A Review of Approaches to Fisheries Management Based on Ecosystem Considerations, With Particular Emphasis on Species Interactions

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Executive Summary

This report is organized around five interlocking questions regarding ecosystem based approaches to managing aquatic living resources, based on the understanding that management in natural resource contexts is the control of human intervention in ecosystems, and with the general focus of fished stocks that may be both predators and prey for other species in the system.

What are the distinctions between single-species and ecosystem-based management?

Single-species approaches to management are based in 19th century determinism in which one assumes that stocks can be viewed outside of their role in the ecosystem and that if one knows enough about the vital information concerning the sock, then it is possible to control the trajectory of the stock . The ecosystem-based approach to management recognizes that target stocks sit in food webs, that non-human predators of stocks are competitors with fishing, and that the abiotic environment is part of the milieu in which organisms live and fishing occurs. Ecosystem-based management attempts to bring more of the lessons from ecological sciences (such as thresholds, uncertainty and surprise) into consideration and focuses on both the target stock and non-target species. The ecosystem-based approach has three broad goals: a sustainable yield of products for human consumption and animal foods, maintenance of biodiversity, and protection from the effects of pollution and habitat degradation.

What problems have been attributed to failure of single-species and ecosystem-based management?

In general, three ratchet processes have resulted from single species approaches and have led to problems in fishery management. The first is *Odum's ratchet*, which is the failure to recognize that harvesting acts as a selective force on ecosystems by culling long-lived and slow growing stocks and individuals in favor of faster growing ones. The second is *Pauly's ratchet*, which is the tendency for scientists to relate changes in the ecosystem to what things were like at the start of their careers (so that accounts of higher levels of abundance are discounted as anecdotes). The third is *Ludwig's ratchet*, which is positive feedback between fishing mortality and over-capitalization.

Specific problems attributed to single-species approaches to management include stock collapses, shifts in trophic structure, habitat degradation, depletion of non-target species, and bioeconomic conundrums such as the optimality of extinction of a stock. The ecosystem-based approach to management is limited by the requirements of data and the requirements of several levels of administration, and considerable amounts of monitoring.

What goals or principles have been espoused for management of marine ecosystems?

At least a dozen set of principles concerning management of marine ecosystems were published in the last 22 years. Perhaps the most fundamental principle is the rejection of the Cornucopian myth concerning the ability of the oceans to provide unlimited resources.

Review of the published principles leads to the following synthesis:

1. Expect change in ecosystems and that the change will likely be due to multiple causes.

<u>The implication:</u> human intervention must be flexible rather than fixed and one must proceed cautiously when increasing the level of intervention. Humans must act like upper tropic level predators.
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2. Assume that human intervention will have an effect on parts of the ecosystem other than the target stock.

The implication is that monitoring is essential and that stock assessment of a targeted species is not enough. One must monitor its prey, competitors and predators.

3. Recognize that no amount of monitoring or sampling will fully reduce the uncertainty that one faces.

The implication: Sensible discussion must thus focus on the risk of overfishing (to both the target stock and other species) and the missed economic opportunities from lower levels of fishing mortality. A consensus cannot be achieved by averaging positions, but one may be able to apply methods of risk analysis.

4. Be prepared to answer Rothschild's five questions: how can we separate the effects of fishing from naturally induced variation; how can we predict the magnitudes of routine fluctuations in recruitment; how can we predict sustained changes or trends in abundance; how can we appraise the influence of pollution, eutrophication, or habitat modification on the abundance of particular fish stocks; what is the nature of interactions among species of fish in an ecosystem, with specific reference to predicting change in abundance of one species as a function of change in abundance of another species?

The implication is that management must be cognizant of the levels of ignorance in which it is working.

What efforts have been made to achieve those goals or implement those principles?

The earliest, and still best, example of an effort made to achieve the goals of ecosystem-based management is in the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), where the goal is to account for the indirect effect of fisheries for Southern Ocean krill on the marine mammals and birds that are krill predators. Although CCAMLR takes a precautionary approach, most of the models used by its scientific committee are still rooted in single-species approaches.

The CCAMLR experience suggests that a successful management procedure requires:

1. Specification of clear operational objectives, including performance criteria for evaluating management procedures and actions.

Examples of such objectives include:

- species-oriented objectives (for example, by how much can the probability of collapse of the stock be altered?);
- habitat-oriented objectives (for example, how much habitat is required to remain unaltered?);
- trends or shifts in state variables (for example, what deviations in environmental state variables can occur before considering the system has changed from its current state and a reevaluation of the monitoring program and catch controls is required?); and
- process-oriented objectives (for example, how much change in ecosystem productivity can be tolerated before changes in the distribution of production between the fishery and the ecosystem need to be made?)

2. Prospective evaluation of the management procedures, which includes fishing controls, monitoring, and decision rules for altering fishing controls or monitoring, to determine those which satisfy the performance criteria.

The evaluation of management procedures prior to their implementation provides the opportunity to eliminate management options that would fail to meet the objectives, thereby potentially avoiding a trial and error approach. Such prospective evaluation allows for the implementation of a management procedure that is most likely to achieve the objectives despite uncertainties in the various parts of the system, including the limitations of a monitoring program, such as incomplete data and low power in assessments. It can also be used to ensure that the costs of management are commensurate with the value of the fishery. Another advantage of the prospective evaluation of management procedures is that it allows one to understand how views of the world affect thinking about management.

Monitoring, both the targeted stock and other components of the ecosystem, is a crucial feature of ecosystem-based approaches to management, especially because fisheries are best viewed as large-scale perturbation experiments. CCAMLR established one monitoring program and one standing working group to deal with monitoring issues: the CCAMLR Ecosystem Monitoring Program (CEMP) and the Working Group on Ecosystem Monitoring and Management (WG-EMM).

What methods have been developed for monitoring/assessing ecosystem effects of fisheries?

Methods for assessing the effects of fishing on populations and assemblages of populations or the effects of fishing on ecosystems include monitoring catch of the target species and incidental take (which is best done through independent observers on fishing vessels), documenting features of the environment and food web, research independent of the fishery, and assessing how the volume of catch may affect the environment.

There is no conclusive study on the resilience or resistance of marine ecosystems to fishing or whether fishing diminishes the resilience of a system as a result of synergies with other environmental interventions. There are currently no measures of ecosystems that can be unambiguously interpreted as to whether the structure and function of the ecosystem has been significantly altered, except in obvious cases of complete habitat destruction. This is particularly problematic when marine systems can naturally shift from one state to another without human intervention.

1 Introduction

This document contains a review of approaches to fisheries management based on ecosystem considerations. A particular emphasis is given to species interactions, with a focus on a fished species that is intermediate in a food web. That is, the focal species is both predator and prey for a species at a higher trophic level; the predator of the focal species may be a competitor with human fishers for the focal stock.

The report is organized around five interlocking questions:

1. What are the distinctions between single-species and ecosystem-based management?

2. What problems have been attributed to failure of single-species and ecosystem management?

3. What goals or principles have been espoused for management of marine ecosystems?

4. What efforts have been made to achieve those goals or implement those principles?

5. What methods have been developed for monitoring/assessing ecosystem effects of fisheries?

After answering these questions, we synthesize the implications for an ecosystem-based approach to management.

Before discussing the distinctions between different approaches to resource management, and their various characteristics and relative merits, we present below a brief explanation of our understanding of “management” in the natural resource context.

Management in natural resource contexts is not management of species or ecosystems. It is actually control of human intervention in ecosystems. Nobody can profess to manage regime shifts, changes in food webs, or climate change. The point is that the ability of humans to control ecosystems is limited to modifications of human interventions. Hence there is a tendency when something goes wrong to try to lay blame (Taylor 1999), rather than to seek explanations via a series of multiple causes. The tendency is to blame fishing pressure, because it is often the only visible human activity.

The goal of management is now often described as “sustainable fisheries”, although Pitcher (2000) argues that this goal should be “rebuilding ecosystems”. Sustainability is essentially an economic and social concept with biological constraints.[1] The best way to sustain fish populations from a biological and ecosystem perspective is to leave them alone. Social and economic imperatives, however, determine that management must focus on rational utilization of resources and simultaneous protection of the stock and the ecosystem; these goals are usually, but not always, in conflict (an exception is described in Mangel 1998). Christy (1986) concludes that the answer to the question “can large marine ecosystems be managed for optimum yield?” is “yes, in cases where wealth distribution decisions can easily be made” (pg. 266); that is, it is social not biological issues that determine the success or failure of such management.

[1] In commentary at a meeting with the subject “Biological Objectives in Fishery Management”, Fox (1988, pg 105) wrote: “It seems to me that there are really no primary biological objectives in fishery management. The real objectives in fishery management are economic and social. There are, of course, biological constraints to how a fishery system can behave with or without management, but these are not objectives. I also distinguish a secondary objective, say avoiding some biological constraint by some margin with some degree of certainty. One can legitimately argue over the timeframe to use in determining which path or set of alternatives is optimal or over how certain one wants to be in achieving specific attributes in the system. All too frequently, however, objectives with longer time-frames and higher degrees of certainty are confused with or are actually termed as being “biological”. Apparently in doing so the proponents hope to obtain some deference to the motherhood concept and the opponents hope for denigration because of it -- both just make it more difficult to resolve the issue.”

Charles (1992) notes that there are three main paradigms in fisheries management:

> *The conservation paradigm*: the purpose of management is to conserve fish stocks. This paradigm is often associated with preserves, no-take areas, and removing humans from nature (also see Mangel *et al*. 1996).
>
> *The economic rationalization paradigm*: the purpose of management is to maximize economic return to society. This paradigm has lead to varying concepts of "optimality" including Maximum Sustainable Yield (MSY), Maximum Economic Yield (MEY), and Optimal Yield (OY).
>
> *The social/community paradigm*: the purpose of management is to maintain communities, social structure, and ways of life. This paradigm supports notions of ecotourism and connections between urbanites and charismatic megafauna.

These paradigms are in potential and very real conflict. But a start is to identify the sources of the conflict. None of the paradigms described by Charles explicitly focuses on the ecosystem, although the ecosystem itself is a component of each paradigm and thus crosses the otherwise conflicting notions. Also see Cochrane (2000).

2 What are the distinctions between single-species and ecosystem-based management?

2.1 An approach to management based on single species considerations

Rothschild *et al.* (1997) write: "Single-species management typically is applied to individual stocks of a wide-spread species such as Atlantic cod or haddock...its goal is to specify optimal levels of size specific fishing mortality for a particular species". To do this requires one to assess the state of the stock (e.g. size and reproductive output). But often this is difficult to do, and different groups may view the same information in different ways, because of different assumptions (Starr *et al.* 1998).

Single-species management is based on the assumption that stocks can be viewed out of the context of their role in the ecosystem, that density dependence is the main regulating factor in population dynamics, and that if one simply knows enough about the vital information of the stock, then it is possible to fully control the trajectory of the stock. This is true whether one uses surplus production models, dynamic pool models, stock-recruitment models VPA, or other more sophisticated tools.

Fox (1988) described the "state of the art" assumptions in single species management:

1. an equilibrium exists;
2. recruitment is independent of stock size or its characteristics over the range of interest;
3. the relationships among control variables, observational variables and fishing mortality are linear;
4. there are no exogenous relationships that can not be represented as random, independent errors with zero means; and
5. one must "prove" an effect of fishing before giving advice to take some management action or before management action is taken even if the advice is given to do so. [This point deals with questions regarding burden of proof of changes or potential changes in ecosystems and the general misuse of statistical hypothesis testing (see below).]

In the dozen years since Fox wrote, alternative assumptions have been introduced. For example, there are now approaches that do not assume equilibrium conditions, that include stock recruitment relationships and that allow for precautionary management measures. However these are far from common, and deterministic, equilibrium thinking is still prevalent.

Mangel (1991) notes the following assumptions that underlie single species models:

1. <u>Stationarity</u>. Fluctuations are assumed to be weakly correlated in time. This allows one to draw a stock-recruitment relationship. [Spencer (1997) discusses alternatives to this assumption and their implications for management.]

2. <u>Linkage</u>. It is assumed that linkages are one-way: the environment affects the stock, but the stock does not affect the environment. Thus there are no effects of history. Multispecies models usually make this assumption too, but for many species instead of just one.

3. <u>Time and space scales</u>. It is assumed that only one temporal and spatial scale is sufficiently important to be included in the model. Also see Levin (1991).

4. <u>Genetics</u>. The population is composed of genetically identical subunits.

Single species models lead to the concepts of "recruitment overfishing" (fishing reduces the recruitment because too many fish are taken) and “growth overfishing” (in which fish are taken before they have achieved sufficient size.) Cushing (1981) notes: “Both conditions are cured by reducing fishing, but whereas growth overfishing merely leads to misuse of the eternally regenerating stock, recruitment overfishing will reduce it so far that the fishery is extinguished”.

The concept of overfishing is applied in management terms through an evaluation of the Maximum Sustainable Yield (MSY OR Y_S). This is a biological concept, which Ricker (1975) defined as “the largest *average* catch or yield that can *continuously* be taken from a stock under *existing environmental conditions*. (For species with fluctuating recruitment, the maximum might be obtained by taking fewer fish in some years than in others).” (italics added).

However, too often Ricker’s notion of MSY is applied without serious thought given to the words in italics. That is, what kind of average should be used (arithmetic, or one that is more risk sensitive such as a geometric or harmonic average), what does “continuously” mean, and does “existing environmental conditions” refer to the current environment or the mixture of potential environmental states associated with the present regime?[2]

Single species models, therefore, take no account of the role of the stock as it interacts with other species or the population dynamical processes (Rothschild 1986). That is, yield-per-recruit is a function of biomass and biomass is a function of growth and mortality. Growth changes the pattern of recruitment so that recruitment and growth are intimately linked. Furthermore, stock-recruitment functions may show depensation, thus leading to multiple steady states and the failure of a stock to recover following a decline. Lierman and Hilborn (1997) use meta-analysis to determine the likelihood of such depensatory behavior. They conclude that the most likely estimates for parameters suggest no depensation, but that parameter distributions are very broad and that “analysis of stock recruitment data should incorporate spawner-recruit curves that include the possibility of depensation”. Even with these caveats, single-species models are still used to conclude stocks may be “growth-overfished” (e.g. Barbieri et al. 1997).

Furthermore, ecosystem components are clearly related to growth and recruitment. Laurence (1991) identifies a number of factors affecting feeding and growth in the early life stage of fish (Table 1). Laurence (1991) also classifies hypotheses concerning recruitment in a number of large marine ecosystems, and the role of fishing mortality (Table 2).

Table 1. Environmental Factors that Affect Feeding and Growth in the Early Life Stages of Fish (Laurence 1991)

Factor	Effect
Temperature	Physiological rates

[2] Under the Magnuson-Stevens Fishery Conservation and Management Act (the Act), US Federal Fishery Management Plans (FMPs) are specifically required to define and remedy overfishing in the single species context. These requirements have gone through two distinct phases. The first phase began in 1989 with the publication of NMFS’ guidelines to National Standard 1, which required definitions of recruitment overfishing and corresponding management plans to avoid recruitment overfishing and/or rebuild stocks that had been reduced in size as a result of recruitment overfishing. The second phase began in October 1996 with the reauthorization and revision of the Act. In the revised Act, the definition of optimum yield (OY) was changed from "...MSY as *modified* by" relevant factors to "...MSY as *reduced* by" relevant factors (Section 3, Definitions, 104-297 28(B)). This was interpreted to mean that MSY, or more correctly, the fishing mortality at MSY (F_{MSY}) should be an upper bound (limit) on fishing mortality; i.e. that overfishing limits or thresholds should be based on F_{MSY} or relevant proxies. Thus, in the new National Standard Guidelines, published in 1998, the emphasis changed from avoiding recruitment overfishing to avoiding fishing mortalities higher than the fishing mortality at which MSY is achieved (F_{MSY}). Both are single species concepts, however. The challenge is to include ecosystem effects explicitly in the consideration of the effects of fishing in FMPs.

Light	Visual perception of prey
Prey type-quality	Preference and nutrition
Prey density-distribution	Prey encounter
Search ability	Prey encounter
Capture rate	Prey intake success

Table 2. **Distribution of Physical and Biological Hypotheses Concerning Recruitment (Laurence 1991)**

Type	Primary Hypothesis	Secondary Hypothesis
Physical	11	4
Biological	4 (3 fishing)	6

2.2 An approach to management based on ecosystem considerations

Belsky (1993, pg. 229) writes that "The ecosystem model [in the sense of a conceptual framework] is nothing more than a shorthand for holistic or comprehensive ocean management. The mandate for use of this model seeks to force government leaders to apply scientific principles to domestic and international law and policymaking". M'Gonigle (1997) notes that in times of great transition "--such as that brought on by widespread ecological decline --- individuals search for new principles to guide them. What might be called a *new naturalism* is such a principle". The essence of this new naturalism is that "management and economic structures must be tailored to fit within the constraints of the natural world, rather than the reverse" (also see Mangel et al. 1996). However, this framework needs to be elaborated before it can be operational.

Defining the Ecosystem

Smith (1994, pg. 8) quotes Margalef that "Ecosystems result from the integration of populations of different species in a common environment. They rarely remain steady for long, and fluctuations lie in the very essence of the ecosystems and of every one of the...populations [that comprise the system]". Sherman (1993) identifies 49 "large marine ecosystems" that are foci for fishing activity (contributing 95% of the world catch of 1987) and that could be foci for management.

Mangel and Hofman (1999) define the ecosystem as the community of organisms, the physical environment and the interactions between and among organisms and between the biotic and abiotic environments. This definition avoids a description of the physical boundaries of the ecosystem. For the practical questions regarding fisheries and their effects, the boundaries will perforce be vague and determined to some extent by the kinds of questions being asked.

This definition leads to two crucial questions: what is a community and what is the nature of the interactions? There is still some disagreement about the meaning of a community of organisms (e.g., Price et al. 1984, Diamond and Case 1986). We adopt Fager's (1963, pg. 415) concept that communities are "recurrent organized systems of organisms with similar structure in terms of species presence and abundances". In other words, communities consist of mixtures of organisms. A mixture can vary over time or space, but there will be a consistent pattern to the mixture, even if it can only be described in terms of probabilities (Fager 1957, 1963; Hubalek 1982). There are alternative definitions. For example Stenseth (1985, pg. 61) defines a community on the basis of systemic integrity or stability as "being such that neither a mutant strategy of an existing species nor any new species can invade". This is a fundamentally static viewpoint, whereas Fager's concept is fundamentally dynamic.

Ecosystem-based or an Ecosystem-approach to management

The ecosystem-based approach recognizes that stocks sit within a food web (almost all species are both predators and prey; Pauly and Christensen 1995, Pauly *et al.* 1998), that non-human predators of stocks are competitors with fishing (e.g. Punt 1997, Fryer 1998), and that the abiotic environment is part of the milieu in which organisms live and fishing occurs.

Grumbine (1994) defined "ecosystem management" as having the following objectives (see also Section 4):

1. maintain viable populations of all native species in situ;
2. represent, within protected areas, all native ecosystem types across their natural range;
3. maintain evolutionary and ecological processes;
4. manage over periods of time of sufficient duration to maintain evolutionary potential of species and ecosystems; and
5. accommodate human use and occupancy within these constraints.

The ecosystem-based approach, therefore attempts to bring more of the lesson from ecological sciences (such as thresholds, uncertainty, and surprise) into scenarios used to inform management decisions. This approach considers not only the target stock, but the effects of intervention on predators and prey of the target stock and on other non-target species; generally these are called indirect effects. This approach also recognizes that fishing may have effects on abiotic components of the ecosystem (e.g. bottom trawlers changing bottom characteristics).

This approach also recognizes that changes in the physical (Kullenberg 1986) components of the ecosystem and flips in biomass availabilities may occur, independent of fishing activity (Sherman 1989) and that the physical abiotic environment may have profound effects on biological interactions (Rothschild and Osborn 1990).

An example is provided by Atlantic salmon *Salmo salar* L. Oceanographic oscillations will affect the abundance of salmonids (e.g. Dunbar 1993) and it is important to understand these if we are to draw correct conclusions about the effects of fishing pressure on the abundance of fish. Dunbar and Thompson (1979) studied the qualitative pattern of Atlantic salmon (*Salmo salar* L) abundance in west Greenland waters (Table 3). Gaps in the sequence were caused by gaps in the literature used for the reconstruction, not the presence or absence of salmon. It is not known if this long-term variation was due to temperature fluctuations, the Great Salinity Anomaly (GSA), or other factors (Dunbar 1993). Similarly, factors contributing to the stock declines of Pacific salmon include (Stouder et al 1997) genetic factors, competition and predation, loss of habitat and dams, harvest, ocean productivity. An ecosystem-based approach leads one to ask for the relative ranking and interactions of these in the salmonid decline, rather than to attribute the decline to only one factor.

Furthermore, ecosystem approaches tend to embrace variation and uncertainty. Bakun (1996) and Spencer and Collie (1997) give examples of dome-shaped time series of stocks that include waxing, waning, and crashing stocks. For example, stocks that rose from the mid 1970s to mid 1980s including sardines (Japan, Peru-Chile, California), anchovy (Benguela), and north Pacific groundfish. Stocks in the opposite phase were anchovies (Japan, Peru-Chile, California) and north Pacific albacore. The Gulf of Guinea sardine population expanded in the mid-1970s and has not yet peaked while the Brazilian sardine and northern cod stocks declined following the mid-1980s. These are patterns that may occur even in the absence of fishing and thus represent the range of results obtained by the interaction of stocks and their natural environment.

Table 3. Long-term abundance of Atlantic salmon in West Greenland Waters (Dunbar and Thompson 1979)

Period	Abundance of Salmon
1576-86	Salmon probably present
1605-25	Salmon probably abundant
18th century	Salmon scarce
1806-12	Salmon present, perhaps abundant
1820-50	Salmon scarce
1890-1928	Salmon scarce
1928-31	Salmon observed in increasing numbers
1935-58	Salmon becoming more abundant
1958-79	Salmon very abundant

2.3 Management targets in single species and ecosystem based approaches

The application of target levels for species and ecosystems in management is important for determining at what stage fishing controls (deployment of effort and/or total catch) may need to be altered in order to ensure the objectives are being met (de la Mare 1998; Cooke 1999). Without such targets and a process for assessing how well the fishery is doing against the targets, agreement over whether the harvest controls should be altered will rarely occur (Holt 1998).
Furthermore, the targets and methods for assessing them need to be agreed in advance (Mangel *et al.* 1996).

The use of biological reference points (BRPs) in some fisheries aims to provide quantitative benchmarks for judging whether mortality of species as a result of fishing is well above sustainable levels (see, for example, Sissenwine and Shepherd, 1987, Gabriel and Mace 1999). BRPs are specific target, threshold and limit levels expressed in terms of management and population variables, such as the total catch from a fishery in the long-term, estimates of fishing mortality or abundance of the target stock. They are defined solely using biological criteria associated with the productivity of the stock, but may be modified into management or technical reference points by incorporating social and/or economic criteria to define optimum yield (OY). The concepts of target “conditions” for ecosystems has been discussed by Gislaason *et al.* (2000) but remain to be developed in practical management terms.

The Ecosystem Advisory Panel (Fluharty *et al.* 1999) recognized that ecosystems are likely to have thresholds, which, when exceeded, may cause the system to shift to a new, potentially irreversible state. However, such thresholds are difficult to define. For example, defining these levels for ecosystems is more difficult than for single species due to complex interactions and greater uncertainties associated with larger numbers of parameters (e.g. the Ecosystem Advisory Panel note that the ability to predict ecosystem behavior is limited).

For some ecosystem objectives, particularly relating to the conservation of specially protected species and/or species threatened with extinction, management measures may need to be similar to those for target species. Ideally, these species require a “no take” policy. However, some mortality may be tolerated although not necessarily explicit in the management of by-catch in this case. A danger of not specifying a limit to such by-catch could result in no action being taken to control harvesting even though the populations of these by-catch species may not be sustainable at those mortality levels.

For example, albatross in the Southern Ocean are incidentally killed by longline fishing (see for example Ashford and Croxall 1998). In this case, there is a trade-off between the maintenance of a lucrative fishery and the conservation of seabird populations. Clearly the best way to mitigate against incidental seabird mortality is to eliminate the interaction between them and the fishing gear, but this may imply the complete closure of the fishery. The question becomes “what level of seabird by-catch is tolerable while undertaking the fishery?” In the case of endangered species the question needs to be asked as to what fishing controls are necessary to reduce to zero the threat of continued mortality through fishing, combined with the former question.

The issue of defining what is tolerable is part of defining the public interest in these cases. If it is generally agreed that nothing above zero by-catch is tolerable then the fishing controls need to be sufficiently restrictive to achieve this objective. On the other hand, if some by-catch can be tolerated then flexibility in the arrangements may be possible.

The difficulty in clarifying what by-catch mortality can be tolerated is illustrated by some management policies which are drafted in ways that do not clearly define the objectives and are open to subjective interpretation. For example, in the U.S., the Endangered Species Act (ESA) (Section 7(a)(2)) requires “ every Federal agency, in consultation with and with the assistance of the Secretary, to insure that any action it authorizes, funds, or carries out… is not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat.” In the event that a Fishery Management Plan (FMP) is found to jeopardize or adversely modify critical habitat of a species listed under the ESA, the National Marine Fisheries Service (NMFS) is required to develop “reasonable and prudent alternatives” (RPAs) to the fisheries managed under the FMP, in order to mitigate these effects.

In many cases it is not currently possible to determine categorically (or even qualitatively) what the effects of fishing are on listed species. It therefore becomes problematic to propose immediate “reasonable and prudent” modifications to fisheries that can be guaranteed to remove any threat of jeopardy or destruction/adverse modification of habitat. To complicate matters, the terms “jeopardy”, “adverse modification of habitat” and “reasonable and prudent alternatives” presently have no widely accepted scientific basis. Their meaning is interpreted instead through legal, regulatory and policy usage and through precedent. As the recent European experience with BSE showed, politicians and the public often seek riskless solutions (Ridley 1999). However, no amount of science can convert an inherently uncertain and stochastic situation into a riskless one.

Science can contribute, however, through comparative studies (management practice in various places in the world where protected species and fisheries interact) and through identification of the key questions that must be addressed. In this review, we generally take a descriptive and qualitative approach (see Appendix for an alternative).

2.4 Ecosystem-based management is difficult

Langton and Haedrich (1997) note that development of ecosystem-based management will require several levels of administration, considerable amounts of monitoring, understanding the behavioral relationships among fishers, the fish they catch and the prey of the harvested species. Furthermore, ecosystem-based management is more difficult to model because of the needs of data estimation (Ludwig and Walters 1985, Ludwig 1995). Rothschild (1986 pg. 97 ff) notes that the modeling complexity grows quickly: a two-species ecosystem has 48 quantities to be studied or modeled, but a 5 species ecosystem has 300.

Because of limited predictive ability for complex systems, we may have to think of ecosystem based management as experiments, in the sense of adaptive management (Collie 1991, *Parma et al.* 1998). For example, Sissenwine and Cohen (1991) note that, in the northeast shelf ecosystem, two important questions meriting scientific investigation are 1) the socio-economic implications of reducing fishing mortality on groundfish and flounders and 2) the effect of reducing dogfish (an economically unimportant predator of finfish) on finfish biomass. Doing the latter involves difficulties, since there would be a need for rapid reduction in biomass of dogfish in order to maximize the chance of seeing an ecosystem response. But they also note that (pg. 120) "The response of the ecosystem to a reduction in dogfish biomass cannot be predicted without a much better knowledge of dogfish diet composition...unexpected responses are also possible. For example, dogfish may be holding down the abundance of another low-valued species that may turn out to be a significant predator of larvae and post-larvae of a valuable species". Abrahams and Healey (1993) demonstrate that such manipulative experiments with fishing vessels are possible. Sainsbury *et al*. (1997) and Thrush *et al*. (1998) demonstrate a similar experimental approach in the context of habitat modification by trawlers or dredges.

3 What problems have been attributed to failure of single-species and ecosystem-based management?

In this section, we review the problems that have arisen in the context of fisheries management based on single species approaches. These problems include stock collapses, shifts in trophic structure, habitat degradation, incidental take and bioeconomic conundrums. In general, fisheries target organisms that are relatively high in the food web (Pauly and Christensen 1995, Pauly et al 1998). This implies that one needs to consider indirect effects as well as direct effects. We then describe some of the difficulties with an ecosystem-based approach to management.

Pitcher (2000) identifies three ratchet processes that have resulted from single species approaches and which have led to problems in fishery management. The first is *Odum's ratchet*, which is the failure to recognize that harvesting acts as a selective force on ecosystems by culling long-lived and slow growing stocks and individuals in favor of faster growing ones. The second is *Pauly's ratchet*, which is the tendency for scientists to relate changes in the ecosystem to what things were like at the start of their careers (so that accounts of higher levels of abundance are discounted as anecdotes). The third is *Ludwig's ratchet*, which is positive feedback between fishing mortality and overcapitalization, which is the source of the fishing mortality: declining stocks (due to fishing mortality) require more effort (more capitalization) to catch the remainder of the stock, which then declines further, and so requires more effort.

3.1 Stock collapses

Many stock collapses can be tied to single-species management; although there is little point in providing a litany of such collapses here. Clark (1985, pg. 6) provides an effective summary through 1981 (Table 4). Even if stocks do not crash to the point that the fishery is economically (or biologically) extinct, fluctuations in catch by factors of 3-7 are common (Freon and Misund 1999, pg. 13). In some cases, fishery collapses can be attributed to an environmental variable, the effects of which were ignored, such as the snow crab collapse in Newfoundland (Taylor *et al*. 1993).

Specific examples

California sardine: The collapse (and recovery, see below) of the Pacific sardine *Sardinops sagax* in the California current was one of the early and spectacular failures of fisheries management (McEvoy 1986, Smith 1994). Indeed, John Gulland once remarked that it was a monument to the refusal to act before all the scientific information was available. Even though the California current has been extensively studied subsequent to the collapse, MacCall (1986, pg. 46) notes "Our knowledge of multispecies interactions is surprisingly limited in view of our extensive knowledge of the California Current...it is difficult to assess the impact of the loss of sardine from the ecosystem". He also shows a highly significant and very linear relationship between pelican productivity and anchovy spawning biomass (anchovy having replaced sardine). For example, the anchovy stock size declined from 3.6 million tons to 1.3 million tons between 1975 and 1978 (McEvoy 1986) and the fledgling rate of pelicans dropped also.

The case of pelicans and anchovies is probably the rule, not the exception. In a different system, Norris et al. (1998) demonstrate that the abundance of oystercatchers in Burry Inlet in Wales was positively correlated with biomass of cockles at the beginning of the winter and negatively correlated with fishery landings of cockles during the winter. They conclude that the likely cause is an early dispersal of birds from the Burry Inlet; this could, of course, have unforseen affects on bird reproductive success. A similar example is provided by Fertram and Kaiser (1993), regarding rhinoceros auklet and Pacific sand lance.

Interestingly, too, single species approaches may make it difficult to understand management successes. For example, a fishery moratorium for Pacific sardine between 1973, the start of a small directed fishery in

1986 and careful control of fishing mortality since 1986 has lead to the recovery of the sardine to high levels of abundance. However, as Smith *et al*. (1992) show, the rapid recovery of the sardine stocks is not likely to have been due to the fishing moratorium alone; something else happened (also see Lo *et al.* 1995). What exactly the something else was is still unknown, although Smith *et al*. provide some suggestions (migration of stocks from elsewhere, decline of natural predators, changes in life history parameters of the fish.) With a focus only on the target stock, one cannot address these other issues.

Northern cod: The collapse of the northern cod *Gadus morhua* is another spectacular story of management failure (a popular account is provided by Kurlansky 1997). Contrary to common assertions (e.g. Parsons and Beckett 1997), Hutchings and Myers (1994) conclude that this collapse was solely due to overfishing that resulted from inappropriate estimates of spawning stock size and that their work "should provide ample justification for politicians, policy-makers, industry and management to limit the urge to attribute resource collapses to vaguely understand or even imagined environmental causes" (pg. 2144). Myers et al. (1995) show that previous estimates of large scale correlations (thousands of kilometers) of recruitment in cod -- suggesting large scale environmental effects -- were inaccurate and that the scale of recruitment correlations generally was less than 500 km. Hutchings (1996) reviews the various assumptions about the collapse of the cod stock and concludes that excessive fishing mortality was the sole significant cause of the collapse. He further notes (Hutchings 1997, pg. 960) that the fundamental question to ask is "what are the effects of fishing on the behavior, life history, and population biology of exploited fishes?"

Table 4. Examples of stock collapses through 1981 (Clark 1985)

Stock	Peak Catch (Year)	1981 Catch
Antarctic blue whale	29,000 whales (1931)	Nil
Antarctic fin whale	27,000 whales (1938)	Nil
Hokkaido herring	850,000 tons (1913)	Nil
Peruvian anchovetta	12.3 million tons (1970)	0.3 million tons
Southwest African pilchard	1.4 million tons (1968)	Nil
North sea herring	1.5 million tons (1962)	Negligible
California sardine	640,000 tons (1936)	Nil
George's bank herring	374,000 tons (1968)	Nil
Japanese sardine	2.3 million tons (1939)	17,000 tons

3.2 Shifts in trophic structure

Removal of predators by fisheries may change the entire trophic structure of an ecosystem (Table 5; Parsons 1992). Hughes (1994) documents changes in coral reefs in the Caribbean Ocean, where overfishing (1960s - present) combined with hurricane damage (1980) and disease (1983-present). Roell and Orth (1998) use a food web model to predict that the MSY-based (i.e. single species) harvesting of crayfish in the New River, West Virginia, USA would likely result in substantial declines in the biomass, production, and harvest of smallmouth bass and other crayfish predators. They conclude "that fishery management outcomes are strongly dependent on the key prey-predator interactions and that distinct harvest regimes produce unique food web structures with varying potentials for sustainable yields".

Table 5. Examples of shifts in trophic structure due to fishing (Parsons 1992)

Fishery	Trophic Effect
North sea herring and mackerel	Increase in the abundance of smaller fish
Eastern North Atlantic herring and mackerel	Increase in the abundance of smaller fish
Antarctic blue whales	Change in whale abundance
Bering sea pollock	Changes in abundance of mammal and bird populations
Peruvian current anchovy	Decrease in birds; increase in sediment carbon
Great Barrier Reefs sports and commercial	Increase in the population of crown-of-thorns; demise of coral.

Specific examples

Peruvian anchovy: In this case, there is evidence (Cushing 1981) there are two food webs, depending upon environmental conditions:

Algae C Anchovy/Zooplankton C Carnivores

or

Algae C Zooplankton C Anchovy C Carnivores

and that zooplankton are destroyed by warm El Nino waters.

Regarding the anchovy, Hilborn and Walters (1992, pg. 19) write: “the stock assessment work prior to the 1972-83 (sic) El Nino concentrated almost exclusively on trying to predict the MSY; the government of Peru wanted to know what level of harvest was sustainable, and the stock assessment experts tried to provide a number. This is probably the greatest failing of fisheries scientists. When decision-makers ask the wrong question, try to convince them to ask a better question instead of providing them with a silly answer that will eventually lead them astray. In the 1960s, it was widely recognized that species such as the Peruvian anchovetta are prone to major fluctuations; after all, the California sardine had recently collapsed. At the time, it was felt that if we just kept catches low enough it might be possible to avoid such collapses. The stock assessment biologists should have emphasized more forcefully that they could not predict the sustainable yield with any reliability (perhaps plus or minus 50%). They should have told the Peruvian government that the time would come when the stock would decline, they should have insisted on helping work out a management plan for that contingency”.

Cushing notes that when anchovy were exploited, birds and fishermen competed for the fish and that the effect of El Nino was to reduce the population of birds and thus augment recruitment. However, three successive years (1971, 1972, 1973) reduced the stock so much that under continued exploitation it did not recover. It is thus possible that the failure of a recruitment class, leading to a decline in predators, may lead to enhanced recruitment in subsequent years, as indicated by general ecological theory. The problem, of course, is that without careful consideration of the entire system, this subsequent increase is likely to be misinterpreted.

Benguela Current Stocks: The Benguela is a boundary current in the south east Atlantic. Evidence (Crawford et al. 1989) suggests that environmental factors favor either groundfish species of epipelagic species, but not both simultaneously. Strong year classes of epipelagic species have often occurred as a result of warm periods, and have been managed using single-species approaches. This leads to a mis-match of fishing mortality and stock abundance in periods of environmental change.

Red grouse in Scotland: This is an instructive terrestrial failure of a single species approach. Game-keepers noted that grouse were preyed upon by carnivores and hawks, so made efforts to cull these predators. After this, the grouse population crashed. Subsequent investigation revealed that the predators had captured the weakest individuals, who generally had a high parasite burden. Removal of the predators allowed the parasites to spread in the population.

Fishery-related examples of this phenomenon are not common. However, Mesa *et al*. (1998) show, by experiments with infected juvenile chinook salmon, that pathogens may play a clear role in predator-prey relations that goes beyond direct mortality due to the pathogenic agent.

3.3 Habitat degradation

Habitat modification is a common result of management based on either single or multispecies approaches in which the stock and catch are the only foci (e.g. Sainsbury *et al*. 1997). Fisheries may cause habitat degradation that ranges from marine pollution (Laist *et al*. 2000) to destruction of seabed communities (Koslow & Gowlett-Holmes, 1998).

Specific example

Chesapeake Bay Oysters: Oyster (*Crassostrea virginica*) catches from Chesapeake Bay, MD, USA peaked in the late 1800s and have declined since then. Rothschild *et al*. (1994) demonstrate that a single-species approach, which focussed solely on catch with little attention to habitat effects of fishing, is almost surely responsible for the decline. Interestingly, in recent years other ecosystem effects (reduced water quality and disease) have been attributed as causes.

3.4 Effects on non-target species

Similarly, a single species approach fails to take into account the effects of fishing mortality on non-target species encountering the fishing gear. Incidental take or by-catch occurs in virtually all fisheries (Alverson 1997, Crowder and Murawski 1998) and in some cases the incidental take dominates the directed take (e.g. Alverson 1997, Northridge 1995).

The drift gillnet fishery for sharks and swordfish in California captured marlin, sea lions, elephant seals, harbor seals, common dolphins, northern right whale dolphins, Pacific white-sided dolphins, Risso's dolphin, finback whale, gray whale, beaked whale, minke whale, short-finned pilot whale, loggerhead turtle and Ridley turtle (Hanan *et al*. 1993). Pennoyer (1997, pg 147) shows that the 1993 trawl fishery for rock sole in the Bering Sea and Aleutian Islands landed 26,144 tonnes and discarded 58,438 tonnes; the hook-and-line fishery for sablefish landed 23,000 tonnes and discarded 5,391 tonnes.

These examples show that incidental take is a common occurrence and affects many different species other than the target species. The effects of these removals are often unknown, although some authors have suggested that they can be considerable (e.g. Northridge 1995). Incidental take of key predators in reef communities, for example, can change the entire structure of the community (Roberts 1995).

Shrimp trawlers in South Carolina incidentally take king mackerel at a rate of 0.244 fish/hour and Spanish mackerel at a rate of 0.7 fish/hour and age-0 king mackerel are vulnerable to shrimping gear for at least half of the season (Harris and Dean 1998). Shrimp or prawn trawl fisheries incidentally take between 2 kg and nearly 15 kg in incidental (discarded) catch per landed kg catch (Alverson 1997, pg. 118). The failure to consider the mortality of juvenile reef fish in shrimp trawls in the Gulf of Mexico has been cited as an important factor in the decline of the red snapper population and the directed fishery for adult red snapper. A reduction in by-catch in shrimp trawls is considered to be a fundamental part of the recovery of the population and the directed fishery (MRAG Americas 1998).

3.5 Bioeconomic conundrums with single-species approaches

When coupled with economic analyses (Smith 1988, Clark 1985, 1990), the single species approach often leads to unsettling results. Examples include 1) the economic optimality of extinction stocks whenever the maximum per capita growth rate is less than the discount rate, 2) bioeconomic equilibria (levels of fishing effort and population size at which there is no aggregate profit) that are independent of biological parameters such as maximum per capita growth rate or carrying capacity.

Regardless of how bioeconomic analyses are employed, there is a tendency towards over-capitalization of fishing fleets and the development of excess fishing capacity (Clark 1985, 1990). For example, between 1970 and 1989, world fishing fleet capacity increased by a factor of more than 4; at the same time, landing rates declined by a factor of more than 3 (Garcia and Newton 1997).

Because many fisheries are multispecies and/or incidental catch occurs, attempts to choose fishing effort to maximize economic benefit on a single species may interfere with the management of other weaker stocks (e.g. Barbieri et al. 1997).

Finally, bioeconomic calculations are deceptively simple because of their mathematical formulation. However, it is often the case that we are highly uncertain about the details of some of the basic operational (let alone biological or economic) assumptions. For example, Freon and Misund (1999, pg. 196) list 29 models that can be used to relate Catch Per Unit Effort (CPUE) to environmental variables and fishing effort; the usual assumption that CPUE is proportional to stock abundance (possibly depending upon the environment) is only one of many.

3.6 Complexities of data analysis in ecosystem-based management

To some extent, ecosystem-based management is limited by the data requirements. Scarcity of data and fluctuating environments mean there are commonly substantial uncertainties in analyzing the effects of fishing on the ecosystem.

For example, data concerning incidental take (in the sense described above) are sufficiently sparse that one often cannot draw a firm statistical conclusion about the effect of incidental take on the non-targeted stock and the incorrect conclusion of “no biological effect” is drawn without a consideration of statistical power (Mangel 1993). That is, incidental take data are often evaluated by testing a hypothesis such as “ the incidental take had no effect on the state of the stock”. The most common error in interpretation is to draw an inference from failure to reject a null hypothesis. Failure to reject is often taken as evidence in favor of the null hypothesis; some even believe that the truth of the null hypothesis is thereby established (Brook *et al* 2000). However, the significance level only addresses the issue of false rejection of the null hypothesis, assuming its truth. If the null hypothesis is not rejected, the quantity of interest is the probability of accepting the null hypothesis when it is in fact false. This quantity is termed the power of the test, and it depends upon which alternative to the null hypothesis is in fact true (Peterman 1990, Peterman and M'Gonigle 1992, Osenberg *et al.* 1994, Steidl *et al*. 1997). Management based on hypothesis testing without consideration of the power of the test may be disastrous. A second error in interpretation of hypothesis testing is to interpret the significance level as the probability that the null hypothesis is true. Such an inference is nonsensical in standard (frequentist) statistics, since hypotheses are either true or false in that framework: they do not have probabilities attached to them.

On the other hand, Bayesian statistics does assign probabilities to hypotheses (Apostolakis 1990, Howson and Urbach 1993, Hilborn and Mangel 1997, Press 1997, Malakoff 1999). There are various objections to Bayesian inference. Some concern technical difficulties in implementing it, and about the manner in which the value of the prior probability of hypothesis is chosen. Dennis (1996) claims that Bayesian methods are not useful for ecological research. He objects (rightly in our view) to Bayesian neglect of methods such as randomization, examination of residuals, and design of sample surveys. Anderson (1998) presents psychological evidence that people find it difficult to reason about probabilities attached to hypotheses. She recommends standardization and improved methods of presentation to overcome some of these difficulties. It is fair to say that Bayesian methods avoid some common pitfalls of scientific inference and interpretation, but they should be used with insights that are not part of that framework. Methods of choosing and implementing appropriate statistical methods are undergoing vigorous development: see Mayo (1996) and references therein.

Real fisheries take place in systems in which there are uncertainties and fluctuations. Ludwig (1995) proposed that natural resource management involves at least two paradoxes connected to such uncertainties:

1. Management for sustained yield cannot be optimal.
2. Effective management models cannot be realistic.

The source of these paradoxes is "statistical issues and the relationship between models and data" The implication of these paradoxes, is that "statistical considerations generally invalidate any but the simplest aggregated models as management tools". For example, in order to estimate parameters in a Ricker relationship (or any other form of parent-offspring dynamics), one needs variation in the spawning stock. Thus, the stock cannot be maintained at a single "optimal" level if one needs to learn about parameters.

There are different kinds of models that one can use for analysis in ecosystem-based fisheries management.

Statistical models are the ones that arise in the analysis of data (e.g. regression, ANOVA, etc). They are used to make inferences about properties of the data. There may be lack of relationship between the ecological variables, but the model cannot be wrong. On the other hand, *theoretical* models posit mechanisms and may lead to predictions that disagree with the data.

There are two main reasons for exploring theoretical rather than statistical models: i) a wish to understand nature or ii) the environment is variable so that statistical relationships will not hold. When mechanistic models lead to predictions that disagree with the data, one must rethink the logic of the model or question the quality or validity of the data. Empirical relationships are valuable in situations with low variability, i.e. when the model may be expected to work also in other situations and populations other than in the situation of measurement. For instance, the way temperature affects growth rate may be studied in a laboratory and will also apply to temperatures in other laboratories and in the field. However, empirical equations must be treated with much caution as soon as the relationship may be influenced by individual behavior. This is particularly true for estimates of natural growth, reproduction, and mortality rates, which are heavily influenced by the activity level and habitat selection of the individuals (Aksnes 1996). To model such phenomena in natural environments, theoretical considerations are needed.

Logical models are mathematics motivated by the natural world. An example of the distinction between a logical and a theoretical model is the Euler-Lotka model, which states that if a population consists of equal individuals for whom fecundity and survival are deterministic variables of age, then the population will grow by a constant rate and reach a stable age distribution. This was first proved by mathematical arguments by Lotka (1925). As a logical statement it is not open to experimental verification, and it is true within the realms of mathematics. However, biologists may investigate whether this model is a good approximate theory for real populations. So for biologists, the Euler-Lotka model is a theory for population dynamics. Since it does not fit well with observations, a rich alternative theory for population dynamics including variable environments, individual variability and stochasticity developed (e.g. Tuljapurkar 1990, Tuljapurkar

& Caswell 1997).

When using theoretical models, we posit mechanisms that connect the independent and dependent ecological variables. Among theoretical models it is fruitful to treat "why" (ultimate) and "how" (proximate) questions separately. Models dealing with ultimate questions address the causes of a phenomenon, which for biology means that these models should be founded on the theory of evolution by natural or artificial selection. Models dealing with proximate questions address how a mechanism operates, and will resolve the process to a desired level. For example, in mortality estimation, the first step is to construct mechanistic models of the environmental impact on factors that influence mortality risk (e.g. visibility, smell, sound, density-dependencies). The next step is to construct theoretical models of how individuals would act in response to a mortality risk (e.g. find the trade-off between predation risk and feeding rate, Werner & Gilliam 1984); by combining these models the mortality rate may be calculated. Functional models (asking why things are as they are) address problems or environments only found in idealized (artificial) worlds. When applied in this world, they only cover parts of the whole.

There are a number of ways to deal with the issues raised by Ludwig:

- ***Avoid too many uncertain parameters***.

Ludwig (1995) points out the dangers of overfitting data by interpolation (e.g. cubic splines) or regression and notes "Having the correct model is not enough: the associated parameters must be well determined" (pg. 521). Picking the right size for a model is a developing art (reviewed in Hilborn and Mangel 1997). This applies to statistical models and to theoretical models for which parameters must be estimated. Furthermore, if the physical or biological parameters are not known or measured with much uncertainty, it is even more important to keep the number of parameters low; with well defined and independently measured parameters this is less critical. There is always a trade-off between simplicity and the level of mechanistic description. In general, simpler models are attractive because of tractability and transparency, and should not be easily abandoned due to dissimilarities with empirical studies (although the unease with the model may increase). For example, a mechanistic model of the functional response in fish may clarify the importance of the optical properties of water in understanding the distribution and dynamics of fish and zooplankton.

- ***Always try to compare multiple models with data***.

The geologist Thomas C. Chamberlain argued that we should always have multiple working hypotheses (his classic 1897 paper is republished in Hilborn and Mangel 1997, pg. 281-293). Theoretical models almost immediately lead to multiple models, as different mechanistic formulations are envisioned. Myers *et al*. (1995) confronted four different models of recruitment and two different models of uncertainty with more than 250 sets of stock-recruitment data. This allowed them to determine the most appropriate description of the functional relationship between recruits and spawners and the most appropriate conceptualization of the variability in recruitment.

- ***Test models appropriately***.

Logical models are tested with mathematics, functional theoretical models are tested by evolutionary theory (i.e., other, more basic functional models), mechanistic theoretical models by careful experimentation and observation. The models we use in management and ecology are often complex. For these, it is better to test each of the major assumptions rather than to try to test the predictions of the models. This has to do with the only partial overlap between model and environment, and the hopeless task of measuring the relevant environmental complexity in an instant. However, a statistical model cannot be broken down to subsets that may be tested independently. In any case, we should always recognize that the model may miss a key feature of the natural system, even one that drives the full behavior of the system.

An example of testing assumptions is from the study of eutrophication in the North Sea (cf. Aksnes et al.

1995). Starting with the Holling equation describing the feeding rate in animals, they used a mechanistic model for nutrient uptake in phytoplankton. Parameters were estimated for two groups of algae (diatoms and flagellates) such that the parameters (which have precise biological interpretations) were fixed from measurements (Aksnes *et al*. 1995). Simultaneously, many series of enclosure experiments were conducted with a wide range of nutrient forcing (Egge and Aksnes 1992, Egge *et al*. 1994), and time series of phytoplankton development compared with model simulations. No tuning of the parameters was allowed as the intention was to develop a general application tool for the study of eutrophication, although the goodness-of-fit may have been improved by this. The model has been incorporated into a three dimensional physical model of the North Sea, and applied to investigate issues related to eutrophication and management (Aksnes *et al.* 1995, Baliño 1996).

- ***Be very careful when going where the data aren't***.

Both theoretical and statistical models may enter intellectual quicksand when applied to situations in which there are no data

- ***Don't confuse statistical and theoretical models***.

The error of mixing the two was called 'the error of pseudo-explanation' in Loehle (1987); Dunham and Vinyard (1997) make a similar point. It is possible to conduct an excellent and elegant study using a statistical model, but then to wrongly conclude that one has constructed a theoretical model. For example, forcing the regression through the origin adds mechanism to a statistical model and thus makes it an implicitly theoretical model. Very often a good statistical model will identify relationships that then lead us to think about the mechanisms underlying them. To be sure, all kinds of models are needed for an ecosystem-based approach to management. As theoretical models become larger and more computationally intensive, they require more parameters and a blend between a theoretical and statistical model is obtained.

4 What goals or principles have been espoused for management of marine ecosystems?

4.1 Synthesis, implications and a pragmatic context

In this section, we review various published goals and principles concerning ecosystem-based management. We begin, however, with a synthesis of the material that we summarize and a brief discussion how a pragmatic approach to using principles can proceed.

Perhaps the most fundamental principle is a change of attitude in which there must be the rejection of the Cornucopian myth concerning the ability of the oceans to provide resources (Regier 1997). T. H. Huxley, at the turn of the last century, formulated it as follows: “Any tendency to over-fishing will meet with its natural check in the diminution of the supply; this check will always come into operation long before anything like permanent exhaustion has occurred” (from 1997, pg. 122). This myth guided much of the development of fishery management in the 20th century. It is still with us, as Kurlansky (1997, pg. 186) notes: “Furthermore, the Kirby report [a Canadian government report ca. 1990 to assess the future of Atlantic fisheries] was still being influenced by Huxley’s teaching about the resilience of indestructible nature. The idea itself seems to have more resilience than nature. As with the sixteenth-century belief in a westward passage to Asia, the theory cannot be killed by mere experience”.

Although the principles described below vary in detail, there are commonalities that can be synthesized. There are three broad goals in the ecosystem approach (Larkin 1996):

1) a sustainable yield of products for human consumption and animal foods;
2) maintenance of biodiversity; and
3) protection from the effects of pollution and habitat degradation.

The following synthetic principles and their implications apply (Table 6).

1. Expect change in ecosystems and that the change will likely be due to multiple causes. <u>The implication:</u> human intervention must be flexible rather than fixed and one must proceed cautiously when increasing the level of intervention. Humans must act like upper tropic level predators.

2. Assume that human intervention will have an effect on parts of the ecosystem other than the target stock. <u>The implication</u> is that monitoring is essential and that stock assessment of a targeted species is not enough. One must monitor its prey, competitors and predators.

3. Recognize that no amount of monitoring or sampling will fully reduce the uncertainty that one faces. <u>The implication</u>: Sensible discussion must thus focus on the risk of overfishing (to both the target stock and other species) and the missed economic opportunities from lower levels of fishing mortality. A consensus cannot be achieved by averaging positions, but one may be able to apply methods of risk analysis (e.g. Francis 1992, Restrepo *et al.* 1992, Rosenberg and Restrepo 1994, Kokko *et al*. 1997, Lane and Stephenson 1997; Peterman *et al*. 1999) to the situation. Similarly, the decision to completely close a fishery, while appearing sensible, is in itself not necessarily sufficient to change things.

4. Be prepared to answer Rothschild’s five questions: how can we separate the effects of fishing from naturally induced variation; how can we predict the magnitudes of routine fluctuations in recruitment; how can we predict sustained changes or trends in abundance; how can we appraise the influence of pollution, eutrophication, or habitat modification on the abundance of particular fish stocks; what is the nature of interactions among species of fish in an ecosystem, with specific reference to predicting change in abundance of one species as a function of change in abundance of another species? The *implication* is that management must be cognizant of the levels of ignorance in which it is working.

Regardless of the particular synthesis of principles that one achieves, it is important to consider how they can be used in a charged political environment. Farber (1999) is guided by legal precedents and the history of legislative enactments of environmental protection rather than abstract arguments in favor of protection. He includes a critical discussion of cost-benefit calculations, the roles of models and discounting of future costs and benefits in the context of a pragmatic approach:

"Being pragmatic does not mean the rejection of rules or principles in favor of *ad hoc* decision making or raw intuition. Rather, it means a rejection of the view that rules, in and of themselves, dictate outcomes. ...Hard policy decisions can't be programmed into a spreadsheet... But we also need an analytic framework to help structure the process of making environmental decisions. Intuition is often an unhelpful guide because environmental law concerns issues outside our normal, everyday experience.... Rather than rigid rules or mechanical techniques, we need a framework that leaves us open to the unique attributes of each case, without losing track of our more general normative commitments" (pg. 10-11).

Table 6. A synthesis of published principles for ecosystem-based management

Principle	Implication
Expect change in ecosystems and that the change will likely be due to multiple causes.	Human intervention must be flexible rather than fixed and one must proceed cautiously when increasing the level of intervention. Humans must act like upper tropic level predators.
Assume that human intervention will have an effect on parts of the ecosystem other than the target stock.	Monitoring is essential. Stock assessment of a targeted species is not enough. One must monitor its prey, competitors and predators.
Recognize that no amount of monitoring or sampling will fully reduce the uncertainty that one faces.	Sensible discussion must thus focus on the risk of overfishing (to both the target stock and other species) and the missed economic opportunities from lower levels of fishing mortality. A consensus cannot be achieve by averaging positions, but one may be able to apply methods of risk analysis to the situation. Similarly, the decision to do completely close a fishery, while appearing sensible, is in itself not necessarily sufficient to change things.
Be prepared to answer Rothschild's five questions: 1. How can we separate the effects of fishing from naturally induced variation? 2. How can we predict the magnitudes of routine fluctuations in recruitment? 3. How can we predicted sustained changes or trends in abundance? 4. How can we appraise the influence of pollution, eutrophication, or habitat modification on the abundance of particular fish stocks? 5. What is the nature of interactions among species of fish in an ecosystem, with specific reference to predicting change in abundance of one species as a function of change in abundance of another species?	Management must be cognizant of the levels of ignorance in which it is working.

4.2 Published principles

We now summarize a variety of goals or principles that have been published and that deal with management in the context of marine ecosystems.

The principles for the conservation of wild living resources published in 1978 by Holt and Talbot were:

1. The ecosystem should be maintained in a desirable state such that:

   a. consumptive and non-consumptive values could be maximized on a continuing basis;
   b. present and future options are ensured; and
   c. the risk of irreversible change or long-term adverse effects as a result of use is minimized.

2. Management decisions should include a safety factor to allow for the fact that knowledge is limited and institutions are imperfect.

3. Measures to conserve a wild living resource should be formulated and applied so as to avoid wasteful use of other resources.

4. Survey or monitoring, analysis, and assessment should precede planned use and accompany actual use of wild living resources. The results should be made available promptly for critical public review.

May *et al.* (1979), in a precursor to the principles adopted in the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), proposed:

1. For populations at the top of the trophic ladder, the concept of maximum sustained yield (MSY) will often remain useful.
2. Other population should not be depleted to such a level that their productivity or that of other species dependent upon them is significantly reduced.
3. Monitoring should be set to the slowest population process time scale.
4. Harvesting levels should be set conservatively to safeguard against the combined effects of environmental variation and harvesting.
5. The intersections of these considerations with economic and political factors imply consequences and management implications that “defy crisp summary”.

Rothschild (1986, pg. 25) puts potential principles as a series of questions:

1. How can we separate the effects of fishing from naturally induced variation?
2. How can we predict the magnitudes of routine fluctuations in recruitment? [This prediction is made more difficult by evidence suggesting that year-class strength in marine fish is most likely determined by mortality during the pre-juvenile stages (Leggett and Deblois 1994), which are rarely monitored in fisheries management.]
3. How can we predicted sustained changes or trends in abundance, such as the decline of herring and increase of cod in the North sea?
4. How can we appraise the influence of pollution, eutrophication, or habitat modification on the abundance of particular fish stocks?
5. What is the nature of interactions among species of fish in an ecosystem, with specific reference to predicting change in abundance of one species as a function of change in abundance of another species?

Olver *et al*. (1995) identify a set of conservation principles:

Fundamental principle

Aquatic ecosystems should be managed to ensure long-term sustainability of native fish stocks.

Ecosystem protection

1. The sustainability of a fish stock requires protection of specific physical and chemical habitats utilized by the individual members of that stock.
2. The sustainability of a fish stock requires maintenance of its supporting native community.

Population utilization

1. Vulnerable, threatened, and endangered species must be rigidly protected from all anthropogenic stresses.
2. Exploitation of populations or stock undergoing rehabilitation will delay, and may preclude, full rehabilitation.
3. Harvest must not exceed the regeneration rate of a population or its individual stocks.
4. Direct exploitation of spawning aggregations increases the risk to sustainability of fish stocks.

Apollonio (1994) notes that fishing vessels, if they replace apex predators, must have characteristics of a K-selected species. These characteristics include (Pitcher and Hart, 1982, pg. 84) a fairly constant and/or predictable habitat, a narrow niche, density dependent mortality, populations that are fairly constant in time and at or near carrying capacity, intense inter-specific competition, long -lived and efficient. For such species, selection favors slow development, low per-capita reproduction, delayed reproduction, and large body size. On the other hand, the tradition with fisheries has been overcapitalization met by government subsidies (Clark 1985, 1990). This is the equivalent of no control on the top predator and has lead to serial depletion of stocks and fishing down the food web (Pauly *et al*. 1998)

Mangel *et al*. (1996) articulated the following principles:

1. Maintenance of healthy populations of wild living resources in perpetuity is inconsistent with unlimited growth of human consumption of and demand for those resources
2. The goal of conservation should be to secure present and future options by maintaining biological diversity at genetic, species, population and ecosystem levels; as a general rule neither the resource nor other components of the ecosystem should be perturbed beyond natural boundaries of variation.
3. Assessment of the possible ecological and sociological effects of resource use should precede both proposed use and proposed restriction or expansion of ongoing use of a resource.
4. Regulation of the use of living resources must be based on understanding the structure and dynamics of the ecosystem of which the resource is a part and must take into account the ecological and sociological influences that directly and indirectly affect resource use.
5. The full range of knowledge and skills from the natural and social sciences must be brought to bear on conservation problems.
6. Effective conservation requires understanding and taking account of the motives, interests, and values of all users and stakeholders, but not by simply averaging their positions.
7. Effective conservation requires communication that is interactive, reciprocal, and continuous.

Mangel *et al*. also give a series of mechanisms for implementing these principles.

Harwell (1997) identifies the following principles of ecosystem management:

1. Use an ecological approach that recovers and maintains the biological diversity, ecological function, and defining characteristics of natural ecosystems.
2. Recognize that humans are part of ecosystems and that they shape and are shaped by the natural

system -- that is, the sustainability of ecological and societal systems are mutually dependent.
3. Adopt a management approach that recognizes that ecosystems and institutions are characteristically heterogeneous in time and space.
4. Integrate sustained economic and community activity into the management of ecosystems.
5. Develop a shared vision of desired conditions for societal systems and ecological systems.
6. Provide for ecosystem governance at appropriate ecological and institutional scales.
7. Use adaptive management [see below] as the mechanisms for achieving both desired outcomes and new understandings regarding ecosystem conditions.
8. Integrate the best science available into the decision-making process, while continuing scientific research to reduce uncertainties.
9. Implement ecosystem management principles through coordinated government and non-government plans and activities.

Lee (1997) proposes three rules:

Cooperation principle: Decide in light of the practical limitations of one's powers, collaborating when necessary.
Bioregional principle: Plan and act at biologically appropriate scales of space, time and function.
Adaptive principle: Act and learn so as to expand practical knowledge of the ecosystem.

Pitcher and Pauly (1998) argued that rebuilding ecosystems, not sustainability, is the appropriate goal for fishery management and that only this goal will successfully deal with the ratchets described above. Pitcher (2000) clarifies this position in the answers to five questions concerning what is needed to rebuild ecosystems.

First, a wide variety of data – much broader than stock abundance or assessment – needs to be collected, in a fully georeferenced manner. These data must, of course, include information about the target stock, but also about prey and predators of the target stock. Although the fishing industry and special government surveys can and will continue to contribute to collection of data, Pitcher envisions that a wider scope (including the coastal dwelling public, bird and whale watchers, and secondary schools) is possible.

Second, Pitcher proposes that data should be openly available, so that any individual with the right skills can perform and confirm analyses; assessments and their implications should no longer be the sole perogative of government employees.

Third, Pitcher reinforces that peer-reviewed publications, rather than gray literature or other un-refereed formats such as the Internet, remain the means for communicating information.

Fourth, Pitcher recommends the use of Bayesian and Monte Carlo methods as means for incorporating uncertainty in the models that are used to describe ecosystems.

Fifth, Pitcher follows Policansky (1998) and argues that the scientific evaluations should be separated from management decisions and enforcement issues.

The Ecosystem Advisory Panel (Fluharty *et al*. 1999) identified the following principles to achieve the goal of maintaining ecosystem health and sustainability.

- The ability to predict ecosystem behavior is limited.
- Ecosystems have real thresholds and limits which, when exceeded, can effect major system restructuring.
- Once thresholds and limits have been exceeded, changes can be irreversible
- Diversity is important to ecosystem functioning

- Multiple scales interact within and among ecosystems
- Components of ecosystems are linked
- Ecosystem boundaries are open
- Ecosystems change with time

Policies to achieve this goal include

- Change the burden of proof
- Apply the precautionary principle
- Purchase “insurance” against unforseen, adverse ecosystem impacts
- Learn from management experiences
- Make local incentives compatible with global goals
- Promote participation, fairness, and equity in policy and management

Mangel and Hofman (2000) identify 10 concepts relevant to ecosystem-based management:

1. Patchiness and variability in space and time are characteristics of most ecosystems.
2. Ecosystems are characterized by multiple cause-effect relationships among biotic and abiotic ecosystem components.
3. The consequences of events at one trophic level often will be manifested across many other trophic levels.
4. Organisms do not recognize political boundaries and management should plan accordingly.
5. Ecosystems should be viewed as the current state of an ongoing process of selective extinction and differential speciation.
6. Change is the rule, not the exception, in ecosystems.
7. Interactions between components of ecosystems may be both one-way and two-way.
8. Marine food chains are complex and in many species the trophic level varies with life stage.
9. Competition and predation both contribute to the structuring of food webs, but their relative importance varies.
10. Top predators such as marine mammals may have population dynamics that prohibit using their abundance and productivity as effective indicators of the current health of ecosystems, although they may be good indicators of long term effects.

5 What efforts have been made to achieve those goals or implement those principles?

5.1 Implementing principles

Concern over the direct and indirect effects of fishing on non-target species has been present for at least the last 20 years, although most attention has been given to the effects on by-catch species and habitat. A number of recent publications highlight the effort by managers of fisheries in defining and, in some cases, implementing ecosystem principles: Australia (Sainsbury *et al.* 2000), CCAMLR (de la Mare 1996; Constable *et al.* 2000; Parkes 2000), FAO (FAO, 1995; Sainsbury *et al.* 2000); ICES (Gislaason *et al.* 2000; Murawski 2000), South Africa (Butterworth & Punt 1999), USA (Fluharty *et al.* 1999). Implementation has involved the formulation of operational ecosystem objectives, the evaluation of management procedures (strategies) for achieving the objectives and incorporating safeguards to take account of uncertainty (de la Mare 1998; Cooke 1999; Smith *et al.* 1999). Inevitably, this needed to account for methods to apply the precautionary approach (FAO 1995).

The earliest institutional example of applying the precautionary approach along with implementing ecosystem principles in fisheries was through the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) (Constable *et al.* 2000; Parkes 2000). The Commission has grappled with the problems of taking an ecosystem approach to management since the inception of the krill convention and the obligations contained therein (Article II) to ensure that fisheries do not jeopardize the maintenance of ecological relationships as well as providing for the recovery of depleted populations, notably the great whales (Constable *et al.* 2000).

In cases where marine mammals are predators of the same fish or shellfish that humans harvest (Alverson 1992), as in the case of the fishery for Antarctic krill, *Euphausia superba*, how should we account for this competition? The krill fishery in the southern ocean is an interesting example of taking account of competition between a fishery and predators of the target species. Nearly all fish, birds and mammals in the Southern Ocean are no more than one or two steps in the food chain away from krill (Everson 1992, Hunt *et al*. 1992 or Nicol and de la Mare 1993 and references therein). Thus, a fishery for krill will potentially have effects on at least three trophic levels. The first effect is potentially releasing other zooplankton from competition with krill. The other effects concern the relationships between a variety of predators of krill and other higher predators. Fish eat krill and in turn are eaten by seals, whales and birds. In a special case at the sub-Antarctic island of South Georgia, the linkages may be affected by fishing in a variety of ways but, also, environmental variability may cause annual variation in trophic shifts, making it difficult to attribute changes in the trophic relationships to fishing alone.

Around South Georgia, there are fisheries for krill and mackerel icefish, *Champsocephalus gunnari*. Both these species are eaten by fur seals. Bottom trawl surveys on the South Georgia shelf indicate episodic declines in the abundance of *C. gunnari*, which since 1990 are not directly attributable to commercial fishing (Agnew *et al*. 1998, Everson *et al*. 1999), similar to the observations on the Kerguelen Plateau (de la Mare *et al.* 1998). The greatest declines have been observed in years when krill are known to have been scarce on the South Georgia shelf. It is thought that *C. gunnari* survivorship is closely related to, but indirectly influenced by krill availability. When krill abundance is low fur seals seem more likely to feed on fish, including *C. gunnari* and also *C. gunnari* condition is markedly reduced at these times because of their dependence on krill as a food source (Everson *et al.* 1999).

This emphasizes how knowledge of the variability in the ecosystem and trophic relationships may be important in managing fishing mortality, particularly for target species that are short-lived and when trophic linkages are not strictly hierarchical.

The fishery for krill has developed in the last twenty to thirty years and reached a peak catch of more than

500,000 tons of krill in the early 1980's. There is interannual consistency and predictability in the fishing locations, suggesting that there is some constancy to the spatial and temporal patterns in the abundance of krill. It is clear that the standing biomass of krill is enormous. Indeed, estimates of annual production range from 75 to more than 1500 million metric tons. The issue regarding krill harvest is how much of that can be taken from highly localized areas near the breeding colonies of land based predators, such as marine mammals and birds (Nicol and de la Mare 1993; Butterworth *et al.* 1991, 1992; Everson & de la Mare 1996) without affecting those colonies in measurably negative ways.

A motivation for the fishery for krill was the presumption that over-harvesting and decline of the krill-eating whale stocks left a vast "surplus" of krill "unaccounted for" or "going to waste" each year and which could be harvested (Mackintosh 1970). The argument presumed that the mortality of krill could be partitioned, with a fraction going to the whales, a fraction to the fish, a fraction to the other marine mammals, etc., and that when the whales disappeared, there was no response on the part of the other predators in response to the availability of additional prey. Such partitioning violates the concepts identified by Mangel and Hofman (2000) (see Section 4.2) about the complexity of food chains (Concept #8), expecting multiple effects (Concept #2), adaptability of organisms (Concept #5) and change (Concept #6). Furthermore, consistent with our concept of multiple causes, in at least one case, it has been proposed that the increase in the abundance of stocks of krill predators has nothing to do with the "krill surplus". Fraser *et al*. (1992) argued that the increase in penguin populations was due to a slow decrease in the frequency of cold years.

Pending the development of a full procedure for managing the krill fishery (de la Mare 1996, 1998), CCAMLR has taken a precautionary approach to protecting predators of target species by adopting the krill yield model (CCAMLR 1994) and setting precautionary catch limits in the krill and some finfish fisheries (Constable *et al*. 2000). In the first action, a cap of 1.5 million tons harvest was placed on the annual catch in one of the most important fishing areas. It was further specified that if catch in any area exceeded the previous high commercial take of 620,000 tons, then sub-area quotas would be established. In the second action, a precautionary limit of 390,000 tons was established for the South Indian Ocean, where there is currently exploratory fishing for krill. The krill yield model (Butterworth *et al.* 1991, 1992) is implicitly based on one-way linkages between krill and their predators, in which there is no effect of the predators on krill mortality, and uses a management goal of keeping the krill population size above a critical level. The assumption of one way linkages need not be true (Mangel and Hofman 2000, Concept #7), especially in cases where fishing effort targets not the entire Antarctic krill population, but those stocks that pass close to the land-based breeding colonies of predators (see Murphy *et al*., (submitted), for discussion of natural mortality for krill near to the Antarctic Peninsula). CCAMLR maintains a program of ecosystem monitoring and assessment (see below) to provide data that can be used to assess the status of the krill and their predators, but as of now there is insufficient information to ascertain the detailed nature of the linkages (Constable *et al.* 2000).

Thus, the precautionary approach in CCAMLR takes into account the large-scale relationships between krill, its predators and the fishery. However, it has not yet taken specific account of the potential for localized effects on some land-based krill predators (Everson & de la Mare 1996), nor the need for recovery of some species. Nevertheless, models have been proposed for monitoring local overlaps between predator foraging areas and fishing activities and evaluating the potential competition between the fishery and krill predators (see SC-CAMLR, 1997 for review).

CCAMLR is in the process of implementing the approach for developing management procedures, which is now becoming common-place in fisheries around the world (e.g. Cooke 1999).

The development of management procedures requires:

(i) specification of clear operational objectives, including performance criteria for evaluating management procedures and actions; and

(ii) prospective evaluation of the management procedures, which includes fishing controls, monitoring, and decision rules for altering fishing controls or monitoring, to determine those which satisfy the performance criteria (de la Mare, 1998; Cooke 1999; Sainsbury *et al.* 2000).

Christensen *et al*. (1996) indicate that "ecosystem management is management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure, and function". They concluded that ecosystem management must include the following:

1. Long-term sustainability as a fundamental value.
2. Clear, operational goals.
3. Sound ecological models and understanding
4. Understanding complexity and interconnectedness
5. Recognition of the dynamic character of ecosystems
6. Attention to context and scale.
7. Acknowledgment of humans as ecosystem components
8. Commitment to adaptability and accountability.

For the most part, these points can be addressed during the development of a management procedure. However, points 3 and 4 may be open to differences of opinion as to which models are appropriate as well as being difficult to achieve in practice. Point 7 is the only controlling parameter available to managers in many cases. While we are part of the system we can choose how to be a part of it rather than being considered as an uncontrollable organizing variable.

The approach of developing management procedures described above can explicitly address the uncertainties in the management system arising from a variety of sources such as natural variability, sampling error and hypotheses concerning the causes of observed variation. Such recognition provides opportunities to proceed even if the uncertainties are great.

5.2 Operational objectives

The important ecosystem-oriented objective for CCAMLR is contained in Article II, paragraph 3(b) that requires "maintenance of the ecological relationships between harvested, dependent and related populations of Antarctic marine resources and the restoration of depleted populations to the levels defined in sub-paragraph (a) above". Sub-paragraph (a) refers to a population level that ensures stable recruitment. Beyond this, CCAMLR has not provided an operational interpretation of this objective or determined the critical status of the ecosystem that can be used as a benchmark for ensuring that the general ecosystem objective is being met.

For many harvesting systems, management has centered on single species or "multi-species" assemblages (as distinct from ecological assemblages), where the multiple species have economic interest, particularly in fisheries where the species are all exploited or "managed" in some way (May *et al.* 1979; Beddington *et al.* 1982; Punt & Butterworth 1995; Larkin 1996). Operational objectives based on critical reference points for species that are "ecologically-related" (assemblages) but not directly affected by the fishing operation have been much more difficult to enunciate (e.g. Beddington *et al.* 1982; May *et al.* 1979).

In CCAMLR, this ecosystem objective has been made operational as part of the reference points for individual fished species (the predator criterion – Constable *et al.* 2000) rather than specifically for the related species or assemblages. This criterion provides for krill escapement from the fishery to be 0.75 rather than 0.5 as in the usual single-species approach. This is important because, even though predators are accounted for in part by the natural mortality rate of krill, the total amount consumed (predator food requirements) is contingent on the total abundance of krill. The Scientific Committee of CCAMLR recognizes that the predator criterion of 0.75 may need to be altered as more information on the food requirements of predators becomes available (de la Mare, 1996). CCAMLR is now considering how this objective may be made operational through specifying how much production can be lost from the system through fishing, however, this is only in its formative stages (Constable 2000).

Generic objectives and principles provide the general sentiments to underpin decisions in ecosystem based management. However, impasses over interpreting scientific results and what is an appropriate degree of risk has led to inertia in decision making (Constable *et al.* 2000) and the collapse of fisheries (Ludwig *et al.* 1993). Operational objectives provide clear, quantifiable rules (reference points or target levels) that provide the basis for choosing between alternative management approaches and to provide guidance when management action is required (de la Mare 1998). They also provide the foundation for choosing between different monitoring strategies through the specification of the types of information that need to be included in the decision making process.

A difficulty with determining operational objectives for ecosystems is that theoretical ecology has provided little indication of the important attributes of systems (emergent properties, *sensu* Lewin 1999) despite many studies examining the theoretical roles of diversity in maintaining the resilience and resistance of an ecosystem. This indicates that the choices for operational objectives need to be made in direct relation to the fishery-ecosystem interaction rather than for the ecosystem as a whole (see also Section 5.3). In the absence of being able to define useable target states for ecosystems, threshold conditions need to be elaborated, at the very least, to circumscribe the state or states to be avoided for species or the system as a whole. The operational definition of when a species should be considered threatened and/or endangered is one such example.

Fishing controls, feedbacks and decision rules

Methods for elaborating management procedures for fisheries are now well described (see reviews de la Mare 1998; Butterworth & Punt 1999; Cooke 1999; Sainsbury *et al.* 2000). One of the most important components of these, based on the objectives, is to have a set of rules that use information from assessments (monitoring) to adjust fishing controls when required. Uncertainties in the assessment and management process can lead to stock failures (Ludwig *et al.* 1993; Holt 1998). Protection against such outcomes, along with powerful tests of the effects of fishing, could be achieved through the establishment of closed areas at a scale that provides for ecosystem behaviors in the open and closed areas, independent of the level of fishing in adjacent areas (Mangel 1998, Walters 1998; Mangel 2000, Sainsbury *et al.* 2000). Such an approach could also be used to help discriminate between the effects of fishing and the effects of other types of human intervention.

An important component of elaborating a feedback management procedure is for the indicators to signal that a change in fishing controls is required before it is too late to take action. Some measures, such as species diversity, abundance of higher predators, or recruitment, may be too variable, may not be sensitive to the presence of the fishery, may not be measured with great precision or may have too much inertia to make them inappropriate as feedback signals. The decision rules need to be constructed in such a way that they take account of these difficulties and uncertainties.

Prospective evaluation of management procedures

The evaluation of management procedures prior to their implementation provides the opportunity to eliminate management options that would fail to meet the objectives, thereby potentially avoiding a trial and error approach that has led to stock collapses (e.g. whales – Holt 1998; finfish - Ludwig *et al.* 1993). Methods for the elaboration of new fisheries and for managing existing fisheries while introducing a precautionary approach that accounts for uncertainty have been developed by CCAMLR (Constable *et al.* 2000) and the FAO (FAO 1995). The elements to consider in a management procedure and its evaluation are well described (de la Mare 1998; Sainsbury *et al.* 2000; Smith *et al.* 1999). Such evaluation allows for the implementation of a management procedure that is most likely to achieve the objectives despite uncertainties in the various parts of the system, including the limitations of a monitoring program, such as incomplete data and low power in assessments. It can also be used to ensure that the costs of management are commensurate with the value of the fishery.

Another advantage of the prospective evaluation of management procedures is that it allows one to understand how views of the world affect thinking about management. Hilborn and Walters (1992) identify the following four world-views about ecosystems:

1. "that in the absence of man, there is a balance of nature (also see Pimm 1991); i.e. there is an assumption of global stability ;

2. that the "world is mostly random" with no orderly patterns;

3. that although the world is locally random, it achieves balance through spatial averaging;

4. that ecosystems have a number of intrinsic possible conditions or states, and they periodically bounce between these states depending upon environmental conditions or other external perturbations (such as fishing)...In some cases managers are now including risk of collapse estimates as one indicator to examine in choosing regulations, but the implications of this [world view] are so frightening that most managers simply prefer to ignore it"

The process of developing management procedures can help achieve consensus despite the differences in these views and the variety of plausible hypotheses about how the ecosystem may function (de la Mare, 1998).

5.3 Monitoring

Fisheries can be considered as large-scale perturbation experiments. Importantly, the degree to which the outcomes may be declared significant is dependent upon the magnitude of effect considered to be important, the types of data available to estimate the effect, the spatial and temporal scales of the sampling/monitoring program and the power of the sampling design to detect effects of interest despite the uncertainties present (Peterman 1990; Mangel 1993). The overall effects of fishing are likely to be detectable only when considered at scales consistent with the full spatial extent of the fishery (Thrush *et al.* 1998) and at temporal scales consistent with the time required for effects to become manifest in the system. In ecology, the issue of matching the scale of the program to the scale of the question has been a recurring theme of difficulty over the last 15 years (e.g. Schneider 1994) but has taken considerable time to become a central issue in designing ecological experiments (Constable 1999). More fundamentally, the types of monitoring or experiments required to examine effects and to provide for strong inferences about the factors influencing ecological systems remains hotly debated (Peters 1991; Resetarits & Bernardo, 1998).

It is difficult to study communities at any large scale in a fully experimental manner (Hairston 1991). We must rely upon good thinking, clever experiments (when possible), seizing the opportunities provided by natural perturbations ("natural experiments"), use management as a means of providing information as well as catch (Parma *et al*. 1998), and thoughtfully interpret data to discern relationships. Thus, we must adopt a pluralistic approach to understanding communities and ecosystems, which means "using a diversity of methodologies to obtain data, and a diversity of models to interpret data" (Diamond and Case 1986).

The issue of scale also extends to the biological scale of interest. In some circumstances, sub organismal (non-lethal) conditions may be as important as the state of populations or communities (Underwood & Peterson, 1988). However, effects over large spatial scales are usually considered in terms of the effects on populations.

As discussed earlier, fisheries may alter habitat or modify food webs. Most attention to testing for the effects of fishing using manipulative experiments has been on habitat modification (Jennings & Kaiser, 1998); the spatial effect is mostly fixed to the fishing location and the temporal changes are relatively easy to monitor. Perturbations of food webs, however, may become manifest in locations far away from the fishing location and natural variability in productivity may create difficulties in detecting the effects of reductions in the abundance of one or more species through fishing, except after a number of generations of the resident species. As for determining the limits to marine populations (Rothschild, 1986), the challenge in this regard is to circumscribe the spatial extent of the ecosystem in which the effects of a fishery are likely to be mostly confined.

Despite continuing debate on the efficacy of different scientific methods and the potential difficulty in identifying the scale of effects, these issues and those discussed in section 3.6 of this report, provide the baseline principles for determining whether a monitoring program will be sufficient for addressing questions about the effects of fishing:

i) statements concerning the types and magnitude of effects of interest are necessary before designing full-scale monitoring programs;

ii) interpretation of the results of a monitoring program will be less controversial if its design provides strong inferences (high power) about the effects of fishing; and

iii) the design of the monitoring program must be commensurate with the spatial and temporal scales of the question and the results must not be confounded by other human interventions in the marine ecosystem or, at the very least, *a priori* decisions must be made concerning the tradeoff between making Type I and Type II errors (Mapstone 1992).

Monitoring communities or ecosystems often involves measuring as many features as possible and over many years. A difficulty with this approach is that changes in any one of the parameters may be interpreted in light of the management question regardless of whether it is relevant to the problem at hand. The danger in this for management is that the accumulation of unnecessary parameters may mask the changes in the few parameters essential for making decisions. This is not to say that the monitoring should focus on only a few indicator parameters, but the inclusion of parameters in a monitoring program needs to be a conscious choice as to what role they may play in the final assessments.

Objectives for the management system (see below) need to be couched in operational terms that specify the circumstances (feedbacks) under which catch controls will need to be varied (see below). In so doing, the objectives for the monitoring program will be specified because the overarching objectives will identify the parameters (performance measures) that need to be monitored and the types of assessments required to judge whether catch controls need to be altered. The following types of operational objectives may be considered, consistent with the ecosystem objectives discussed earlier:

1. Species-oriented objectives, e.g. by how much can the probability of collapse of the stock be altered?
2. Habitat-oriented objectives, e.g. how much habitat is required to remain unaltered?
3. Trends or shifts in state variables, e.g. what deviations in environmental state variables can occur before considering the system has changed from its current state and a reevaluation of the monitoring program and catch controls is required?
4. Process-oriented objectives e.g. how much change in ecosystem productivity can be tolerated before changes in the distribution of production between the fishery and the ecosystem need to be made?
5. Fishery-related objectives e.g. has the fishing pattern altered (spatial distribution and level of effort and/or catch)?

Design of monitoring programs

The principles for monitoring the effects of human intervention in the marine environment are becoming well developed and generally involve the approach of replicate sampling before and after the beginning of the intervention and in areas without (controls) and with the specific intervention (Stewart-Oaten *et al.* 1992; Osenberg *et al.* 1994; Underwood 1994; Keough & Mapstone, 1995). These designs endeavor to ensure that conclusions about effects of a particular kind of intervention are not confounded by other factors. They have only rarely been applied to fisheries – e.g. examining the effects of trawl fishing on benthic habitats in northwestern Australia (Sainsbury *et al.* 1997) and the effects of line fishing on coral reef fish assemblages on the Great Barrier Reef (B. Mapstone, James Cook University, Australia, manuscript in preparation). These approaches can provide for feedbacks to management on how catch controls can be adjusted to achieve objectives concerned with the status of the ecosystem (Sainsbury *et al.* 1997; Sainsbury *et al.* 2000) and the performance of the fishery. However, the choice of dependent (indicator) variables and statistical analysis needs to be undertaken carefully (Rice 2000).

In contrast, many fisheries operate at scales that may not be amenable to unambiguous tests of the effects of fishing, particularly effects on food webs. Despite the logical difficulties that may arise in attributing causes of change (Underwood 1997), the objectives for the monitoring program and the accuracy and precision of the assessments need to be considered in the context of making the right decisions to meet the management objectives despite the uncertainties in parameter estimates, limitations of the monitoring program and in the potential hypotheses concerning the causes of change (de la Mare 1998; see below).

Distinguishing anthropogenic and "natural" effects

The importance of identifying whether changes in the food web are caused by fishing or by changes in the ecosystem may be diminished if the management action will be the same, irrespective of the cause. For example, if a change in production of the ecosystem occurs then how is that change (loss or gain) to be shared between the fishery and competing predators? If the decline in competing predators is a product of a long-term decline in ecosystem production and not because of overfishing (though competition may well be occurring under the changed circumstances) then maintaining catches at the existing level would be weighting the objectives towards the fishery. Conversely, if the catches remained the same when ecosystem production increased then the objectives would be weighted towards the ecosystem.

If fishing is the only human intervention in the system, making the operational objectives clear is relatively straightforward. The monitoring program may then only need to identify when trigger points have been reached, including if productivity shifts dramatically, rather than endeavoring to attribute the cause of change to the fishery or to some natural forcing variables. For cases where multiple human interventions overlap, it is important (i) to identify, where necessary, when different kinds of human intervention are affecting productivity in the system and (ii) to ensure that the management responses are appropriately weighted according to the relative importance of the different interventions along with the relative importance of the ecosystem and that such responses occur with sufficient time to ensure the objectives will be met (see below).

The frequency of sampling over time is contingent on how natural variation will be accounted for in the assessment process. For example, in eastern Antarctica, Adelie penguins will have years where reproductive success will be almost zero in the absence of fishing (Mangel and Switzer 1998, Irvine *et al.* submitted). Adelie penguins feed on krill, a commercially exploited species, and reproductive success of penguins is considered an important indicator of the availability of krill (Agnew 1997). The probability of such events occurring will need to be considered in the assessment of the effects of krill harvesting on the Antarctic marine ecosystem. Similarly, reductions in krill availability can cause substantial changes in the relationships amongst species, such as fur seals eating substantial quantities of mackerel icefish around South Georgia when krill are not available (Agnew *et al.* 1998, Everson *et al.* 1999). This adds complexity to achieving ecosystem objectives when there are fisheries for both krill and mackerel icefish in that region and these events are currently unpredictable from one year to the next (see Constable *et al.* 2000 for review).

5.4 Methods of assessing effects

The predominant methods for examining effects of fishing on the ecosystem have relied on multi-species models, mostly focussed on the suite of fish species associated with fisheries (Pope *et al.* 2000; Hollowed *et al.* 2000, Whipple *et al.* 2000). In some cases, the interaction between fish and marine mammals have been explored when culling or harvesting of marine mammals has been a focus of the management system (May *et al.* 1979; Butterworth & Punt, 1999). The application of these models to predict the effects of fishing on other elements of the ecosystem through aggregate or mass-balance models are only in their formative stages and such predictions are extremely uncertain (see Hollowed *et al.* 2000 for review).

In the absence of models that link components of the system with a reasonable degree of certainty, assessments need to rely on suitable metrics of the status of the system and methods for statistically analyzing trends in those metrics and comparing such trends between areas. Most existing community or ecosystem metrics and statistical analyses are difficult to use. To be useful, metrics must both relate to the management tasks and processes of interest, and have statistical procedures that provide strong inferences on changes in the system (see Rice 2000 for review).

CCAMLR has taken steps to resolving some of these issues through two approaches:

First, a program to monitor for the effects of fishing on the Antarctic ecosystem was established in 1986 (Agnew 1997). This “CCAMLR ecosystem monitoring program” (CEMP) targets a set of critical prey and predator species. The former were selected for their key positions in Antarctic ecosystems and their potential as harvestable resources (krill, Antarctic silverfish and early life stages of fish). Selection of predators was based on the criteria that they feed predominantly on the prey species identified, have a wide geographical distribution, represent important ecosystem components, that their biology was sufficiently understood and that sufficient baseline data exist to construct a scientific monitoring program. The present list contains seals, penguins and two species of flying birds.

Second, CCAMLR established a Working Group on Ecosystem Monitoring and Management (WG-EMM). Models of the Antarctic ecosystem are unlikely to be developed with much complexity due to the paucity of data for the region (Constable *et al.* 2000). However, monitoring of specific parameters related to changes in production of the system is possible and remains at the foundation of the CEMP. Recent work has elaborated a procedure for combining these parameters in an assessment process that involves comparing time series of observations for the target species (krill), a combined index of the parameters of krill predators sensitive to changes in the abundance of krill and a combined index of important environmental parameters (de la Mare & Constable 2000). An important step in this process is agreement that significant departures of these indices from a baseline set of observations will signal the need for intervention, either through more refined monitoring and/or a re-assessment of catch controls based on changes in productivity, or recognition that fishing may be affecting the ecosystem. A program of work has been adopted for testing the efficacy of this approach and for integrating this monitoring program into a management procedure for krill (de la Mare & Constable 2000; WG-EMM 2000).

WG-EMM is also considering a new metric that focuses on aggregate production (across all predators) arising from species caught in a fishery (target and by-catch species) and how this may alter with the advent of a fishery (Constable 2000). This will potentially provide a means for specifying operational objectives concerning the acceptable changes in overall system productivity, i.e. how far productivity in predators competing with the fishery may be allowed to deviate from the pre-exploitation baseline production. Even though it is only in its formative stages, this type of approach is endeavoring to target that part of the ecosystem directly influenced by the fishery (primary interaction) and reducing the statistical weight in the assessments of interactions distant from the primary interaction. Also, it reduces the potential for natural short-term and long-term shifts in the relative importance of fished species in the diet of a specific predator to confound the assessment and management process.

5.5 Establishing an ecosystem approach for existing fisheries: operational considerations for the recommendations of the Ecosystem Advisory Panel

The Ecosystem Advisory Panel recommends the development of a Fisheries Ecosystem Plan (FEP) that mimics the Fishery Management Plan and which has these components:

1. Delineate the geographic extent of the ecosystem under consideration.

   This will include an evaluation of the land-water interface as well as circumscribing the most important spatial relationships amongst species

2. Develop a conceptual model of the food web.

   This can often be done, but has difficulties because the same individual, at different times in its life plays different roles in the food web. Pitcher and Hart (1982, pg. 37) show how herring interact with different members of the plankton, depending upon the age of the individual herring.

   In other cases, the sheer numbers of species involved makes creating a food web difficult. For example, more than 35 species are involved in the northwest Atlantic groundfish complex (Boreman *et al.* 1997, pg. xxi; Murawski *et al.* 1997, pg. 62-70); the eastern Bering Sea fishery involves more than 15 species of flatfish, 20 of rockfish, and 4 of roundfish, plus squid (Francis *et al*. 1988, pg. 190). One solution, consistent with Fager's notion of communities as recurrent groups, is to focus on species assemblages (e.g. Rothschild *et al*. 1997, pg. 148). Another is to draw webs of increasing complexity (Mangel 1988, pg. 90-91).

   As discussed above, this task needs to focus on the primary interactions between the fishery and components of the food web and the possible interactions that might provide feedbacks to the primary interaction (see Yodzis 2000).

3. Describe the habitat needs of different life history stages of the organisms in the "significant food web" and how they are considered in conservation and management measures.

4. Calculate total removals -- including incidental mortality -- and show how they relate to standing biomass, production, optimum yields, natural mortality, and trophic structure

5. Assess how uncertainty is characterized and what kind of buffers against uncertainty are included in conservation and management actions.

6. Develop indices of ecosystem health as targets for management

7. Describe available long-term monitoring data and how they are used.

   At this stage, an evaluation of habitat condition, oceanographic variability, potential confounding influences (e.g. terrestrial, freshwater, waste disposal) and scales of interactions amongst these factors need to be described and the overall status of the system related to the targets for management.

8. Assess ecological, human, and institutional elements of the ecosystem that most significantly affect fisheries

   Based on the recent experience in other fora, attention needs also to be given to evaluation of the spatial and temporal manifestations of effects. This is required to verify that the assessments, management decisions and future monitoring activities account for the types of effects that might arise and whether the management system is able to respond to these before irreversible changes occur.

10. Prospective evaluation of management procedures (including monitoring, assessment and decision rules) and the implementation of the precautionary approach
    - Objectives for ecosystem, fishery, competing human interventions
    - relative weighting of management responses given observed changes
    - Reference points
    - Performance measures

6 What methods have been developed for monitoring/assessing ecosystem effects of fisheries?

We now explore and evaluate the types of monitoring and other experimental work used to formulate hypotheses and to document and test for the effects of fishing.

6.1 Characterizing the potential fishing-ecosystem interaction

Most attention has been given to describing fisheries and how these may relate to the ecosystem. This involves monitoring catch of target and by-catch species (fished species), preferably through observers deployed on fishing vessels, documenting the general features of the environment (habitat or food web), through research independent of the fishery, and assessing whether the volume of catch may affect environmental values. Such effects may be explored by simply comparing the abundance taken by the fishery and the abundance in the environment (typically habitat related - a static approach) or, for food webs, by more complicated analyses such as using ecosystem mass-balance or multi-species simulation models (e.g. Bax, 1998). For example, a preliminary analysis, which has been used successfully by CCAMLR, is to study the geographical overlay of the target species, its predators, and the fishery.

These latter analyses provide opportunities for drawing together a disparate set of information and formulating a set of plausible hypotheses concerning the potential effects of fishing on the system (Walters *et al.* 1997; Pauly *et al.* 2000). However, they do not provide powerful tests of effects because they can be limited by how well the parameters of the different models are estimated for the actual system of interest. Also, assumptions about the processes limiting production in the system usually need to be tested, such as whether the system is controlled from the 'top down' (predation) or 'bottom up' (primary production) (Walters *et al.* 1997; Pauly *et al.* 2000; Rice 2000; Hollowed *et al.* 2000).

6.2 Effects of fishing on populations and assemblages

Quantitative assessments of effects most often arise from the combination of catch data and fishery-independent data. The latter are used to assess the abundance of populations and can be used to formulate dynamic models of these populations to assess the effects of the fishery on stocks and other species and how fast recovery might occur if fishing should be reduced. This approach is routinely used in fisheries for single-species (Hilborn & Walters 1992) and multi-species assessments (Daan 1986). It can also be used for examining time trends in the abundance of predators of the target species and how they relate to the dynamics of the fishery. For example, Murphy (1977) shows a high correlation between the catch of anchovetta and the decline of guano birds. He concludes "…the Peru anchovy population was yielding *near the maximum before man began fishing heavily*. This is quite different from the usual point of departure" (italics in the original). Similarly, shifts in the dominance of different species can be observed using time series correlations. In the California current, time series correlation provides evidence for the episodic replacement of sardine by anchovy and vice versa (Bakun 1996, pg. 182).

Multivariate analyses of survey data provide opportunities for assessing spatial and temporal differences in the composition of assemblages and whether shifts in assemblage structure may be associated with changes in the pattern of fishing (spatial distribution and/or magnitude of catch) (Rice 2000).

The role of fishing in causing spatial or temporal differences in species abundance or assemblage structure can be difficult to identify because of the potential influences of other factors on the system (e.g. primary production may be affected by changes in nutrient availability due to terrestrial agricultural practices, climate change or some complex mix of factors such as the 'cyclical' oceanographic states associated with el Nino and la Nina). Nevertheless, the adjustments to catch controls that may be required will depend on the objectives concerning the relative importance of environmental values compare to the values of the fishery and how much the fisheries should be altered in response to changes to the environment through natural or other causes (see also Section 5.3).

6.3 Effects of fishing on ecosystems

The abundance of species in an area will vary from one year to the next without fishing and can include shifts in dominance of species at different trophic levels (e.g. sardine-anchovetta relationship, Bakun 1996). Fishing contributes to changing the abundances of species. The degree to which this affects the system depends on the dynamics of species (recruitment, survivorship, growth, reproduction, migration), the dynamic interaction amongst species (competition, predation, habitat etc.) and the manner in which the fishery causes change in these processes, including changes in the relationship between biota and the physical environment such as might arise from modification of the habitat by bottom trawling.

Modification of habitats by fisheries may cause the assemblages of fish and other species to change to those that are better suited to the altered habitat, as has been observed through a large-scale experimental fishing program on the northwest shelf of Australia (Sainsbury *et al.* 1997). Higher predators that depend almost exclusively on species found in the original habitat would be affected not only by the loss of food but also by the length of time before the habitat returns to its original state (its resilience – Underwood 1989), including the succession of the original fish assemblages. For some sponge assemblages, this may take decades (Dayton 1994).

Fishing may affect food webs by reducing the carrying capacity (available production) for predators of fished species resulting in a decline in the populations of those predators or a change in foraging by those species (i.e. changes in space or prey species). Shifts in trophic structure or community function have been implied by a number of indirect measures of the system, including estimates of species richness, diversity, and mean size of fish caught in fisheries (see Rice 2000 for detailed review). While metrics such as the mean size of fish caught, by and large, indicate that the trophic composition in marine food webs have been altered by fishing (Bianchi *et al.* 2000; Rice 2000), the consequences to some predators, such as marine mammals and birds, are not clear, because of the uncertain strengths of interactions between these species, their prey species and those species for which substantial removals have occurred. For example, the survivorship of marine birds is not expected to decrease as a result of reduced food until prey species are extremely low in abundance and the effect may not be observed for some years (Cairns 1987). In contrast, reproductive success in seabirds is likely to be immediately influenced by and positively correlated with food supply (Mangel and Switzer 1998 show how this effect can be modeled).

There is no conclusive study on the resilience or resistance (*sensu* Underwood 1988) of marine ecosystems to fishing or whether fishing diminishes the resilience of a system as a result of synergies with other environmental interventions (e.g. the potential synergies of ecological processes and different human effects on coral reefs - McManus *et al.* 2000 – and on semi-enclosed seas – Caddy 2000). The only field experiments examining how the removal of fish predators from a food web might affect lower trophic levels have been undertaken in freshwater lakes (Kitchell, 1992). Also, there are currently no measures of ecosystems that can be unambiguously interpreted as to whether the structure and function of the ecosystem has been significantly altered (Murawski 2000, Rice 2000). This is particularly problematic when marine systems can naturally shift from one state to another without human intervention.

As described in section 5.4, CCAMLR has established a program to monitor for the effects of fishing on the Antarctic ecosystem (CEMP). A core set of sites was chosen from within three defined Integrated Study Regions – around South Georgia Island and the Antarctic Peninsula in the southeastern Atlantic and Prydz Bay/Mawson Coast in the southern Indian Ocean. A wider network of sites complement the research within these regions. Within each region, sites were chosen so that distinctions between broad-scale and local-scale changes, and changes occurring in fished areas versus non-fished areas, may be detectable. However, the choice was limited by practical considerations and the presence of established bases and long-term data sets. Selection of 'control' sites proved problematic, because the geographical scale of the effects was expected to be large. Finding comparable sites outside such large areas that had similar environmental and biological characteristics, and where the collection of monitoring data was logistically feasible, was extremely difficult.

Several parameters are monitored for each predator species. The temporal and geographic scales over which these parameters are expected to indicate changes in the status of the ecosystem varies from several weeks and local (reflecting the duration of foraging trips: chick diets and growth) to annual/semi-annual, and region-wide (the weight of birds arriving to breed, breeding success, population size).

Sea-ice and hydrographic conditions are both important features governing the distribution, abundance, movements and recruitment of krill as well as the distribution, winter survival and timing and access to breeding colonies of its predators. To date, methods for monitoring environmental parameters of sea ice cover, local weather and snow cover have been agreed. Other parameters for monitoring the environment and prey species condition are currently being developed. Field work and data acquisition are carried out voluntarily by member states of CCAMLR.

For CEMP to be used in managing fisheries according to the aims of CCAMLR, two more specific objectives need to be addressed (Constable *et al.* 2000). The first objective is to detect effects of fishing in sufficient time for decisions to be taken before irreversible damage is incurred. The second objective is to foresee whether changes in the environment may require re-assessment of the controls on fishing. For example, a continued long-term decline in sea ice extent (de la Mare 1997) may affect the demography and productivity of krill (Loeb *et al*. 1997) and, consequently, a re-assessment of catch limits would be required. A major task for CCAMLR is to determine how the data from CEMP may be synthesized in a way that provides a quantitative basis for making decisions. A second task is to identify what variation in these parameters constitutes ecologically significant variation to which the Commission must respond in setting regulations. This program of work is summarized by Constable *et al.* (2000).

7 References

Abrahams, M.V. and M.C. Healey. 1993. Some consequences of variation in vessel density: a manipulative field experiment. Fisheries Research 15:315-322

Agnew, D. J. 1997. The CCAMLR ecosystem monitoring programme. Antarctic Science, 9: 235-242.

Agnew, D. J., Everson, I., Kirkwood, G. P., and Parkes, G. B. 1998. Towards the development of a management plan for the mackerel icefish (Champsocephalus gunnari) in Subarea 48.3. CCAMLR Science, 5: 63-77.

Aksnes, D. L., K. B. Ulvestad, B. M. Baliño, J. Berntsen, J. K. Egge, E. Svendsen. 1995. Ecological modelling of coastal waters: towards predictive physical-chemical-biological simulation models. Ophelia 41:5-36

Aksnes, D.L. (1996) Natural mortality, fecundity and development. time in marine planktonic copepods implications of behaviour. Mar. Ecol. Prog. Ser. 131, 315-316.

Alverson D.L. 1992. A review of commercial fisheries and the Steller Sea Lion (Eumetopias jubatus): the conflict arena. Reviews in Aquatic Sciences 6:203-256.

Alverson, D.L. 1997. Global assessment of fisheries bycatch and discards: a summary overview. pg 115-125 in Global Trends in Fisheries Management (EK Pikitch, DD Huppert and MP Sissenwine, editors). American Fisheries Society, Bethesda, MD

Anderson, E. N. 1996. Ecologies of the Heart. Emotion, belief, and the environment. Oxford University Press, New York.

Apollonio, S. 1994. The use of ecosystem characteristics in fisheries management. Reviews in Fisheries Science 2:157-180

Apostolakis, G. 1990. The concept of probability in safety assessments of technological systems. Science 250:1359-1364

Ashford, J.R. and J.P. Croxall. 1998. An assessment of CCAMLR conservation measures employed to mitigate seabird mortality in longlining operations for Dissostichus eleginoides around South Georgia. CCAMLR Science, Vol. 5 (1998): 217-230.

Bakun, A. 1996. Patters in the Ocean. Ocean processes and marine population dynamics. California Sea Grant Program, La Jolla, CA

Baliño, B. M. 1996. Eutrophication of the North Sea, 1980-1990. An evaluation of anthropogenic nutrient inputs using a 2D phytoplankton model. Ph.D thesis, University of Bergen, Bergen..

Barbieri, L.R., Chittenden, M.E., and C.M.Jones. 1997. Yield-per-recruit analysis and management strategies for Atlantic croaker, Micropgonias undulatus, in the Middle Atlantic Bight. Fishery Bulletin 95:637-645.

Bax, N. J. 1998. The significance and prediction of predation in marine fisheries. ICES Journal of Marine Science, 55: 997-1030.

Beddington, J. R., and May, R. M. 1982. The harvesting of interacting species in a natural ecosystem;. Scientific American, 247: 42-49.

Bertram, D.F. and G.W. Kaiser. 1993. Rhinoceros auklet (Cerorhinca monocerata) nestling diet may gauge Pacific Sand Lance (Ammodytes hexapterus) recruitment. Canadian Journal of Fisheries and Aquatic Sciences 50:1908-1915.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Jin, X., Koranteng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F., and Zwanenburg, K. 2000. Impack of fishing on size composition and diversity of demersal fish communities. ICES Journal of Marine Science, 57: 558-571.

Boreman, J., Nakashima, B.S., Wilson, J.A., and R.L.Kendall. 1997. Northwest Atlantic Groundfish: Perspectives on a Fishery Collapse. America Fisheries Society, Bethesda, MD

Brook, B.W., O'Grady, J.J., Chapman, A.P., Burgman, M.A., Akcakaya, H.R. and Frankham, R. 2000. Predictive accuracy of population viability analysis in conservation biology. Nature 404:385-387

Butterworth, D.S., Gluckman, G.R. and S. Chalis. 1992. Further computations of the consequences of setting the annual krill catch limit to a fixed fraction of the estimate of krill biomass from a survey. Preprint, Working Group Krill, 92/4. Commission for the Conservation of Antarctic Marine Living Resources, Hobart, TAS, Australia.

Butterworth, D. S., and Punt, A. E. 1999. Experiences in the evaluation and implementation of management procedures. ICES Journal of Marine Science, 56: 985-998.

Butterworth, D.S., Punt, A.E. and M. Basson. 1991. A simple approach for calculating the potential yield from biomass survey results. CCAMLR Document Working Group Krill 91/23, Commission for the Conservation of Antarctic Marine Living Resources, Hobart, TAS, Australia.

Caddy, J. F. 2000. Marine catchment basin effects versus impacts of fisheries on semi-enclosed seas. ICES Journal of Marine Science, 57: 628-640.

Cairns, D. K. 1987. Seabirds as indicators of marine food supplies. Biological Oceanography, 5: 261-271.

CCAMLR. 1994. Report of the Thirteenth Meeting of the Commission. CCAMLR, Hobart.

Charles, A.T. 1992. Fishery conflicts. A unified framework. Marine Policy, September: 379-393

Christensen, N. L., Bartuska, A. M., Brown, J. H., Carpenter, S., D'Antonio, C., Francis, R., Franklin, J. F., MacMahon, J. A., Noss, R. F., Parsons, D. J., Peterson, C. H., Turner, M. G., and Woodmansee, R. G. 1996. The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. Ecological Applications, 6: 665-691.

Clark, C.W. 1985. Bioeconomic Modelling and Fisheries Management. Wiley-Interscience, New York

Clark, C.W. 1990. Mathematical Bioeconomics. 2nd ed. Wiley-Interscience, New York

Cochrane, K.L. 2000. Reconciling sustanability, economic efficiency and equity in fisheries: the one that got away? Fish and Fisheries

Constable, A. J. 1999. Ecology of benthic macro-invertebrates in soft-sediment environments: a review of progress towards quantitative models and predictions. Australian Journal of Ecology, 24: 452-476.

Constable, A. J. 2000. The ecosystem approach to managing fisheries: achieving conservation objectives for predators of fished species. Paper WG-EMM-00/60 submitted to the CCAMLR Working Group on Ecosystem Monitoring and Management, Sicily, Italy. 17 pp.

Constable, A. J., de la Mare, W. K., Agnew, D. J., Everson, I., and Miller, D. 2000. Managing fisheries to conserve the Antarctic marine ecosystem: practical implimentation of the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR). ICES Journal of Marine Science, 57: 778-791.

Cooke, J. G. 1999. Improvement of fishery-management advice through simulation testing of harvest algorithms. ICES Journal of Marine Science, 56: 797-810.

Crowder, L.B. and S.A.Murawski. 1998. Fisheries bycatch: implications for management. Fisheries 23:8-17

Cushing, D.H. 1981. Fisheries Biology. University of Wisconsin Press, Madison, WI

Dayton, P. K., Mordida, B. J., and Bacon, F. 1994. Polar marine communities. American Zoologist, 34: 90-99.

de la Mare, W. K. 1996. Some recent developments in the management of marine living resources. In Frontiers of Ecology, pp. 599-616. Ed. by R. B. Floyd, A. W. Sheppard and P. J. D. Barro. CSIRO Publishing, Melbourne.

de la Mare, W. K. 1997. Abrupt mid-twentieth-century decline in Antarctic sea-ice extent from whaling records. Nature, 389: 57-60.

de la Mare, W. K. 1998. Tidier fisheries management requires a new MOP (management-oriented paradigm). Reviews in Fish Biology and Fisheries, 8: 349-356.

de la Mare, W. K., and Constable, A. J. 2000. Utilising data from ecosystem monitoring for managing fisheries: Development of statistical summaries of indices arising from the CCAMLR Ecosystem Monitoring Program. CCAMLR Science, in press.

de la Mare, W. K., Williams, R., and Constable, A. J. 1998. An assessment of the mackerel icefish (Champsocephalus gunnari) off Heard Island. CCAMLR Science, 5: 79-102.

Dennis, B. 1996. Discussion: should ecologists become Bayesians? Ecological Applications 6(4),1095-1103.

Diamond, J. and T.J. Case. 1986. Community Ecology. Harper and Row, New York

Dunham, J.B. and G.L. Vinyard. 1997. Incorporating stream level variability into analyses of site level fish habitat relationships: some cautionary examples. Transactions of the American Fisheries Society 126:323-329

Egge, J. K. and D. L. Aksnes. 1992. Silicate as regulating nutrient in phytoplankton competition. Mar. Ecol. Prog. Ser. 83:281-289.

Egge, J. K., Ø. Fiksen, D. L. Aksnes, T. Harboe, A. Jacobsen and Ø. Rykkelid. 1994. Summary of data from enclosure experiments at Espegrend in 1986-1993. IFM-Rapport no. 36/1994, University of Bergen, Bergen 1:3-21

Everson, I. 1992. Managing southern ocean krill and fish stocks in a changing environment. Philosophical Transactions of the Royal Society B 338:311-317

Everson, I., and de la Mare, W. K. 1996. Some thoughts on precautionary measures for the krill fishery. CCAMLR Science, 3: 1-12.

Everson I., G. B. Parkes, K.-H. Kock and I.L. Boyd. 1999. Variation in standing stock of the mackerel icefish Champsocephalus gunnari at South Georgia. Journal of Applied Ecology 1999, 36, 591-603.

Fager, E.W. 1957. Determination and analysis of recurrent groups. Ecology 38:586-595

Fager, E.W. 1963. Communities of organisms. pg 415-437 in The Sea, Volume 2 (M.N. Hill, editor). Interscience, New York

FAO. 1995. Technical Consultation on the Precautionary Approach to Capture Fisheries and Species Introductions. FAO, Rome. 54 pp.

Farber, D.A. 1999. Eco-pragmatism. Making sensible environmental decisions in an uncertain world. University of Chicago Press, Chicago, IL.

Fluharty, D., Aparicio, P., Blackburn, C., Boehlert, G., Coleman, F., Conkling, P., Costanza, R., Dayton, P., Francis, R., Hanan, D., Hinman, K., Houde, E., Kitchell, J., Langton, R., Lubchenko, J., Mangel, M., Nelson, R., O□Connell, V., Orbach, M and M. Sissenwine. 1999 Ecosystem-Based Fishery Management. A report to Congress. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service

Fowler, C.W. and J.A. MacMahon. 1982. Selective extinction and speciation: their influence on the structure and functioning of communities and ecosystems. American Naturalist 119: 480-498

Fox, W.W. 1988. Comments in biological assumptions in fishery management. Springer Lecture Notes on Coastal and Estuarine Studies 28:103-105

Francis, R.C., Adlerstein, S.A., and R.D. Brodeur. 1988. Biological basis for management of commercial fishery resources of the Eastern Bering Sea. Springer Lecture Notes on Coastal and Estuarine Studies 28:187-209

Francis, R.I.C.C. 1992 Use of risk analysis to assess fishery management strategies: a case study using orange roughy (Hoplosthethus atlanticus) on the Chatham Rise, New Zealand. Canadian Journal of Aquatic Sciences and Management 49:922-930

Fraser, W.R., Trivelpiece, W.Z., Ainley, D.G. and S.G. Trivelpiece. 1992. Increases in Antarctic penguin populations: reduced competition with whales or a loss of sea ice due to environmental warming? Polar Biology 11:525-531

Freon, P. and O.A. Misund. 1999. Dynamics of Pelagic Fish Distribution and Behaviour: Effects on Fisheries and Stock Assessment. Fishing News Books, Oxford, UK

Fryer, J.K.1998. Frequency of pinniped-caused scars and wounds on adult spring-summer chinook and sockeye salmon returning to the Columbia River. North American Journal of Fisheries Management 18:46-51

Gabriel, W.L. and P. M. Mace. 1999. A review of biological reference points in the context of the precautionary approach. In V.R. Restrepo (Editor), Proceedings of the Fifth National NMFS Stock assessment Workshop: Providing Scientific Advice to Implement the Precautionary Approach Under the Magnuson-Stevens Fishery Conservation and Management Act, p. 34-45. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-F/SPO-40.

Garcia, S. M. and C. Newton. 1997. Current situation, trends and prospects in world capture fisheries. pg 3-27 in Global Trends in Fisheries Management (EK Pikitch, DD Huppert and MP Sissenwine, editors). American Fisheries Society, Bethesda, MD

Gislason, H., Sinclair, M., Sainsbury, K., and O'Boyle, R. 2000. Symposium overview: incorporating ecosystems objectives within fisheries management. ICES Journal of Marine Science, 57: 468-475.

Grumbine, R.E. 1994. What is ecosystem management? Conservation Biology 8:27-38

Hairston, N. G. 1991. Ecological Experiments: Purpose, Design, and Execution. Cambridge University Press, Cambridge.

Hanan, D.A., Holts, D.B., and A.L. Coan. 1993. The California drift gill net fishery for sharks and swordfish 1981-82 through 1990-91. Fish Bulletin 175, State of California, The Resources Agency, Department of Fish and Game, Long Beach, CA

Harris, P.J. and J.M. Dean. 1998. Characterization of king mackerel and Spanish mackerel bycatches of South Carolina shrimp trawlers. North American Journal of Fisheries Management 18:439-453

Harwell, M.A. 1997. Ecosystem management of South Florida. BioScience 47:499-512

Hilborn, R. and C.J.. Walters. 1992. Quantitative Fisheries Stock Assessment. Choice, dynamics and uncertainty. Chapman and Hall, New York

Hilborn, R. and M. Mangel. 1997. The Ecological Detective. Confronting models with data. Princeton: Princeton University Press.

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

Holt, S. J. 1998. Points of View. Fifty years on. Reviews in Fish Biology and Fisheries, 8: 357-366.

Howson, C. and P. Urbach. 1993. Scientific Reasoning: The Bayesian Approach. Open Court Publishing. Co., La Salle, Ill.

Hubalek, Z. 1982. Coefficients of association and similarity, based on binary (presence-absence) data: an evaluation. Biological Reviews 57:669-689

Hughes, T.P. 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. Science 265:1547-1551

Hunt, G.L., Heinemann, D., and I. Everson. 1992. Distributions and predator-prey interactions of macaroni penguins, Antarctic fur seals, and Antarctic krill near Bird Island, South Georgia. Marine Ecology Progress Series 86:15-30

Hutchings, J.A. 1996. Spatial and temporal variation in the density of northern cod and a review of hypotheses for the stock□s collapse. Canadian Journal of Fisheries and Aquatic Sciences. 53:943-962

Hutchings, J.A. and R.A. Myers. 1994. What can be learned from the collapse of a renewable resource? Atlantic cod, Gadus morhua, of Newfoundland and Labrador. Canadian Journal of Fisheries and Aquatic Sciences 51:2126-2146

Irvine, L. G., Clarke, J. R., and Kerry, K. R. submitted. Poor breeding success of the Adelie penguin at Bechervaise Island in the 1998-99 season. Antarctic Science, .

Jennings, S., and Kaiser, M. J. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology, 34: 201-352.

Keough, M. J., and Mapstone, B. D. 1995. Protocols for Designing Marine Ecological Monitoring Programs Associated with BEK Mills. Pages 185 pp;. National Pulp Mills Research Program, Canberra.

Kitchell, J. F. 1992. Food web management: a case study of Lake Mendota. Springer-Verlag, New York, N.Y. 553 pp. pp.

Kokko, H., Lindström, J, and E. Ranta. 1997. Risk analysis of hunting of seal populations in the Baltic. Conservation Biology 11:917-927

Koslow, J. A., and Gowlett-Holmes, K. 1998. The Seamount Fauna off Southern Tasmania: Benthic Communities, their Conservation and Impacts of Trawling. Pages 104. CSIRO Division of Marine Research, Hobart, Australia.

Kurlansky, M. 1997. Cod. Penguin Books, New York

Laist, D.W., Coe, J.M., and K.J. O'Hara. 1999. Marine debris pollution. Pg 342-366 in Conservation and Management of Marine Mammals (JR Twiss, Jr. and RR Reeves, editors). Smithsonian Press, Wasington, DC

Lane, D.E. and R.L. Stephenson. 1997. Decision analysis. Pg 203-209 in Northwest Atlantic Groundfish: Perspectives on a Fishery Collapse. (Boreman, J., Nakashima, B.S., Wilson, J.A., and R.L.Kendall, editors). America Fisheries Society, Bethesda, MD

Langton, R.W. and R.L. Haedrich. Ecosystem-based management. Pg 153-158 in Northwest Atlantic Groundfish: Perspectives on a Fishery Collapse. (Boreman, J., Nakashima, B.S., Wilson, J.A., and R.L. Kendall, editors). America Fisheries Society, Bethesda, MD

Larkin, P.A. 1996. Concepts and issues in marine ecosystem management. Reviews in Fish Biology and Fisheries. 6:139-164

Larkin, P. A. 1996. Concepts and issues in marine ecosystem management. Reviews in Fish Biology and Fisheries, 6: 139-164.

Lee, K.N. 1997. Sustaining salmon: three principles. pg 665-675 in Pacific Salmon and Their Ecosystems (DJ Stouder, PA Bisson, and RJ Naiman, editors). Chapman and Hall, NY

Leggett, W.C, and E. Deblois. 1994. Recruitment in marine fishes: is it regulated by starvation and predation in the egg and larval stages? Netherlands Journal of Sea Research 32:119-134

Lewin, R. 1999. Complexity. 2nd edition. University of Chicago Press, Chicago, IL

Liermann, M. and R. Hilborn. 1997. Depensation in fish stocks: a hierarchic Bayesian meta-analysis. Canadian Journal of Fisheries and Aquatic Sciences 54: 1976-1984

Lo, N.C.H., Smith, P.E., and J.L. Butler. 1995. Population growth of norther anchovy and Pacific sardine using stage-specific matrix models. Marine Ecology Progress Series 127:15-26

Loeb, V., Siegel, V., Holm-Hansen, O., Hewitt, R., Fraser, W., Trivelpiece, W., and Trivelpiece, S. 1997. Effects of sea-ice extent and salp or krill dominance on the Antarctic food web. Nature, 387: 897-900.

Loehle. 1987. Errors of construction, evaluation, and inference: a classification of sources of error in ecological models. Ecological Modelling 36:297-314.

Lotka, A. J. (1925) Elements of Physical Biology. Williams & Wilkins Company, Baltimore. 460 pp.

Ludwig, D., Hilborn, R., and Walters, C. 1993. Uncertainty, resource exploitation, and conservation: lessons from history. Science, 260: 17, 36.

MacCall, A.D. 1986. Changes in the biomass of the California Current ecosystem. pg 33-54 in Variability and Management of Large Marine Ecosystems (K. Sherman and L.M. Alexander, editors). AAAS Selected Symposium 99, Westview Press, Boulder, CO

Malacoff, D. 1999. Bayes offers a 'new' way to make sense of numbers. Science 286 16 Nov : 1460-4.

Mangel, M. 1988 From egg surveys to ecosystem models: Biological assumptions in fisheries management. Springer Lecture Notes on Coastal and Estuarine Studies 28:75-102

Mangel, M. 1993. Effects of high-seas driftnet fisheries on the northern right whale dolphin Lissodephis borealis. Ecological Applications 3:221-229

Mangel, M. 2000. On the fraction of habitat allocated to marine reserves. Ecology Letters 3:15-22

Mangel, M. and 42 coauthors 1996.. Principles for the conservation of wild living resources. Ecological Applications 6:338-362

Mangel, M. and P.V. Switzer. 1998. A model at the level of the foraging trip for the indirect effects of krill (Euphausia superba) fisheries on krill predators. Ecological Modelling, 105:235-256

Mangel, M. and R.J. Hofman. 1999. Ecosystems. Patterns, processes and paradigms.

Mapstone, B. D. 1992. Scalable decision criteria for environmental impact assessment: effect size, Type I, and Type II errors. Ecological Applications, submitted.

May, R.M., Beddington, J.R., Clark, C.W., Holt, S.J., and R.M. Laws. 1979. Management of multispecies fisheries. Science 205:267-277

Mayo, D. G. 1996. Error and the Growth of Experimental Knowledge.University of Chicago Press, Chicago, IL

McManus, J. W., Menez, L. A. B., Kesner-Reyes, K. N., Vergara, S. G., and Ablan, M. C. 2000. Coral reef fishing and coral-algal phase shifts: implications for global reef status. ICES Journal of Marine Science, 57: 572-578.

Mesa, M.G., Poe, T.P., Maule, A.G., and C.B. Schreck. 1998. Vulnerability to predation and physiological stress responses in juvenile chinook salmon (*Oncorhynchus tshawytscha*) experimentally infected with *Renibacterium salmoninarum*. Canadian Journal of Fisheries and Aquatic Sciences 55:1599-1606

M'Gonigle, R.M. 1997. A new naturalism in international fisheries law? Fisheries 22:24-25.

MRAG Americas. 1997. Consolidated report on the Peer Review of Red Snapper Lutjanus campechanus research and management in the Gulf of Mexico. NMFS Office of Science and Technology, December 1997.

Murawski, S. A. 2000. Definitions of overfishing from an ecosystem perspective. ICES Journal of Marine Science, 57: 649-658.

Murawski, S.A., Maguire, J-J, Mayo, R.K., and F.M. Serchuk. 1997. Groundfish stock and the fishing industry. Pg 27-70 in Northwest Atlantic Groundfish: Perspectives on a Fishery Collapse. (Boreman, J., Nakashima, B.S., Wilson, J.A., and R.L.Kendall, editors). America Fisheries Society, Bethesda, MD

Murphy, E.J., Constable, A.J. and D.J. Agnew, Modelling the dynamics of krill populations in the Antarctic Peninsula region. ICES Journal of Marine Science. submitted

Myers, R.A., Bridson, J. and Barrowman. 1995. Summary of Worldwide Spawner and Recruitment Data. Canadian Technical Report of Fisheries and Aquatic Sciences 2020. Department of Fisheries and Oceans, Northwest Atlantic Fisheries Centre, St. John's Newfoundland.

Myers, R.A., Mertz, G., and NJ. Barrowman. 1995. Spatial scales of variability in cod recruitment in the North Atlantic. Canadian Journal of Fisheries and Aquatic Sciences 52:1849-1862

Nicol, S. and W. de la Mare. 1993. Ecosystem management and the Antarctic krill. American Scientist 81:36-47

Norris, K, Bannister, R.C.A., and P.W. Walker. 1998. Changes in the number of oystercatchers Haematopus ostralegus wintering in the Burry Inlet in relation to the biomass of cockles Ceastoderma edule and its commercial exploitation. Journal of Applied Ecology 35:75-85

Northridge, S. 1995. Environmental mismanagement on the high seas: a retrospective analysis of the squid and tuna driftnet fisheries of the North Pacific. Report to US Marine Mammal Commission PB95-238945

Olver, C.H., Shuter, B.J., and C.K. Minns. 1995. Toward a definition of conservation principles for fisheries management. Canadian Journal of fisheries and Aquatic Sciences 52:1584-1594

Osenberg, C. W., Schmitt, R. J., Holbrook, S. J., Abu-Saba, K. E., and Flegal, A. R. 1994. Detection of environmental impacts: natural variability, effect size, and power anlaysis. Ecological Applications, 4: 16-30.

Parma, A., Amarasekare, P., Mangel, M., Moore, J., Murdoch, W.W., Noonburg, E., Pascual, M.A., Possingham, H.P., Shea, K., Wilcox, C. and D. Yu. What can adaptive management do for our fish, forests, food and biodiversity? 1998 Integrative Biology 1:16-26

Parsons, L.S. and J.S. Beckett. 1997. Fisheries management in Canada: The case of Atlantic groundfish. Pg 73-79 in Global Trends in Fisheries Management (EK Pikitch, DD Huppert and MP Sissenwine, editors). American Fisheries Society, Bethesda, MD

Parsons, T.R. 1992. The removal of marine predators by fisheries and the impact of trophic structure. Marine Pollution Bulletin 25:1-4

Pauly, D. and V. Christensen. 1995. Primary production required to sustain global fisheries.Nature 374:255-257

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and F.C. Torres, Jr. 1998. Fishing down marine food webs. Science 279- 860-863

Pauly, D., Christensen, V., and Walters, C. 2000. Ecopath, ecosim, and ecospace as tools for evaluating ecosystem impact fisheries. ICES Journal of Marine Science, 57: 697-706.

Pennoyer, S. 1997. Bycatch management in Alaska groundfish fisheries. pg 141-150 in Global Trends in Fisheries Management (EK Pikitch, DD Huppert and MP Sissenwine, editors). American Fisheries Society, Bethesda, MD

Peterman, R. M. 1990. Statistical power analysis can improve fisheries research and management. Canadian Journal of Fisheries and Aquatic Sciences, 47: 2-15.

Peterman, R.M.,and J.L. Anderson. 1999. Decision analysis: a method for taking uncertainties into account in risk-based decision making. Human and Ecological Risk Assessment 5:231-244

Peters, R. H. 1991. A Critique for Ecology. Cambridge University Press, Cambridge. 365 pp.

Pimm, S. 1991. The Balance of Nature? University of Chicago Press, Chicago, IL

Pitcher, T.J. 2000. Ecosystem goals can reinvigorate fisheries management, help dispute reslution and encourage public support. Fish and Fisheries 1:99-103

Pitcher, T.J. and D. Pauly. 1998. Rebuilding ecosystems, not sustainability, as the proper goal of fishery management. Pg 311-329 in Reinventing Fisheries Management (TJ Pitcher, PJB Hart and D. Pauly, editors), Chapman and Hall, London, UK

Pitcher, T.J. and P.J.B. Hart. 1982. Fisheries Ecology. Chapman and Hall, London, UK

Policansky, D. 1998. Science and decision making in fisheries management. Pg 57-71 in Reinventing Fisheries Management (TJ Pitcher, PJB Hart and D. Pauly, editors), Chapman and Hall, London, UK

Pope, J. G., D.S., M., Daan, N., Reynolds, J. D., and S., J. 2000. Gauging the impact of fishing mortality on non-target species. ICES Journal of Marine Science, 57: 689-696.

Press, W.H. 1997. Understanding data better with Bayesian and global statistical methods. Pg 49-60 in Unsolved problems in Astrophysics (Bahcall, J.N. And Ostriker, J.P., editors) Princeton University Press, Princeton, NJ

Price, P. 1984. Alternative paradigms in community ecology. pg 353-383 in A New Ecology. Novel Approaches to Interactive Systems (P.W. Price, C.N. Slobodchikoff, and W.S. Gaud, editors). John Wiley and Sons, New York

Price, P.W., Slobodchikoff, C.N., and W.S. Gaud. 1984. A New Ecology. Novel Approaches to Interactive Systems (P.W. Price, John Wiley and Sons, New York

Punt, A. 1997. The effect of future consumption by seals on catches of Cape Hake off South Africa. pg 134-140 in Global Trends in Fisheries Management (EK Pikitch, DD Huppert and MP Sissenwine, editors). American Fisheries Society, Bethesda, MD

Punt, A. E., and 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 bilogical interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the Cape hakes *Merluccius capensis* and *M. paradoxus*. South African Journal of Marine Science, 16: 255-285.

Regier, H.A. 1997. Old traditions that led to abuses of salmon and their ecosystems. Pg 17-28 in Pacific Salmon and Their Ecosystems (DJ Stouder, PA Bisson, and RJ Naiman, editors). Chapman and Hall, NY

Resetarits, W. J., and Bernardo, J., eds. 1998. Experimental ecology: issues and perspectives. Oxford University Press, New York.

Restrepo, V.R., Hoenig, J.M., Powers, J.E., Baird, J.W., and S.C. Turner. 1992. A simple simulation approach to risk and cost analysis, with applications to swordfish and cod fisheries. Fishery Bulletin US 90:736-748

Rice, J. 2000. Evaluating fisher impacts using metrics of community structure. ICES Journal of Marine Science, 57: 682-688.

Ridley, M. 1999. Genome. Harper Collins, New York.

Roberts, C.M. 1995. Effects of fishing on the ecosystem structure of coral reefs. Conservation Biology 9:998-995

Roell, M.J. and D.J. Orth. 1998. Indirect effects of fishery exploitation and pest control in a riverine food web. North American Journal of Fisheries Management 18:337-346

Rosenberg, A.A. and V.R. Restrepo. 1994. Uncertainty and risk evaluation in stock assessment advice for U.S. Marine fisheries. Canadian Journal of Fisheries and Aquatic Sciences 51:2715-2720

Rothschild, B. J. 1986. Dynamics of marine fish populations. Harvard University Press, Cambridge, Ma. 277 pp.

Rothschild, B.J., Ault, J.S., Goulletquer, P., and M. Heral. 1994. Decline of the Chesapeake Bay oyster population: a century of habitat destruction and overfishing. Marine Ecology Progress Series 111:29-39

Rothschild, B.J., Sharov, A.F., and M. Lambert. 1997. Single-species and multispecies management. Pg 141-152 in Northwest Atlantic Groundfish: Perspectives on a Fishery Collapse. (Boreman, J., Nakashima, B.S., Wilson, J.A., and R.L.Kendall, editors). America Fisheries Society, Bethesda, MD

Sainsbury, K.J., Campbell, R.A., Lindholm, R., and A.W. Whitelaw. 1997. Experimental management of a Australian multispecies fishery: examining the possibility of trawl-induced habitat modification. pg 107-112 in Global Trends in Fisheries Management (EK Pikitch, DD Huppert and MP Sissenwine, editors). American Fisheries Society, Bethesda, MD

Sainsbury, K. J., Punt, A. E., and Smith, A. D. M. 2000. Design of operational management strategies for achieving fishery ecosystem objectives. ICES Journal of Marine Science, 57: 731-741.

SC-CAMLR. 1997. Report of the Sixteenth Meeting of the Scientific Committee. Pages 438. CCAMLR, Hobart.

Schneider, D. 1994. Quantitative Ecology: spatial and temporal scaling. Academic Press. 395 pp.

Sissenwine, M.P. and J.G. Shepherd. 1987. An alternative perspective on recruitment overfishing and biological reference points. Can. J. Fish. Aquat. Sci. 44:913-918.

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

Smith, P.E., Lo, N. C-H., and J.L. Butler. 1992. Life-stage duration and survival parameters related to interdecadal population variability in Pacific sardine. Cal COFI Reports 33:

Smith, T.D.. 1994. Scaling Fisheries. The science of measuring the effects of fishing, 1855-1955. Cambridge University Press, New York

Spencer, P.D. 1997. Optimal harvesting of fish populations with nonlinear raters of predation and autocorrelated environmental variability. Canadian Journal of Fisheries and Aquatic Sciences. 54:59-74

Spencer, P.D. and J.S. Collie. 1997. Patterns of population variability in marine fish stocks. Fisheries Oceanography 6:188-204

Starr, P., Annala, J.H., and R. Hilborn. 1998. Contested stock assessment: two case studies. Canadian Journal of Fisheries and Aquatic Sciences 55:529-537

Stenseth, N. 1985. Darwinian evolution in ecosystems: the Red Queen view. pg 55-72 in Evolution: Essays in Honour of John Maynard Smith (P. J. Greenwood, P. H. Harvey and M. Slatkin, editors), Cambridge University Press, New York.

Stewart-Oaten, A., Bence, J. R., and Osenberg, C. W. 1992. Assessing effects of unreplicated perturbations: no simple solutions. Ecology, 73: 1396-404.

Taylor, D.M., O□Keefe, P.G., and C. Fitzpatrick. 1993. A snow crab, Chionoecetes opilio (Decapoa, Majidae), fishery collapse in Newfoundland. Fishery Bulletin 92:412-419

Taylor, J.P. 1999. Making Salmon. An environmental history of the Northwest fisheries crisis. University of Washington Press, Seattle.

Thrush, S.F., Hewitt, J.E., Cummings, V.J., Dayton, P.K., Cryer, M., turner, S.J., Funnell, G.A., Budd, R.G., Milburn, C.J., and M.R. Wilkinson. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecological Applications 8:866-879

Trippel, E.A. 1995. Age at maturity as a stress indicator in fisheries. BioScience

Tuljapurkar, S. & H. Caswell (eds.) 1997. Structured-population models in marine, terrestrial, and freshwater systems. Population and community biology series 18, Chapman & Hall, 643 pp.

Tuljapurkar, S.D. (1990) Population dynamics in variable environments. Lecture notes in Biomathematics 85. New York: Springer Verlag.

Underwood, A. J. 1989. The analysis of stress in natural populations. Biological Journal of the Linnean Society, 37: 51-78.

Underwood, A. J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecological Applications, 4: 3-15.

Underwood, A. J. 1997. Experiments in Ecology: Their Logical Design and Interpretation using Analysis of Variance. Cambridge University Press, Melbourne, Australia. pp.

Underwood, A. J., and Peterson, C. H. 1988. Towards an ecological framework for investigating pollution. Marine Ecology Progress Series, 46: 227-234.

Walters, C. 1998. Designing fisheries management systems that do not depend upon accurate stock assessment. In Reinventing Fisheries Management, pp. 279-288. Ed. by T. J. Pitcher, P. J. B. Hart and D. Pauly. Kluwer Academic Publishers, London.

Walters, C., Christensesn, V., and Pauly, D. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries, 7: 139-172.

Werner, E. E. and Gilliam, J. F. (1984). The ontogenetic niche and species interactions in size-structured populations. Ann. Rev. Ecol. Syst. 15: 393-425.

WG-EMM. 2000. Report of the Working Group on Ecosystem Monitoring and Management. Scientific Committee for the Conservation of Antarctic Marine Living Resources, Hobart. 204 pp.

Whipple, S.J., Link, J.S., Garrisn, L.P., and M.J. Fogarty. 2000. Models of predation and fishing mortality in aquatic ecosystems. Fish and Fisheries 1:22-40

Winemiller, K.O and K.A. Rose. 1992. Patters of life-history diversification in North American Fishes: Implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences 49:2196-2218

Yodzis, P. 2000. Diffuse effects in food webs. Ecology, 81: 261-266.

Appendix I. Quantitative Approaches and a Conceptual Framework for Thinking About Interactions

Humans may be direct competitors with other species for the same resource. For example, Murphy (1977) showed the take of Peruvian anchovetta viewed solely from the perspective of human harvesting was sustainable, but when both human and bird take was included, the level of harvest was not sustainable. Also see Pitcher and Hart (1982, pg 231). Whipple et al. (2000) recently reviewed models of predation and fishing mortality in aquatic ecosystems.
Here we illustrate how there may be unanticipated indirect effects due to fishing. Imagine a food web

Zooplankton --> Small fish ---> Large fish ---> Mammals and birds

Denoting the population size of small fish by K, of large fish by V, and by mammals and mammals and birds by P, the population dynamics of the latter two may be modeled as

$$\frac{dV}{dt} = rV(1-\frac{V}{K}) - mPV$$

$$\frac{dP}{dt} = cPV - dV$$

where r is the maximum per capita growth rate of the large fish, mP is the per capita mortality of large fish due to mammals, cV is the per capita birth rate of large fish, and d is the per capita death rate of mammals. The steady state of the system is determined as the solution of

$$P = \frac{r}{m}(1-\frac{V}{K})$$

$$P = \frac{d}{c}$$

A simple analysis shows that there is a minimal population size of the small fish for the persistence of the predators: predators can persist only if K > d/c. Thus, a fishery for small fish (e.g. a bait-fish fishery) that consequently reduces K can lead to extinction of the predators two trophic levels above. Most fisheries focus on large rather than small fish (P rather than V) and thus often compete directly, rather than indirectly, with other top predators.