relevant subsystems will guide the potential scope of the model and what data may need to be collected if the model is to be used in more than a "theoretical world" capacity.

The definition of the relevant subsystem and from there the model specification should be achieved following a clear, logical and consistent process. For each dimension or attribute of the model the complexity or added detail being suggested must be evaluated in terms of what contributions it makes to the model and overall analysis. This not only dictates what components are included in the model, but can dictate what type of model is used (e.g. aggregate system models vs. spatial dynamic systems models – see Figure 1) and the data required. There is a potentially long list of model attributes to be considered when deciding on a model specification (see section 4.2 for details and guidance), which can be extended further by the specific question being asked of the model and details highlighted in conceptual models. Nevertheless the following captures the broad steps required in defining a typical multispecies or ecosystem model (more detail on each step can be found in the Appendix):

i. Define the question to be addressed.
ii. List the important potential features and use conceptual models and the following steps to drill down to necessary components for inclusion in the final model.
iii. Scales (and distribution) of each process and component (see section 4.2):
   – Spatial scale
   – Temporal resolution
   – Taxonomic resolution
   – Process resolution
   – Forcing
iv. Fisheries model resolution.

2.1.2 Model validation and performance evaluation

Model validation is the process of checking that the model is useful in that it addresses the problem posed and provides accurate information about the system being modelled. Model validation is different from model verification which relates to checking that the model is correctly programmed. Although traditionally the parameters of ecosystem models have not been estimated using standard statistical methods, nor have these models been subjected to validation, best practice now is to use a structured approach for both estimation of parameters and model validation. Parameter estimation and model validation can, however, be extremely difficult for ecosystem models which have a large number of parameters that must be estimated and many submodels to validate. Given this, there will be substantial uncertainty associated with model outputs, so consideration must be given to both quantifying parameter uncertainty as well as uncertainty about the structure of the model. How parameter estimation and model validation will be achieved, as well as uncertainty quantified, should be identified during model scoping.

Ideally, model validation should be based on using the model to predict data that were not included when the model was designed and its parameters were estimated (e.g. cross validation). However, this is rarely possible in practice because there are generally far fewer data than desirable, so that all of the data are used for parameter estimation. Instead, the predictions of the model should be compared with the data used during parameter estimation and standard regression diagnostics considered (e.g. the residuals should be checked for systematic patterns). Although it may be simpler to validate each submodel of an ecosystem model in turn by comparing its predictions with data, this is inappropriate because the estimates of the parameters of submodels that are not independent may be inconsistent if these submodels are fitted to the same data. In many cases there is qualitative information about the system being modelled and this can be used during model validation.

2.2 TECHNICAL CHALLENGES

There can be considerable computing requirements for some of the moderate to more complex ecosystem models. This is particularly the case if there is high spatial, temporal or taxonomic resolution. While this should not be the only reason for avoiding excessive detail in model specification and development, it may require further compromise or the use of alternate representations (e.g. statistical models of fine scale spatial interactions of a fleet and a patchy resource within a larger spatial cell of the model).

All existing ecosystem models in use in fisheries run on standard desktop computers, though larger models do require higher speed processors and memory requirements. If fitting a spatial model or stochasticity is an important part of the analysis, then execution of the model on a cluster of computers is highly desirable in order to reduce computing time. Operating system specificity was once a major barrier to the use of certain existing ecosystem models, but many are now available for at least

Windows and Linux and the existence of efficient emulators means that even those that are not cross platform can still be run on machines with either operating system.

2.3 CANNED MODEL OR FRESH PRODUCE?

It has been traditional in fisheries science that modellers should design, program, and implement their own programs. This is, as a rule, a good practice and the model construction process is indeed both valuable and informative. There are, however, cases, notably related to data access, reporting and infrastructure overheads that make using an existing approach and software package a wise choice. The increasing flexibility of a number of the existing models means that they have become a framework for model creation and use rather than a monolithic model. This means that the user can benefit from the package's overhead handling while not being simultaneously locked into rigid assumptions. Careful design and application, raises the stakes for and capabilities of the modelling programs, while making ecosystem modelling more accessible for a wider range of scientists. In the current era where EAF is being introduced and developed, this is indeed a facilitating factor.

That being said, care must be taken when using pre-existing packages. It is necessary with these to carefully examine assumptions and requirements, and to investigate how different parameterizations and implementations impact model findings. There is never one model formulation that is "correct"; alternatives must be examined. Importantly, models should not be used as simple black-box formulations. Ecosystem models are tools, and as such are valuable only if used with thought.

3. ISSUES TO BE ADDRESSED BY ECOSYSTEM MODELLING

Ecosystem models can be used to assist in addressing several ecological issues pertaining to EAF. It is recognized that a range of different model constructions are needed to address the full range of issues, with no one model capable of addressing all aspects. A summary of the issues, grouped into three categories that could be addressed by ecosystem models is given below.

3.1 ISSUES PERTAINING TO THE MANAGEMENT OF TARGET AND RELATED SPECIES

- The impact of a target fish species on other species in the ecosystem. For example, does the removal of the target species negatively impact other species which depend on it as prey?
- The ecosystem considerations to be taken into account to rebuild depleted stocks.
- Is a single-species-based assessment of the status and productivity of a target species non-trivially biased or wrong because of a failure to consider multispecies interactions?
- Are there relatively unexploited species about which something is known and which could be targeted without having a detrimental effect on other components of the ecosystem?
- The impacts of retained bycatch.
- The effect on top predators of removing the predators themselves as well as their prey.
- The extent of competition between fisheries and species of concern such as marine mammals, turtles, seabirds and sharks. This includes consideration of both “direct competition”, which involves reduction (by consumption or utilization) of a limited resource but with no direct interactions between the competing species, and “indirect competition” in which the competitors may target different resources but these are linked because of a foodweb effect.

3.2 ISSUES PERTAINING TO SPECIES

- The impacts of fishing on biodiversity.
- The impacts of commencing fishing on a previously unexploited species about which little is known.

- The effects of the introduction of non-native species.
- The impacts of non-retained bycatch.

3.3 ENVIRONMENTAL AND UNINTENTIONAL IMPACTS ON ECOSYSTEMS

- The effects of physical/environmental factors on the resources on which fisheries depend.
- The consequences of changes in ecosystem state, for example, regime shift considerations, and whether fishing on particular stocks drive the ecosystem to a less productive/less desirable state.
- The importance of other anthropogenic effects besides fisheries.
- The effects of habitat modification. This includes consideration of effects such as trawling damaging benthic habitats, and hence perhaps having an indirect negative effect on fish stocks.

4. MODEL TYPES AND ATTRIBUTES

4.1 EXISTING MODEL TYPES

A comprehensive summary of existing ecosystem models is given in Plagányi (2007). The different models can broadly be categorized according to the framework presented in Figure 1. Models which represent only that subset of the ecosystem important for the issue under consideration are termed Minimum Realistic Models (MRMs) in contrast to whole ecosystem models that attempt to represent all trophic levels in an ecosystem in a balanced way.

Models that focus on inter-species interactions only are termed dynamic multispecies models. In contrast, dynamic system models incorporate the environment and lower trophic levels, although this is often at the expense of not representing the higher trophic levels in sufficient detail (when considered in a fisheries management context). In classifying models further, it is important to differentiate between models that take age structure and spatial aspects into account.

As stressed in section 2.1 and in the Appendix, definitive conclusions cannot be drawn from a single model structure and ideally a range of complementary models should be used.

There is a full continuum from qualitative conceptual-type models to fully quantitative detailed and statistically-based models. Conceptual models play an important role in consolidating understanding of a system as well as guiding the potential scope of later models together with their data needs. Ecosystem models are currently mainly used for strategic purposes, to assist in understanding a system, evaluating trade-offs and exploring a broad range of management-related questions.

If available resources do not allow for a full quantitative modelling (or MSE) exercise, some insight may still be possible using a more simplified analysis or qualitative or statistical methods.

As discussed in section 1.4, the MSE (or analogously MP) approach has been identified as best practice in ecosystem modelling because of its focus on the identification and modelling of uncertainties, as well as through balancing different resource dynamics representations and associated trophic dependencies and interactions. It has already been used in this role in Australia and in the development of ecosystem models for the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR).

4.2 ATTRIBUTES

Below follows a summary of which model attributes are considered important in developing models for informing an EAF, together with discussion as to why they are important, some guidelines as to best practice as regards the attribute, and some selected examples.

4.2.1 Ecological-related attributes

4.2.1.1 Model aggregation

Taxonomic resolution

Taxonomic resolution, and the best means of deciding upon it, are fundamental to all forms of multispecies and ecosystem models and are discussed in detail in the Appendix. To summarize, the number of groups must be dictated by the question being addressed and the relevant subsystems involved. If the subsystem is small then explicit representation of all members is feasible, otherwise some form of aggregation is advisable. The use of functional groups (defined based on predator and prey connections, size and rates, role, habitat use, behaviour, other non-trophic interactions, and spatial structure) is the most effective means of accomplishing that aggregation. Clear methods for this definition of group membership, such as clustering or regular colouration (network theory), are essential. Excessive aggregation as well as excessive detail will both result in a degradation of performance and should be avoided. One area in which it is often customarily assumed that more is better is in the representation of biodiversity. It may be the case that more groups are required, but rather than immediately jumping to that conclusion it is important to consider what is the appropriate form of representing biodiversity for the question in hand (a statistical or analytical index associated with trends in group biomass may be a more effective and tractable means of representing diversity). As the taxonomic resolution is such a key source of model uncertainty, it must be considered in conjunction with uncertainty itself, and it is important to trial different levels of taxonomic complexity. Whether it is best to start large and simplify or start with a few groups and increase is partly a matter of personal taste. Both approaches have been used successfully, though the latter is the more common.

Data availability may also be an important dictate of taxonomic resolution. section 4.3 of this report discusses the overall issue of data demands for ecosystem modelling.

Best practice: When developing conceptual models err towards a finely resolved taxonomic resolution. Once model development progresses to strategic or tactical model uses, it is important to aggregate based on shared characteristics of the species and to omit the least important if the food web is becoming large and unwieldy.

Age/size/stage structure

Age, size, or stage structure is considered to be an essential component of models if there are major ontogenetic shifts in the behaviour of the species of interest through the course of its life. It is particularly important when answering questions concerning predation and fishing because these effects are usually size specific. It is also important when spatial models are included where different parts of the population inhabit different areas or different parts of the water column.

Careful consideration should be given to the number of age/length classes. Experience has shown that using a smaller number of age or size groups will greatly reduce computing time with minimal loss of information. Most MRMs will need to include the age, size, or stage structure of the target species for providing management advice.

Best practice: Age, size or stage structure of the species of interest should be included if this feature is of importance to the issue of concern and could affect recommendations for management.

4.2.1.2 Spatial considerations

Spatial structure (explicit spatial cells)

Spatial structure should be modelled to the degree required to address the management issues and ecological aspects of concern. The following are a few examples of when spatial structure may need to be included: (i) when there are major ontogenetic shifts in location through the course of a species life history; (ii) when space is needed to capture the stock structure of a species (e.g. sedentary species) or species dependencies on critical habitat; (iii) when biological interactions or anthropogenic impacts are spatially localized. Spatial structure is obviously essential when the management question involves evaluation of spatial strategies such as the placement of MPAs or reproductive reserves.

A good example of the importance of spatial structure is in the south Atlantic krill fishery (Watters *et al.*, 2006). Although the krill population is large, fishing vessels mainly target localized krill swarms close to the island groups which are also the main foraging areas for land-based predators.

Careful consideration should be given to the number of spatial cells because of the associated costs in computing time. Further discussion on the selection of an appropriate spatial scale is given in the Appendix, together with examples of features useful for defining spatial cells. Spatial resolution can dictate broad processes represented, but that does not mean that important processes on finer scales can or should be ignored. Instead consideration should be given to how important processes on finer scales are, and whether analytical or statistical formulations should be used to represent these sub-grid scale processes. For instance, statistical models can be used to capture the impacts on habitat of a fleet interacting with a patchy resource (Ellis and Pantus, 2001).

Many models will need to include some degree of spatial resolution depending on the complexity of the physical environment, the species in question, and the questions being addressed. However, it may be possible to reduce the level of spatial resolution in some cases when providing management advice.

Best practice: Spatial structure should be included to the degree required to address the management issues and ecological aspects of concern.

Seasonal and temporal dynamics

Seasonal and temporal structure is considered to be an essential component of models if there are large seasonal differences in species movements or production. It is particularly important when answering questions concerning predation and the negative impacts of temporal (and spatial) location of fishing because these effects are often season specific. It is also important when considering temporally differentiated environmental and anthropogenic impacts, and fishing on spawning fishes.

A good example of the importance of temporal structure is provided by the Antarctic ecosystem models. Because of the huge seasonal shifts in primary productivity, associated changes in krill, and migrations of many large predators, it is essential that these models contain a seasonal component, which matches the scale of environmental variability.

Careful consideration should be given to the number of temporal cells, because of the associated costs in computing time. Further discussion on the selection of an appropriate temporal scale is given in the Appendix together with different ways to handle time.

Some models may need to include some degree of temporal resolution depending on the temporal variability of the physical environment, the species in question, and the questions being addressed. However, it may be possible to reduce the level of temporal resolution in some cases when providing management advice on the species of interest.

Best practice: Seasonal and temporal structure should be included if this feature is of importance for the issue of concern and could affect recommendations for management.

Flexible boundary conditions

In constructing a model, it is important first to identify the core spatial domain and then decide how to handle links with external domains. Boundary conditions are an important consideration if there are: a) important major immigration and emigration components such as seasonal movements of species; b) other substantial import / export processes such as occur around seamounts, or c) exchanges as a result of ontogenetic changes in habitat use. Models need to be sufficiently flexible to take account of these boundary conditions adequately.

The general consensus is that best practice involves basing boundaries on biological rather than anthropogenic considerations such as national boundaries. This may introduce additional complications if there are different jurisdictions in different regions, such that a range of alternative scenarios of anthropogenic impacts for these regions may need to be considered. Hence whereas basing boundaries on biological considerations is essential from strategic perspective, practical considerations may necessitate restricting the model domain when applied for tactical purposes.

Best practice: Boundaries should be based on biological rather than anthropogenic considerations such as national boundaries.

Multiple stocks

If it is possible that a fishery may be harvesting more than one stock of a particular species, models need to distinguish such different stocks when the harvesting practice is such that it might impact these stocks to different extents. The presence, number and distributions of different stocks are typically difficult to determine, so that the possibility that multiple stocks are present should not be dismissed lightly. Management should aim to conserve all stocks when more than one may be present, particularly because heavy depletion of some stocks can reduce genetic diversity and make the species as a whole more susceptible in the event of environmental change. Improved management given the possible presence of multiple stocks is generally achieved by ensuring that catches are spread widely, so that including a spatial component in model structure to allow advice in this regard to be refined becomes essential in these circumstances.

The ATLANTIS-SE model used in the Australian Alternative Management Strategy (AMS) project (Fulton, Smith and Smith, 2007) is an example of a whole ecosystem model that includes multiple stocks for the target species (e.g. *Hoplostethus atlanticus*, *Genypterus blacodes* and *Seriolella brama*). This was necessary to capture their biology and ecology (with some of their ecological parameters differing among stocks) as well as the range of management options for the system, such as stock specific assessments and actions, spatial management and regional TACs.

Best practice: If it is possible that a fishery may be harvesting more than one stock of a particular species, models need to distinguish such different stocks when the harvesting practice is such that it might impact these stocks to different extents; this will necessitate spatially structured models.

Multiple fleets

Models need to distinguish different fleets if, for the same mass of catch, they make substantially different impacts on target and bycatch species or on the habitat and/or when such distinctions have important social or economic ramifications. The reasons for this may be related to the fleets operating in different areas or at different times, or using different gears, which can lead to different species mixes and to differing size compositions of the same species. Examples include longliners and trawlers, or commercial and artisanal fishers targeting the same species. The need to take these differences into account may require models incorporating spatial resolution. Furthermore, analysis outputs will need to distinguish performances by the different fleets, as prices and costs per ton may differ, and the benefits accrue to different social groups.

Best practice: Models need to distinguish between different fleets if, for the same mass of catch, they have different impacts on target and bycatch species or on the habitat and/or when such distinctions have important social and economic ramifications.

4.2.1.3 Model components

Primary productivity/nutrient recycling

The inclusion of primary productivity and explicit nutrient cycling is far more common in strategic models and models for understanding than in tactical models. Specifically, the representation of these processes is required to address questions that relate to bottom-up forcing, the microbial-loop and the role of anoxia, as well as whole ecosystems rather than restricted parts of them (such as adult life history stages of higher trophic levels, which is why primary productivity and explicit nutrient cycling is often not a concern for multispecies models). In that context the explicit inclusion of these processes provides the potential to look at a wider range of potential hypotheses regarding forcing and alternative stable states (e.g. system dynamics under different nutrient loads). This is particularly useful in “what-if” gaming for conceptual understanding and hypothesis generation. For example, variation of primary productivity can be seen to ripple through the web and impact target species (such as cod) and higher trophic levels (such as toothed whales). Including such processes can give insight into mechanisms that may need further exploration and data collection in reality.

An example is provided by the North Sea, where in terms of a summation of the assessments of single-species assessments, the total equilibrium biomass if fishing were to cease would be much higher than has ever been evident in the past. Implicit account of primary productivity limitations could be taken by placing a realistic cap on the total biomass or production of all the major species.

Even when the processes of primary productivity and nutrient cycling (or anoxia) are considered to have an important role in shaping the dynamics of the system under consideration, explicit representation may not be necessary. In the case of primary production, as long as careful thought is given to alternative scenarios regarding the production of the basal resource group represented, it is not necessary to explicitly represent the mechanisms of primary production. This is important as explicit inclusion of those processes can mean moving to finer time and spatial scales (or more careful handling of those dimensions), with associated computational costs. This is also an important issue when dealing with processes like anoxia: depending on the question it may be more effective to represent the impact of the event rather than the detailed processes leading to the event (e.g. system dynamics under different nutrient loads).

Best practice: *Careful thought must be given to how production in a system is represented: explicit representation of primary productivity and nutrient cycling may only is necessary when bottom-up forces or lower trophic levels are of key concern. In such cases, inclusion of these processes can be highly informative for some strategic modelling exercises.*

Recruitment models

Recruitment is often a fundamental process in multispecies and ecosystem models. The degree to which the process is represented explicitly will be a decision to be made during model formulation. Applying a standard stock-recruitment relationship (e.g. Beverton-Holt) is the traditional approach that is appropriate for some questions (particularly in multispecies models), but will often be in a modified form (e.g. fecundity is dependent on condition of the spawning adult) to avoid "double counting" of processes represented explicitly in the model that are also implicitly represented in the standard formulation of the relationship. There are commonly-used approaches for statistical estimation of the parameters of these kinds of relationships for many assessed stocks, even if it can be difficult to distinguish the type of relationship where the range of spawning stock biomass observed is limited. It is important to have time series data of stock and recruitment in order to evaluate such relationships, and the reader is cautioned that without such data it is not possible to verify these relationships. For instance, recruitment can be correlated with environmental variables, typically in the form of temperature relationships. However, the reader is strongly cautioned against casting a wide net to determine such relationships through uncritical correlation studies of recruitment and environmental parameters.

The other most commonly used representation of recruitment is as an emergent property that is obtained by explicitly modelling early life history processes of relevant species or functional groups. For example

NWS-InVitro (a model of the Northwest shelf of Australia, Gray *et al.*, 2006) has an option to explicitly represent four phases of larvae and juvenile fish development: free-floating larvae, settlers, juveniles, and maturing sub-adults. This is particularly useful if issues of larval supply are in question, as is often the case in climate impact scenarios. Even without explicitly representing these phases, specification of a pre-defined stock-recruit relationship may be avoided by allowing recruitment to be an emergent property that can arise from modelling parental abundance, feeding conditions and climatic factors, combined with early life history processes such as larval advection, settlement, predation, and food conditions.

Variation in recruitment may also need to be modelled and is important in models where capturing observed interannual variation is important (e.g. tactical models). In strategic models direct incorporation of recruitment variability can be less critical, particularly as it may result from the impact of environmental forcing on the system acting in combination with trophic interactions to vary growth and juvenile mortality. As recruitment variation affects risks in strategic evaluations, if recruitment is variable then alternative forms of representing recruitment variability must be considered when evaluating robustness and model uncertainty. This is also the case when considering its impact on tactical decisions and catch variation. An important problem is that without measures of recruitment this component of the model cannot be verified.

Best practice: *Recruitment may be included either as an emergent property or as a derived relationship (which should not be based on uncritical correlation studies of recruitment and environmental parameters). Recruitment variability is likely to be important for tactical and risk analyses, but is not a strict requirement for many strategic models.*

Movement

Incorporating movement into a model can fall into one of two categories. Immigration into the model domain can be dealt with fairly simply and straightforwardly, such as by using an empirical formulation based on data from surrounding areas. In some instances, movement of species or other ecosystem components into a model domain can also be represented by using simple forcing functions. On the other hand, representing movement explicitly within a model is a challenging topic with several alternative methods for consideration, such as whether to assume movement is density dependent or habitat dependent. It may also be necessary to consider vertical migration. Including a spatial component in model structure becomes essential in these circumstances.

Movement may be implemented by either directly specifying migration matrices, or calculating these based on migration rate input information describing the proportions of a stock that will migrate between different areas. These matrices can, for example, be used to capture broad seasonal patterns, even if the finer details are not known. Moreover, an addition that can be useful is the inclusion of a tagging experiment feature that can keep track of the number and proportion of fish in an age-length cell that have been tagged. Other approaches may use decision rules such as that fish move to adjacent cells with the highest biomass of potential prey.

In cases where movement is considered important, best practice involves testing sensitivity to a range of movement hypotheses. If data are available, best practice involves parameterising movement matrices by fitting to data or at least including penalty functions to guard against nonsensical resultant changes in distribution. Where relevant, the outputs of circulation models may be used to assist in parameterising movement matrices, but consideration needs to be given to possible errors both with respect to the outputs from the circulation model and the extent to which model components can be assumed to be passive drifters.

Best practice: This includes testing sensitivity to a range of movement hypotheses, and where possible, parameterising movement matrices by fitting to data. If decision rules are used to drive movement, attention should be focused on whether the resultant changes in distribution are sensible. As with other complicated model features, best practice involves including only as much detail as necessary.

Fleet dynamics

Fleet dynamics become important to consider if substantial changes to the spatial distribution of fishing may result from, for example, the declaration of an MPA (leading perhaps to a concentration of fishing effort close to the MPA boundaries), or environmental changes leading to a different distribution of target species. The population model will need to incorporate a spatial component and it may be necessary to develop a model of fleet dynamics to predict how fishing patterns will shift in response to other changes. Changed fishing patterns may impact the economic performance of the fishery, and also have implications for localized fishing communities, so that model outputs need to include statistics that provide information on these aspects. An example is provided by the necessary consideration of limitations on a relatively small spatial scale for krill fishing in the Scotia Sea, this being aimed at enhancing the reproductive success of land-breeding krill predators where the consequential changes in the distribution of krill fishing effort need to be taken into account.

Best practice: Fleet dynamics are important to consider if substantial changes to the spatial distribution of fishing may result from, for example, the declaration of an MPA. The population model must include spatial components in these circumstances, and it may be necessary to develop a model of the manner in which fishing effort patterns will change in response.

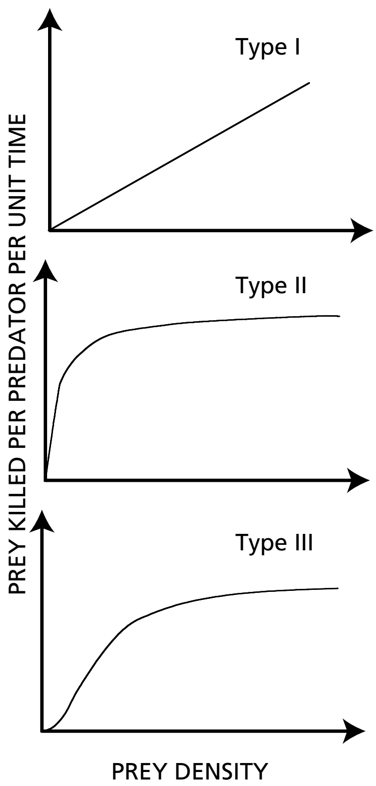
4.2.1.4 Modelling predator-prey interaction (see also 4.2.1.7 Non-trophic interactions)

Predator prey bi-directional feedback

Whereas most ecosystem models include predator-prey feedback dynamics, MRM's are often constructed with only a uni-directional predator-prey link. For example, in response to concerns related to the impact of Cape fur seals (*Arctocephalus pusillus)* on hake, Punt and Butterworth (1995) developed a model that incorporated the effect of seals on hake, but did not include any feedback between a paucity of hake and a population-dynamic response in (for example) weight-at-age, survival and/or reproduction of seals, i.e. it was assumed that there was always sufficient "other" food for such predators. This differs from a scenario such as current concerns by CCAMLR that a potentially increasing krill fishery may negatively impact land-based predators in the Antarctic Peninsula region, whilst simultaneously recognising that these predators in turn exert considerable mortality on the krill there. A third case involves situations in which a predator may not exert a substantial impact on a prey species that is also targeted by a fishery, but may be particularly sensitive to reductions in the level of prey abundance. For example, the breeding success of African penguins may depend critically on the abundance of their pelagic fish prey, but penguin abundance itself is too low for their overall predation impact on pelagic fish to be substantial.

Best practice: Predator-prey interactions should be represented in models as bi-directional unless sufficiently strong motivation can be provided that it is adequate to include a one-way interaction only. Bi-directional interactions are desirable at the strategic level, but may not be relevant at the tactical level if the associated interaction strengths are low.

FIGURE 3
Schematic summary of conventional functional response relationships



Predator-prey functional relationships

General consensus is that best practice involves acknowledging the paramount importance of the appropriate form for functional responses (the prey-predator interaction terms) (Figure 3) and feeding selectivities/suitabilities. Progress in this field is primarily impeded by a lack of suitable data and experimental studies. Simulation exercises are helpful to systematically and thoroughly explore the issue, and have clearly demonstrated the sensitivity of model results to the choice of functional relationship (Fulton, Smith and Johnson, 2003). It is therefore essential that model robustness be examined to alternative interaction representation hypotheses.

Thus care needs to be taken to test the appropriateness of default parameter settings. Models need to be closely scrutinized to understand the extent to which underlying model assumptions predetermine or have implications for the results obtained. An example is the foraging arena model where, except in the limits of very high vulnerability, the model necessarily yields additional sustainable catch of a forage species that is less than the reduction in consumption achieved by reducing the abundance of a predator of that forage species. A further caution to be borne in mind occurs if one has a particular functional form at the microscale, and the parameters of that form vary spatially; this does not mean that when that form is integrated over space the resultant functional form will necessarily lie within the set of forms covered by varying the parameters of the original form. Considerable computational savings can be made by using a formulation appropriate to the spatial and ecological resolution of the model. For instance, trials performed as part of the development of NWS-InVitro showed that the results of an Individual-Based Model (IBM) implemented over regional scales are effectively identical to the functional forms produced by Holling Type II and III formulations. Moving to a more spatially aggregated model where explicit use of these Holling functional responses was possible saw the computation costs reduced by three orders of magnitude for no loss in model performance when considering regional scale questions.

Best practice: Acknowledge the paramount importance of the appropriate form for functional responses (the prey-predator interaction terms) and feeding selectivities/suitabilities, and test sensitivity and robustness to alternative forms.

4.2.1.5 External forcing

Environmental forcing

Environmental forcing is considered an essential component of strategic models designed to answer questions on the effects and the relative role of climate change, regime shifts and anthropogenic effects. Experience to date shows that in a number of cases, inclusion of environmental forcing improves fits to historical trends or time series such as recruitment, growth and spatial distribution of catches, in which case it is clear that those models need to accommodate the forcing (see for instance Christensen and Walters, 2005). For example environmental forcing of lower trophic levels and production has been found to produce a much better fit across the system (e.g. Preikshot, 2007). Environmental forcing of other components of the system may also be necessary for capturing their driving forces (e.g. conditioning of recruitment of prawns on rainfall in NWS-InVitro, Gray *et al.,* 2006). There are few examples of the use of environmental forcing in tactical models. Near real time information on the spatial distribution of habitat for southern bluefin tuna off the east coast of Australia is inferred from a high resolution ocean model and used to set management zones. The boundaries of these zones are moved as oceanographic conditions change, and fishermen must own quota to fish in these zones, with quota being most costly in the zone where bluefin are most likely to occur (Hobday and Hartmann, 2006). In another example, three-year moving average water temperatures measured at Scripps Pier, La Jolla, California are used to adjust annual harvest rate for Pacific sardine in the California Current. Below a fixed threshold, cooler temperatures lead to near-linear reductions in the harvest rate, down to an agreed minimum (Pacific Fishery Management Council, 1998).

Environmental forcing is much more common in strategy evaluations. In the context of closed loop simulations and strategic models, if environmental forcing is required to capture historical patterns, then these patterns must be continued into the future for projections. This requires careful thought on how future forcing time series are generated – whether by obtaining future trends from climate models, repeating the historical time series in full, drawing time periods from it, drawing from a statistical distribution based on historical data, or using scenarios to depict a much high/lower frequency or magnitude of the environmental driver. When statistical fits alone have been used to deduce the existence of an environmental forcing effect, then it is again important to consider alternative models (with and without the driver, or with different forms of the driver) during the evaluation.

Best practice: Carefully consider whether environmental forcing is required to capture system dynamics. Care must be exercised in selecting the basis to generate future forcing for use in predictions and closed loop simulations.

Other process error (i.e. random variation)

Other process error arises from natural variation in model parameters, such as variability in survival, movement rates, fishing selectivity, availability of fish to the fishery and catchability. Often this variability cannot be captured explicitly in an assessment model and contributes to the residual error of the assessment. Similarly, the processes that cause such variability may be poorly understood with the variability in a parameter being represented as a simple probability density function. Such process error, drawn from an appropriate statistical distribution, needs to be considered for inclusion in projections, whether they be strategic or tactical, in order to account for stochasticity in these parameters, particularly when that variation contributes substantially to uncertainty in the model outcomes. An example of how such process error could impact on conclusions or decisions is in determining the relative importance of an area, such as an MPA, compared to surrounding areas. A large degree of variability in rates of movement between these areas will potentially cause the relative importance to the population of the protected area to vary over time.

Best practices: Other process error, arising from natural variation in model parameters, needs to be included in projections, whether they be strategic or tactical, when that variation contributes substantially to uncertainty in model outcomes.

Other anthropogenic forcing

Other anthropogenic pressures on marine ecosystems include all the major non-fisheries anthropogenic influences such as nutrient pollution (which may cause eutrophication) or other contaminant pollution (e.g. oil or heavy metal); large scale changes in freshwater flow or water properties (e.g. bitterns from salt production can change the temperature and salinity profile of the local area); and habitat degradation (e.g. due to dredging, land clearing and coastal development). If any of these impacts the system of interest then it is highly desirable to include them in the model representation. This is usually done via forcing (e.g. with time series of loading) rather than by using a detailed model of the process (unless the model is part of a larger multiple-use management analysis). This form of forcing is more likely to be seen in strategic than tactical models, and even then is really only seen in shallow or coastal waters rather than deepwater systems. There is no doubt that in shallow systems the inclusion of anthropogenic forcing has improved the fit to historical trends or time series in many cases, because the impacts of these other processes can have a major role in determining system dynamics. This makes their inclusion an important component of coastal and estuarine models because without them it would be impossible to determine the relative importance of fishing impacts and the robustness of potential management actions. For example the confounding of eutrophication and depletion of filter feeding bivalves in Chesapeake Bay has meant that simply acting on water quality or fishing alone could not return the bay to less perturbed states typical in the past. It is also important to include this sort of model forcing if it affects trade-off or management decisions, such as catch versus risk of contaminant contact in inshore fisheries around outfalls, or the risk of contamination of mussel farms by *E. coli.* The very nature of the impacts and often high site association of these other anthropogenic pressures means that their inclusion is tightly linked to the spatial structure of the model. They are also often linked to social and economic components of the model.

Best practice: Anthropogenic forcing on shallow coastal and estuarine systems should be considered in conceptual models and if found to lead to appreciable pressures on the system then this forcing should be included empirically (e.g. simply as a forcing term) in any strategic models and MSEs for the system.

4.2.1.6 Model structure

Potential for alternative stable states

"Alternate states" in models and ecosystems encompass two, distinct concepts which are often confounded in the literature. It is important to distinguish "Regime Shifts", defined as a change in external parameters that force the system, from "Phase Shifts", defined as a qualitative change in the behaviour of the system (Duffy-Anderson *et al.*, 2005). A "qualitative" change refers to a change in the organization or structure of the ecosystem that occurs through crossing a threshold into the domain of a new stable state, from which return may not be possible even if the force driving the transition (e.g. climate or fishing) is reversed. A phase shift seems to occur in nonlinear steps rather than in linear relationship to a climate regime (Overland, Percival and Mojfeld, 2006).

Crossing a threshold may involve changing the dominant controlling process. For example, a threshold switch in controlling process from bottom-up to top-down control has been hypothesized for pollock recruitment in the Gulf of Alaska (Ciannelli *et al.,* 2005). A model calibrated/fit to data from a single phase, or containing in its model structure the potential for only one mode of control (e.g. bottom-up only), may not capture the threshold behaviour and could vary greatly in predictive power for such an ecosystem interaction.

Techniques for analysing nonlinear switch responses are important for ecosystem modelling as some time series analyses (e.g. Hsieh *et al.*, 2005) have distinguished climate time series (linear with many controlling variables, or "regimes") from biological time series (nonlinear with few controlling variables).

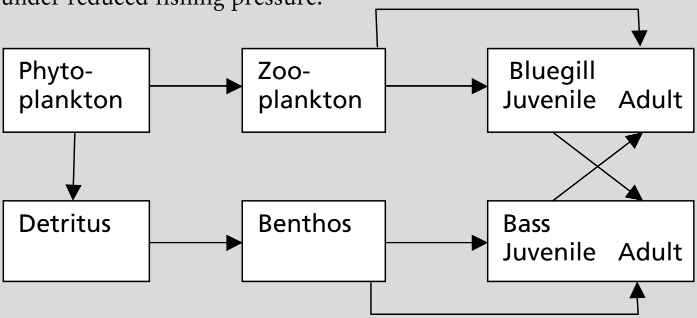
Examples of models which may contain potential alternate stable states are Type III functional responses (Figure 3), certain types of predation on age structure (see Box 1), and the "predator pit" formulation or models that include depensation (Bakun, 2006).

Box 1

Alternate stable states: bass and bluegill

Stable alternate states are known to occur in freshwater lakes with bass and bluegill, where different initial conditions may lead to dominance of one or the other. This can be modelled using a simple age-structured model with cross-linkage between adult and juveniles of bass and bluegill (see diagram below). Even with only a very small part of the diet being juveniles of the other species, e.g. 1 percent for adult bass–juvenile bluegill and 0.01 percent for adult bluegill–juvenile bass, the model can produce alternate stable states if the adult biomass of one of the species is perturbed by a brief fishing pulse.

The bass – bluegill interactions are an example of the cultivation/ depensation hypothesis, presented by Walters and Kitchell (2001). They describe how fishing down of a dominant predator may lead to an alternate stable state, where a prey/competitor species may take over and become dominant in the ecosystem due to released predation pressure. The new dominating species may in turn be keeping the previous dominant species at a low level by feeding of the young of that species, making recovery difficult even under reduced fishing pressure.



Best practice: Include consideration of models, especially strategic models forecasting the consequences of environmental change, which contain the capacity (e.g. flexibility in choice of functional relations) which allow for plausible phase shifts, either directly (in accordance with past observations) or as an emergent property of the functions of the model. Even if such a functional form is used, it must be recognized that, until a threshold is crossed by the system, it may not be possible to parameterize the threshold point: given such uncertainty, possible thresholds may need to be evaluated on either a theoretical or an empirical basis.

4.2.1.7 Technical and non-trophic related

Technical interactions

In the context of the use of ecosystem models to inform fisheries management, technical interactions refer to the effects of fisheries that catch species other than the primary target. This includes multi-stock fisheries and fisheries that nominally have a single target species but take bycatches of other fish stocks that are the target of other fisheries. Many fisheries have bycatches of threatened species such as sharks, turtles, seabirds or marine mammals – these are also technical interactions. Bottom trawls or dredges fisheries that damage the habitat of benthic and epibenthic communities also have technical interactions. Technical interactions thus describe direct effects (i.e. removals) on other species or habitats as an, often unintended, consequence of fishing.

Technical interactions should be included in tactical models that directly inform management decisions if the bycaught species are themselves also subject to management, including stock rebuilding, or if the model aims to inform the level of bycatch of a threatened species. Technical interactions are also essential to include in strategic models if the aim is to explore tradeoffs in how different management actions affect bycatch.

If technical interactions are included in a model, other features are also likely to be necessary including: age/size/stage structure because bycatch is often of juvenile fish, spatial structure because different ages of fish may be found in different areas, fleet dynamics and multiple fleets because the species or impacts of interest will be respectively taken or caused by more than one specific fleet, and social and economic aspects.

Best practice: Technical interactions need to be included in a model if the question that the model is aiming to address relates to the direct impact of a fishery on another species or habitat.

Non-trophic interactions

Conceptual models of ecosystems, particularly subsystems associated with the benthos, frequently include non-trophic interactions. These often concern habitat dependency or spatial refugia, but other forms do exist. Whether or not these interactions need to be included in any model of the system is dependent on the importance of their role in system dynamics and the specific question being asked. For example, if habitat is a critical determinant of the biomass or distribution of the main groups of interest, or if management could be based around habitat (e.g. effective spatial control of vulnerable bycatch groups by closing all rough ground to trawls and dredges) then inclusion of habitat dependency and habitat mediated interactions and processes would be highly desirable. Habitat dependent non-trophic interactions have tight links to the spatial resolution required of a model (or at least how spatial processes are represented within a model), as well as to anthropogenic forcing (due to the potential impacts of habitat degradation), technical interactions (which may modify the habitat), fleet dynamics (which may see differential pressure across the spatial domain of the model), and the decision to represent age, stage or size structure in the modelled populations (as only specific life history phases may be habitat dependent, or conversely the habitat itself may need to be represented in a size- or age-based way to capture recovery lags; Fulton *et al.*, 2006).

The importance of habitat dependency may be so great that it overwhelms the typically trophic focus of multispecies and ecosystem models. In some circumstances, statistical correlations related to trophic or environmental causes may be used to avoid having to represent trophic inter-species links explicitly. For example, in NWS-InVitro (Gray *et al.*, 2006) computational constraints meant those difficult decisions had to be made regarding the taxonomic resolution of the model. Because habitat dependency was a dominant determinant of the presence and abundance of key groups in the system (large target species that were reef-associated such as *Lutjanus sebae* or large Lethrinids and less discerning Saurida species), it was possible to model the key technical and ecological interactions in the system without explicitly representing trophic connections (instead a habitat model was included and it was assumed that if the habitat was of a suitable form then all trophic connections supporting the species in question were functional).

The most recent interest in this form of non-trophic interaction has arisen through their potential role in enhancing or mediating the impacts of climate change. Considerable attention is being focused on the potential impacts on target and endangered species of shifts or loss of biogenic habitat due to climate change.

Not all non-trophic interactions, however, involve habitat. There is the potential for trophic interactions between two (or more species) to be mediated by the action of a third group (see review by Dill, Heithaus and Walters, 2003). For instance, prey fish may escape predation by large tunas by moving into warmer surface waters. However this makes them available to diving seabirds which could not normally access them at depth (Ramos, 2000). Another form of this kind of mediation is when marine mammals raid fishing gear and consume part (or even all) of the catch.

Best practice: If conceptual system understanding indicates that a non-trophic interaction is a critical determinant of the dynamic of interest (e.g. biomass or abundance of a target group), or if management could be based around this interaction, then its inclusion is highly desirable.

4.2.2 Model specification-related attributes

4.2.2.1 Dealing with uncertainty

Ability to fit to data
Fitting multispecies models to data is best practice and essential in both strategic and tactical contexts. Fitting to data is, *inter alia*, important for 1) estimating model parameters and providing diagnostics that may be used to improve model formulation; 2) quantifying parameter uncertainty; and 3) weighting alternative hypotheses represented by alternative models, including identification of those that are not supported by the observations. In many cases there is not enough information to estimate all model parameters, and some have to be fixed.

Best practice in fitting models to data requires careful specification of likelihoods, which involves making assumptions about the processes involved in the collection of data. Model predictions and observations must be compatible, meaning that they must have the same taxonomic, temporal and spatial resolution. When data exist at a finer level of resolution than the model variables, some of the data may be aggregated in order to avoid having to increase the complexity of the model to accommodate all available data. Ideally, multispecies models should be fitted to actual data (e.g. survey data) rather than outputs from single-species assessment models (e.g. estimated biomass series). For whole ecosystem models, however, such an approach may entail increases in model complexity and computational burden beyond what is practical, especially when many species are included and considerable data exist for many of them. In such cases the only practical approach may be to treat outputs from assessment models as data while assuring that issues of consistency between the assessment and the ecosystem model assumptions are addressed. The following are examples of the types of inconsistencies that would lead to problems:

i. single species assessments are commonly based on a constant natural mortality (M) while ecosystem models will imply variable Ms as a result of predator-prey interactions;
ii. estimates derived from single species models with an aggregated biomass (e.g. Schaefer model) are not comparable, at least in absolute terms, with biomasses predicted from age-structure models.

The inconsistency in the modelling of M could be addressed by iterating between fitting the ecosystem model to abundances estimated by the single-species assessment, and re-estimating abundances using the single-species model conditioned on the M trends predicted by the ecosystem model, until convergence is attained.

In addition to addressing consistency issues, when outputs from single-species assessments are used as input data for fitting ecosystem models, the likelihoods should recognize the uncertainties in these outputs, their correlation structures, and their effective sample sizes.

Best practice: Fitting to data is best practice, and this requires careful specification of likelihoods.

Parameter uncertainty

Best practice requires clear statements about uncertainties in model parameters. Similar to uncertainty in model structure, evaluating the sensitivity of model outcomes to parameter uncertainty is essential for all of the strategic and tactical questions and issues that may be addressed with ecosystem models. Predictions from ecosystem models can be conditioned on prior parameter uncertainties, but it is generally preferable to quantify these uncertainties by fitting to data. Stakeholders often request complex models with many parameters, but it is important to remember that such requests may be inappropriate when there is no information to quantify the uncertainty in these additional parameters.

Bayesian methods and bootstrapping are considered best practice for quantifying parameter uncertainties in extended single-species models and MRMs. With Bayesian methods, the plausibility of alternative parameter values can be evaluated based on objective criteria derived from the likelihood of the available data for the system in question, complemented with information from other regions for the same or related species. The latter type of information can be used to derive prior density functions or to set bounds on some parameters. Various methods (e.g., Markov Chain Monte Carlo) can then be used to estimate the joint posterior probability of all model parameters conditioned on all the data, and parameters sampled from the joint posterior can then be used to simulate future trends. These simulations will provide results that integrate across the range of hypotheses represented by the different parameter values. With bootstrap methods, the model can be fitted, *via* maximum likelihood, to multiple data sets that are themselves developed by resampling the original data. Uncertainties in the parameters are quantified by the distribution of estimates derived from these multiple fits.

In practice, the absence of data or concerns about the validity of the likelihood function may lead to the use of less rigorous approaches to describing parameter uncertainty, for example by assigning distributions of parameters, with appropriate correlation structures, developed from "expert judgment." When this is the case, it is still best practice to fully explore parameter uncertainty with sensitivity analyses or likelihood-based approaches as a first stage, evaluate their performance using formal diagnostics, and then provide an explicit rationale for the final choice of weights.

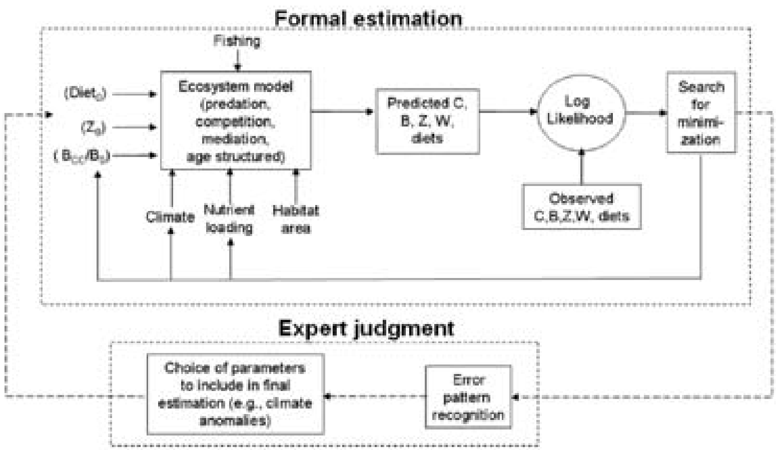
While Bayesian and bootstrap approaches used for single-species can be readily applied in moderate-size multispecies models, best practice for quantifying parameter uncertainties in more complex ecosystem models is currently not clear (this is an area of active research). Nevertheless, the current practice of adopting default parameter values, and their distributions if such are available should be improved so that careful attention is paid to all parameters. At a minimum, this requires: 1) that there is an explicit accounting of the number of parameters that are being estimated and fixed, 2) qualitative estimates of the uncertainty in every parameter, and 3) sensitivity analyses. While exploring the sensitivity of modelling outcomes to parameter uncertainty is standard practice in single-species models, and there is abundant experience about what are the critical parameters, their possible confounding and their likely effects, the problem is much more difficult to handle in ecosystem models. This difficulty arises both because there are often large numbers of parameters in ecosystem models, and because there are complex constraints imposed by the coupled dynamics in such models (e.g., the mass-balance in Ecopath with Ecosim [EwE]). A useful place to begin such sensitivity analyses is with the parameters that govern interactions among species (e.g. the vulnerabilities in EwE) and with parameters that were assigned a high qualitative level of uncertainty.

Whole ecosystem models tend to contain a large number of parameters and care must be taken in their interpretation. Static models are not fitted outside of obtaining the best individual data points and satisfying mass-balance constraints; for these models (used primarily for understanding), sensitivity/perturbation routines have been developed and used (e.g. Ecoranger and the Trophic Impact Matrix; Christensen, Walters and Pauly, 2005), but it is the primary responsibility of the user to assess data quality through a formal data "pedigree" (quality ranking) process (see section 4.3), and to use the assessed range of uncertainty in sensitivity analyses.

When progressing to dynamic whole ecosystem models, the problem is compounded, as a result of the addition of interaction terms. Here, sensitivity analyses may be ad hoc, but have been formally developed, for example in Aydin *et al.*, 2005. Even if used only as a conceptual model, sensitivity analyses themselves are generally insufficient for dynamic models. The models should be fitted to time trend data before being used for providing strategic or tactical advice.

A typical EwE model may contain 8 parameters per species (for 40-50+ species) plus 2-4 parameters per trophic link and additional parameters for juvenile/adult groups, while having only a fraction of this number of data points available for fitting. A strategic reduction of parameters is necessary, either by grouping parameters (e.g. reducing per-trophic link parameters to functions of a predator or prey, or fitting by species type), or fixing certain parameters. It is important to note that these decisions may greatly affect the results. For example, "standard" EwE fitting allows estimation of vulnerability parameters while fixing handling time, unexplained mortality, and juvenile/adult parameters, leaving residuals to be fit or tuned from external anomalies (e.g. primary production). Figure 4 shows an example of a fitting procedure used in EwE. On the other hand, a method for freeing and fitting more parameters in EwE (e.g. Aydin *et al.*, 2005) may reduce these residuals and the need for external forcing, while producing poorly converging fits and wide error ranges. This trade-off in fitting methods for these models is an area of continued and active research.

FIGURE 4
Example of fitting procedure to minimize residuals between observed and estimated parameters (modified from Christensen and Walters, 2005). Here illustrated for time-dynamic Ecosim models in EwE and involving a formal estimation procedure and an expert judgment. It is necessary to include the expert judgment, notably to minimize the number of parameters that are to be estimated



Best practice requires explicit evaluation of the effects of uncertainties in model parameters for management advice. Bayesian methods and bootstrapping are considered best practice for quantifying parameter uncertainties in extended single-species models and MRMs.

Best practice for quantifying parameter uncertainties in more complex ecosystem models is currently not clear. At a minimum, improving current practices requires: 1) that there is an explicit accounting of the number of parameters that are being estimated and the number fixed, 2) qualitative estimates of the uncertainty in every parameter, and 3) sensitivity analyses.

Best practices for mass-balance/static models are to develop and fully document a formal data "pedigree" (quality ranking), and if possible include error ranges for estimates, with input from data providers as to potential biases. Further, sensitivity analyses may be conducted using available routines.

For dynamic models, best practice is to fit to as much data as possible using appropriate likelihood structures, while being clear about both potential biases arising from fixing parameters, as well as fully reporting error ranges resulting from freeing parameters. In the case of fixing parameters, additional sensitivity analyses (e.g. resampling, Monte Carlo routines) should be used to assess model sensitivity to the assumptions. An important component of best practice is using results of sensitivity analyses to guide future data collections and the continuation of key time series.

Model structure uncertainty

Model structure uncertainty relates to the choice of the hypotheses and associated functional forms to be included in an analysis, be it tactical or strategic. Alternative hypotheses for the processes governing the dynamics of the ecosystem, the fishery, etc. need to be carefully considered because the results for most of the issues identified in Chapter 3 will be sensitive to the selection of some of these functional forms, and treating model structure uncertainty inappropriately may lead to a false sense of certainty.

Best practice in the treatment of model structure uncertainty in complex models (including ecosystem models) is still an active research area (Hill *et al.*, in press). However, general consensus is that best practice involves first identifying alternative qualitative hypotheses for all of the processes considered likely to have an important impact on the model outputs, and then formulating these hypotheses mathematically (or as the values for parameters of a general relationship). Section 4.2.1 of this report provides details of some of these processes (e.g. taxonomic complexity and the choice of the feeding functional relationship). Other areas where it is important to consider alternative hypotheses are the processes that impact the mortality on juvenile fishes and whether (rare) predation links exist among components of the foodweb. Alternative hypotheses can also relate to different interpretations of existing data: for example, whether a decline in the abundance of a particular species can be explained by the impact of fishing or that of environmental forcing.

It is likely that a very large number of alternative hypotheses will be identified and best practice is to use techniques for rejecting models that are inconsistent with existing information and then assigning weights to the remaining models. In principle, a formal scheme such as that developed by Butterworth, Punt and Smith (1996) can be used to assign weights to alternative hypotheses, i.e.:

1. how strong is the basis for the hypothesis in the data for the species or region under consideration?
2. how strong is the basis for the hypothesis in the data for a similar species or another region?
3. how strong is the basis for the hypothesis for any species? and
4. how strong or appropriate is the theoretical basis for the hypothesis?

Care must be taken not to focus too much on the application of methods for weighting models based on the likelihood function, such as AIC weights or the Bayes factor (i.e. point #1 of the above scheme) unless there is confidence in the likelihood function used. Therefore, in many cases, best practice is to use a form of "Delphi method" to assign weights to hypotheses, for example by using expert judgment to assign "high", "medium" and "low" weights to each alternative hypothesis (perhaps using an approach such as that outlined in steps 2–4 of the scheme above). From a practical point of view, it would not be unreasonable to ignore hypotheses that are assigned "low" weight when conducting analyses using ecosystem models, to save time and resources.

Best practice: Consideration of model structure uncertainty involves first identifying alternative qualitative hypotheses for all of the processes considered likely to have an important impact on the model outputs, *formulating these hypotheses mathematically (or as the values of parameters of a general relationship), and then assigning weights to each hypothesis.*

Features to include in closed loop simulations

Closed loop simulations (as used in MSE exercises) involve management actions being set using a harvest strategy that is based on data generated by an operating model (Walters, 1986). These types of simulations therefore account for the feedback arising from an improved understanding of the system state resulting from the collection of future data. Simulations into the future can be based on constant catch strategies, constant fishing mortality strategies, or some other form of harvest strategy. In all cases, account needs to be taken of parameter, model, and implementation uncertainty when conducting such simulations. However, account also needs to be taken of error in the results from stock assessments when harvest strategies that involve the application of a stock assessment method are being simulated.

Ideally, the harvest strategy that is used to determine management actions should be that which will be actually applied in practice. Therefore, if the harvest strategy involves the application of a particular stock assessment method, best practice is to simulate the application of that stock assessment method. However, simulation of some actual stock assessment methods can be computationally prohibitive if either the stock assessment method is very complicated (e.g. is based on Bayesian techniques) or if there are several species for which the harvest strategy involves the application of a stock assessment method. In such cases, it is either necessary to approximate the results from the stock assessment method or to consider basing tactical advice on simpler harvest strategies that can be fully simulated (and hence comprehensively evaluated).

Assuming an *a priori* level of assessment error (e.g. the estimate of biomass is the true biomass multiplied by log-normally-distributed error with a coefficient of variation of 20 percent) should be avoided. Rather, if a stock assessment method is to be approximated, best practice is to use a limited number of simulations in which the actual stock assessment method is used to determine the properties of the estimates from that method. When approximating stock assessment methods, consideration should be given to the possibility of assessment bias (systematic differences between the true (i.e. operating model) and estimated (i.e. assessment method) stock biomass) as well as imprecision in the estimates. In general, it should be expected that assessment errors will be temporally auto-correlated (i.e. if the estimate of biomass is larger than the true value this year, it should be expected that the estimate of biomass next year will also exceed the true value).

If the stock assessment method is to be based on indices of abundance, consideration should be given to the possibility that there is a trend in the catchability coefficient. This is particularly important if the assessment method is based on commercial fishery CPUE data. Data generated from operating models should account for all sources of error. Specifically, it is often the case that the sampling error associated with indices of abundance under-estimate the true extent of uncertainty because sampling error does not capture, for example, variation in catchability and/or availability. In this case, the error variance used when generating the indices of abundance should reflect both sampling error and the impact of additional sources of uncertainty. One way to determine the extent of additional variance is to subtract the extent of sampling error from the variance of the residuals arising from fitting the model to the historical data for the index to be simulated.

The Australian AMS project (Fulton, Smith and Smith, 2007) is an example of a case in which computational constraints led to the biomass estimates on which harvest strategies were based being generated by adding noise to the true biomasses. A series of simulations was conducted comparing the actual assessment methods and simply adding error to the true values in terms of both estimates of biomass and TACs. There were differences in the estimates of biomass (by as much as 15 percent), but the differences in the TACs were less than 10 percent. Given that the extent of TAC change was constrained in the simulations, and the other sources of error and variation in the operating model, these differences were considered acceptable.

Punt and Butterworth (1995) simulated the use of the assessment model on which the (then) actual harvest strategy for the Cape hakes (*Merluccius capensis* and *M. paradoxus*) was based in computations using their Minimum Realistic Model of the hake-seal-fishery system. In contrast, a simpler harvest strategy (a constant fishing mortality strategy based on the assumption of perfect information) was used for the other predatory fish component of that model, and the simulated future harvests of Cape fur seals were treated as alternative management scenarios.

Best practice: Evaluation of feedback control harvest strategies should involve simulating the scheme (including any stock assessment method) that is likely to be used in practice to determine management actions.

Implementation uncertainty

Predictions of ecosystem responses to management measures depend on the successful implementation of the recommended measures i.e. the degree of compliance with management plans. Uncertainties about the implementation arise from a variety of practices both by fishers and managers. There is a substantial current focus on this as evidenced, for example, by the FAO International Plan of Action to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated fishing (IPOA–IUU). Illegal activities include fishers' use of illegal gear such as undersize mesh in nets, removal of bycatch reduction devices or the use of explosives or poisons. Other violations may include fishing in prohibited areas, e.g. MPA's or management time/area closures, or taking of undersized or protected individuals. Misreporting or non-reporting of catches (by species, by area, by size) will, in addition to undermining the management measures, have an additional impact by contaminating the fishery data used in the model. In addition, fisheries may be unregulated, or essentially so, due to lack of either political will or resources to adequately enforce the provisions of the plan.

Implementation failures introduce biases in fishery data which will impact assessment and tactical models. They also create biases in the expected impacts of simulated management measures within an MSE.

Implementation uncertainty needs to be linked to consideration of fleet dynamics and is largely driven by, and must be included in, economic considerations.

Best practice: Identify, and quantify if possible, the type and extent of implementation failures to be expected through consultations including fisheries managers and knowledgeable fishers during the model development process.

4.2.2.2 Use and outputs

Social and economic outputs

Economic and social outputs are required to relate measures of fishery performance to management objectives (e.g. employment and foreign exchange). They can also be used to include consideration of measures of fishing success generated in a model as factors explaining fishers' decision-making, e.g. increased or decreased investment or effort. The social and economic components are generally poorly or incompletely considered, if at all in modelling at present. Implementations of economic models within ecosystem models are often subject to overly simple assumptions or lack required data, and social considerations in models are, in general, even less well-developed and harder to simulate. There is a need to have economic experts collaborating with fisheries ecologists when designing a model incorporating economic factors and similarly social experts for social factors.

Economic outputs will have an important link to fleet dynamics in strategic models or MSE.

Best practice: Have economic and social experts collaborating with fisheries ecologists when integrating economic and social factors into ecosystem models.

Ease of modularization

The concept of "modularity" has several important meanings in the context of computer programs to implement ecosystem models. In terms of computer programming, object-oriented design is considered best practice, and this includes modular programming. Such design can be handled within many existing programming languages, and is currently implemented by models such as EwE, Gadget and InVitro.

For ecosystem modelling, modularity in the sense that each attribute of an ecosystem model is handled separately is important. When a feature such as a growth curve is implemented only in one place in a computer program (as a "module"), this module can be extended to include any one of several growth functions. The same applies to several other functions, such as those describing recruitment and relationships for species interactions.

This basic approach implies that the appropriate functions can be used for each species or stock in a model and different model structures can be explored, even when there is no simple parametric function which links these structures. For example, within the same framework one can test whether assuming growth depends on the biomass of a prey species is better than assuming that average growth follows a specified growth curve.

Object–oriented programming also facilitates linkages between different models, e.g., linking a Nutrient-Phytoplankton-Zooplankton (NPZ) model to an upper-trophic level. This practice should be encouraged.

Best practice: Object-oriented design in the programming of ecosystem models.

Ease of use and communication

An analyst or modeller should keep in mind that models and quantitative analysis tools can be difficult to use and understand. For such tools to be useful, the analyst/modeller must make principled quantitative arguments about how a system operates while making the methods and results accessible to colleagues, co-developers, and stakeholders. While some multispecies models may be quite small and easy to demonstrate and explain in full, the task grows increasingly difficult as the model complexity and size grows. This means that supporting materials for the model, such as documentation and freely accessible code, must be available if it is to be reviewed and understood. Not only is it good to avoid a "black box" impression of a model, it is also important for removing suspicion regarding the model. Distrust, generated through a lack of clarity or understanding, can result in the potential insights that a model provides being ignored by, or lost on, key stakeholder groups (including ecologists). In addition to documentation, model clarity and familiarity can be increased via stakeholder inclusion and interaction during model scoping, evaluation and publication (consideration of co-authorship with data providers is highly recommended). It is always a good idea to communicate with data providers and bring modellers and ecologists or other stakeholders together so that there can be discussion of the model and whether it is meeting stakeholder objectives. A good stakeholder relationship makes it much easier to communicate trade-offs, both in terms of trade-offs highlighted by the model, but also in terms of model structure itself (though it is not possible or desirable to include every detail that may be identified in early stakeholder conceptual models).

Models are tools and as such are prone to misuse. While highly complex models are difficult to use and mistakes in understanding can lead to misuse, models that are easier to use are often more prone to misuse as people find them easy to execute without careful thought. Good documentation can help avoid this problem. In addition, models that are relatively easy to implement allow for a smoother learning curve and will encourage more users (e.g. ecologists) to begin model explorations. While this may be a springboard for further model development and expansion, or a means to an end in its own right, the wider benefit is that a larger clientele encourages more cooperation between groups of data providers as there is a broader shared understanding upon which to build. In this way ease of use is key to education and understanding.

A final concern when considering model communication is whether outputs reflect uncertainty. That is, is the model output designed so that the numbers are meaningful and do not create a false sense of confidence in the results. Regardless of the currency used within the model it is typically much better to present relative performance than absolute values. As confidence intervals for parameters are often not particularly meaningful to stakeholders (and may be ignored by them) it is more effective to present the range of outcomes reached over the scenario(s).

Best practice: Model developers must keep in mind that communicating with colleagues (ecologists, etc.) to develop models and communicating system trade-offs with stakeholders are essential for developing models that are valued and useful for EAF. Ease of use is desirable for education and understanding, but may lead to misuse. To achieve these communication goals and avoid misuse, modellers should provide models with 1) clear documentation, 2) freely accessible source code, and 3) effective model input and output interface systems.

4.3 DATA DEMANDS OF ECOSYSTEM INTERACTION MODELLING

Data demands for modelling ecosystem interactions will vary greatly depending on the questions asked. Scoping the data requirements will be an iterative process involving: i) examining the question and hypotheses; ii) determining taxonomic/spatial/temporal resolution and coverage of data and the model needed to distinguish the hypotheses; iii) collecting or gathering the data; and iv) re-assessing the hypotheses (Figure 2).

At a conceptual level, even an extremely small amount of data, collected with limited resources, may build an understanding of the important interactions that will need to be modelled to develop an ecosystem approach for a particular region. Data gaps can be identified as areas of uncertainty, and guide the data gathering required constraining the problem or distinguishing between hypotheses. A "finished" model based upon limited data can be used (for example, through a sensitivity analysis) to determine the most critical data gaps or most likely sources of strong ecosystem interactions, even if the magnitudes or directions of the interactions are not known.

An important component of the data gathering process will be facilitating access to sources of disparate data which have not been collected with a single purpose in mind. This should be seen as an opportunity for assessing available data and improving availability and analysis. An important part of this process is data "pedigree", i.e., the careful documentation of data quality and sources (see Box 2). Documentation should include consultation with data or ecological experts on the representativeness of the data. Especially in examining interspecies interactions, questions to be asked involve what bias might arise from limited geographic or seasonal coverage. A part of documentation involves choosing data based on specificity. Data should preferably be collected from the ecosystem and species under study before choosing parameter values from other species or ecosystems (see Box 2).

Box 2

Data "pedigree" in EwE

It is a daunting, intensive, and perhaps impossible task to describe the probability distributions for all input parameters in a complex ecosystem model. To facilitate this task and to make the process more transparent, EwE implements a simplified approach (the "pedigree" routine) that serves the dual purpose of describing data origin and assigning confidence intervals to data based on their origin.

The pedigree is a coded statement (see below for examples), which categorizes the origin of a given input (i.e., the type of data on which it is based), and specifies an approximate uncertainty associated with the type of input. The key criterion is that input estimated from local data is as a rule better than input from data from elsewhere, be it a "guesstimate", derived from empirical relationships or derived from other models.

Specifying the "pedigree" of input data is useful particularly to make users aware of the danger of parameterising a model mainly from input taken from other models, pertaining to different areas and/or periods, to provide parameter-ranges for analysis of uncertainty, and to provide an overview of the model parameter "quality".

Box 2 *continued*

Example of elaborated criteria for ranking data quality ("pedigree") for biomass, production rate (P/B), consumption rate (Q/B), catch, and diet input parameters between multiple types of models (Aydin *et al.,* in press). 1=best data, 8 = worst.

Rank & corresponding data characteristics	
1. Data are established and substantial, include more than one independent method (from which best method is selected) with resolution on multiple spatial scales.	
2. Data are direct estimate but with limited coverage/corroboration, or established regional estimates are available while subregional resolution is poor.	
3. Data are proxies, where such proxies may have known but consistent bias.	
4. Direct estimate or proxy with high variation/limited confidence or incomplete coverage.	
Biomass and Catch	**P/B, Q/B, and Diet**
5. Estimate requires inclusion of highly uncertain scaling factors or extrapolation.	5. Estimation based on same species but in "historical" time period, or a general model specific to the area.
6. Historical and/or single study only, not overlapping in area or time.	6. For P/B or Q/B, general life-history proxies. For diets, same species in a neighbouring region, or similar species in the same region.
7. Requires selection between multiple incomplete sources with wide range.	7. General literature review from wide range of species, or outside of the region.
8. No estimate available (estimated by model itself with no prior information)	8. Functional group represents multiple species with diverse life history traits.

To move to a strategic or tactical level of advice, a reasonable amount and range of data will be required, but determining what constitutes "reasonable" will be an iterative process which includes formal fitting procedures (see section 4.2.2.1), and also clarification from managers concerning the questions they require to be addressed. For example, an initial trophic model may indicate that for some species, habitat is more important than predation, and thus further diet collections would not be necessary, while for other species, a simple model may determine that predation is probably a key controlling factor so that further data collection is required to provide good projections. However, care must be taken that such scoping is not merely confirming preconceived notions built into the model structure.

For best practices, a distinction should be made between conceptual uses and providing strategic/tactical advice. For conceptual understanding, even extremely limited data may be sufficient, provided expectations and uncertainties are documented and described as above. For strategic or tactical advice, validation of models/hypotheses should be required. In this case sufficient data to parameterize processes and to appropriately quantify relative differences between model components is a necessary prerequisite for the associated models.

Time trends of data become important for data fitting and making predictions, with special focus on contrasts that occur in the data over time. For interspecies interactions, a balance will need to be found between synoptic studies (a single survey of many species at a single time and place) compared to isolated but extended time series (surveys of a single species over time). Differences in predator diets between two regimes can be extremely informative for fitting functional responses and interaction terms, but only if both the predator and prey are surveyed. Measurements of two or three strongly interacting species at two or three points in time may be better than measurements of many species at one point in time, and also better than measurements of a single species over a long time period.

Further, best practices demand that the models be used to guide and adapt future data gathering, for example for designing a balanced survey, collection, or experiment, or for identifying specific data or scales of resolution that would be needed to distinguish between specific model-generated hypotheses. While opportunistic use of patchy historical data may be informative, especially in cases with strong contrasts, data collection should proceed with as balanced a design as possible with existing resources, as specifically guided by the models. Specification of the data requirements for a model should be clear as to the extent and quality of the collections and the potential biases arising from limited coverage.

Types of data that will tend to be required, or at least considered, for collection are as follows:

1. **Removals**: Human interactions remain a dominant issue and should remain a focus of data procurement, and the requirements are similar to those for traditional stock assessments. Further, for ecosystem interactions, incidental/bycatch (e.g. interactions between gears and non-target/non-assessed species) may take a central role and should be

a focus of new data gathering, analysis, and reporting of uncertainty (especially with respect to implementation error).

2. **Indices of abundance**: In general, best practices for single-species stock assessments apply equally to ecosystem models, with the addition that indices of non-target species (e.g. plankton, or upper trophic levels) may be extremely desirable. Time trends in upper trophic levels (e.g. birds) may serve as indicators of forage abundance where direct data are not available. A common missing piece in many ecosystem models is time series of target species: forage fish, small squid, and predatory zooplankton (e.g. krill) abundances. Many models are sensitive to forage fish variation, while data on temporal variability in these species are extremely limited. Furthermore, the value of long time series of comparable indices of abundance of key target species for use to determine unbiased estimates of trend cannot be overemphasized.
3. **Vital rates** (production, mortality, consumption, growth, and migration): Measuring vital rates is a long-standing problem in stock assessment, and the difficulty and requirements are compounded in modelling interactions. Interactions may relate closely to variation in vital rates (e.g. habitat-based, climate, or predation-based mortality and growth), yet taking direct measurements is extremely costly and subject to error and bias. When direct measurements are not available, sources from other studies may be used and ranked or error ranges applied according to specificity (see Box 2) with the strong expectation that formal fitting procedures shall be needed to adjust and validate these parameters.
4. **Diet/interaction data**: Measures of interaction strength between components are the fundamental addition which extends stock assessment modelling into ecosystem interactions. A crucial and little understood aspect relates to predation mortality on juvenile fishes, where there is often only very poor or little information about the identity of predators. Diet interactions are perhaps the most well-known, but interaction terms include relationships, indices of habitat, climate, or other driving variables. For the last, it is important to limit "data dredging" to prevent fitting to spurious correlations. While diet data have been collected and published throughout the history of ecology, standard practices in the statistical fitting of these data to multispecies models, including sample sizes, bias, coverage, and variability, may vary from system to system, and species to species,

and are an area of active investigation (e.g. Jurado-Molina, Livingston and Ianelli, 2005). In general, the key to best practice is to design the data collection to be adaptable and iterative between model fitting and improving data collection.

5. BEST PRACTICES IN ECOSYSTEM MODELLING

5.1 BEST PRACTICES

The previous chapters included discussion of model types and the role of models for developing understanding of ecosystem processes and for strategic and tactical management. There is a continuum of model categories, and there is no law of nature stating that all steps in the conceptual – strategic – tactical model sequence must be visited in all cases. Indeed, valuable information for informing an EAF can be obtained from a simple conceptual model or from a static food web model, and for many nations adopting such an approach may be a first, worthwhile and feasible strategy.

Likewise, these guidelines do not state that complex is better. Valuable insight may be gained from simple models; thus, for instance, adding spatial structure is not necessary if the questions to be addressed in a given management context do not require explicit spatial representation. If a simple model can be developed to address a given question it may well be better than developing a more complex model. A problem may be that often it is not known *a priori* if a model indeed is reaching the minimum required level of complexity to allow reliable inferences to be drawn. Consideration of the question posed based on alternative conceptual model formulations (and on more refined models if needed), may provide guidance for evaluating the degree of detail required.

In all cases a best practice modelling approach must include specification, implementation, evaluation, reporting and review steps. Model scoping undertaken during model specification must include the iterative construction of conceptual models that are used to define the relevant subsystem to be modelled. Once this subsystem is identified its representation in the final model must be defined based on the question being considered, available data, the important system features (including forcing) and the appropriate scales (regarding space, time, taxonomic and fisheries resolution) and process representations.

Table 2 shows recommended best practices for modelling. These are not benchmarks, but rather are an achievable set of practices that should guide thinking as to the importance of different model attributes and suggested approaches for handling each of these. It is recommended that these practices should be followed to the extent possible.

5.2 STRATEGIC MODEL CONSIDERATIONS WITH REGARD TO KEY ATTRIBUTES

Table 2 summarizes some of the key attributes to be considered in model development and suggests the current best practice for handling each of these, noting that this may not be practically achievable in many circumstances.

TABLE 2
Best practices for developing models for an ecosystem approach to fisheries

Numbers as in section 4.2	Consideration in model development	Best practice approach
4.2.1.1	***Model aggregation***	
	How many species or groups?	Aggregate based on shared characteristics of the species and omit the least important to keep the foodweb tractable.
	Include age, size or stage structure of the species of interest?	Include if this feature is of importance to the issue of concern and could affect recommendations for management.
4.2.1.2	***Spatial considerations***	
	Include spatial structure?	Include to the degree required to address the management issues and ecological aspects of concern.
	Include seasonal and temporal structure?	Include where there are large seasonal differences in species' movement or production that are important for the management issues and ecological aspects of concern.
	Defining boundary conditions	Base boundaries on biological rather than anthropogenic considerations such as national boundaries.
	Is fishery harvesting more than one stock of a particular species?	Modelling needs to distinguish such different stocks when the harvesting practice is such that it might impact these stocks to different extents; this will necessitate spatially structured models.
	Distinguish different fleets?	Important in the context of provision of advice at the tactical level, if for the same mass of catch, the fleets have substantially different impacts on target and bycatch species or on the habitat and/or when such distinctions have important social or economic ramifications.

4.2.1.3	***Model components***	
	Explicitly represent primary productivity and nutrient cycling	This may only be necessary when bottom-up forces or lower trophic levels are of key concern. Inclusion of these processes can be highly informative for some strategic modelling exercises.
	How to model recruitment?	Recruitment may be included either as an emergent property or as a derived relationship (which should not be based on uncritical correlation studies of recruitment and environmental parameters). Recruitment variability is likely to be important for tactical and risk analyses, but is not a strict requirement for many strategic models.
	How to model movement?	This includes testing sensitivity to a range of movement hypotheses and, where possible, parameterising movement matrices by fitting the associated model to data. If decision rules are used to drive movement, attention should be focused on whether the resultant changes in distribution are sensible.
	Explicitly consider fleet dynamics?	It is important to consider if substantial changes to the spatial distribution of fishing may result from, for example, the declaration of an MPA. The population model must include spatial components in these circumstances, and it may be necessary to develop a model of the manner in which fishing effort patterns will change in response.
4.2.1.4	***Predator-prey interactions***	
	How much detail in representing predator-prey interactions?	Represent as bi-directional unless strong motivation can be provided that it is adequate to include a one-way interaction only. Bi-directional interactions are desirable at the strategic level, but may not be relevant at the tactical level if the interaction strength in question is low.
	Which functional response?	Acknowledge the paramount importance of the appropriate form for functional responses (the prey-predator interaction term) and feeding selectivities/suitabilities, and test sensitivity and robustness to alternative forms.

4.2.1.5	***External forcing***	
	Include environmental forcing?	Carefully consider whether environmental forcing is required to capture system dynamics. Care must be exercised in selecting the basis to generate future forcing for use in predictions and closed loop simulations.
	Other process error considerations?	Other process error, arising from natural variation in model parameters, needs to be included in projections, whether they be strategic or tactical, when that variation contributes substantially to uncertainty in model outcomes.
	Other anthropogenic forcing?	Influence on shallow coastal and estuarine systems should be considered in conceptual models, and if found to lead to appreciable pressures on the system then this forcing should be included empirically (e.g. simply as a forcing term) in any strategic models and MSEs for the system.
4.2.1.6	***Model structure***	
	Alternative stable states?	Strategic models in particular need to include forecasting of the consequences of environmental change and must contain the capacity (e.g. flexibility in choice of functional relations) to allow for plausible phase shifts, either directly (in accordance with past observations) or as an emergent property of the functions of the model. Even if such a functional form is used, it must be recognized that, until a threshold is crossed by the system, it may not be possible to parameterize the threshold point: given such uncertainty, possible thresholds may need to be evaluated on either a theoretical or an empirical basis.
4.2.1.7	***Technical and non-trophic interactions***	
	Technical interactions	Technical interactions need to be included in a model if the question that the model is aiming to address relates to the direct impact of a fishery on another species or habitat.
	Non-trophic interactions	If conceptual system understanding indicates that a non-trophic interaction is a critical determinant of the dynamic of interest (e.g. biomass or abundance of a target group), or if management could be based around this interaction, then its inclusion is highly desirable.

4.2.2.1	***Dealing with uncertainty***	
	Should the model be fit to data?	Fitting to data is best practice, and this requires careful specification of likelihoods.
	Taking account of parameter uncertainty	• Explicitly evaluate the effects of uncertainties in model parameters for management advice. • Bayesian methods and bootstrapping are considered best practice for quantifying parameter uncertainties in extended single-species models and MRMs. • Improving current practices for more complex models requires: 1) explicit accounting of the number of parameters that are being estimated and the number fixed, 2) qualitative estimates of the uncertainty in every parameter, and 3) sensitivity analyses. • For mass-balance/static models: 1) develop and fully document a formal data "pedigree" (quality ranking); 2) sensitivity analyses may be conducted using available routines. • For dynamic models: 1) fit to as much data as possible using appropriate likelihood structures; 2) be clear about both potential biases arising from fixing parameters, as well as fully reporting error ranges resulting from freeing parameters; 3) in cases of fixing parameter values, additional sensitivity analyses should be used to assess model sensitivity to the assumptions; and 4) use results of sensitivity analyses to guide future data collections and the continuation of key time series.
	Model structure uncertainty	Identify alternative qualitative hypotheses for all of the processes considered likely to have an important impact on the model outputs and then formulate these hypotheses mathematically (or as the values for parameters of a general relationship), assigning weights to each hypothesis.
	What features to include in closed loop simulations?	Evaluation of feedback control harvest strategies should involve simulating the scheme (including any stock assessment method) that is likely to be used in practice to determine management actions.
	Implementation uncertainty	Identify, and quantify if possible, the type and extent of implementation failures to be expected through consultations including fisheries managers and knowledgeable fishers during the model development process.

4.2.2.2	***Use and outputs***	
	Social and economic outputs	Have economic and social experts collaborate with fisheries ecologists when integrating economic and social factors into ecosystem models.
	Ease of modularization	Object-oriented design in the programming of ecosystem models.
	Ease of use and communication	• Provide models with 1) clear documentation, 2) freely accessible source code, and 3) effective model input and output interface systems. • Clear communication of model outputs, including tradeoffs, to stakeholders. • Documentation and source code must be freely available to allow for review and understanding of the model. Using existing models can be of great help in learning, but careful thought is required when using a pre-existing model so that the tool is not misused.

5.3 WHAT TO DO WHEN THERE IS INSUFFICIENT DATA, INFORMATION AND EXPERTISE

The processes of developing and evaluating ecosystem models, conducting a MSE and deriving potentially simpler tactical model demands time, expertise and data resources that are simply unavailable in many parts of the world. Thus the identified best-practices (Table 2) are not a feasible option in many cases. In spite of this, the officials responsible in such places need to recognize and account for ecosystem effects in their decision-making regarding fisheries and coastal development, and many have recognized the need but lack the capacity to respond fully.

A process is needed to make the information about, and benefits of, ecosystem models available in resource and data limited situations. This should be based on the compilation of a library of completed ecosystem models and MSEs with analyses of applicability and limitations. Meta-data with each model should provide a basis for classification which facilitates identification of closest matches to a given new situation. There should also be an assessment of strengths in each case, the areas of greatest uncertainty and the risks associated with decisions based on the model. Making use of the model library to provide ecosystem-based advice in a new situation would require a modelling expert to work in collaboration with knowledgeable local staff and stakeholders. Box 3 provides a hypothetical example of a procedure that could be adopted in a data limited situation.

The first steps would include compiling an inventory of the ecosystem components, and identifying the competing interests, research activities, management activities and agencies, and stakeholders. Issues and questions are then identified and a conceptual model must be constructed based on the ecosystem structure and the issues identified.

At this point there should be sufficient information to consult the library of ecosystem models, identify several appropriate analogues and, from them, potential management responses to address the issues identified. A draft management plan should be developed with a strong emphasis on precautionary application of the proposed measures. Because the models are being applied by analogy, the associated uncertainties are expected to be larger than those identified in the model library. A final, valuable step would be to have the draft plan reviewed by one or more external experts in ecosystem modelling and management to identify hazard points or provide a risk analysis.

Box 3

Accounting for ecosystem considerations in a hypothetical small island developing State

A small island developing State is in the process of developing an ecosystem-based fisheries management plan. A round of discussions with the fisheries and coastal zone management officials reveals an obvious problem of reef degradation including lack of large fish and algal over-growth. Stakeholder consultations confirm this and provide a more detailed understanding of the fishing practices including gears used, effort expended, and preferred fishing areas. A species list including relative abundance estimates from current and historically recorded catches and related information is compiled and augmented from published and online sources as well as fishers' and other stakeholders' knowledge.

In collaboration with an ecosystem modelling expert, the information obtained thus far on ecosystem components and issues is developed into a conceptual model. The ecosystem model library is searched for analogous systems, based on locality, size, latitude, species composition, fisheries and identified issues. Ideally several models will be available as useful analogues, and these models together with the associated results and analyses are obtained. A careful evaluation of the model and MSE results obtained from the library is used to identify appropriate (and inappropriate) management measures for the case at hand. Because the applicability of the management measures is based on analogy, the uncertainties are greater than those represented in the model and MSE results. Thus, the managers must work with stakeholders to be cautious in using the model information when developing their management plans.

6. CONCLUDING REMARKS

Ecosystem model applications range from basic understanding to providing information for making tactical management decisions. Such decisions will be enhanced by exploring the same issue with different models; confidence in the decisions will increase when the models independently converge on the same management decisions and when uncertainties in the results have been adequately considered. As such, the development of alternative models is encouraged. However, the uncertainties that usually arise in model results can lead to conflicting advice on which management decisions might be preferable. In undertaking the process of evaluating model results, models need to be appropriately weighted for their plausibility so that results from the most plausible models are given more weight in the decision-making process than those that are less plausible.

In practice, because ecosystem/multispecies models can be complex and data and resources for data collection and model development are usually limited, the actual uncertainty involved in model application may be greater than would ideally be tolerated. Nevertheless, an important principle for scientists and managers is that decisions have to be made and actions implemented to ensure sustainable and optimal utilization of marine living resources. These decisions must be informed by the best available scientific advice and, in the context of EAF, this scientific advice must include ecosystem considerations. Ecosystem models, adhering as far as possible to the best practices described here, will frequently be the best sources of such information, and can lead to advice that rests on explicit and principled arguments. In their absence, managers and decision-makers will have no choice but to fall back on their own mental models which may frequently be subjective, untested and incomplete, a situation which is clearly to be avoided.

Ecosystem models are not at the stage where a single such model could be selected as a "management" model (i.e. within, say, a management procedure) and reliably used at the tactical level to provide management recommendations in a particular case. However, the use for this purpose of simple models with an ecosystem foundation could become more widespread in the near future. Such a foundation would be provided by evaluating these simpler "management" models using MSE, where the operating models reflecting alternative possible underlying dynamics that are used in this evaluation process would include a range of ecosystem models. These ecosystem models would incorporate foodweb and other ecosystem processes to be able to ascertain whether ecosystem objectives wider than purely target species concerns should be met were the simpler management models to be applied in practice. The tactical management models would not themselves necessarily incorporate these ecosystem features, but might be single species assessment models linked to control rules whose parameters are tuned to meet ecosystem as well as target species objectives in the evaluation process. Alternatively they could comprise simple empirical decision rules using both target species and ecosystem indicators as inputs.

The following step in this development process would be consideration of the use of less complex forms of ecosystem models, such as simpler examples of Minimum Realistic Models, for these tactical management models. These simple models could prove to be viable options, but they will require longer development times because they will need a more detailed associated MSE process.

7. REFERENCES

Aydin, K.Y., McFarlane, G.A., King, J.R., Megrey, B.A., & Myers, K.W. 2005. Linking oceanic food webs to coastal production and growth rates of Pacific salmon (*Oncorhynchus* spp.), using models on three scales. Deep-Sea Res II: 52:757–780.

Aydin, K., Gaichas, S., Ortiz, I., Kinzey D. & Friday, N. (in press). A comparison of the Bering Sea, Gulf of Alaska, and Aleutian Islands large marine ecosystems through food web modelling. NOAA NMFS Technical Memorandum.

Bakun, A. 2006. Wasp-waist populations and marine ecosystem dynamics: navigating the "predator pit" topographies. Progress in Oceanography. 68:271–288.

Boldt, J. 2006. Ecosystem Considerations for 2007. Appendix C of the Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, AK 99501. 360p. http://www.afsc.noaa.gov/refm/stocks/assessments.htm.

Butterworth, D.S., Punt, A.E. & Smith, A.D.M. 1996. On plausible hypotheses and their weighting, with implications for selection between variants of the Revised Management Procedure. Reports of the International Whaling Commission 46:637–640.

Butterworth, D.S., Cochrane, K.L. & De Oliveira, J.A.A. 1997. Management procedures: a better way to manage fisheries? The South African experience. In: Pikitch, E.L., Huppert, D.D. and M.P. Sissenwine (Eds). American Fisheries Society Symposium no. 20, Bethesda, Maryland: pp. 83–90.

Christensen, V. & Walter, C.J. 2005. Using ecosystem modelling for fisheries management: Where are we? ICES C.M.: M19.

Christensen, V., Walters, C. & Pauly, D. 2005. Ecopath with Ecosim; A user's guide. Fisheries Centre, University of British Columbia, Canada. 154 p.

Ciannelli, L., Bailey, K.M., Chan, K.S., Belgrano, A. & Stenseth, N.C. 2005. Climate change causing phase transitions of walleye pollock (*Theragra chalcogramma*) recruitment dynamics, Proceedings of the Royal Society, Series B 272:1735–1743.

Dambacher, J., Fulton, E.A., Wild-Allen, K., Gray, R., Parslow, P.S., Matear, R. & Okey, T. 2007. Ecological Modelling Workshop Ecosystem Based Management Framework Project, Hobart November 28–29, 2006. CSIRO Marine and Atmospheric Research, Hobart.

Dambacher, J., Luh, H.-K., Li, H. & Rossignol, P. 2003. Qualitative stability and ambiguity in model ecosystems. American Naturalist, 161:876–888.

De Oliveira, J.A.A. & Butterworth, D.S. . 2004. Developing and refining a joint management procedure for the multispecies South African pelagic fishery. ICES Journal of Marine Sciences. 61:1432–1442.

Dill, L., Heithaus, M.R. & Walters, C.J. 2003. Behaviorally mediated indirect interactions in marine communities and their conservation implications. Ecology 84(5):1151–1157.

Dorn, M., Aydin, K., Barbeaux, S., Guttormsen, M., Megrey, B., Spalinger, K. & Wilkins, M. 2005. Gulf of Alaska Walleye Pollock. In: Appendix A of the Stock assessments and fishery evaluation reports for the groundfish resources of the Gulf of Alaska region. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, AK 99501. http://www.afsc.noaa.gov/refm/stocks/assessments.htm.

Duffy-Anderson, J.T., Baileya, K., Ciannellia, L., Cury, P., Belgranoc, A. & Stenseth, N.C. 2005. Phase transitions in marine fish recruitment processes. Ecological Complexity Vol.2(3):205–218.

Ellis, N. & Pantus, F. 2001. Management Strategy Modelling: Tools to evaluate trawl management strategies with respect to impacts on benthic biota within the Great Barrier Reef Marine Park area. CSIRO Marine Research, Cleveland, Australia. ISBN 0 643 06241 6.

FAO. 1996. Precautionary approach to capture fisheries and species introductions. FAO Technical Guidelines for Responsible Fisheries. No. 2. Rome, FAO. 1996. 54p.

FAO. 2003. Fisheries Management. 2. The ecosystem approach to fisheries. FAO Technical Guidelines for Responsible Fisheries. No. 4, Suppl. 2. Rome, FAO. 112 p.

Fulton, E.A., Smith, A.D.M. & Johnson, C.R. 2003. Effect of complexity on marine ecosystem models. Marine Ecology Progress Series, 253:1–16.

Fulton, E.A. 2001. The effects of model structure and complexity on the behaviour and performance of marine ecosystem models. School of Zoology, University of Tasmania, Hobart, Tasmania, PhD thesis.

Fulton, E.A., Hatfield, B., Althaus, F. & Sainsbury, K. 2006. NWS Benthic Habitat Dynamics Data and Models. North West Shelf Joint Environmental Management Study Technical Report – Vol 17, CSIRO, Hobart, Tasmania.

Fulton, E.A., Smith, A.D.M. & Smith, D.C. 2007. Alternative Management Strategies for Southeast Australian Commonwealth Fisheries: Stage 2: Quantitative Management Strategy Evaluation. Report to the Australian Fisheries Management Authority and the Fisheries Research and Development Corporation. 330pp.

Gray, R., Fulton, E.A., Little, L.R. & Scott, R. 2006. Operating Model Specification Within an Agent Based Framework. North West Shelf Joint Environmental Management Study Technical Report – Vol 16, CSIRO, Hobart, Tasmania.

Gulland, J.A. 1983. Species interactions. (pp. 186–201) In: Fish Stock Assessment: A manual of basic methods. Gulland, J.A. (Ed.) John Wiley and Sons, Chichester. 223 pp.

Hill, S.L., Watters, G.M., Punt, A.E., McAllister, M.K., LeQuere, C. & Turner, J. (in press). Model uncertainty in the ecosystem approach to fisheries. Fish and Fisheries 00:00–00.

Hobday, A.J. & Hartmann, K. 2006. Near real-time spatial management based on habitat predictions for a longline bycatch species. Fisheries Management and Ecology. 13(6):365-380.

Hollowed, A.B., Bax, N., Beamish, R., Collie, J., Fogarty, M., Livingston, P., Pope, J. & Rice, J.C. 2000. Are multispecies models an improvement on single-species models for measuring fishing impacts on marine ecosystems? ICES Journal of Marine Sciences, 57:707–719.

Horbow, J. 2005. The dynamics of Baltic fish stocks based on a multispecies stock production model. J. Appl. Ichthyology 21: 198-204.

Hsieh, Sugihara, C.H., Glaser, S.M., Lucas, A.J. & Sugihara, G. G. 2005. Distinguishing random environmental fluctuations from ecological catastrophes for the North Pacific Ocean. Nature 435, 336–340.

Jurado-Molina, J., Livingston, P.A. & Ianelli, J.N. 2005. Incorporating predation interactions in a statistical catch-at-age model for a predator-prey system in the eastern Bering Sea. *Can.J.Fish.Aquat.Sci.* Vol. 62 (8):1865–1873.

Laws, R.M. 1977. Seals and whales of the Southern Ocean. Phil. Trans. R. Soc. Lon. B, *Biol.Sci.* 279:81–96.

Livingston, P.A. & Methot, R.D. 1998. Incorporation of predation into a population assessment model of eastern Bering Sea Walleye Pollock. In Fishery Stock Assessment Models, Alaska Sea Grant College Program, AK-SG-98–01.

Murawski, S.A. 1984. Mixed-species yield-per-recruitment analyses accounting for technological interactions. Canadian Journal of Fisheries and Aquatic Sciences 41:897 916.

Overland, J.E., Percival, D.B., & Mofjeld, H.O. 2006. Regime shifts and red noise in the North Pacific. Deep Sea Research Part I: Oceanographic Research Papers. Vol. 53 (4):582–588.

Pacific Fishery Management Council. 1998. Amendment 8 ([to the northern anchovy fishery management plan] incorporating a name change to the coastal pelagic species fishery management plan. Pacific Fishery Mangement Council, Portland, OR, USA. Available at http://www.pcouncil.org/cps/cpsfmp.html).

Pinnegar, J.K., Blanchard, J.L., Mackinson, S., Scott, R.D. & Duplisea, D.E. 2005. Aggregation and removal of weak-links in food-web models: system stability and recovery from disturbance. Ecological Modelling, 184:229–248

Plagányi, É.E. 2004. Walking the bridge from single- to multispecies approaches in southern African fisheries management. PhD thesis, University of Cape Town, Rondebosch, South Africa.

Plagányi, É.E. 2007. Models for an ecosystem approach to fisheries. FAO Technical Paper. No. 477. Rome, FAO. 108p.

Preikshot, D. 2007. The influence of geographic scale, climate and trophic dynamics upon North Pacific oceanic ecosystem models. Ph.D. Thesis, University of British Columbia, Vancouver, xv+208 pp.

Punt, A.E. & Butterworth, D.S. 1995. The effects of future consumption by the Cape fur seal on catches and catch rates of the Cape hakes. 4. Modelling the biological interaction between Cape fur seals Arctocephalus pusillus pusillus and Cape hakes Merluccius capensis and M. paradoxus. South African Journal of Marine Science, 16:255–285

Rademeyer, R.A., Plagányi, É.E. & Butterworth, D.S. 2007. Tips and tricks in designing management procedures. ICES Journal of Marine Sciences. 64:618–625.

Ramos, J.A. 2000. Characteristics of foraging habitats and chick food provisioning by tropical Roseate Terns. Condor 120, 795–803.

Sainsbury, K.J. 1991. Application of an experimental approach to management of a tropical multispecies fishery with highly uncertain dynamics. ICES Marine Sciences Symposium. 193, 301–320.

Sainsbury, K.J., Campbell, R.A., Lindholm, R. & Whitelaw, A.W. 1997. Experimental management of an Australian multispecies fishery: examining the possibility of trawl-induced habitat modification. In: Global Trends: Fisheries Management. Pikitch, E.L., Huppert, D.D. & Sissenwine, M.P. (Eds). American Fisheries Society Symposium no. 20, Bethesda, Maryland: pp. 107–112.

Smith, A.D.M., Sainsbury, K.J. & Stevens, R.A. 1999. Implementing effective fisheries-management systems: management strategy evaluation and the Australian partnership approach. ICES Journal Marine Sciences. 56:967–979.

Tjelmeland, S. & Lindstrøm, U. 2005. An ecosystem element added to the assessment of Norwegian spring spawning herring: implementing predation by minke whales. ICES Journal of Marine Sciences 62:285–294.

Walters, C. & Kitchell, J.F. 2001. Cultivation/depensation effects on juvenile survival and recruitment: implications for the theory of fishing. Canadian Journal of Fisheries and Aquatic Sciences, 58:39–50.

Walters, C.J. 1986. Adaptive management of renewable resources. McMillan, New York [Reprint edition, Blackburn Press, 2001] s8u. 374 pp.

Watters, G.M., Hinke, J.T., Reid, K. & Hill, S. 2006. KPFM2, be careful what you ask for – you just might get it. Submission to SC-CAMLR WG-EMM, WG-EMM-06/22.

APPENDIX
TECHNICAL ASPECTS OF MODEL SPECIFICATION

The key steps in best practice model use are outlined in Figure 2 in the main part of these guidelines. While the stakeholder focus will almost undoubtedly be on model results the most critical steps from the modellers perspective are at the other end of the modelling loop during the model specification stage. It is key throughout the modelling loop, but never more so than during model specification, to maintain focus on the reason for modelling (the question to be addressed); this should be the primary guide during modelling decisions and will guide many of the specific decisions discussed further below. It will be a key consideration when identifying validation criteria used to check model performance and developing conceptual models.

Conceptual models are descriptive (often box and arrow) models of "how the system works" and should capture understanding of the system structure, interactions and drivers. This should be done in consultation with all stakeholders so that a complete set of their knowledge is captured in full (rather than just focus on selected "expert" advice). Where there is uncertainty or contention or speculation, where additional information or hypotheses can be proposed, but this should be presented to stakeholders for comment in an iterative process. This makes sure that stakeholders have a full understanding of what is potentially being represented or considered in later modelling steps.

Considerable effort should be put into developing conceptual models, as these are important for identifying relevant subsystems, appropriate resolutions and essential processes for inclusion in the final model. Without them there is considerable risk of adopting a model that is not fit for the purpose or inappropriately complex (either because it does not contain sufficient detail or it is excessive). Complexity is a key concern for ecosystem models as there is the potential for it to derail the modelling process. The work that has been done on model complexity shows that there is a humped relationship between model performance and complexity. Very simplistic models that do not capture the most critical interactions and components of the system are not helpful. At the other extreme very complex models are not necessarily useful either, as they are particularly impacted by uncertainty and the danger for large models to be prescriptive rather than predictive. Moreover there are considerable computation issues with using very large models. For all these reasons (computational cost, uncertainty and performance issues) inclusion of details beyond those absolutely required to address the specific issue in question should be avoided. Models are sufficiently detailed if all critical processes, drivers and components under scrutiny are captured.

The full form of the conceptual model does not automatically need to be taken further into prototype or full models, but it should be used to define relevant subsystems. It has been found that the use of qualitative modelling in the form of loop analysis (Dambacher *et al.*, 2003) has been extremely useful in making the transition from conceptual models to defining relevant subsystem and checking for the potential magnitude of model structure uncertainty. Using this approach positive and negative signs are assigned to each interaction in the conceptual model (trophic or non-trophic) using the conventions of signed diagraphs (from network theory). From this it can be determined (using the algebraic methods described by Dambacher *et al.*, 2003) what the gross results of system perturbations are likely to be, the identity of key system components and what the chance of significant structural error is if particular components are omitted or aggregated. Another benefit of this approach is that it is exceedingly flexible and allows for a rapid exploration of alternative model structures and configurations. The speed with which it can be done makes it quite easy to incorporate stakeholder advice and input (which can ultimately improve uptake and ensure stakeholder requirements are understood and being addressed). There are limits to the usefulness of this approach however. While it may still be used as an infrastructure for considering conceptual models no matter how large the network grows, its ability to inform on potential gross perturbation responses and structural error fails once you reach large model size, as the results become ambiguous. If such large systems need to be used, breaking the overall model into more subsystems, each of which is considered in turn, is more useful.

Relevant subsystems to be considered in later modelling stages should be drawn from the conceptual models, preferably using methods like loop analysis. From there the model specification should be completed using a clear, logical and consistent process. For each dimension or attribute of the model, the proposed complexity must be evaluated in terms of what contributions it makes to the model and overall analysis. This will dictate model structure and potentially the type of model used (see Figure 1 in the body of the Guidelines). In turn this will determine data requirements.

The list of potential model attributes is quite long (Chapter 4 of this report) and may be extended further by the specific question being asked of the model and details highlighted in conceptual models. Regardless of the list of attributes finally considered, a useful template for model definition would include the following:

1) Define question to be addressed

2) List important potential features

The list of attributes discussed in section 4 is an excellent starting point for this consideration, though question specific considerations should also be put forward (even if it is later decided to drop them for that case). As an example, the list of system components to be considered when defining important potential features of the model could include oceanography and climate, biogeochemistry, biogeography, biological components (dominant, keystone, vulnerable groups, age or size structuring required), links (trophic and otherwise, weights, multiple pathways), ecological processes, anthropogenic pressures and activities. The conceptual models and the following steps should be used to reduce the full list of potential components to those that (i) must and (ii) should be included in the final model.

3) Determine scales (and distribution) of each process and component

Spatial scale: To determine the spatial scale of the model start with the bounds of the core (or basic) domain (where the majority of ecological components exist or overlap). Then consider whether the range of the ecological components means that the domain needs to be extended to cover the majority of their distributions; and then whether it needs to be extended still further to capture seasonal or ontogenetic shifts, or whether these should be treated instead as import or export from the system or as transients.

Once the boundary is defined, decide on the internal spatial resolution (both vertical and horizontal), including whether internal division is required and if it is whether a homogeneous grid or heterogeneous network of polygons or sites (nodes) is used to define the system. If a heterogeneous patchwork of polygons or nodes are used then these should be tailored to match how system properties change within the system (do not put breaks through transition zones, instead put them to either side) and the strength and speed of the change. For instance, a few large boxes can be used to cover broad homogeneous areas (like the centre of large bays or open ocean areas), while a series of small boxes should be used to represent areas where conditions change rapidly (such as in shallow waters, estuaries and around seamounts).

The resolution used should be dictated by the ecological, environmental and anthropogenic (including jurisdictional) length scales, though it is not always necessary for all components of a model to use the same spatial resolution. It is important that the spatial resolution captures the major characteristics (e.g. physical fronts or boundaries) of the system, but be careful of defaulting to a fine resolution as that has a high computational cost and is often not necessary in the context of considering fisheries questions. Taking this to the extreme, this does not mean that multiple cells are automatically required, but if explicit internal subdivisions are omitted then any internal divisions that exist within the modelled area must be captured implicitly (e.g. by including an "inshore fish" and "offshore fish" group) or erroneous dynamics will result; this is because space is itself an important system resource, particularly where benthic groups are important. Trophic self-simplification of the food web (when 1 or more components of the web are consistently lost) is often a good indicator that spatial representation is overly restricted (Fulton, Smith and Johnson, 2003). An example of the features of the system that may be used to aid in the definition of the bounds and resolution are oceanographic properties (like currents and fronts), depth-structured (rivers, inshore, shelf, shelf break, slope, deep water), biological distributions (benthic, pelagic, oceanic vs. coastal, migratory groups), bottleneck locations, and major human input sites. Further pros and cons are discussed under section 4.2.1.2.

Temporal resolution: Decide on an appropriate temporal resolution (e.g. snapshot, tidal, daily, weekly, monthly, seasonal, and annual) and whether all system components are handled in the same way or whether different resolutions are used for different model parts. It may be the case that some groups are represented on finer time scales than others (e.g. lower trophic levels with faster rates of turnover maybe considered on a different temporal scale than higher trophic levels which change more slowly).

Also the form of handling of time should be considered, as it can have computational and numerical implications. The three most common forms are synchronous (when all components move in lock step together), adaptive (where the rate of instantaneous change of any one group can dictate the size of a sub-step which is then iterated and accumulated until the full time-step is reached), and asynchronous (where the time step shifts for each component depending on what actions they are taking, so attention can focus on critical events, and there is no requirement for all components to be using the same time step at any one time). Each has its advantages and disadvantages, but the common critical consideration is that whatever form is used it is imperative that no process bias is introduced by execution order (e.g. a prey group should not be allowed to consistently escape predators because of the time step it uses or the position it sits in within the model loop; effectively simultaneous execution is a must).

Taxonomic resolution: Decide on the taxonomic resolution to use (the number of groups and degree of aggregation). The number of groups is dictated by the question to be addressed and the relevant subsystems involved. If the subsystem is small enough (of order 10 or less) then explicit representation of all members is feasible (this is the case in many Minimum Realistic Models). Beyond this however some form of omission or aggregation is advisable in the majority of cases. It can be argued it is easier to be inclusive earlier and simplify later in the model development cycles, but this has its own drawbacks as large models have large computational and data requirements and are much harder to work with and often do not provide a clear improvement in performance. With this in mind it must be decided which components will be omitted ("chopped") and which will remain within the model, and of those remaining in the model which will be at the species level and which will be aggregated into functional groups ("lumped"). Functional groups (or guilds) should be defined based on predator and prey connections, size and rates, role, habitat use, behaviour, other non-trophic interactions, and spatial structure. Species are often pulled out separately due to human interest (targeting or conservation). Where possible use clear methods for this definition of group membership (e.g. clustering, regular colouration [network theory]). Aggregation beyond the level of functional groups is ill-advised in most cases, as it can lead to aberrant behaviour (such as markedly different recovery dynamics [Pinnegar *et al.*, 2005]). Omission of the least important groups is a better strategy if further simplification is necessary (Fulton, Smith and Johnson, 2003). As a guide, simplifying an ecological web of the subsystem to less than 20 to 25 percent of its original size is rarely beneficial, as representing the distinctions between large and small or mobile and sedentary groups is usually crucial (Fulton, Smith and Johnson, 2003). Moreover, while the inclusion of all system components is not necessary and there is much to recommend the Minimum Realistic Modelling approach (e.g. Punt and Butterworth, 1995), careful thought must be given to any skew in the taxonomic resolution. Models aggregated with emphasis placed on particular parts of the food-web (fish, marine-mammals or invertebrates) can exhibit markedly different system indices to models that have the same number of components overall but are more evenly resolved (Fulton, 2001; Pinnegar *et al.*, 2005). Models in which invertebrates, primary producers and detritus are heavily aggregated tend to be particularly resilient to system disturbances. Similarly models focusing on marine-mammals that heavily aggregated the rest of the food web also prove resilient to disturbance (due to the slow turnover rates and low biomasses of these top-predatory consumers compared to all other functional groups in the model). This is an important illustration of why care must be taken that model construction decisions do not ultimately dictate model results (in terms of its performance and predictions).

Once an initial web has been drawn up consider if substructure is required for each group (and the same resolution need not be used for each component). If the number of individuals is low or individual variation is important then representation of individuals may be necessary, more commonly pools or patches are sufficient. If there are major shifts in behaviour through the course of the life history then age or size-structure is probably needed.

Be alert to the implications of the connections and level of aggregation chosen and try alternatives (this is true for all of the model structure, but particularly so for this dimension of the model), as it can impact results (e.g. model responsiveness) quite strongly. For example in a system where the relevant subsystem has a single predator and two potential prey (which in turn may compete or consume each other), very different (often contradictory) results are obtained if the prey are kept separate and the connections are kept in place compared to when everything is reduced to a simple one-predator-one-prey application (e.g. Punt and Butterworth, 1995). At the other extreme of model complexity (where hundreds of groups are included in a model) parameter uncertainty can cause model performance to degrade significantly and result in pathological problems such as numerical instability. Moreover, even in those cases where they can be fit to data (and do fit well) and can be made numerically stable, the size of the system being represented can make the results ambiguous and of little value. Consequently, there must be a practical compromise between even handed detail, available data and the focus of the question.

Process resolution: Decide on what processes must be included and the detail to associate with the process. For instance, two-way coupling of predation (predator impacts prey and vice versa) will not always be necessary; as was the case when evaluating the impact of the fishing of prey fish on higher predators in South Africa (see section 4.2.1.4). Similarly explicit representation of primary productivity processes is often not required unless lower trophic levels and bottom-up forcing are significant components of the relevant subsystem. In that case this does not mean primary productivity can not still be represented via a forcing or other function (in fact it is always important to consider the way in which the production of basal groups is represented – see section 4.2.1.3).

Typically elaboration of process detail only occurs if it has a major impact on the process (e.g. nutrient, light and oxygen or space limitation of growth) but alternative forms should be tested. Consideration should also be given to whether other forms of model can be linked in to represent the form of the process without getting into minute detail. For instance it may be possible to use a statistical or other type of model to represent the impacts or gross form of a key process even if the fine mechanics cannot be represented explicitly.

Forcing: Decide on whether environmental forcing is required and which anthropogenic pressures need to be represented (either as an impact or in detail). Best practice on this can be found in section 4.2.1.5, but as a rule of thumb environmental drivers should be included if they are defining feature of the system's driving forces or current state. Anthropogenic processes to be considered include: inputs and pollution, tourism, shipping, clearing and coastal development, ports and dredging, economics and markets, management, ports and shipping, habitat degradation.

Fisheries model resolution: Decide which fisheries should be included, whether multiple fleets are required, and whether explicit splits between commercial, charter, artisanal and recreational sectors are needed. Also decide on the resolution of the fisheries model used, which may differ for different fleets in the same way as the taxonomic resolution could differ across ecological groups (e.g. a simple fixed F may be used in some models, or for some fleets in larger models, while full socio-economically driven fleet dynamics models are used in other models, or for fleets of particular interest or impact in larger models).

4) Final model form

At the end of this process the necessary components will need to be represented at the appropriate scales in prototype or final model(s). It is important to reemphasize here that there is no one single right model. All models have problems and it is best (where possible) to use a range of models that can address the question in different ways. These models will overlap in resolution or form, but can complement each other and provide more robust advice. There is a tension between prediction and understanding, but experience shows that uncertainty associated with ecosystem-level questions means the greatest leverage is gained by considering ensembles of models (multiple models of different form or with alternative structures or formulations). The continuum of model types from qualitative models (simple network models) to statistical inference models (determined from relationships in data) and quantitative mechanistic process models (where process understanding is captured in a decision tree or series of questions) can be used very effectively to inform each other and resolve different aspects of reality. Use of all of these model types is not an absolute requirement, but if they can all be used then it has been found that significant benefit can be drawn from their mutual implementation. This may sound imposing or overwhelming, but the final word on ecosystem modelling should be that of the precautionary principle. The absence of resources and information should not be used as a reason to do nothing. Matching the question to be addressed with the resources available is the best way forward. This means that even on relatively small resources great insight can be drawn from the use of qualitative or simplified quantitative multispecies and ecosystem models. The greatest contribution of the ecosystem modelling approach is to expand the thought of all involved (stakeholder and modeller) to consider larger system interactions than tightly confined single species considerations.

These guidelines were produced as an addition to the FAO Technical Guidelines for Responsible Fisheries No. 4, Suppl. 2 entitled *Fisheries management. The ecosystem approach to fisheries* (EAF). Applying EAF in management requires the application of scientific methods and tools that go beyond the single-species approaches that have been the main sources of scientific advice. These guidelines have been developed to assist users in the construction and application of ecosystem models for informing an EAF. It addresses all steps of the modelling process, encompassing scoping and specifying the model, implementation, evaluation and advice on how to present and use the outputs. The overall goal of the guidelines is to assist in ensuring that the best possible information and advice is generated from ecosystem models and used wisely in management.

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