Scientific Review of the Harvest Strategy Currently Used in the BSAI and GOA Groundfish Fishery Management Plans

Draft Report

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1 SCIENTIFIC REVIEW OF THE HARVEST STRATEGY CURRENTLY USED IN THE BSAI AND GOA GROUNDFISH FISHERY MANAGEMENT PLANS

1.1 Executive Summary

1.1.1 Introduction

The North Pacific Fisheries Management Council (NPFMC) convened this panel to provide an independent scientific review of the current harvest strategy embodied in the NPFMC fisheries management plan (FMP) for the Bering Sea/Aleutian Islands/Gulf of Alaska groundfish fisheries, with particular attention to the role played by the $F_{40\%}$ reference point, and to determine whether changes should be made to account for particular species, or ecosystem needs in accordance with the National Standards of the Magnuson-Stevens Fisheries Conservation and Management Act (MSFCMA).

1.1.2 Charge to the Panel

The review panel was charged specifically to carry out three tasks.

1. To define and explain the harvest strategy currently used in the management of the BSAI and GOA groundfish fisheries; i.e., develop an educational primer on the Council's current procedure.
2. To determine if the current quota setting approach (Tier ABC determination, OFL derivation, and TAC specification) is consistent with the Magnuson-Stevens Act. In particular, determine if F_{40} is an appropriate MSY substitute for all species, and if not, to explain the alternative(s) and describe the data available to implement them.
3. To determine if the current quota setting approach is considerate of ecosystem needs in the BSAI and GOA, and if not to explain how it should be changed, what data are available for implementation of the changes, and how the transition to the changed approach might be carried out.

1.1.3 Explanation of the Current Harvest Strategy

The current harvest strategy is essentially a maximum sustainable yield (MSY) single-species approach, modified by some formal safeguards incorporated to ward against *overfishing* as defined from the single-species standpoint, and with opportunities of a less-structured nature for reducing harvest rates further in response to perceived social, economic and ecological concerns. No quantitative standards or specific decision rules are stated for these latter considerations, except as they are imposed, from outside the MSFCMA, by the Endangered Species Act (ESA) or the Marine Mammal Protection Act (MMPA), and only for particular populations.

The overfishing level (OFL) set for each stock is an estimate either of the fishing mortality rate associated with MSY (F_{MSY}) or an estimate of a surrogate for F_{MSY}. The OFL is treated in the management system as a limit that should not be exceeded except with a very low probability. The acceptable biological catch (ABC) set for each stock is an estimate of a target rate, which is intended to establish some margin between it and the OFL. The hope is that managing so as to achieve this target on average will accomplish the desired compliance with exceeding the limit (OFL) only rarely. The ad hoc downward adjustments of harvest in response to other social, economic, and ecological considerations takes place in the deliberations where the total allowable catch (TAC) is set subject to the constraint that it be less than or equal to the ABC.

The formulaic component of the reduction of harvest rate from the theoretical MSY harvest rate (from OFL to ABC) is by an amount that is often modest, when expressed as a fraction of the harvest rate; but in terms of the total tonnage involved, or its dollar value, the amount is considerable. The margin is also small relative to real natural variation, and small relative to the practical uncertainty about stock status or population parameters for many of the target stocks and indeed for most of the ecosystem. By contrast, in actual practice, the reduction of the TAC from the ABC has for some stocks and some years been quite large, but there is no explicit and general formula for this reduction.

The formal and standardized quantitative portions of the process of determining OFL and ABC begin with the assignment of each stock to one of six "Tiers" based on the availability of information about that stock. Tier 1 has the most information, and Tier 6 the least. The so-called $F_{40\%}$ construct, which is one focus of our review, plays a prominent role in some of the Tiers (2, 3, and 4) but not the others. Notably, in Tier 3 (which is where many of the major BSAI/GOA stocks are assigned) and Tier 4, the estimate of $F_{40\%}$ is used as a surrogate for a fishing mortality rate that is somewhat below F_{MSY}.

$F_{40\%}$ is the calculated fishing mortality rate at which the equilibrium spawning biomass per recruit is reduced to 40% of its value in the equivalent unfished stock. This is an esoteric, but useful, measure of the amount by which the associated fishing rate reduces the stock size, in the long run. The useful features of this particular measure are two-fold. First, its calculation is less sensitive to the details of the stock-recruitment relationship than is the calculation of F_{MSY}, so it is practical to estimate $F_{40\%}$ for stocks that are not well enough studied for estimation of F_{MSY}. The second is that, for a range of dynamics encompassing many, but not all, of the BSAI/GOA target groundfish stocks, modeling studies have shown that harvesting at $F_{35\%}$ accomplishes about the same thing as harvesting at F_{MSY}, so harvesting at the slightly lower rate, $F_{40\%}$, establishes a modest margin of safety.

In fact, the dynamics of only one stock covered by the FMP, BSAI pollock, are well-enough quantified to qualify for Tier 1. In Tier 1 the limiting F_{OFL} is the equivalent of the point estimate of F_{MSY} (that is to say, roughly, the "best" estimate without adjusting for uncertainty), and the target F_{ABC} is the harmonic mean of the distribution of the estimate for F_{MSY}. The harmonic mean has the mathematical property that it is less than the simple average (roughly, the point estimate) by an amount that increases with the spread of the distribution, so this establishes a margin that increases with the uncertainty in the estimate. However, this mechanism for adjusting the F_{ABC} downward from the F_{OFL} does not have the statistical property of ensuring a *constant specified* *confidence* that the F_{ABC} does not exceed the true F_{MSY}, as would be ensured by using a lower confidence limit of the estimate of F_{MSY} for the F_{ABC}.

Tier 2 differs from Tier 1 in that only point estimates of the key population parameters are available, so the *distribution* of the estimate for F_{MSY} is not known. In this Tier, the limiting F_{OFL} is the point estimate of F_{MSY}, much as in Tier 1, but a different formula (based on the adjustment used in Tier 3) is used for adjusting the F_{ABC} downward from F_{OFL}. The mathematics of the different formulas used for adjusting the F_{ABC} downward from F_{OFL} in Tier 1 and Tier 2 does not guarantee that the margin so established in Tier 2 will be wider than the margin in Tier 1.

Tier 3 differs from Tier 2 in that information is insufficient for any estimation of MSY. In this Tier, the limiting F_{OFL} is the point estimate of $F_{35\%}$ and the target F_{ABC} is the point estimate of $F_{40\%}$. The width of the margin between F_{ABC} and F_{OFL}, in this Tier, therefore, will be essentially the same as in Tier 2, and the relation to the width of the margin in Tier 1 is variable. Most of the major target stocks in the BSAI/GOA are in Tier 3.

Tier 4 differs from Tier 3 in that information is insufficient for estimation of target biomass levels. In this Tier, the limiting F_{OFL} is the point estimate of $F_{35\%}$, and the target F_{ABC} is the point estimate of $F_{40\%}$, both as in Tier 3. The width of the margin between F_{ABC} and F_{OFL}, in this Tier, therefore, will be identical to that in Tier 3, and essentially the same as in Tier 2, and the relation to the width of the margin in Tier 1 is variable.

Tier 5 differs from Tier 4 in that information is insufficient for estimating $F_{40\%}$ or $F_{35\%}$, so the limits and targets use different surrogates to attempt to approximate management for MSY. In this Tier, the limiting F_{OFL} is the point estimate of the natural mortality rate of the stock, and the target F_{ABC} is three fourths of that value. The limiting F_{OFL} in this Tier maybe either conservative or aggressive relative to the limiting F_{OFL} of $F_{35\%}$ in the three Tiers above. Theoretical work [Deriso 1982 among others and Thompson] has shown that M is often higher than F_{MSY}, so it would be a better as a limit than a target. The margin between F_{ABC} and F_{OFL} in this Tier, corresponding to a 25% reduction of fishing mortality rate, is wider than the margin in Tiers 2 through 4. Most of the minor target stocks in the BSAI/GOA are in Tier 5.

Tier 6 differs from Tier 5 in that information is insufficient for estimating any of the stock parameters, and all that is known is the catch history. In this Tier, the limiting F_{OFL} is the average historic catch, and the target F_{ABC} is three fourths of that value. In practice, without estimates of stock size, the control is exerted simply through a limit on amount of catch. The margin between F_{ABC} and F_{OFL}, in this Tier, considered as a fractional reduction, is the same as in Tier 5.

In Tiers 1 through 3 there are provisions for rapid rebuilding of stocks from an overfished condition, by reductions in the target fishing mortality rate triggered whenever the estimate of stock biomass is below the target biomass. There is no such provision in Tiers 4 through 6.

In Tiers 1 through 5, the information on the stock is sufficient to give clear indications if the stock status is departing substantially from the management goals. In Tier 6, this is not the case.

We see that for the most part there is not a clear systematic progression in increasing conservatism in the targets or in the width of the margin between target and limit, in moving from the Tiers with more information to those with less. Similarly, there is not, for the most part, a clear systematic incentive, in terms of potential for greater harvest, to improve the information base in order to move a stock from Tiers with less information to Tiers with more. Finally, the control rule provisions to accelerate rebuilding of stocks from an overfished condition do not apply to the 3 Tiers with the least information, and which, therefore, are subject to the greatest uncertainties. Within Tier, almost all the inputs to the control rule are point estimates, and so these do not adjust in response to uncertainty either.

Over time, the evolution of this management system has been in the direction, overall, of greater conservatism. By the standards of most of the world's large commercial fisheries, this management system is conservative.

The adequacy (and safety) of $F_{35\%}$ as a surrogate for F_{MSY} depends on the inherent productivity of the stock. For most of the BSAI/GOA target stocks this surrogate appears to be adequate, though the case of the GOA pollock stock, which has declined from its 1985 stock size under this management system, warrants a closer look. This surrogate is now believed to be inappropriate for less productive stocks, such as sharks and rockfish, and it is now thought that considerably lower harvest rates (considerably lower than $F_{40\%}$ as well) should be applied for those stocks.

In practice, this management system seems to have worked well, judged simply by the continuing productivity of the target stocks, for the bulk of the BSAI/GOA stocks in recent decades, most of which period has corresponded to a regime phase which began in 1976 and is thought to have ended only recently. The definite exceptions to this empirical record of success are the rockfish, which were overfished early on, and have not recovered (except that GOA Pacific ocean perch have rebuilt above the $B_{40\%}$ level). A further possible exception is the GOA pollock which has declined since 1985. The robustness of the management system to large regime changes is largely untested in practice, and has been explored in models only in a limited way. If the regime has in fact recently changed it is possible that some of the stocks are entering a period of lower productivity, which may itself cause some populations to decline. Overall, there has been only limited modeling analysis of the theoretical performance of the system as a whole, in realistic scenarios. Realistic scenarios should include realistic representation of the spatial distribution of stock abundances and the spatial distribution of fishing, with various possible underlying stock-recruitment relationships, and various kinds of uncertainty in the input information that becomes the basis for the stock assessments which in turn are the sources of the estimates that are used to assign stocks to Tiers and to generate the values for F_{OFL} and F_{ABC} according to the rules for that Tier.

1.1.4 Single Species Considerations

The $F_{35\%}$ and $F_{40\%}$ proxies for MSY used in the groundfish FMPs are defensible, for this purpose, in that these values are supported by a body of scientific literature as being reasonable F_{MSY} proxies for "typical groundfish" species. However, the Council should be aware that harvests taken at these levels may be too high for species that have very low productivity and that are characterized by highly episodic recruitment. The Tier system could improve if allowances were made for the different life history types covered by the FMPs.

The management system contained in the groundfish FMPs is generally consistent with the single-species/target-stock components of the MSFCMA. While the FMPs specify only one of the two status determination criteria that are required by NMFS' National Standard Guidelines, the FMPs are sufficiently conservative, with respect to the target stocks evaluated from a single-species perspective, and incorporate automatic rebuilding plans to such a degree for stocks in Tiers 1 through 3 (the Tiers with the better availability of information) that this lack of conformity with the Guidelines should not pose a conservation danger from a single species viewpoint, except possibly for Tiers 4 through 6.

In terms of Optimum Yield, there is uncertainty about the conformity of the FMP definitions with the MSFCMA. The Council should review and revise its OY specifications in order to make more explicit links with environmental considerations and to more directly specify the relationship between OY and MSY for GOA groundfish.

In a single-species/target-stock context, the TAC-setting process employed by the Council is a very conservative one, at least for Tiers 1 through 5, and the in-season monitoring and management system seems adequate for implementing the TACs with little risk of exceeding them.

We recommend that a management strategy evaluation (MSE) analysis, along the lines described in Section 3 of this report, be undertaken to provide additional assurance that the current NPFMC ABC harvest strategy is a robust one and is likely to continue to meet the objectives of MSFCMA and of NPFMC itself (noting that the actual harvest strategy is difficult to define except to say that it is ≤ABC). We recognize that an MSE analysis can be potentially a time-consuming and technically difficult undertaking. Sufficient resources in time and people would need to be allocated to undertake the work. The skills and expertise to undertake the work already reside within AFSC.

There is obviously a wide range of alternative harvest strategies that might be considered, and MSE methods are a useful way to design and evaluate alternatives. If this "comparative" approach is used, a wider set of performance measures, including utilization as well as conservation objectives, should be evaluated and the tradeoffs across objectives highlighted. We suggest that wider stakeholder discussion on alternative approaches be held before embarking on a major exercise to evaluate alternatives.

Apart from exploring and evaluating generic harvest strategies, several of the target species in the BSAI/GOA groundfish fishery are of sufficient value (and importance) to warrant the effort to formally evaluate species-specific harvest strategies (e.g., for pollock). This would allow more of the detailed knowledge and understanding about these species and associated fishery to be incorporated in the operating models, and could potentially lead to better performing harvest strategies for those species. It would also allow changes to harvest strategies that occur for other reasons to be more formally evaluated. An example is the recent change to the pollock harvest control rule to set zero ABCs if the stock falls below the MSST. This change was brought in because of concerns about food chain impacts of the fishery on Steller sea lions.

Overall, the current NPFMC approach to advising on ABCs appears to meet the requirements of MSFMCA, from a single-species/target-stock management perspective for most of the target stocks (the exceptions are primarily the rockfish). Precautionary elements in the current NPFMC approach derive from the additional constraints in the overall management system that often result in catches well below ABCs. Nevertheless, the review panel recommends that additional work be undertaken to more formally test the robustness of the current NPFMC harvest strategy to various uncertainties, and to explore alternative harvest strategies that may be more appropriate for some groups of species or individual species. Existing staff at AFSC have the expertise and a range of suitable models to undertake the MSE approach suggested, but time and resources will need to be allocated for such a task.

1.1.5 Ecosystem Considerations

The panel was asked to consider two basic questions about the ecosystem aspects of the present NPFMC groundfish fishery management plan and the role of $F_{40\%}$ in it. These are (1) Is the approach "considerate" of ecosystem needs in the BSAI and GOA? and (2) Are data available to implement an alternative approach for satisfying ecosystem needs? Our brief response is that the MSY based approach in the setting of F_{ABC} in the current NPFMC system for groundfish management, which is consistent with the explicit OY goals of the MSFCMA, makes only a slight adjustment for *possible* ecosystem needs; while the TAC setting adjustment downward from ABC allows for considerable reduction in harvest, but the procedure for doing so is *ad hoc*. The available data could be used for a more ambitious, and more formalized, decision system that might be more protective of ecosystem considerations. However, the available data have not, to date, proven sufficient to demonstrate conclusively that more protection is or is not needed. Present legislative policy mandates in the MSFCMA are not explicit enough about the burden of proof in deciding between utilization and protection goals to determine how much protection of ecosystem considerations is legally required when the uncertainty about the needs for such protection is great. Other legislation, notably MMPA and ESA, is much clearer about the burden of proof and the required standards of protection for special species, and actual FMPs have been modified to conform when those regulatory frameworks have come into play. Resolution of this question for other non-target species, and for the ecosystem as a whole, will require the articulation of more specific policy.

These comments are not peculiar to the $F_{40\%}$ driven aspects of the FMP. They would apply to any single species MSY-based, or MSY-surrogate, approach, as indeed they apply to the management of Tier 1, Tier 5 and Tier 6 stocks in the BSAI/GOA FMP where $F_{40\%}$ does not play a role. Regardless of the use of $F_{40\%}$ as a F_{MSY} surrogate, fishing so as to achieve MSY-related objectives will inevitably reduce the equilibrium biomass very substantially from the unfished condition, and will inevitably shift considerably the age and size structure of the target stock. These changes to the target stock *could* propagate through the food web, and effect large changes in the populations of other species. However, the theoretical models for predicting such effects in practice have low predictive power, and the intensity of monitoring required to document such changes for particular species, and to attribute causation convincingly, require a major undertaking. Furthermore, with the exception of species listed under the ESA, there are no general policy standards for whether effects of this kind, or of any particular magnitude, are acceptable consequences of management.

The $F_{40\%}$ approach to estimating the ABC, by itself, is inherently a single species approach. It is thought that for most of the target species in the FMP, a fishing mortality rate of $F_{35\%}$ would be appropriate for achieving long-term catches near MSY, under the condition of an unchanged oceanographic regime. The main exceptions among the target species are the rockfish, which apparently need a considerably lower fishing mortality rate to avoid overfishing. That the actual target fishing rate is $F_{40\%}$ rather that $F_{35\%}$ creates some additional margin of safety, from a single species perspective, for target species excluding rockfish. The decision to use $F_{40\%}$ rather than $F_{35\%}$ was deliberately protective, and was intended to function as a buffer against several sources of uncertainty, including the concern that theoretical models have shown that managing each species for its single species MSY will not achieve MSY for the aggregate. Nevertheless, it is not clear *how much* of the margin between $F_{35\%}$ and $F_{40\%}$ was "allocated" to ecosystem considerations. Nor was a calculation carried out to demonstrate what amount of escapement is needed for ecosystem purposes, or to assess whether the margin between fishing at $F_{35\%}$ and $F_{40\%}$ supplies this amount.

The TAC setting process has provisions for adjusting the allowed catch downward from the ABC, and in practice the TAC is adjusted downward. Such adjustments are made for considerations of by-catch, protected species, and general concern about the ecosystem. Again, except for the adjustments in response to the very specific requirements of ESA, it is not clear how the magnitude of this downward adjustment of the TAC from a $F_{40\%}$-based ABC is chosen, how much of it is attributed specifically to ecosystem considerations, and whether there are specific grounds for believing the magnitude is enough for those purposes.

It is easy enough to say that a management system could be made more protective of ecosystem properties by building additional margins of safety into a fishing mortality rate rule (such as shifting to $F_{50\%}$ or $F_{60\%}$ for example) or stipulating a more stringent threshold on the total allowed depression of equilibrium biomass (such as the limit adopted in the Commission for the Conservation of Antarctic Marine Living Resources Convention). But current knowledge does not allow precise scientific specification of what margin or threshold would be appropriate to achieve what level of protection of various ecosystem properties.

Modeling can offer up hypothetical scenarios to illustrate various possible outcomes, but multispecies ecosystem modeling has not yet developed to the point where it has documented predictive power in real applications. Nevertheless, this modeling is very interesting on several grounds, and continued investment in developing and testing such models is warranted.

At present, we essentially face a sliding scale of possible ecosystem protective measures, where the choices are largely policy choices. Current policy guidance is insufficiently specific, and the available science is insufficiently conclusive about the precise magnitudes of expected effects. Given the scientific uncertainty, there is merit in approaching ecosystem management in the spirit of cautious experimentation supported by a large investment in carefully-designed monitoring.

In chapter 4, this report explores a variety of frameworks for expressing ecosystem goals, and a spectrum of management approaches that might be conducive to achieving those goals. The large uncertainties, and the overt appeal to experimental management, put a high premium on continuing and expanding the regular monitoring in this ecosystem, along with surveys of the fishery resources, and oceanographic survey programs.

Currently available data might well be adequate for implementing imaginable ecosystem control rules. But currently available data almost certainly are not sufficient for specifying the quantitative details of such general ecosystem control rules in the absence of more explicit policy formulations. We can hope that continued research and monitoring will improve our general understanding of the BSAI/GOA ecosystems. There is reason to expect that the present increases in research directed specifically at population dynamics of the Steller sea lion will bring more satisfactory resolution to the vexing outstanding questions about causes of the decline of *that* population and its possible relation to the fishery. Elucidation of broader aspects of the ecosystem, and their relationship to the fishery, may prove to be an even greater challenge.

In the context of fishery management that takes ecological and ecosystem considerations into account, reserves (marine protected areas) play two extremely important roles. First, a no-take marine reserve of sufficient size will allow one to maintain a source of baseline data for components of the ecosystem. This is important because we should expect change to occur in ecosystems. Without having a source of baseline data in which there is no (or at least limited) human intervention, it will often be difficult to ascertain whether changes are due to fishing or other factors. Second, for stocks that have complicated social structure (eg sex-changing fish or harem or lek breeding marine mammals or birds), a no-take marine reserve will allow a full representation of the social structure of that stock; such social structures might otherwise be truncated by either direct or indirect effects of fishing. The effectiveness of a reserve for conservation purposes will depend on the relationship between the reserve size, and the natural spatial structure and dispersal rates of the populations. If these spatial scales coincide, the results could be counter productive: then closed areas may result in protection within the area but an *increased* chance of depression outside.

Monitoring plays a crucial role in making less tractable problems more tractable. Monitoring of catch, by-catch and fishing effort is of course critical to the data gathering that supports the assessments of status of the target stocks. Thus we recommend that the Observer Program be maintained and improved to provide even more precise and accurate information about directed catches and bycatch of all species. Systematic and well-designed monitoring is also essential for determining the magnitudes and timing of real environmental variation, such as regime shift, and it is at the heart of all experimental approaches to ecosystem management which hopeful will increase our knowledge about the ecosystem and reveal which management strategies work and which do not. It is important that the program of surveys in the BSAI/GOA ecosystem be continued, and perhaps extended even further to provide adequate information for addressing the ecosystem question.

1.2 Introduction

At its October 2001 meeting, the North Pacific Fishery Management Council passed a "final motion on Steller sea lions" (Council Newsletter, October 2001, Attachment 1). As part of this action, the Council moved "to seek an independent scientific review of the $F_{40\%}$ harvest policy relative to national standards." At its December 2001 meeting, the Scientific and Statistical Committee interpreted the subject of the review to be "the current groundfish harvesting strategy" and requested that terms of reference be developed, to include the following features: 1) a description of the issue, 2) the purpose of the review, and 3) a list of charges to be addressed.

Harvests in the BSAI and GOA fisheries are governed by the respective fishery management plans (FMPs). Identification of an explicit "harvest strategy" in these FMPs is somewhat problematic. In a broad sense, the FMPs themselves *are* the harvest strategy. However, the FMPs allow for a wide range of possible harvests for any given stock in any given year, meaning that, in a narrower sense, the plans are consistent with a large number of particular harvest strategies. Of course, any harvest allowed by the FMPs is required to be consistent with the National Standards described in the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA). Of particular relevance in this regard is National Standard 1, which states, "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry" (Title III, Section 301(a)(1)). Optimum yield, in turn, is defined as that which (Section 3(28)):

A. will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems;
B. is prescribed as such on the basis of the maximum sustainable yield from the fishery, as reduced by any relevant economic, social, or ecological factor; and
C. in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the maximum sustainable yield in such fishery.

The review panel was charged specifically to carry out three tasks:

1. To define and explain the harvest strategy currently used in the management of the BSAI and GOA groundfish fisheries; i.e., develop an educational primer on the Council's current procedure.
2. To determine if the current quota setting approach (tier ABC determination, OFL derivation, and TAC specification) is consistent with the Magnuson-Stevens Act. In particular, determine if F_{40} is an appropriate MSY substitute for all species, and if not, to explain the alternative(s) and describe the data available to implement them.
3. To determine if the current quota setting approach is considerate of ecosystem needs in the BSAI and GOA, and if not to explain how it should be changed, what data are available for implementation of the changes, and how the transition to the changed approach might be carried out.

The exact charge to the panel is reproduced in the next section.

The panel met in Seattle, at the Alaska Fisheries Science Center, for 3 days of briefings and discussions, on June 17-19, and developed this report over the subsequent 3 months.

1.3 Terms of Reference

Scientific Review of the Harvest Strategy Currently Used In the BSAI and GOA Groundfish Fishery Management Plans

Terms of Reference

At its October 2001 meeting, the North Pacific Fishery Management Council passed a "final motion on Steller sea lions" (Council Newsletter, October 2001, Attachment 1). As part of this action, the Council moved "to seek an independent scientific review of the F40 harvest policy relative to national standards."

At its December 2001 meeting, the Scientific and Statistical Committee interpreted the subject of the review to be "the current groundfish harvesting strategy" and requested that terms of reference be developed, to include the following features: 1) a description of the issue, 2) the purpose of the review, and 3) a list of charges to be addressed. These features are provided sequentially below.

1) Description of the Issue

Harvests in the BSAI and GOA groundfish fisheries are governed by the respective fishery management plans (FMPs). Identification of an explicit "harvest strategy" in these FMPs is somewhat problematic. In a broad sense, the FMPs themselves are the harvest strategy. However, the FMPs allow for a wide range of possible harvests for any given stock in any given year, meaning that, in a narrower sense, the plans are consistent with a large number of particular harvest strategies. Of course, any harvest allowed by the FMPs is required to be consistent with the National Standards described in the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA). Of particular relevance in this regard is National Standard 1, which states, "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry" (Title III, Section 301(a)(1)). Optimum yield, in turn, is defined as follows (Section 3(28)):

The term "optimum", with respect to the yield from a fishery, means the amount of fish which—

A. *will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems;*
B. *is prescribed as such on the basis of the maximum sustainable yield from the fishery, as reduced by any relevant economic, social, or ecological factor; and*
C. *in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the maximum sustainable yield in such fishery.*

In recent months, concern has been expressed regarding the extent to which harvests allowed under the FMPs are consistent with protection of marine ecosystems, as required implicitly by National Standard 1.

2) Purpose of the Review

The purpose of the independent scientific review is as follows: to critically review the current harvest strategy as applied to our FMP fisheries, and determine whether changes need to be made to account for particular species, or ecosystem needs.

3) Charges to be Addressed

The independent scientific review shall address the following:

a) *Define and explain the harvest strategy currently used in the management of the BSAI and GOA groundfish fisheries; i.e., develop an educational primer on the Council's current procedure.*
b) *Determine if the current quota setting approach (tier ABC determination, OFL derivation, and TAC specification) is consistent with the Magnuson-Stevens Act. Determine if F40 is an appropriate MSY substitute for all species? If not, what are the alternative(s) and are data available to determine the value(s) of the substitute?*
c) *Is the approach considerate of ecosystem needs in the BSAI and GOA?*
    i. *If not, how should it be changed?*
    ii. *Are sufficient data available to allow implementation of the alternative approach?*
    iii. *How would the transition from the current approach to the proposed revised one be handled?*

In addressing the above questions, the reviewers shall:

a) *use whatever scientific information or methodology is appropriate and practicable within the time allotted for the review;*
b) *describe the role played by the $F_{40\%}$ reference point in their findings; and*
c) *relate their findings to the MSFCMA's National Standards, particularly National Standard 1.*

1.4 Glossary

ABC	Acceptable Biological Catch. As a starting point, scientists set ABC equal to F_{MSY} applied to the exploitable biomass, and if necessary, decreased to incorporate "safety factors and risk assessment due to uncertainty". This starting maxABC may be subsequently modified by the Plan Team, by the SSC, or by the Council
ADF&G	Alaska Department of Fish and Game
AFSC	Alaska Fisheries Science Center (NOAA-NMFS)
AP	Advisory Panel. A panel made up of industry representatives and members of various interest groups which advises the Council on matters such as TACs based on the ABC recommendations made by the SSC, modified by other (non-scientific) considerations
B	Biomass level
B_{init}	pre-fishing biomass level
B_{MSY}	Biomass level associated with MSY
BRP	Biological Reference Point (see Reference Points). A benchmark against which stock abundance or fishing mortality can be measured, in order to determine its status. BRPs can be categorized as limits or targets, depending on their intended use
BSAI	Bering Sea and Aleutian Islands
$B_{40\%}$	Biomass level associated with average (normal) recruitment and fishing at rate that gives spawning biomass per recruit reduced to 40% of that of the unfished population (similar meaning for other % values)
Control Rule	Describes a plan for pre-agreed management actions as a function of variables related to the status of the stock. For example, a control rule can specify how F should vary with biomass. The NPFMC Tier System prescribes different control rules for stocks, depending on the quality of information available
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CPUE	Catch per Unit of Effort
CRAWDAD	Control Rule Alternatives Workshop: Design, Analysis, Decision
EA	Environmental Assessment
EA/RIR	Environmental Assessment / Regulatory Impact Review
EBS	Eastern Bering Sea
ESA	Endangered Species Act

F	Fishing mortality rate
F_{ABC}	Fishing mortality associated with the Acceptable Biological Catch (ABC)
$F_{0.1}$	Fishing mortality at which the slope of equilibrium yield per recruit (YPR) is reduced to 10% of the slope when F=0
$F_{40\%}$	Fishing mortality rate at which the spawning biomass per recruit is at 40% of the unfished value (similar for other % values)
F_{max}	Fishing mortality that maximizes the yield per recruit
F_{med}	Fishing mortality rate corresponding to an equilibrium SPR equal to the inverse of the median observed ratio of recruits to spawning biomass
F_{MSY}	Fishing mortality rate which, if applied constantly, would result in MSY
F_{OFL}	Fishing mortality rate which, if applied constantly, would just constitute overfishing
$F_{\%SPR}$	Fishing mortality rate that results in x% equilibrium spawning potential ratio
FEP	Fisheries Ecosystem Plan. Similar to an FMP, a plan intended to assess and monitor fishery impacts on an ecosystem
FMP	Fishery Management Plan
GAO	General Accounting Office. Reports to U.S. Congress
GOA	Gulf of Alaska
IFQ	Individual Fishing Quota
ITQ	Individual transferable quota
K	Carrying capacity. Equilibrium number of individuals for that environment
L	Fish length
M	Natural mortality rate
maxABC	Maximum target catch
Management Strategy	A combination of data collection, assessment and decisions (control rules) that follow pre-specified rules
MRB	Maximum Retainable By-Catch
MSE	Management Strategy Evaluation. A formal evaluation and comparison of management strategies, with respect to how well they meet management objectives

MFMT	Maximum Fishing Mortality Threshold. Status Determination Criterion of the NSGs intended for determining if overfishing is occurring
MMPA	Marine Mammal Protection Act
MSA	MSFCMA. Magnuson-Stevens Fisheries Conservation and Management Act. Magnuson Fisheries Conservation and Management Act (MSFCMA) as amended by the Sustainable Fisheries Act (SFA**)**
MSFCMA	Magnuson-Stevens Fishery Conservation and Management Act. U.S. Public Law 94-265, as amended through October 11, 1996. Available as NOAA Technical Memorandum NMFS-F/SPO-23, 1996
MSST	Minimum Stock Size Threshold. Status Determination Criterion of the NSGs intended for determining if a stock is in an overfished condition
MSY	Maximum Sustainable Yield. Largest long-term average yield (catch) that can be taken from a stock (or stock complex) under prevailing ecological and environmental conditions
National Standards	A series of ten Standards (objectives) that the MSFCMA requires of Fishery Management Plans
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NPFMC	North Pacific Fisheries Management Council
NRC	National Research Council, of the US National Academy of Science
NSGs	National Standard Guidelines. Advisory guidelines developed by NMFS, based on the National Standards of the MSFCMA, intended to assist in the development of FMPs. Guidelines on National Standard 1 were published in the Federal Register on May 1, 1998
OFL	Overfishing level. Equivalent to the MFMT of the NSGs
OY	Optimum Yield
PBR	Potential Biological Removal. A limit for incidental take of marine mammals under the Marine Mammal Protection Act (MMPA)
pdf	probability density function, a description of a distribution
PSC	Prohibited Species Catch
PSEIS	Programmatic Supplemental Environmental Impact Statement
PT	Plan Teams. Groups of experts who review the SAFE reports with a principal focus on methodological issues. The Plan Teams provide recommendations to the SSC

Reference Points	Values of parameters (e.g., $B_{40\%}$, $F_{35\%}$) that are useful benchmarks for guiding management decisions. Biological reference points are typically limits that should not be exceeded with significant probability (e.g., OFL) or targets for management (e.g., OY); see also BRP
SAFE	Stock Assessment and Fishery Evaluation. Reports prepared by a Plan Team
SDC	Status Determination Criteria. Objective and measurable criteria used to determine if a stock is being overfished or is in an overfished state according to NSGs
SEIS	Supplemental Environmental Impact Statement
SPR	Spawning Potential Ratio: The expected lifetime spawning output per recruit relative to the spawning output that would be realized in the absence of fishing, expressed as a percentage
SSB	Spawning Stock Biomass: A measure of the reproductive output of a stock
SSC	Statistical and Scientific Committee. This body reviews the SAFE reports and Plan Team recommendations for OFL and ABCs, which it then forwards to the Advisory Panel with possible adjustments
TAC	Total Allowable Catch. The intended target catch for each stock
UNCLOS	United Nations Convention on Law of the Sea
UNFA	Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks
W	fish weight
α	Use 1. Recruits per egg at very small spawning density. A term in both the Beverton-Holt and Ricker density dependent models of the stock-recruitment relationship Use 2. Threshold biomass level, specified as a fraction of the target biomass level (B_{MSY} or $B_{40\%}$), below which the NPFMC control rule sets the allowed fishing mortality to zero. In the current (1998) version of the control rule, this fraction is 0.05

2 A PRIMER ON FISHERIES MANAGEMENT AS CONDUCTED IN THE BSAI GOA GROUNDFISH FISHERIES

2.1 Introduction

Fisheries are complex dynamic systems, involving physical, biological and human dimensions. Within those dimensions, innumerable elements inter-relate and change through time. Observing those elements and understanding the relationships between them is difficult, and being able to predict the fate of all system elements accurately is impossible.

And yet, despite this complexity and limited predictability, the goal of fisheries management is, as far as possible, to make sense of the various dimensions and elements, and to make decisions on alternative policies in the face of uncertainty. How decisions are made, what research is done in support of those decisions, and which dimensions and elements are considered most important, varies from one region of the world to another.

Before considering how fisheries are managed, it is worth thinking a bit more about the dimensions and elements. The physical dimension consists of elements of the oceanographic environment; its relationship to weather systems, and linkages with coastal and terrestrial systems. The elements of the physical dimension are themselves dynamic, thus creating a highly variable backdrop to the biological processes that underpin fishery resources. The biological dimension consists not only of the fish that are harvested, but it includes also the fish that eat and are eaten by those fish, and plants, animals and microorganisms that are indirectly related through the food chain. The human dimension consists of the fishers, processors and consumers of fish, as well as wider economic, social, political and cultural linkages.

If we were to represent all the elements of any fishery system in a schematic, it would be hugely complex and daunting. It is obvious, however, that whilst management, whether voluntary or regulated, is achieved through the human dimension alone (by modifying how, when and where fishers are allowed to fish, directly or through incentives), the effects of management propagate through all dimensions of the system.

Given our limited understanding of the world, and limited ability to collect and interpret information, fishery management can only serve to modify limited aspects of fishers' behavior, and can react only to perceptions and inferences about limited elements of the fishery system.

2.2 Approaches to Fishery Management

A common approach to fishery management around the world has been to monitor and assess the status of individual, commercially exploited species (fish stocks) and to adjust the intended amount of catch, each year. The aim of this approach is to maintain each individual stock at a level that is safe for that stock (allowing it to continue to reproduce effectively in sufficient numbers), and which is biologically productive in an ongoing (sustainable) way [see BOX 1].

Coupled with adjustments to catch, it is common also for fishery management to restrict catches to certain times and places, or to use of particular gears. These additional restrictions are usually intended to ensure that not too many young fish are caught, that incidental damage to other species or habitats is controlled to an acceptable level, and perhaps to reduce conflict between fishers.

Increasingly, however, there is a move to consider not just individual fish stocks, but in addition to consider the wider consequences of fishing activities. Those consequences may be due to direct biological relationships between fish species (multi-species interactions) or other species such as birds and marine mammals (by-catch), relationships between fishing operations and stocks caught (technical interactions), or relationships between any or all of the many elements of the system (ecosystem effects).

Both the traditional single-species approach to fishery management and the more recent multi-species, technical interaction and ecosystem approaches, are faced with the problem of making decisions in the face of uncertainty. As noted above, the ability to monitor, understand and predict the behavior of the systems or particular elements is limited. There has been a growing understanding amongst scientists, the public and decision-makers that, despite best scientific efforts, the world is not totally reducible to simple equations. Rather, as concepts such as chaos have become more familiar through popular books and articles, the realization is that although we can strive constantly to improve understanding, many of the systems we seek to manage need to be treated as knowable only within quite wide bounds, and that management needs to take uncertainty (imperfect knowledge) into account.

2.3 <u>Uncertainty and Risk Management</u>

That we live in an uncertain world is well enough known. Despite scientific and technological advances, the recognition of practical uncertainty, and concomitant risk are quite familiar. Nightly weather forecasts dealing with the *chance* of rain reflect this reality. Even the most sophisticated physical and mathematical models [see BOX 2] of climate and weather, run using the most powerful supercomputers, are able only to make forecasts of the probability of rain; and the specificity of those forecasts decays markedly and quickly beyond one or a few days. So it is with fisheries systems. The forecasts for fisheries systems, or particular elements of them such as the state of a single fish stock, in principle include many more dimensions, and many more unknowns, than those that the weather models and forecasts are based on. Furthermore, the period for which fisheries forecasts are needed is not tomorrow or next week, but next year or the next decade. In reality, models of fisheries systems also have far less data available to them, and far less fundamental research on the various dimensions (including, notably, the human dimension).

How can fishery management deal with the large uncertainty [see BOX 3] and the resulting lack of predictability? One view is that uncertainty should result in *conservative* management decisions that attempt to implement “margins of safety” in the direction of reducing somewhat the amount of exploitation, especially when biological elements and systems are involved. A refinement of this view is that decisions need to be made in full recognition of risks (that is, the probability of something bad happening), but be well supported by careful analyses of the risks involved. This latter approach has been adopted, to some extent, by the majority of fishery management agencies where professional fishery managers attempt to take actions that are intended to control risks to an acceptable level (risk management), and where the estimates of the amount of risk involved are clarified (risk assessment) for them by scientists and others. Notwithstanding this adoption of such a technical, scientific decision framework, controversies and conflicts may arise (and persist) over differences about what risks are considered, how the effects of various possible outcomes are weighed, what level of risk is acceptable, and how the costs are born by various constituencies.

At the heart of any decision making problem faced by fisheries managers is the need to consider the chances of meeting different objectives using alternative, workable management options. Fishery management has to balance the many obvious and hidden objectives that relate to the dimensions of fisheries systems.

2.4 Objectives

That fishery management exists at all is because people want or need to fish. Fishing, whether motivated by commercial, recreational, or cultural considerations, is about utilization. Commercial fishing results in a major contribution to the world's supply of protein; it creates jobs, at sea and onshore, in the fishing and related industries. In some economies it is a major or even principal factor. Depending on why people fish, the culture of the fishing nation and community, national and international markets, and a variety of other factors, the particular objectives for utilization will vary. Objectives for utilization might include maximizing the volume of catch next year; maximizing the total export value of catch over ten years; or reducing the year-to-year variation in the supply of given species.

But utilization alone is not a sufficient objective to be considered in fishery management decision-making. Even in a single-species management regime, the ability to utilize in a sustainable fashion, without causing undue risks to the stock, is essential. Objectives relating to *sustainability,* therefore, need also to be considered. Objectives for sustainability might include maintaining a minimum stock size of adult fish; protecting juvenile fish in an area; or restricting the proportion of fish that can be caught each year.

Most fishery management regimes in the world try to balance objectives relating to utilization and sustainability on a single stock basis. A common approach to this is to monitor and assess the status of individual stocks and to provide decision makers with scientific advice on current stock status (usually by comparing indicators with agreed reference points; see BOX 4) and scientific advice predicting likely future status under different catch scenarios. In making a decision about intended future catches, decision makers usually also consider scientific information on other stocks, perhaps on multi-species or technical interactions, and occasionally on ecosystem considerations or by-catch issues. Other information on economic or social consequences is occasionally available, but often in a less formal manner.

Information on by-catch varies greatly from fishery to fishery. Estimates of by-catch of other fish species are often available, especially where observers are present. The common assumption of fisheries science is that, for non-target species with biological characteristics similar to the target species, the death rate due to fishing is usually less than that of the target stock. If this assumption holds true, managing the target stock in a sustainable manner, should not create any sustainability problem for the non-target stock. Where non-target species in one fishery are also commercially targeted by other fisheries, the non-target catches need to be taken into account in separate single- or perhaps multi-stock assessments. Where by-catch is of less productive or more vulnerable fish species, for example sharks or rays, comparability of biological characteristics no longer holds, and the effects of by-catch require a specific analysis.

By-catch of non-fish species such as turtles, dolphins, birds or bottom-dwelling creatures is usually considered as a separate issue. In many fisheries around the world, the issue of by-catch of such species is a major problem. For some species, by-catch can pose a conservation threat in the classic sense of reducing their sustainable utilization potential. For some species, however, the issue has less to do with this aspect of conservation, and is based instead on very different valuation systems that accord certain taxa, such as marine mammals, a special status, or that attach a seriousness to the risk of extinction that goes far beyond a concern over the lost harvest potential that an extinction might represent. As an economic activity in the natural environment, it is inevitable that fishing will impact ecosystem attributes and species beyond the commercially harvested fish. The degree to which this is sustainable for individual species can in principle be assessed and considered in the same way as target fish stock sustainability is addressed, if the valuation systems are commensurate. But the way in which some species are valued in some nations results in the sustainability objectives for by-catch species being very different from those for the target fish stocks.

Dealing with objectives relating to the ecosystem, rather than component species, is more difficult still. Little is known about the structure, function and dynamics of ecosystems. Indeed, it is often hard to agree on definitions of ecosystems. Nevertheless, there is a growing desire to adopt an "ecosystem approach" (see chapter 4) to fishery management that will in the coming years take a more solid shape. At present, there are few cases where the scientific knowledge is deep enough, and the policy mandate is explicit enough, that a true ecosystem approach might readily be adopted, but there are many instances where consideration of effects on non-target species are incorporated in the management decisions. Most notable amongst these "bottom up" approaches, that build on traditional management and deal pragmatically with specific policies, are attempts to control the level of by-catch of marine mammals and seabirds and indirect, food-chain-mediated effects on marine mammals.

2.5 The Role of Science

The by-catch issue, as much as any issue in fishery management, highlights the need for clear policy objectives in order for the role of science to be effective in supporting decision-making. Science can monitor, assess, forecast within bounds, and generally inform and support decision-making. But science has only a limited role in determining the objectives and their relative importance (importance is sometimes called ‘weight’ or ‘utility’). The best that science can do is to use models to calculate the expected amount of utility, so defined, that will result from a proposed management plan. If there is an agreed upon utility measure that can apply to all the various objectives, the science can also *optimize* the management plan by seeking plans that maximize utility, within the stated constraints. If the various objectives cannot be reduced to a common currency, the best that the science can do is to explore and report the various trade-offs between satisfying the different objectives that will result under different management actions. To pursue these kinds of optimizations or explorations, scientists have to formulate models of the systems and develop indicators [see BOX 4] to estimate how well different objectives might be met.

As the complexity of information requested from fisheries science increases, and as more and more complex models are utilized, predictability and certainty do not necessarily increase. Indeed, the more complex the models, the more they have to depend, in practice, on assumptions and presumptions rather than data. Science is quite good at giving robust advice based on simple models, but the robustness of the advice may deteriorate, as models get more complicated. The bottom line is that the more that the decision-makers want by way of scientific advice on complex issues, the more scope is created for scientists to influence decisions by inventing objectives and indicators, making assumptions, and inadvertently taking on a policy role rather than just attending to the technical components of risk assessment and risk management.

The traditional approach to fisheries science has been to assess the state of stocks on a single-species basis, using catch and biological data as input to the models, and then to forecast (in the same sense that weather forecasts are made, though farther into the future) what will happen if things (usually total catches) stay as they are or get changed somewhat. This leads to decisions being made based on expected outcomes. Some sense of the robustness of decisions can be made by running the forecasts with different assumptions or from different starting places, but this sort of exploration has traditionally been limited and ad hoc.

A more recent approach (see chapter 3) is instead to create models of the fishery system and to use computer simulations to test systematically what would happen if different management strategies (combinations of data collection, assessment and decisions following specified rules) were adopted. This is somewhat like building a wide range of test cars, with a range of safety features costing different amounts of money, and crashing them in every conceivable way to find out which safety features work best at preventing certain types of injury. The purpose of this type of work is to find car safety designs (equivalent to fishery management options) that perform better overall than others and that best meet the range of safety (sustainability) objectives balanced against the costs (other objectives). This sort of analysis, which is aimed at systematically revealing how different management approaches compare in meeting sets of objectives (but which does not necessarily forecast an expected outcome for any particular approach), in principle allows a better integration of risk assessment and risk management with clear roles for scientists and managers.

2.6 Fisheries Management in the USA

In the United States, following the Magnuson-Stevens Fishery Conservation and Management Act (MSFMCA,; last amended by the Sustainable Fisheries Act in 1996, 16 U.S.C. 1801 et seq.), most decisions on fishery management are made by the Regional Fishery Management Councils. The Councils have considerable autonomy but must prepare fishery management plans (FMPs), create regulations and generally make decisions that are consistent with the provisions of the Act and other pertinent legislation and regulation. The Act sets out a clear Purpose, and importantly provides fishery management objectives through ten National Standards for Fishery Conservation and Management. Those ten National Standards cover a range of biological, social and economic issues, as well as addressing uncertainty, by-catch and habitat protection. Operational guidance on the National Standards is provided by Guidelines (NSGs) established by the Secretary of Commerce. Those Guidelines are intended to help the Councils develop and implement FMPs.

Although there are ten National Standards, National Standard 1 (*Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry*) is undoubtedly the most influential in terms of most FMPs and management decisions. Considerable effort has been expended by the National Marine Fisheries Service (NMFS) to develop Guidelines on the definitions of overfishing for all stocks, and on "precautionary" approaches to implementing the Standard. National Standard 2 (*Conservation and management measures shall be based upon the best scientific information available*) gives a dominant role to science in ensuring that National Standard 1 is followed.

2.7 National Standard 1

National Standard 1 introduces two fundamental objectives for United States fishery management. The first objective derives from the language: *shall prevent overfishing.* The second objective derives from the language: *achieving, on a continuing basis, the optimum yield from each fisher.* The word "optimum" is later explained to include ecological and even ecosystem considerations, among others. The two objectives are linked by the word *while*; this introduces the requirement to do three things simultaneously. Essentially, National Standard 1 says that the objectives are to catch the greatest amount of fish possible but taking account of the need to ensure long-term viability of the stock and other ecological considerations.

National Standard 1 does not explicitly discuss Maximum Sustainable Yield (MSY), but rather it calls for achieving the optimum yield (OY) from each fishery. In defining OY, the Act says that it is the amount of fish that will provide greatest benefit to the nation with respect to food production and recreational opportunities, and with respect to ecosystem protection. For commercially harvested species with little or no recreational interest, and with no ecosystem implications, this equates simply to greatest food production - tonnage. In the definitions within the Act, however, MSY is prescribed as the basis for OY, but as reduced by social, economic or ecological factors. In the United States, therefore, under the Magnuson-Stevens Fishery Conservation and Management Act, biomass and fishing mortality reference points associated with MSY might be taken as target reference points if the only objective to be considered were maximizing yield. If other objectives (ecological, social, economic) that depend on the biomass of the target stock being greater than the biomass associated with MSY (B_{MSY}) are also to be considered when managing fisheries, fishing mortality target reference points should be less than the fishing mortality rate associated with (F_{MSY}) and biomass target reference points above B_{MSY}. Alternatively, if other objectives put a high premium on ensuring that the biomass of the target species not decline below B_{MSY} it would be possible still to define target reference points based on achieving MSY, but to manage such that the probability of overshooting the target yield (and falling short of the target remaining biomass) was appropriately low.

The other condition imposed by National Standard 1 is to prevent overfishing. In terms of the Act, this means not fishing in a manner that will prevent the stock from producing MSY on a continuing basis. In other words, this condition itself puts a premium on not fishing so hard that the stock size is reduced below B_{MSY}.

Taken together, the conditions of National Standard 1, for a stock without ecosystem considerations, lead to a management framework that should ideally define target reference points based on MSY and associated quantities (or proxies therefore) but which should also set limit reference points to control risk by requiring that management ensure with a high probability that these limits are not exceeded. The 1998 Guidelines for National Standard 1 (Optimum Yield) of the Magnuson-Stevens Fishery Conservation and Management Act, 50 CFR Part 600 try to do just that. Generally, target reference points are set sufficiently below the limit reference points to provide this assurance. Furthermore, the margin of safety embodied in the distance between the target reference point and the limit reference point can be increased in response to greater uncertainty about the status of the stock, its productive capacity, the precision of management control, or the constancy of the relevant environmental conditions. This shifting of the burden of proof to require, effectively, some evidence that the limit reference point will not be exceeded, and some evidence that satisfying the limit reference point is adequate to meet the biological objectives, constitute key components of the *Precautionary Approach*. The Precautionary Approach is not invoked or defined in the MSA, but is understood more broadly as according resource conservation a priority as a management objective—but this can mean different things to different constituencies. The MSA does state a requirement to manage so that "irreversible or long-term adverse effects on fisheries and the marine environment are avoided," which implies a precautionary spirit both with respect to the fishery resource and other ecological considerations. The Precautionary Approach is referred to explicitly in the 1998 Guidelines, where technical guidance is provided on the use of a precautionary approach to implementing National Standard 1 with respect to conservation of the fishery resources. There is not equivalent detail, in the Guidelines, on how to be "precautionary" with respect to other goals.

The National Standard Guidelines identify two types of limits. In fact, in the Guidelines, these limits are referred to as thresholds. The two thresholds referred to are a maximum fishing mortality threshold (MFMT) and a minimum stock size threshold (MSST). These thresholds are reference points, used to judge whether or not stocks are in an overfished state (below the MSST) or are being overfished (at a rate above the MFMT), and can be used in conjunction with control rules to prescribe catches. For example, a control rule designed to meet the objectives embodied in the MSFCMA at National Standard 1 might use F_{MSY} as the MFMT for stock sizes above a target biomass reference point of B_{MSY}, but could reduce this MFMT in response to stock size so that the fishing mortality rate would approach zero at the limit biomass reference point (the MSST, set say at half of B_{MSY}). This is just an example; the Guidelines leave open to Councils the exact specification of target and limit reference points, and of control rules. The important thing is that overfishing definitions are made and that prescribed actions are agreed upon.

2.8 Other Legislated and Regulatory Constraints

FMPs are federal actions, and must conform to the requirements of other environmental legislation and regulations besides the MSFCMA and the regulations which derive from it. The most consequential of these are the Endangered Species Act (ESA), the Marine Mammal Protection Act (MMPA) and the National Environmental Policy Act (NEPA). NEPA very broadly requires federal agencies to give "appropriate consideration" to environmental factors so as to prevent damage to the "environment and biosphere," and it specifically requires documentation of the process whereby this is taken into account in arriving at pertinent decisions. The ESA sets extremely stringent standards for protecting populations that are classified as "endangered." The protection applies to actions with the potential for direct and indirect effects, ranging from direct kill (as in by-catch), through disturbance, to "adverse modification" of habitat. MMPA sets extremely stringent standards for protecting marine mammal populations that are classified as "depleted." The protection applies to actions with the potential for direct effects ranging from incidental take to disturbance. Neither ESA nor MMPA invoke a Precautionary Approach by name, but the implementation and interpretation of both ESA and MMPA employ formally precautionary elements, often in decision theoretic language—more so than MSFCMA itself. The ESA legislation uses probabilistic language ("likelihood") and risk-related language ("jeopardy") and is interpreted as placing a high standard for the burden of proof that the protected population will not be harmed. In practice, critical ESA decisions are often based on probabilistic analysis, with uncertainty taken into account in explicit technical calculations. MMPA is interpreted as placing a burden of proof on showing that protection is not needed. Regulations for implementation of decisions about permitted incidental kill levels, called Potential Biological Removal (PBR), under the MMPA, define a formula that responds to uncertainty through use of a specified confidence limit. The development of that formula stated specific performance criteria that the formula was expected to meet, and these criteria are stated in terms of specified probabilities of outcomes.

Of course, both MMPA and ESA are almost wholly protective in their objectives, whereas MSA sets forth utilization objectives and protective objectives, with only limited guidance on how these are to be balanced if they should be in conflict.

2.8.1.1 BOX 1. Sustainable Fisheries and MSY

In the context of fisheries management, the question of whether or not a fishery is sustainable is the same as asking whether taking catches continuously at the current rate is possible without compromising the stock's ability to replace itself at around it's current stock size.

The average biomass (weight of the stock) at which a stock persists depends on the relationship between the spawning (breeding) stock biomass and the average production of new fish, reduced to take account of how well those recruits survive after they enter the fishery. The relationship between stock biomass and production is of major importance in fisheries management. If there was no stock then there could not be any production. At the other end of the spectrum, for some very high stock biomass, such as that in a pristine environment, there would also be zero or negative production because regardless of how many recruits were produced, there would not be enough food for them to all grow and survive through to an age at which they would be caught. Between the zero and high values of stock biomass where production is also zero, there are intermediate values of stock biomass at which production increases to a maximum and then decreases again.

The important thing to realize about this stock-production relationship is that in principle it is possible to have sustainable fishing at almost any level of stock biomass, so long as the catch that is taken balances the production. In principle, therefore, sustainable fishing could take place anywhere between very low or very high stock sizes. The ability to manage with confidence a stock to any given stock size would depend, however, on how well the stock size is known, how well the relationship between stock size and production is understood, how well catches can be controlled to match production, the dynamics of the stock's response to deviations from the intended level of catch, and a variety of other difficult and uncertain factors.

The level of stock size that produces the maximum possible production is the so-called *maximum sustainable yield* (MSY). In practice, because of economic and social objectives, as well as uncertainty, there are good reasons for trying to manage fish stocks near to, but somewhat below the stock size that confers MSY,

The MSY is the highest theoretical production (yield, or catch) that can be continuously taken from a stock under constant environmental conditions without affecting the production of new recruits. It is estimated from surplus production models [see BOX 2] and other methods. In practice, MSY, and the level of fishing effort needed to take it are difficult to assess [see BOXES 2 and 3]. Nevertheless, MSY is a benchmark in fisheries theory, international agreements and national legislation; as such, it is the basis for important reference points [see BOX 4] used in fishery management.

2.8.1.2 BOX 2. Models in Ecology and Fisheries

Models can take a large variety of forms, but in essence they all serve the same purpose–they allow thoughts, theories, and data (observations of the world) to be organized and simplified such that complicated issues can be cut through and clear logic applied. Theoretical models may be used to follow through to logical conclusions. Statistical models may be used to "fit" data and estimate parameter values (fixed numbers) to be used elsewhere. Simulation models may be used to combine theory, knowledge and data to consider what might be and to ask "what if?" questions. Models as used in ecology and fisheries are often highly complex, using state-of-the-art mathematics, statistics and computing approaches, but they always represent major simplifications of real systems.

In a *deterministic* model all processes are treated as completely predictable in principle. Therefore, if all parameters are known and fixed, a deterministic model run repeatedly from the same starting point will repeatedly result in the same sequence of outputs. In fact, this is only partially true–some deterministic models can behave chaotically (apparently randomly within bounds) for certain parameter inputs and can actually be used as "random number" generators. In a *stochastic* model, there is random variability in some of the parameters or processes. Running a stochastic model many times will, therefore, result in different outcomes. Stochastic models are in principle closer to reality, but only if the variability can be properly incorporated; this is very difficult and makes stochastic models difficult to set up and apply. A stochastic model may be fitted to data from the history of a population, but it will not predict a unique future for that population.

Stock assessment models used in fisheries are standard tools of fisheries science. Single species stock assessment models are used to consider the data collected from fisheries or research on fish stocks. Those data contain information on how fish age, grow and mature, how fish die and how fisheries select fish of different sizes or ages. The data, though, are never perfect and there are always many things that assessment scientists have to make assumptions about, often based on experience elsewhere. What the assessment models do, given data, assumptions and prior knowledge, is allow inferences to be made about the past and present state of stocks. This allows scientists to advise managers as to the status of stocks: whether or not stocks have been, or are currently, overfished, and whether or not overfishing is taking place. In addition to assessment models to determine stock status, it is common also to forecast the future state of stocks under different catch levels or rates. Forecasting involves updating the estimated current status using assumptions or models to determine how many new fish (recruits) there will be in the future.

Assessment models and forecasts may be deterministic or stochastic and they take many different forms. *Production models* represent the state of a stock by a single variable (stock biomass) and estimate production (yield, or catch) from its relationship with biomass. *Age (or size) structured models* represent the state of a stock by the number of fish in each age (or size) class. They differ from production models in that whilst a stock may have had the same biomass at different points in history, the yields produced would have been different because the stock would have been made up of differently aged (or sized) fish.

A deterministic model is in e*quilibrium* when all of the variables stay the same from year to year. This kind of constancy doesn't obtain in real the world; it is an attribute of a model. Although equilibrium results are hypothetical, they are nevertheless widely used to obtain reference points [see BOX 4] for fishery management. A production model would be in equilibrium once the catch equals the yield, because this will maintain the biomass at a constant value. An age (or size) structured model, however, would only be in equilibrium when the numbers of fish in each age (or size) group in both the stock and the catch is the same each year. The equivalent to equilibrium for a stochastic model is a *stationary distribution*, where the relevant variables exhibit a kind of consistent range of variation over time, though they are not constant. Analysis of stochastic models is more involved than analysis of deterministic models, and even the definition of appropriate indicators of good performance requires much more thought with stochastic models. Nevertheless, variability is a feature of the world, so there is merit to examining reference points from the perspective of stochastic models.

2.8.1.3 BOX 3. Uncertainty

The well-known fisheries scientist John Gulland described three successive phases of fisheries management: *unthinking optimism*, *naive belief in Science* and *confronting uncertainty*. With many notable fishery collapses in recent decades, the potential for unthinking optimism has long since passed. Naive belief in science is more apparent than real, but the most common approaches to fisheries management appear still to operate as though science can deliver accurate forecasts. There are many sources of uncertainty, however, that need to be taken account of and dealt with in providing risk assessments, and which managers need to be aware of when making risk-based management decisions.

Fisheries systems are complex and the relationships between their various elements are not simple; this would make fisheries systems hard or impossible to predict even if we had complete understanding of them. Over and above this problem, though, uncertainty prevents predictability on at least three counts: structure of systems, the way structures are modeled, and extrapolation. It is a truism that the structures of the complex fisheries systems we seek to manage are poorly understood. The way to model structures can only be achieved through careful analyses of data collected at appropriate scales. Historically it has been very difficult to obtain quantities of oceanographic and population data at the right scales for purposes of fisheries modeling.

There are four main sources of uncertainty in mathematical models of biological and other systems:

Process error is a consequence of the effects of underlying demographic (population) and environmental stochastic (random) variability on the dynamics of the system.

Observation (*measurement*) *error* is a consequence of the way in which observations are made of the system. This may be due to the chosen sampling strategy, or errors in data collection.

Estimation error is the inaccuracy and imprecision in the estimates of system parameters, which can result from all other sources of uncertainty and the statistical methods used to make inferences.

Model error all models are caricatures of reality, and thus fail to represent the system dynamics in full. This has two consequences. First, model mis-specification will contribute to estimation error when making inferences. Second, model misspecification will cause systematic errors in forecasting (sometimes referred to as *forecast error*).

For managed systems, *implementation error* is often regarded as an additional source of uncertainty–for example, failure to achieve an intended catch. However, it is essentially a combination of all other types of error within a management strategy (see Section 3).

2.8.1.4 BOX 4. Objectives, Indicators and Reference Points

Management decisions need to take account of how well different *objectives* are likely to be met by alternative policies and actions. In order, however, that decisions can be based on quantified risk analyses, the objectives, which can be somewhat abstract, need to be translated into quantifiable *indicators* and *reference points*. A concrete linkage between risk analyses and risk management decisions can be made if *performance measures* and *control rules* (sometimes called *decision rules*) are also defined.

Reference points may correspond to a situation considered as desirable. Such reference points are described as *Target Reference Points* (*TRP*s). Reference points corresponding to undesirable situations, and perhaps requiring immediate management action, are described as *Limit Reference Points* (*LRP*s), or sometimes as *Threshold Reference Points* (*ThRP*s)

For example, the objective that a fish stock should be maintained at a "safe level" can be represented in a risk analysis by an indicator of stock size derived from a stock assessment (e.g., spawning stock biomass, SSB) and a defined limit reference point (e.g., a fixed tonnage, or a percentage of SSB in the pristine environment) that is theoretically considered to be sufficient for continued reproduction. If the indicator were above the reference point, the stock would be considered at a safe level. Alternatively, for the objective that a stock should be rebuilt from a currently low level to an "optimal" one, the same indicator of stock size might be used, but now with a target reference point (e.g., the biomass associated with MSY).

The difference between an indicator and a reference point can be viewed as a performance measure of how effective management is or how much management is needed. It is possible to set up control rules that specify management actions depending on the value of one or more performance measures. For example, a control rule might specify a zero catch if stock biomass is estimated to be below a limit reference point, a catch of twenty per cent of estimated biomass for stock biomass between limit and target reference points, and a fixed catch of twenty per cent of the target reference point for all stock biomasses greater than the target reference point.

Control rules have been set up in some management regimes directly, whilst in other management regimes the approach has been to select a management strategy from among a defined set of alternatives based on an evaluation of how well the different strategies (a combination of data collection, assessment method and control rules) are likely actually to perform in meeting objectives given the many uncertainties that need to be faced in making inferences and providing advice.

In practice, many fishery management decisions in different parts of the world are based on stock assessments that provide results on sustainability indicators and reference points. Many of the reference points relate to theoretical considerations of MSY. Importantly, because indicators and reference points usually derive from stock assessment models, they primarily deal just with biological objectives (sustainability in particular) and are subject to all of the uncertainties and errors associated with modeling. In most places, the decisions taken are concerned also with less formal (not quantified) consideration of ecological, social, and economic objectives. In principle, these other objectives might also be considered using appropriate models and might be included in a multi-objective risk assessment.

MSY reference points

In 1982, the United Nations Convention on the Law of the Sea (UNCLOS) specified MSY [see BOX 1] as a target reference point for yield, and B_{MSY} (the biomass associated with MSY) as a target reference point for stock biomass. Associated with those target reference points (MSY and B_{MSY}) is a target reference point for the fishing mortality, F_{MSY}. (Fishing mortality, F, and natural mortality, M, are used in mathematical models; they relate to the proportion of the stock that is killed each year due to fishing or which dies through natural causes.) Although UNCLOS does not define MSY with reference to a deterministic production model, this is how it is sometimes interpreted. However, stochastic variability and various sources of uncertainty make it difficult to obtain actual estimates of MSY, $B_{MSY,}$ and F_{MSY} or estimates of the stock and fishery status in relation to these quantities. If B_{MSY} and F_{MSY} are estimated ignoring stochastic variability, and if estimates of current biomass and fishing mortality are in error, it is possible that the value of MSY so estimated would not actually be sustainable. For this reason, other more conservative quantities have been suggested as target reference points.

The United Nations Agreement on Straddling Fish Stocks and Highly Migratory Fish Stocks (UNFA 1995) states that two types of precautionary reference points should be used: conservation (that is, limit) reference points and management (that is, target) reference points. The Agreement states that the risk of exceeding limit reference points should be low, and that target reference points should not be exceeded on average. The agreement further states that F_{MSY} should be used as a minimum standard for a fishing mortality limit reference point.

Sometimes, indeed often, the data available are not sufficient for fitting models or for estimating the inputs to models. For those occasions, and for more general use, alternative quantities have been suggested as target and limit reference points. These include MSY and F_{MSY} reduced by set proportions, or thresholds for stock biomass of twenty per cent or more of the pristine biomass. These are all ad hoc values intended to protect stocks from recruitment failure. Natural mortality, M, has also been suggested as a conservative target reference point for fishing mortality.

Yield per recruit and Spawning biomass per recruit

One of the most widely used models in fisheries is *yield per recruit* analysis. Yield per recruit analysis recognizes that the biggest unknown and variable factor is the number of new recruits produced each year. Yield per recruit analyses is deliberately simple. It is deterministic and uses only information on average growth, natural mortality and fishing mortality by age estimated using age structured assessment models. An important reference point based on yield per recruit analysis is F_{MAX}. This is the fishing mortality corresponding to the maximum yield per recruit that can be obtained. If recruitment were constant from year to year, and not related to spawning stock biomass (SSB), then the yield per recruit curve would convert to a surplus production curve by multiplying the yield per recruit by the mean recruitment. F_{max} and F_{MSY} would then be equivalent. If, however, recruitment depended on stock biomass, the yield per recruit curve would convert to a surplus production curve by multiplying by the recruitment corresponding to each biomass level from the stock recruitment relationship. This would change the shape of the curve so that F_{MSY} is usually less than F_{max}.

Spawning biomass per recruit (SPR) analysis builds upon yield per recruit analysis. SPR declines as fishing mortality increases. Therefore, in order to maintain constant recruitment as fishing mortality increases, it is necessary for an increasing proportion of eggs, larvae and juvenile fish to survive to the age of recruitment. SPR results are given as the percentage of the SPR in the unfished state, obtained at different fishing mortalities. For example, 100%*SPR* is obtained when fishing mortality is zero (unfished). At a higher fishing mortality, a lower percentage of SPR would be expected, such as 35%SPR. The term $B_{X\%}$ is defined as the stock biomass at X%*SPR*. Target and limit reference points of $B_{20\%}$, $B_{35\%}$ and $B_{40\%}$ have been proposed based on theoretical modeling work in which different assumptions have been made about the biology of the stocks that are so managed.

2.9 NPFMC Harvest Strategy: BSAI and GOA Groundfish Fisheries

The management system for the NPFMC groundfish fisheries is a complex suite of measures comprised of harvest controls—e.g., OY, Allowable Biological Catch (ABC), Total Allowable Catch (TAC), OFL—effort controls (ITQs, licenses, cooperatives), time and/or area closures (also known as habitat protection, marine reserves), by-catch controls (PSC limits, retention and utilization requirements), monitoring and enforcement (observer program, social and economic protections, and rules responding to other constraints (e.g., regulations to protect Steller sea lions and to avoid seabirds). While this review focuses on harvest controls, the efficacy of the management system must necessarily depend on the interaction of all components. Therefore, our answers to the questions posed in the Terms of Reference require broader consideration of the management system as a whole.

In this section, we describe the elements of the harvest control rules and TAC-setting process currently used by NPFMC. We then provide a historical perspective of how this system evolved to show a trend of increasing conservatism, which also has developed around the world in other fisheries management systems. Finally, we give an example to illustrate the theoretical underpinnings of the rules.

2.10 Harvest Control

Harvest control (catch limits, quotas, Total Allowable Catches (TAC)) is one of the primary management measures with proven capability for preventing overfishing of fishery resources. The NPFMC harvest control system is complex and multi-faceted in order to address issues related to sustainability, legislative mandates, and quality of information.

2.10.1 Optimum Yield

The first element is the specification of Optimum Yields (OY) for the groundfish complexes in the Bering Sea / Aleutian Islands (BSAI) and the Gulf of Alaska (GOA) as a range of numbers. The sum of the TACs of all groundfish species (except Pacific halibut) is required to fall within the range. The range for BSAI is 1.4 to 2.0 million mt; the range for GOA is 116 to 800 thousand mt (see Historical Background). In practice, only the upper OY limit in the BSAI has been a factor in altering harvests. Because of high productivity, Acceptable Biological Catches (ABCs) in the BSAI have summed to well above 2.0 million metric tons for several years. Some people believe this OY limit has been the main reason that the fisheries in the BSAI have held up so well. The lower limits in both the BSAI and the GOA have never been approached in recent time, so they have not received recent attention.

2.10.2 *The Tier System*

The second element is the specification of maximum permissible ABCs and of OFLs for each stock in the complex (usually individual species but sometimes species groups). NPFMC inaugurated the Tier system in fisheries management: the harvest control rule depends on the amount of information available [BOX 14, Tier definitions]. In Tier 1, information is abundant enough and compelling enough to determine the statistical distribution of maximum sustainable yield. In this Tier is only one stock: BSAI walleye pollock. Most of the larger and commercially important stocks are in Tier 3, which has sufficient information to determine $F_{40\%}$ and its corresponding biomass $B_{40\%}$. For these stocks, the spawner-recruit relationship is uncertain, so that MSY cannot be estimated with confidence. Hence, a surrogate based on $F_{40\%}$ is used, following findings in the scientific literature in the 1990s. A large number of the remaining stocks (generally of lower magnitude) are in Tier 5, in which natural mortality is the basis of the maximum permissible ABC. A few are in Tier 6, in which biomass and reference points cannot be determined, so that the rule is a function of average catch. As described in Historical Background below, these rules have become more conservative over time.

2.10.3 *ABC, OFL, and TAC*

ABC is a scientifically acceptable level of harvest based on the biological characteristics of the stock and its current biomass level. OFL is a limiting catch level, higher than ABC, which demarcates the boundary beyond which the fishery is no longer viewed as sustainable. The TAC is an adjustment downward from ABC that takes into account social and economic factors and the OY range.

In practice, NMFS attempts to manage a fishery so that total catch (including all discards) is less than, but very close to, TAC. Ideally, the directed fisheries are closed well before TAC is reached, so that when by-catch needs for that stock in other fisheries are factored in, the annual total catch is less than but very close to TAC. When a directed fishery is closed, by-catch of that stock is limited by a Maximum Retainable By-catch amount (MRB), which is determined as a percentage of retained catch (not including arrowtooth flounder). If it appears that the TAC may be exceeded due to unanticipated circumstances, and ABC is being approached, NMFS managers will prohibit retention of that species by all fisheries, in order to eliminate any 'top off' activity for by-catch of valuable species. If ABC is exceeded, and OFL is being approached, NMFS can prohibit or close any fisheries that might possibly take that species as by-catch.

2.10.4 *Form of the Harvest Control Rule*

In Tiers 1–3, sufficient information is available to determine a target biomass level, which would be obtained at equilibrium when fishing according to the control rule with recruitment at the average historical level. The control rule is a biomass-based rule, for which fishing mortality is constant when biomass is above the target and declines linearly down to a threshold value when biomass drops below the target. Fishing mortality is 0 below the threshold, which is currently set to 0.05 of the target biomass. In Tiers 4 and 5, a Biological Reference Point (BRP) cannot be determined, so fishing occurs at a constant fishing mortality, which is chosen to be conservative according to findings in the scientific literature. In Tier 6, such a fishing mortality cannot be determined, so catch is constrained to be 75% of the average historical catch.

2.10.5 Stock Assessment and Harvest Strategy

Each year, scientists from the AFSC and ADF&G collect data, and compile and update databases on catch, age and size composition, and survey biomass. Stock assessment scientists from these agencies analyze the data and calculate estimates of key population parameters. In most cases, contemporary stock assessment models are constructed to integrate the scientific information, except when information is not sufficient for model construction. The techniques of stock assessment are beyond the scope of this review but are adequately summarized in the texts by Hilborn and Walters (1992), Quinn and Deriso (1999), and Haddon (2001). An overview of issues related to stock assessment points out the difficulties and challenges (National Research Council 1998). The processes of stock assessment and harvest strategy development are interrelated. Stock assessment parameters are used in development of the harvest strategy, and the current NPFMC biomass-based harvest strategy utilizes the most recent biomass estimates in determining ABC, OFL, TAC, and whether overfishing is occurring. Nevertheless, the goal of harvest strategy development is to provide a stable, quantitative set of control rules for operating the fisheries, and the goal of stock assessment is to use the best available scientific information to determine the status of the population in reference to the quantities that are inputs to the rules.

2.10.6 Process and Peer Review

An annual process determines the values of ABC, OFL, and TAC. Stock assessment scientists make recommendations about ABC and OFL in their Stock Assessment and Fishery Evaluation (SAFE) documents. A group of scientists also constructs an Ecosystem Considerations chapter. The BSAI and GOA Plan Teams meet in September and November to review the SAFE documents. The two teams meet jointly to discuss common issues and separately about the individual assessments. The major goal of the September meeting is to discuss general methodological issues and applications. The Statistical and Scientific Committee (SSC), Advisory Panel (AP) of industry representatives and interest groups, and the Council meet in October. The SSC discusses methodology, while the AP and Council set preliminary TACs based on the previous year. In December, the SSC reviews the SAFEs and PT recommendations and issues its recommendations about ABC and OFL. The AP recommends values of TAC that are lower than the ABC values of the SSC. The Council then sets final values of ABC, OFL, and TAC. The Council could recommend higher ABCs than those of the SSC, but in the 20 years of TAC-setting the NPFMC has generally chosen not to do so (with a couple of exceptions).

2.10.7 *Bringing in Ecosystem Considerations*

The Ecosystem Consideration chapter in the respective SAFE documents is evolving to be more operational, and other multispecies studies have been undertaken. Ecosystem indicators are being constructed, and multispecies models have been constructed as part of the November 2000 SEIS process, mainly by researchers at AFSC. The multispecies models have allowed consideration of ecosystem impacts in a way that single-species models cannot address. They are not viewed as a replacement of the single-species approach, which remains the determinant of catch control, but rather they add insight into potential ecosystem effects. To date, the multispecies modeling studies reported in the SAFE and SEIS documents have suggested that fishery impacts on *fish* species in the BSAI GOA system seem to be about the same order of magnitude as what is shown in single-species models. [See BOX 5 for further description of the approaches and their results.] Nevertheless, it is known from theoretical models of harvest dynamics in a predator-prey-competition system that harvesting at single-species MSY levels will not achieve MSY for the aggregate because of species interactions. This knowledge is one of the reasons that the BSAI OY cap was set at 85% of the single-species MSYs (see Historical Background).

This same review of ecosystem effects of the BSAI/GOA FMP came to a rather different conclusion about the potential effect of the fishery on the western stock of Steller sea lions, which is listed as "endangered" under ESA. This analysis, in a determination called a "biological opinion," concluded that the FMP, as it then was drafted, posed "jeopardy" to the sea lion population, through the possibility that activities of the fishery in reducing the abundance of the target species, which are also prey items for the sea lions, might adversely affect the sea lion population's prospects for recovery. Much of the focus of this analysis was on the potential for local and temporary effects, called "local depletion," that might arise because of the uneven distribution of fish and fishing activities in space and time. The analysis did not focus on the overall effect of the FMP in reducing the total biomass of the target stocks, which reduction occurs by design in an attempt to achieve OY. The resolution of the conflict with ESA, in the biological opinion, was the modification of the FMP to conform to a "reasonable and prudent alternative" which redistributed some of the fishing effort in space and time, to reduce the expected intensity of local depletions in areas that were thought to be important for Steller sea lion foraging.

2.10.7.1 BOX 5. Recent Multi-Species Modeling by the Alaska Fisheries Science Center

Excerpt from pages 246-247 of the Endangered Species Act–Section 7 Consultation: Biological Opinion and Incidental Take Statement, November 30, 2000, National Marine Fisheries Service, Alaska Region.

> "Since the 1960s, commercial exploitation of groundfish in the action area has significantly reduced populations of some target species and species caught as by-catch. Over time, but prior to the present fishery management regime, prior to the NPFMC and prior to the current FMPs which are being considered in this biological opinion, the fisheries have depleted or overfished yellowfin sole, Pacific Ocean perch, sablefish, walleye pollock, and Pacific halibut. …

Evaluation of the present fishery management regime in the last 20 years does not show such dramatic reductions of individual populations that occurred previously. Most of the work evaluating predator/prey relationships in the EBS/AI and GOA regions in recent years has been done in the eastern Bering Sea. Evidence from retrospective and modeling studies (Hollowed et al. 1998, Livingston and Jurado-Molina, 2000) and examination of trophic guild changes (Anderson and Piatt, 1999; Livingston et al, 1999) suggest that under the present groundfish fishery management regime, there has not been clear evidence of fishing as the cause of species fluctuations through food web effects. Multispecies models have shown that although cannibalism can explain a large part of the density dependent part of the stock recruitment relationship for pollock (that is the decline in recruitment observed at high spawner biomasses), that most of the overall variability in stock and recruitment for pollock is not explained by predation but appears to be more linked to climate events (Livingston and Methot 1998).

Pollock is a key prey species of many target and nontarget species in the Bering Sea and Gulf of Alaska (Livingston 1989, 1994) and has a central position in the food webs of those ecosystems. Modeling of predation on pollock in the eastern Bering Sea and Gulf of Alaska (Livingston and Methot 1998, Livingston and Jurado-Molina 2000, and Hollowed *et al.* 2000) shows that different predators may be the most important source of predation mortality during different time periods. For example, Steller sea lion predation on pollock in the Gulf of Alaska was more important in earlier years but the most important contemporary source of predation mortality on pollock is now from arrowtooth flounder. Population levels of some of these predators such as arrowtooth flounder appear unrelated to fishing removals but are more linked to environmental forces that favor the production of these species (Hollowed et al. 1998). Similarly, the fluctuations observed in species composition of trophic guilds (Livingston et al. 1999) do not appear to be related to fishing removals of competitors or prey, when analyzed at the aggregated level for the whole eastern Bering Sea. Measures of pelagic forage abundance under current fishing practices indicate in the short term that from 2001 to 2005, that the fraction of pollock in the total groundfish biomass is predicted to increase 6% in the BSAI and 29% in the GOA, in the short term. Pollock biomass is predicted to increase 12% and 47%, respectively in these areas. Stability of trophic level of the groundfish biomass and trophic level of the groundfish catch also indicate there has not been a large change due to fishing in the groundfish community structure. These have been relatively steady over the last 20 years and do not indicate successive depletion of populations or fishing down the food web effects observed in more heavily fished ecosystems of the world. This assessment is supported by the stock trajectories shown in Figure 6.16. The stock trajectory in both fished and unfished scenarios indicate similar trends. Some species have shown strong increases even when fished and declining fished stocks also declined when no fishing was assumed, although the absolute biomass level was different."

2.11 **Historical Overview**

2.11.1 Before 1976

The groundfish resources of the BSAI and GOA have long been a component of the subsistence requirements of natives in the area. Otherwise, these resources were of limited interest or use to other humans until the 1950s (except for Pacific halibut and Pacific cod).

Records of Pacific halibut landings date back to 1888, increased to over 50 million pounds by 1910, and peaked at 69 million pounds in 1915 (Thompson and Bell 1934).

> The history of the fishery… shows that the industry has been maintained only by expansion to new banks. (p.18) But we know that in 1911 the fishery spread out of sheltered waters onto deeper banks, and in 1913 the exceedingly heavy winter fishery in the Gulf of Alaska (Area 3) had begun… After the temporary recession of war times, the use of Diesel engines by the fleet enabled them to exploit even the banks along the Aleutians, greatly increasing since 1921 the intensity of the western fishery. This great expansion in area since 1911 has meant the origin of a full half of the yield from a new and separate district. (p.17)… The level of abundance is in each case an economic one. Hence the older banks, nearer the landing ports, have declined to the greater extent despite their originally far larger population, and must be regulated accordingly. (p.18)

Fishing in the Bering Sea was either limited or non-existent until after World War II. After regulatory measures were put into place by the International Fisheries Commission (later International Pacific Halibut Commission) and with favorable environmental conditions, the population and catch continued to rise until the 1960s with a peak catch of 75 million pounds in 1962. Poorer environmental conditions and incidental catches by the foreign trawl fisheries reduced the population until the mid-1970s. The population rebounded thereafter, with help from a conservative directed catch policy and restrictions on by-catch of halibut in foreign and domestic fisheries. The current population is in good condition.

Pacific cod were fished by schooners with dories beginning in 1864 and further developing in the 1880's. Vessels hailed from San Francisco and other ports. The cod were caught on longlines (using halibut for bait), fished from dories, and salted. In the early 1890's they were dried onshore on the Alaska Peninsula, in places like Sand Point. The early cod fishery reached its peak during WWI when demand was high for cod liver oil. A range of 13 to 24 schooners (presumably per year) fished during the period 1915-1920, with annual catches of 12,000-14,000 mt. The number of vessels in the fishery declined after 1920 and the fishery was terminated by 1950.

According to the NRC report *The Bering Sea Ecosystem* (National Research Council 1996), Japanese and Russian trawlers discovered and exploited groundfish resources in the 1950s and 1960s, particularly walleye pollock, Pacific cod, yellowfin sole, and Pacific Ocean perch [see BOX 6]. These catches had their largest impact on Pacific Ocean perch: the Bering Sea and Aleutians stocks have never recovered, and the GOA stock only recovered in the late 1990s.

2.11.1.1 BOX 6. Paragraph from the Bering Sea Ecosystem (NRC 1996), p.159

Exploitation of groundfish resources in the eastern Bering Sea and along the Aleutain Islands changed dramatically in 1959… Following several years of prospecting, foreign fleets from Japan and the Soviet Union (USSR) began harvesting yellowfin sole (on the eastern Bering Sea shelf) and other flatfish, and Pacific Ocean perch (in the Aleutian Islands, and Gulf of Alaska) and other slope rockfish. Harvest of eastern Bering Sea shelf flatfish peaked in 1961 at nearly 700,000 t, and harvest of the Aleutian Island and Gulf of Alaska slope rockfish complex peaked in 1965 at over 450,000 t… Between 1959 and 1964, over 1.8 million t of yellowfin sole was taken from the eastern Bering Sea shelf; between 1962 and 1968, almost 1.7 million t of slope rockfish (primarily Pacific Ocean perch) was taken from the eastern Bering Sea, Aleutian Islands, and Gulf of Alaska (as far south as southeastern Alaska). In the mid 1960s, some non-U.S. fishers switched their target species to pollock and Pacific cod, and catches quickly grew to more than 2 million tons (primarily eastern Bering Sea pollock) per year by 1971. Catch by non-U.S. fishermen peaked at approximately 2.25 million tons in 1972 before declining again… By that time, other nations and additional vessels were joining the fleet in increasing numbers.

Walleye pollock do not seem to have been adversely impacted. It is unclear from the historical record whether the pollock population was low in the 1960s and increased (possibly to fill a niche vacated by the removals of large cetaceans and pinnipeds), or was a large pristine population that declined during the foreign fishery. Japanese catch per unit effort (CPUE) data suggest an increase, but age composition, inferred from length frequencies, suggest a decline.

Management of the foreign fisheries was non-existent during this period. The National Marine Fisheries Service had limited resources and personnel at the time, and there was no law in place to restrict fisheries beyond 12 miles. NMFS did place observers on foreign vessels, which resulted in length frequency data that is used to stock assessments to this day.

2.11.2 The Magnuson Act of 1976

Nevertheless, increasing concern about foreign removals, here and other places in the world, led to increased interest, through the Law of the Sea proceedings, in extended jurisdiction of resources out to 200 miles. The United States essentially "discovered" its fishery resources as a matter for national policy in the mid-1970s. The Magnuson (later Magnuson-Stevens) Fishery Conservation and Management Act of 1976 revolutionized marine fishery management in the United States. With the extension of jurisdiction out to 200 miles, the Act required that Fishery Management Plans be developed. The key objective in mind at that time was to replace foreign fisheries with domestic fisheries, and Congress provided funds for developing a new domestic fleet (which later led to problems of overcapitalization). Rather than put into place the traditional agency management scheme (centralized management with rules generated by the agency), the Act set up eight regional councils, made up of industry representatives and agency personnel, to develop these plans. The Councils were modeled after the International Pacific Halibut Commission and Inter-American Tropical Tuna Commission management systems in its Council and Advisory Panel structure. However, the regional Science Centers of NMFS were to provide the scientific support for management, with assistance from state agencies, and the Act set up Scientific and Statistical Committees to provide for scientific advice and peer review.

The North Pacific Fishery Management Council set to work rapidly: Preliminary Management Plans were in place in 1977 in both the BSAI and GOA. The Council implemented the formal Fishery Management Plan (FMP) for GOA groundfish on December 1, 1978 and the first FMP for BSAI groundfish on January 1, 1982. A change to these Plans required a formal Plan Amendment.

At the same time, the Alaska Fishery Science Center (AFSC) increased its research and monitoring of the groundfish resources and fisheries, in particular developing and expanding a comprehensive triennial groundfish trawl survey based on previous surveys for crab in the Bering Sea. Also, requirements for 100% observer coverage were implemented, and biological collections (otoliths, maturity, length) were undertaken.

The original harvest control rules specified OYs for each species and stock complex in the two major areas. These OYs were derived from MSYs provided by scientists from the AFSC and reviewed by the SSC. Essentially the strategy was a constant catch strategy that would be changed in light of new information. Because data was limited and stock assessment models were at their infancy, the stock assessments were fairly simple and often based mainly on survey data. In BSAI, the early assessments came from data from a period of time that we now believe was a low productivity period. Consequently, the sum of the OYs in the BSAI from this period was below 2.0 mmt, a number that would later in time become a significant upper limit for TACs.

2.11.3 <u>OY Limits</u>

The Council soon learned that the bureaucracy of the federal government could not respond to annual changes in these OYs, because the paperwork and notification requirements took months to construct amendments that made it into federal rules. Yet the assessed status of the population could change dramatically due to new survey and catch data, recruitment variability, and other changes in population parameters. The Plans were amended several times to make these changes, but a better solution had to be found.

A solution was provided in developing a broad OY range for each of the two groundfish complexes, as the definition of OY in the FMPs. The total allowable catch of all groundfish species under NPFMC management would have to be within the range, but the TACs could be adjusted annually through a specifications process (i.e., there were no OYs for individual species; therefore, it was not necessary to file a plan amendment). This framework approach also clarified the biological versus social and economic components of the TACs. Acceptable biological catches (ABCs) were first defined from the biological information, and in particular, MSYs. Adjustments downward or upward could then be made for social and/or economic reasons.

The Council set the OY range for BSAI of 1.4–2.0 mmt in Amendment 1 on January 1, 1984 (3 other amendments 1a, 2, and 4 were actually approved earlier). This range was 85% of the summed MSYs of the species in the BSAI. According to AFSC scientist Loh Lee Low, the range was chosen to provide for a stable series of sustainable catches. Multispecies modeling by

Laevastu and Larkins of AFSC suggested that higher catches might actually be sustainable. But at the same time, it was noted that the MSY of the complex should be less than the sum of the MSYs of individual species because of negative species interactions (predation, competition). It was also noted that the catches above 2 mmt in the 1970s seemed to be having a deleterious effect. So to be conservative, the formula 85% of MSY was chosen.

The Council set the OY range for GOA of 116,000 – 800,000 mt in Amendment 15 on April 8, 1987. This upper end of the range was lower than the average of the summed MSYs of the species over the period 1983–1987 (873,070). The lower end of the range is near the lowest historical catch over the period 1965–1985 (116,053 in 1971, using the information from that time). In 1971, pollock, cod, and Atka mackerel abundances were at low levels and consequently, it was thought unlikely that catches lower than this value would occur even in times of low abundance. The upper limit was selected in consideration of the volatility in pollock and flounder ABCs, the potential for harvesting at MSY, and the desire to allow for some moderate expansion in the future flounder fisheries.

2.11.4 Quantifying Harvest Strategy in the 1980s

With the framework in place for annual TAC-setting, the Plan Teams and SSC moved away from the previous constant catch strategy to a constant fishing mortality (or harvest rate) strategy. The constant fishing mortality strategy adjusted the catch relative to the stock biomass in a (mostly) linear fashion, so that if the stock went up, so did the catch, and vice versa. The default strategy was to set fishing mortality at the level F_{MSY} that corresponds to a catch of MSY at the equilibrium biomass level producing MSY, B_{MSY}. Scientists became more skeptical of the MSY values, so alternatives or proxies became popular. One such alternative was $F_{0.1}$, defined as the fishing mortality for which the marginal change (or slope) in yield per recruit has dropped to 10% of what it was at the origin (at F=0). The use of yield per recruit took away the problem caused by ambiguous spawner-recruit relationships, and the use of a 10% marginal change provided a less aggressive fishing policy than the then-commonly-used F_{max}, which maximized yield per recruit (and thus resulted in the higher fishing mortality necessary to drop the marginal change all the way to 0).

In the late 1980s, the Plan Teams and SSC desired to have more conformity among stock assessments, and the SSC set upon the task of standardizing definitions of ABC, MSY, overfishing, and threshold [BOX 7]. The SSCs of the Pacific and North Pacific Councils met jointly and were able to come up with common definitions, as implemented in Amendments GOA 16 / BSAI 11 (January 4, 1988). These definitions interconnect ABC, MSY, threshold, and overfishing.

The headquarters of NMFS became interested at this time in developing National Standards for all the Councils to follow, with the hope of reining in overfishing. The SSCs of the North Pacific, Pacific, and Western Pacific Councils held a joint meeting with the NMFS director to attempt to standardize these definitions. The Western Pacific did not use ABCs, so the National Standards had to allow for overfishing definitions that did not rely on ABC. At one point, adjustments for risk and uncertainty, included in the NPFMC definition of ABC, were removed and made part of the OY/TAC adjustment. The National Standards that came out in 1989 ignored the issue of risk and uncertainty altogether. The definition of ABC replaced F_{MSY} with natural mortality M, and the definition of overfishing was not linked to the threshold [BOX 8]. The main focus of the National Standard for overfishing was to require an operational definition of overfishing, so that NMFS could determine if Councils were overfishing and what they were doing about it.

The NPFMC considered a variety of alternatives, including constant fishing mortality, biomass-based fishing mortality policies (with reduced F at lower biomass) and fishing mortality/threshold policies. Of the seven or so alternatives, the Plan Teams split between three alternatives, and the SSC opted for a constant F definition. The Council ended up choosing an alternative, in Amendments GOA 21 / BSAI 16 (November 9, 1990), which is essentially the biomass-based strategy now in effect (but with some different values). The corresponding definitions of ABC, overfishing, threshold, and MSY are given in BOX 9.

During the late 1980s, the groundfish fisheries in the North Pacific had nearly completed their transition from foreign to joint venture (American vessels selling to foreign processors) to fully domestic fisheries. Observers were placed on foreign and joint-venture operations, but it was thought to be too demanding for the fledgling domestic fleet. The SSC was alarmed that there would essentially be no information on the catches, once domestication was complete. Therefore, it filed a plan amendment to ban fishing on any stock without an approved observer program. The Council took serious notice of this amendment, Alaska Sea Grant conducted a pilot observer program to demonstrate its feasibility, and the Council passed an observer program in Amendments GOA 18 and BSAI 13 on November 1, 1989. The plan was industry funded, wherein all vessels above 125 feet had to carry observers, and vessels from 60 to 125 feet had to have observers 30% of the time. While there are statistical, sampling, and logistic issues with this program, it is one of the best observer programs in the world, and provides for total catch accounting, including by-catch.

It was clear in the 1980s that the upper limit in the BSAI of 2 million mt was constraining catches. The 1980s were a period of increased productivity (now understood in terms of changing oceanic regimes), which led to large increases in flatfishes in particular. Strong year-classes of pollock in 1978, 1982, and 1984 also increased biomass in BSAI.

Between 1984 and 1990, there were 6 proposals to raise the cap; 3 of these were analyzed, and all were rejected. Of interest is that the plan teams, SSC, and AP all said that the cap could be increased on biological grounds. In 1988, the SSC noted that the new OY range based on 1984 data would be between 2.2 and 2.9 million mt. The proposals to raise the OY limit were based on new calculations of MSY or by using the sum of the ABCs.

In 1988, the Plan Amendment to raise the cap was considered serious enough environmentally and economically that it required a full SEIS, rather than an EA. The Council voted to not change the upper limit, both for conservation reasons and because at least some of the benefits would go to the foreign fishery at the expense of the domestic fishery.

In 1991, the General Accounting Office of the United States, a research arm of the Congress, investigated aspects of the management of the North Pacific groundfish fisheries by the North

Pacific Fishery Management Council and Department of Commerce. The OY cap was one of the issues investigated. From the Executive Summary:

> Purpose: Representative Les AuCoin asked GAO to examine whether the annual fishing cap of 2 million metric tons in the Bering Sea is based on the best available scientific information and on sound principles of fisheries management. (p.2)
>
> Results in Brief: Recent estimates of fish stocks suggest that the 2-million metric ton cap for groundfish in the Bering Sea could be increased. The Council acknowledges the improved accuracy of the biological data, but has consistently decided not to increase the cap in order to (1) Americanize the fishery, (2) protect markets for groundfish, and (3) sustain the ecological balance. In view of the Magnuson Act's multiple objectives and the issues involved in achieving them, the Council has decided to maintain a conservative cap. (p.2-3)
>
> Principal Findings: Views differ on appropriateness of the Bering Sea fishing cap. When the Bering Sea fishing cap was implemented in 1984, the biological information available for estimating existing fish stocks was limited and incomplete. Because of these data limitations, the Council set a conservative groundfish cap of 2 million metric tons. However, by 1987 new information, based on more current, detailed, and accurate data, showed larger stocks of available fish than NMFS had estimated in 1984. Studies indicate that 3 million metric tons of groundfish could have been harvested in 1990. On the basis of these estimates, NMFS biologists concluded that the cap could be increased.
>
> The Council has rejected proposed increases in the cap each year since 1984. Factors other than the amount of available fish are considered in setting the cap. The Magnuson Act requires the Council to balance several sometimes competing objectives–such as preventing overfishing, achieving optimum yield, and Americanizing the fishery–when making decisions about the fishery. (p. 3-4)

2.12 <u>Refining the Biomass-Based Strategy: The 1990s and Beyond</u>

Since July 1990, when the Council approved the overfishing definition in Amendment 21/16, the biomass-based strategy has undergone near-continual, if not unidirectional, evolution, punctuated by a series of difficult-to-predict legislative and policy changes at the national level. A chronology of this evolution follows.

- *September 1990:* The Plan Teams adopted a policy on ABC, the central features of which were as follow: 1) an endorsement of the ABC definition contained in the *602 Guidelines*; 2) an encouragement of assessment scientists to explore new methods of addressing uncertainty, recruitment variability, and multispecies considerations; 3) a constraint that ABC recommendations not exceed OFL; and 4) a clarification that the need for a buffer between recommended ABC and OFL would be determined on a case-by-case basis, depending on factors such as recruitment trends, multispecies interactions, and the degree of uncertainty in data or parameter estimates.

- *January 1992:* The SSC identified a set of concerns arising from application of the overfishing definition established by Amendment 21/16: 1) F_{ABC} should be reduced when biomass is below B_{MSY}, 2) more caution should be required when less information is available, 3) F_{OFL} should exceed F_{ABC} , and 4) OFL should remain constant over time when catch history is the only information available.

- *July 1992:* The SSC submitted a proposed FMP amendment intended to redefine ABC and overfishing, the objectives of which were as follows: 1) to define F_{OFL} as the level of fishing mortality that risks long-term depletion rather than as the level that maximizes long-term average yield, thereby insuring a buffer between OFL and ABC; 2) to eliminate use of F_{MSY} and B_{MSY} as reference points altogether; and 3) to set ABC and OFL according to an unspecified "rule of reason" when information is extremely poor.

- *September 1992:* The Plan Teams held a lengthy discussion regarding the SSC's proposal. However, the teams were unable to reach a consensus at that meeting. In response to further deliberation and comments received from individuals, the SSC directed one of its members to prepare a modification of the original SSC proposal for review by the Plan Teams at their November meeting. Among other modifications, the new proposal reinserted F_{MSY} as a potential management target and dropped the requirement for a buffer between ABC and OFL in cases where F_{MSY} is used to set OFL.

- *November 1992:* The Plan Teams declined to endorse the modified SSC proposal. Instead, the Teams adopted a statement which they hoped would become a "joint memorandum of understanding" between themselves and the SSC and which would be implementable within the existing structure of Amendment 21/16. Its central feature was the establishment of a buffer between ABC and OFL. In cases where F_{MSY} was unknown, this was to be accomplished by capping F_{ABC} at the $F_{35\%}$ level. In cases where F_{MSY} was known, establishment of the buffer was to be accomplished by scaling ABC downward by the ratio of $F_{35\%}$ to $F_{30\%}$.

- *January 1993:* The SSC rejected the Plan Teams' proposed joint memorandum and ended up tabling the amendment proposal. At that time, NMFS was undertaking a nationwide review of overfishing definitions, and the SSC's decision to table was based in part on the hope that this review would result in "a stronger consensus on overfishing."

- *December 1993:* The SSC restated its concurrence with "the Team's aim of providing a margin between ABC and overfishing." The SSC further noted, "The NMFS overfishing review, now nearing completion, will most likely provide a new standard definition of overfishing that will far exceed any of the ABC definitions used by this Council. After this report is available, the SSC wishes to work with the Team on ABC and OFL definitions."

- *1994:* The report of NMFS' Overfishing Definitions Review Panel (Rosenberg *et al.* 1994, known as the "Rosenberg Report") was published. The report did not contain a new standard definition of overfishing. However, it did contain language interpreting the generic overfishing definition contained in the *602 Guidelines* [BOX 10]. In addition, the report suggested modifying the Amendment 21/16 overfishing definition as follows: 1) F_{OFL} should vary directly with biomass, from a value of zero when biomass is at a very low "threshold" reference level to a value greater than F_{MSY} when biomass is at a much higher "precautionary" reference level; 2) a buffer between OFL and ABC should be established; 3) the authority for determining reliability of information should be specified; and 4) ambiguity should be eliminated in any text relating SPR to exploitable biomass.

- *1994-1996:* Research was conducted in anticipation of a future amendment to redefine ABC and OFL. This research focused largely on decision-theoretic methods to adjust harvest rates appropriately in relation to uncertainty.

- *January 1996:* The SSC considered a proposed plan amendment to redefine ABC and OFL, and recommended that an amendment package be developed for initial consideration at the April Council meeting.

- *April 1996:* A draft EA/RIR for Amendment 44/44 was presented, containing proposed revisions to the ABC and OFL definitions. The SSC recommended that the document go out for public review with some minor revisions. The Council concurred.

- *June 1996:* The Council approved Amendment 44/44, which specified revised definitions of ABC and overfishing [BOX 11], including a revised Tier system [BOX 12]. The revisions addressed the concerns identified by the SSC and the Rosenberg Report. The basic functional form of the OFL control rule instituted in Amendment 21/16 was retained but now exhibited an intercept to the right of the origin, and a control rule of similar functional form was now specified for ABC as well. Two other qualitatively new features were also introduced: 1) a buffer was instituted between ABC and OFL in all cases, and 2) the ABC control rule was defined such that greater uncertainty regarding the productive capacity of a stock resulted in a lower ABC (Tier 1 only). In Tier 2, the buffer was established by adjusting the cap on F_{OFL} upward so that it now exceeded F_{MSY}, in keeping with the approach advocated by the SSC and the Rosenberg Report (but opposite to the Plan Teams' approach, which was to adjust F_{ABC} downward).

- *October 1996:* The Sustainable Fisheries Act was signed into law. The Sustainable Fisheries Act consisted of a set of amendments to the Magnuson Fishery Conservation and Management Act (MFCMA). The MFCMA as amended by the SFA became known as the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA). Important features of the MSFCMA included the following: 1) FMPs were now required to include "objective and measurable criteria for identifying when the fishery to which the plan applies is overfished;" 2) any amendments necessary to bring FMPs into compliance with the new requirements of the Act had to be submitted by 10/11/98; 3) the Secretary of Commerce was required to establish advisory guidelines to assist in development of any necessary amendments (but without a deadline); and 4) the relationship between OY and MSY was changed such that MSY was now to serve as an upper limit on OY, implying that the upward adjustment of F_{OFL} in Tier 2 of Amendment 44/44 was no longer permissible.

- *August 1997:* NMFS published a draft revision of the *National Standard Guidelines* as a proposed rule. Important features of the draft revision included the following: 1) the concept of "MSY control rule" was introduced, with a wide variety of options including constant catch, constant fishing mortality, constant escapement, and any functional form relating fishing mortality to stock size; 2) the concept of "status determination criteria" was introduced, the required components of which consisted of a maximum fishing mortality threshold (MFMT) at least as conservative as the MSY control rule and a minimum stock size threshold (MSST) defined in part by the rate of rebuilding expected under the MFMT; 3) "overfishing" and "overfished" were distinguished such that "overfishing" meant any fishing mortality rate greater than MFMT whereas "overfished" meant any stock size less than MSST; 4) a "precautionary approach" was defined, modeled after three central features of the harvest control rules defined in Amendment 44/44; and 5) a promise was made to supplement the *Guidelines* in the "near future" with "additional technical guidance."

- *February 1998:* The 5th National Stock Assessment Workshop was held. Papers and working groups discussed various aspects of implementing a precautionary approach to fisheries management. Work continued on "Technical guidance on the use of precautionary approaches to implementing National Standard 1 of the MSFCMA" (Restrepo *et al*. 1998, known as the "Restrepo report"), which was intended to fulfill the promise of additional help made in the draft revision of the *National Standard Guidelines*.

- *April 1998:* The Council considered a preliminary draft of the EA/RIR for Amendments 56/56. This draft contained three alternatives, including the "no action" Alternative 1. The following characteristics pertained to Alternatives 2 and 3: 1) a set of proxies for the MSY level was listed, of which $B_{35\%}$ was one; 2) the inflection point of the control rules was set at the MSY level in all cases; and 3) an MSST was included. The SSC recommended a number of substantial modifications to the draft. The Council approved release of the draft, as modified by the SSC, for public review. Some concern was expressed over the fact that much of the draft EA/RIR was based on NMFS' draft revision of the *National Standard Guidelines*, which had not yet been published in final form.

- *May 1998:* NMFS published the revised *National Standard Guidelines* in final form. All of the central features pertaining to status determination criteria were retained in the final rule. Among other things, the revised *Guidelines* stated that Secretarial approval or disapproval of a Council's proposed status determination criteria will be based on whether the proposal (a) has sufficient scientific merit, (b) provides a basis for objective measurement of the status of the stock against the criteria, (c) is operationally feasible, and (d) contains both an MFMT and an MSST.

- *June 1998:* The Council approved Amendment 56/56, which added two sentences pertaining to the preferred estimators of B and B_{MSY} in the definition of overfishing [BOX 13] and made changes to the F_{ABC} definition in Tiers 2a and 2b and the F_{OFL} definition in Tiers 2a, 2b, 3a, 3b, and 4 [BOX 14]. The EA/RIR contained the "no action" alternative and Alternative 2, which was crafted by the SSC and was distinguishable from earlier draft alternatives in a number of ways, including the following: 1) no proxies for the MSY level were listed, 2) the inflection point of the control rules was set at the MSY level for Tiers 1 and 2 and at $B_{40\%}$ for Tier 3, and 3) an MSST was not included. The SSC recommended approval of Alternative 2, and the Council did so. The SSC also recommended that the Plan Teams "consider further improvements to ABC and OFL definitions at their September meeting. These potential improvements could include consideration of proxies for biological reference points (along the lines of Alternative 2 and 3 of the previous EA/RIR), adjustments of either biomass or fishing mortality using standard errors, and other approaches to incorporating uncertainty into decision making. The SSC will then review Plan Team findings for possible development of an amendment at its October meeting."

- *July 1998:* The "Restrepo Report" (Restrepo *et al.* 1998) was published. Default control rules (both limit and target) were suggested, incorporating the central features of the "precautionary approach" defined in the *National Standard Guidelines* which were in turn modeled on the harvest control rules defined in Amendment 44/44: a) target harvest rates (such as F_{ABC}) should be set safely below limit harvest rates (such as F_{OFL}), b) a stock that is below an appropriate reference level should be harvested at a lower rate than if it were above that level, and c) criteria used to set target catch levels should be explicitly risk averse, so that greater uncertainty regarding the status or productive capacity of a stock corresponds to greater caution in setting target catch levels. The report suggested that the above features be adopted regardless of the level of information available (i.e., they should not be used only for data-rich cases).

- *September 1998:* The Plan Teams discussed the potential improvements referred to them by the SSC in June, including use of a minimum stock size threshold, and forwarded two documents to the SSC for consideration: "Technical guidance on the use of precautionary approaches to implementing National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act" (NOAA Tech. Memo. NMFS-F/SPO-31 by Victor Restrepo et al.), and "Optimizing harvest control rules in the presence of natural variability and parameter uncertainty" (draft manuscript by Grant Thompson).

- *October 1998:* The SSC declined to address development of a new amendment regarding redefinition of ABC and OFL.

- *January 1999:* Amendments 56/56 received Secretarial approval. This approval was granted with the understanding that Amendments 56/56 contained a proxy for MSST. This proxy involved shifting the intercept of the sloped portion of the OFL control rule such that rebuilding to the MSY level would be expected within 10 years even if catches were set equal to the value associated with the OFL control rule in each year. However, this proxy had not been considered by either the SSC or the Council and had not been tested at the time of approval.

- *April-July 1999:* The MSST proxy envisioned when Amendments 56/56 were approved turned out to be highly impractical, resulting in OFLs of zero for some stocks that were only modestly below $B_{40\%}$. Many alternative methods for interpreting or revising Amendments 56/56 were then examined for each stock managed under Tiers 1-3.

- *August 1999*: NMFS decided upon a strategy to be used in completing the required status determination report (the "Report to Congress"). Major features included the following: 1) an MSST was used for all stocks managed under Tiers 1-3; 2) $B_{35\%}$ was used as the proxy for the MSY level in Tier 3 (this did not involve a change in the control rule, but rather an interpretation as to when a stock would be considered "rebuilt"); and 3) a "regime shift" commencing in 1977 was recognized, meaning that all recruitment time series were standardized such that no year classes spawned prior to 1977 were included. By this time, formal requests for a new FMP amendment relating to MFMT and MSST had been made by the Alaska Marine Conservation Council, the Center for Marine Conservation, and the Alaska Fisheries Science Center.

- *September 1999:* The SSC discussed issues pertaining to Amendments 56/56. The BSAI and GOA Groundfish Plan Teams gave a "high-plus" priority rating to proposals calling for an amendment to the FMPs' current treatment of MFMT and MSST (or lack thereof).

- *January 2000:* The "Control Rule Alternatives Workshop: Design, Analysis, Decision" (CRAWDAD) was convened. The workshop was held for the purpose of developing alternatives to be analyzed in a new amendment package dealing with harvest control rules. Workshop participants consisted of four representatives from the Plan Teams and four representatives from the SSC. A series of 18 issues were addressed. Workshop participants were in agreement that the issues involved in this exercise warranted an especially thorough analysis, even if this meant that an FMP amendment would not be completed in time to be implemented for the 2001 harvest specifications.

- *February 2000:* The SSC received the report of the CRAWDAD and concluded, "The workshop was successful in exploring the scope of alternatives and analysis to be conducted, although much fleshing out of activity remains to be done.... The SSC suggests that the timeline for the new analysis be sufficiently long enough to involve the SSC and Plan Teams in the range of alternatives and analytical approach. It is not necessary to rush this analysis; our current procedures can be used one more year if necessary."

- *Remainder of 2000-Present:* Consideration of another revision of the ABC and OFL definitions has been hampered by a number of factors. First among these is the protracted development of the Programmatic Supplemental Environmental Impact Statement (PSEIS) for the two groundfish FMPs. Because a key component of the PSEIS is an evaluation of the status quo, there has been some reluctance to implement any significant change in the status quo until the PSEIS has been finalized unless such significant change is absolutely necessary. Second is the uncertainty caused by the annual attempts at reauthorizing the MSFCMA. There is understandable desire to avoid repeating the experience of Amendment 44/44, wherein the SFA was passed only four months after Council approval of the amendment, which then started the clock ticking on another revision to the ABC and OFL definitions before the previous revision had even been implemented. Third is the apparent lack of an agency-wide consensus on the part of NMFS as to the importance of complying with the *National Standard Guidelines*. In the spring of 2002 NMFS Headquarters convened a panel to evaluate the compliance of all the status determination criteria currently in use nationwide and since then has been conferring with the regional offices to determine an appropriate course of action for the future, which could involve a new round of plan amendments, another revision of the *Guidelines*, or something else.

2.12.1.1 BOX 7. Definitions in 1988 (after Amendments GOA 16/BSAI 11)

Acceptable biological catch (ABC) is a seasonally determined catch or range of catches that may differ from MSY for biological reasons. It may be lower or higher than MSY in some years for species with fluctuating recruitments. Given suitable biological data and justification by the plan team and/or SSC, ABC may be set anywhere between zero and the current biomass less the threshold value. The ABC may be modified to incorporate safety factors and risk assessment due to uncertainty. Lacking other biological justification, the ABC is defined as the maximum sustainable yield exploitation rate multiplied by the size of the biomass for the relevant time period. The ABC is defined as zero when the stock is at or below its threshold.

Overfishing is a level of fishing mortality that jeopardizes the capacity of stock(s) to maintain or recover to a level at which it can produce maximum sustainable yield on a long-term basis under prevailing biological and environmental conditions. Overfishing is the application of exploitation rates that drive the stock below its threshold. Exceeding acceptable biological catch need not result in overfishing, unless the excess is taken over sufficient time to reduce the population below its threshold.

Threshold is the minimum size of a stock that allows sufficient recruitment so that the stock can eventually reach a level that produces MSY. Implicit in this definition are rebuilding schedules. They have not been specified since the selection of a schedule is a part of the optimum yield (OY) determination process. Interest instead is on the identification of a stock level below which the ability to rebuild is uncertain. The estimate given should reflect use of the best scientific information available. Whenever possible, upper and lower bounds should be given for the estimate.

Maximum sustainable yield (MSY) is an average over a reasonable length of time of the largest catch which can be taken continuously from a stock under current environmental conditions. It should normally be presented with a range of values around its point estimate. Where sufficient scientific data as to the biological characteristics of the stock do not exist or the period of exploitation or investigation has not been long enough for adequate understanding of stock dynamics, a preliminary MSY will be estimated from the best information available.

2.12.1.2 BOX 8. NOAA Section 602 Guidelines (national), July 24, 1989

Overfishing is a level or rate of fishing mortality that jeopardizes the long-term capacity of a stock or stock complex to produce MSY on a continuing basis. Each FMP must specify, to the maximum extent possible, an objective and measurable definition of overfishing for each stock or stock complex covered by that FMP, and provide an analysis of how the definition was determined and how it relates to reproductive potential.

Acceptable biological catch (ABC) is a preliminary description of the acceptable harvest (or range of harvests) for a given stock or stock complex. Its derivation focuses on the status and dynamics of the stock, environmental conditions, other ecological factors, and prevailing technological characteristics of the fishery. When ABC is used, its specification constitutes the first step in deriving OY from MSY. Unless the best scientific information available indicates otherwise, ABC should be no higher than the product of the stock's natural mortality rate and the biomass of the exploitable stock. If a threshold has been specified for the stock, ABC must equal zero when the stock is at or below that threshold. ABC may be expressed in numeric or nonnumeric terms.

Optimum yield (OY): Optimum, with respect to yield from a fishery, is the amount of fish which will provide the greatest overall benefit to the nation, with particular reference to food production and recreational opportunities, and which is prescribed as such on the basis of MSY from each fishery, as modified by any relevant economic, social, or ecological factors.

Maximum sustainable yield (MSY) is the largest average annual catch or yield that can be taken over a significant period of time from each stock under prevailing ecological and environmental conditions. MSY may be presented as a range of values.

Threshold is a minimum level of spawning biomass that may be used in defining overfishing.

2.12.1.3 BOX 9. Revised Definition of Overfishing in 1990 (after Amendments GOA 21/BSAI 16)

Overfishing is defined as a maximum allowable fishing mortality rate. For any stock or stock complex under management, the maximum allowable fishing mortality rate will be set at the level corresponding to maximum sustainable yield (F_{MSY}) for all biomass levels in excess of the level corresponding to maximum sustainable yield (B_{MSY}). For lower biomass levels, the maximum allowable fishing mortality rate will vary linearly with biomass, starting at a value of zero at the origin and increasing to a value of F_{MSY} at B_{MSY}, consistent with other applicable laws. If data are insufficient to calculate F_{MSY} or B_{MSY}, the maximum allowable fishing mortality rate will be set equal to the following (in order of preference): (1) the value that results in the biomass-per-recruit (measured in terms of spawning biomass) falling to 30% of its pristine value, (2) the value that results in the biomass-per-recruit (measured in terms of exploitable biomass) falling to 30% of its pristine value, or (3) the natural mortality rate (M). If data are insufficient to estimate any of the above, the TAC shall not exceed the average catch since 1977.

2.12.1.4 BOX 10. Overfishing Definitions Review Panel (1994)

Because the generic overfishing definition contained in the *602 Guidelines* is subjective, the Panel felt that it would be useful to interpret this definition in a consistent and objective fashion. Since the primary intent of the *Guidelines* is to prevent recruitment overfishing, the Panel's interpretation is as follows:

> *Absent compelling evidence to the contrary, a finding that expected recruitment has fallen below one-half the expected maximum will be taken as sufficient evidence that the stock has been overfished (in the sense of the MFCMA), and any level of fishing that would result in expected recruitment falling below one-half the expected maximum (or remaining there indefinitely) will be taken to constitute overfishing.*

Several points regarding the above interpretation should be noted: (1) For the purpose of this review, this interpretation was intended to provide a conceptual focus rather than a means of empirically evaluating each of the current stock-specific definitions; (2) it frames the overfished condition and the act of overfishing in the common currency of expected recruitment; and (3) because it links overfishing explicitly to expected recruitment, this interpretation provides a natural transition to definitions based on spawning biomass thresholds or on fishing mortality rates that are expressed in terms of equilibrium spawning per recruit.

2.12.1.5 BOX 11. Definitions of ABC and Overfishing in Amendment 44/44 (June 1996)

Acceptable Biological Catch is a preliminary description of the acceptable harvest (or range of harvests) for a given stock or stock complex. Its derivation focuses on the status and dynamics of the stock, environmental conditions, other ecological factors, and prevailing technological characteristics of the fishery. The fishing mortality rate used to calculate ABC is capped as described under "overfishing" below.

Overfishing is defined as any amount of fishing in excess of a prescribed maximum allowable rate. This maximum allowable rate is prescribed through a set of six Tiers which are listed below in descending order of preference, corresponding to descending order of information availability. The SSC will have final authority for determining whether a given item of information is "reliable" for the purpose of this definition, and may use either objective or subjective criteria in making such determinations. For Tier (1), a "pdf" refers to a probability density function. For Tiers (1-3), the coefficient α is set at a default value of 0.05, with the understanding that the SSC may establish a different value for a specific stock or stock complex as merited by the best available scientific information. For Tiers (2-4), a designation of the form "$F_{X\%}$" refers to the F associated with an equilibrium level of spawning per recruit (SPR) equal to $X\%$ of the equilibrium level of spawning per recruit in the absence of any fishing. If reliable information sufficient to characterize the entire maturity schedule of a species is not available, the SSC may choose to view SPR calculations based on a knife-edge maturity assumption as reliable. For Tier (3), the term $B_{40\%}$ refers to the long-term average biomass that would be expected under average recruitment and $F=F_{40\%}$.

2.12.1.6 BOX 12. Tier System Defined by Amendment 44/44 (June 1996)

1) *Information available: Reliable point estimates of B and* B_{MSY} *and reliable pdf of* F_{MSY} .
   1a) *Stock status:* $B/B_{MSY} > 1$
      $F_{OFL} = \mu_A$, the arithmetic mean of the pdf
      $F_{ABC} \leq \mu_H$, the harmonic mean of the pdf
   1b) *Stock status:* $\alpha < B/B_{MSY} \leq 1$
      $F_{OFL} = \mu_A \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
      $F_{ABC} \leq \mu_H \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
   1c) *Stock status:* $B/B_{MSY} \leq \alpha$
      $F_{OFL} = 0$
      $F_{ABC} = 0$
2) *Information available: Reliable point estimates of B,* B_{MSY}, F_{MSY}, $F_{30\%}$, and $F_{40\%}$.
   2a) *Stock status:* $B/B_{MSY} > 1$
      $F_{OFL} = F_{MSY} \times (F_{30\%}/F_{40\%})$
      $F_{ABC} \leq F_{MSY}$
   2b) *Stock status:* $\alpha < B/B_{MSY} \leq 1$
      $F_{OFL} = F_{MSY} \times (F_{30\%}/F_{40\%}) \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
      $F_{ABC} \leq F_{MSY} \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
   2c) *Stock status:* $B/B_{MSY} \leq \alpha$
      $F_{OFL} = 0$
      $F_{ABC} = 0$
3) *Information available: Reliable point estimates of B,* $B_{40\%}$, $F_{30\%}$, and $F_{40\%}$.
   3a) *Stock status:* $B/B_{40\%} > 1$
      $F_{OFL} = F_{30\%}$
      $F_{ABC} \leq F_{40\%}$
   3b) *Stock status:* $\alpha < B/B_{40\%} \leq 1$
      $F_{OFL} = F_{30\%} \times (B/B_{40\%} - \alpha)/(1 - \alpha)$
      $F_{ABC} \leq F_{40\%} \times (B/B_{40\%} - \alpha)/(1 - \alpha)$
   3c) *Stock status:* $B/B_{40\%} \leq \alpha$
      $F_{OFL} = 0$
      $F_{ABC} = 0$
4) *Information available: Reliable point estimates of B,* $F_{30\%}$, and $F_{40\%}$.
      $F_{OFL} = F_{30\%}$
      $F_{ABC} \leq F_{40\%}$
5) *Information available: Reliable point estimates of B and natural mortality rate M.*
      $F_{OFL} = M$
      $F_{ABC} \leq 0.75 \times M$
6) *Information available: Reliable catch history from 1978 through 1995.*
      OFL = the average catch from 1978 through 1995, unless an alternative value is established by the SSC on the basis of the best available scientific information

$$ABC \leq 0.75 \times OFL$$

2.12.1.7 BOX 13. Definition of Overfishing in Amendment 56/56 (June 1998)

Overfishing is defined as any amount of fishing in excess of a prescribed maximum allowable rate. This maximum allowable rate is prescribed through a set of six Tiers which are listed below in descending order of preference, corresponding to descending order of information availability. The SSC will have final authority for determining whether a given item of information is "reliable" for the purpose of this definition, and may use either objective or subjective criteria in making such determinations. For Tier (1), a "pdf" refers to a probability density function. For Tiers (1-2), if a reliable pdf of B_{MSY} is available, the preferred point estimate of B_{MSY} is the geometric mean of its pdf. For Tiers (1-5), if a reliable pdf of B is available, the preferred point estimate is the geometric mean of its pdf. For Tiers (1-3), the coefficient α is set at a default value of 0.05, with the understanding that the SSC may establish a different value for a specific stock or stock complex as merited by the best available scientific information. For Tiers (2-4), a designation of the form "$F_{X\%}$" refers to the F associated with an equilibrium level of spawning per recruit (SPR) equal to $X\%$ of the equilibrium level of spawning per recruit in the absence of any fishing. If reliable information sufficient to characterize the entire maturity schedule of a species is not available, the SSC may choose to view SPR calculations based on a knife-edge maturity assumption as reliable. For Tier (3), the term $B_{40\%}$ refers to the long-term average biomass that would be expected under average recruitment and $F=F_{40\%}$.

2.12.1.8 BOX 14. Tier System Defined by Amendment 56/56 (June 1998)

1) *Information available: Reliable point estimates of B and B_{MSY} and reliable pdf of F_{MSY}.*
    1a) *Stock status: $B/B_{MSY} > 1$*
        $F_{OFL} = \mu_A$, the arithmetic mean of the pdf
        $F_{ABC} \leq \mu_H$, the harmonic mean of the pdf
    1b) *Stock status: $\alpha < B/B_{MSY} \leq 1$*
        $F_{OFL} = \mu_A \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
        $F_{ABC} \leq \mu_H \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
    1c) *Stock status: $B/B_{MSY} \leq \alpha$*
        $F_{OFL} = 0$
        $F_{ABC} = 0$
2) *Information available: Reliable point estimates of B, B_{MSY}, F_{MSY}, $F_{35\%}$,* and $F_{40\%}$.
    2a) *Stock status: $B/B_{MSY} > 1$*
        $F_{OFL} = F_{MSY}$
        $F_{ABC} \leq F_{MSY} \times (F_{40\%} / F_{35\%})$
    2b) *Stock status: $\alpha < B/B_{MSY} \leq 1$*
        $F_{OFL} = F_{MSY} \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
        $F_{ABC} \leq F_{MSY} \times (F_{40\%} / F_{35\%}) \times (B/B_{MSY} - \alpha)/(1 - \alpha)$
    2c) *Stock status: $B/B_{MSY} \leq \alpha$*
        $F_{OFL} = 0$
        $F_{ABC} = 0$
3) *Information available: Reliable point estimates of B, $B_{40\%}$, $F_{35\%}$,* and $F_{40\%}$.
    3a) *Stock status: $B/B_{40\%} > 1$*
        $F_{OFL} = F_{35\%}$
        $F_{ABC} \leq F_{40\%}$
    3b) *Stock status: $\alpha < B/B_{40\%} \leq 1$*
        $F_{OFL} = F_{35\%} \times (B/B_{40\%} - \alpha)/(1 - \alpha)$
        $F_{ABC} \leq F_{40\%} \times (B/B_{40\%} - \alpha)/(1 - \alpha)$
    3c) *Stock status: $B/B_{40\%} \leq \alpha$*
        $F_{OFL} = 0$
        $F_{ABC} = 0$
4) *Information available: Reliable point estimates of B, $F_{35\%}$,* and $F_{40\%}$.
        $F_{OFL} = F_{35\%}$
        $F_{ABC} \leq F_{40\%}$
5) *Information available: Reliable point estimates of B and natural mortality rate M.*
        $F_{OFL} = M$
        $F_{ABC} \leq 0.75 \times M$
6) *Information available: Reliable catch history from 1978 through 1995.*
        OFL = the average catch from 1978 through 1995, unless an alternative value is established by the SSC on the basis of the best available scientific information
        ABC ≤ $0.75 \times$ OFL

2.13 Illustrative Example

We develop a hypothetical prototype of an age-structured population with density-dependence (i.e., a spawner-recruit relationship) to illustrate these concepts (after Quinn and Collie, "Sustainability in single-species population models," in review). Our goal is to illustrate the response of the population to five alternative harvest strategies. The central focus of the policies is related to the NPFMC biomass-based strategy and its major component, $F_{40\%}$. $F_{40\%}$ is calculated for a specified amount of recruitment from essential population parameters, such as natural and fishing mortality, average weight, fecundity, and maturity. Essentially, $F_{40\%}$ is the full-recruitment fishing mortality that results in spawning biomass *per recruit* being reduced to 40% of that under no fishing (Figure 1).

When fishing occurs, there is a continuum of sustainable (meaning positive) yields and populations, starting at F=0 with zero yield and equilibrium population at carrying capacity. There is a fishing mortality, F_{MSY}, which results in maximum sustainable yield, MSY, and a higher value, F_{ext}, for which the population is eventually driven to extinction. For each F between 0 and F_{ext}, there is a corresponding sustainable population. In this example, F_{MSY} is chosen to be the same as $F_{40\%}$ for simplicity. We alter the productivity parameter α for the spawner-recruit relationship (described below) so that this equivalence occurs. We discuss later what happens when less productive or more productive values of α are chosen.

The five strategies are depicted in Figure 2a in terms of full-recruitment fishing mortality (the fishing mortality experienced at older ages for which selectivity of the gear is maximal) and in Figure 2b in terms of the corresponding catch in weight units (yield).The x axis in Figures 2a and 2b is truncated at 1.5 times $B_{40\%}$ (the equilibrium spawning biomass corresponding to $F_{40\%}$, obtained by multiplying spawning biomass per recruit by the anticipated average recruitment over the period of interest) to clearly distinguish the various policies at low biomass levels.

Three constant fishing mortality policies are set: F=0 (no fishing), F=0.26 (fishing at $F_{40\%}$), and F=0.74 (heavy fishing). Two other policies using $F_{40\%}$ are also set: the biomass-based strategy used by NPFMC and a stair-step threshold strategy wherein no fishing is allowed until the population is above the threshold. The threshold occurs halfway between the threshold (0.05 $B_{40\%}$) and the target ($B_{40\%}$) in the NPFMC strategy.

The pristine biomass corresponding to no fishing occurs at about 3 times $B_{40\%}$. The reason that this value is not 2.5 times $B_{40\%}$ (i.e., the inverse of 40%) is that $F_{40\%}$ is calculated on the basis of spawning biomass *per recruit*. When a density-dependent relationship is actually present, the recruitment is on average somewhat lower when fishing at $F_{40\%}$ than when not fishing, so that equilibrium spawning biomass is somewhat less than 40% (namely 33%) of the pristine level.

The parameters for this hypothetical population are shown in Table 1. The population has 10 age classes. Natural mortality M is a U-shaped function of age, with the highest mortality during the early life history and increasing mortality as senescence approaches at the older ages. Fishing mortality F is a logistic function of age, in which 50% selectivity occurs at age 3. Fishing mortality for each year is calculated from the product of the logistic function and the fishing mortality for fully-recruited ages, which comes from the harvest strategy.

Length L is modeled as a typical von Bertalanffy function, and weight W is an isometric (cubic) function of length. Maturity is a logistic function of age, in which 50% are mature at age 5. Fecundity is an isometric function of length. A Beverton-Holt spawner-recruit relationship [$N_1 = \alpha\,\text{Eggs}/(1+\beta\text{Eggs})$] is used to determine the number of age 1 individuals from the number of eggs produced by the spawning population. Because weight and fecundity are both isometric, spawning biomass and egg production are proportional. None of the population parameters is a function of time, so that carrying capacity K is a constant. At the start, the population is at a low population well below the equilibrium level so that various harvest policies can be evaluated in terms of rebuilding success.

When no fishing occurs, recruitment to the population steadily increases, reaching the equilibrium recruitment of about 2000 fish at age 1 (Figure 3, Table 2). The constant $F_{40\%}$ strategy, the NPFMC biomass-based strategy, and the threshold strategy have similar increases, rising to equilibrium of about 1600 fish at age 1, about 80% of pristine. They all fish at $F_{40\%}$ at high population levels. The NPFMC and threshold strategies rebuild similarly to the no fishing strategy until recruitment is about 80% of the pristine. The rate of rebuilding is substantially higher than for the constant $F_{40\%}$ strategy. The heavy fishing strategy results in no increase in recruitment at all.

The corresponding abundance of the population has trends similar to those of recruitment. Under no fishing, the population equilibrates to its carrying capacity K of near 4500 individuals, aged 1 and older (Figure 4, Table 2). For the three $F_{40\%}$-based policies, abundance equilibrates to about 3200 individuals, about 70% of pristine, showing that these policies do not reduce the *overall* population down to 40% of pristine. Heavy fishing results in no rebuilding of the depleted population.

Spawning biomass under no fishing equilibrates to about 3500 biomass units (Figure 5, Table 2). For the three $F_{40\%}$-based policies, spawning biomass equilibrates to about 1200 biomass units, about 33% of pristine, showing that these policies do reduce spawning biomass lower than 40% of pristine. As mentioned before, $F_{40\%}$ is based on spawning biomass *per recruit* being 40% of pristine; the reduction to 33% comes from the decreased recruitment to 80% of pristine. Heavy fishing results in no rebuilding of the depleted spawning biomass. Spawning biomass and egg production are directly proportional in this example, as shown by the equilibrium values in Table 2, so they equally measure reproductive value.

There is no harm to this population with the reduction to 33% of pristine. Indeed, this population is then at its most productive state, because $F_{MSY} = F_{40\%}$ and the equilibrium biomass is hence B_{MSY}. In a multi-species setting, the reduction could either be beneficial or detrimental to other species, depending on whether they are predators or prey, and which age classes are of importance.

For all five-harvest strategies, the population also has a stable age distribution at equilibrium (Figure 6). These results are independent of whether the population starts low or high. The initial population size influences the trajectory of the approach to equilibrium but not its final outcome. The resulting stationarity of abundance and stability of its age distribution is a result of density dependence and is not an assumption.

This example suggests that fishing mortality is an important control variable in managing a fishery, with drastic consequences on the population if it is allowed to be too high. In this example a population at a low level can rebuild quickly to the target B_{MSY} or $B_{40\%}$ level, even in the face of recruitment being lower at lower biomass levels through the spawner-recruit relationship. Only highly depensatory spawner-recruit relationships prevent rebuilding, and there is not much evidence of depensation in actual fish populations, as shown by Myers and his colleagues at Dalhousie University.

Policies that diminish fishing mortality at lower biomass levels can help to speed the rebuilding process and protect the population if it becomes depleted. Even though the constant $F_{40\%}$ strategy is conservative, additional conservatism through the NPFMC biomass-based strategy or a threshold strategy helps guard against the risk of overharvesting. Such overharvesting could be due to changes in population parameters that go undetected, errors in data or assessment models, random variability and stochasticity, and of course, recruitment variability.

We constructed this example to show the theoretical underpinnings of NPFMC's harvest strategy. The example is not meant to be an exact rendition of how a population responds to fishing but rather to show the underlying tendency of the population response. In reality, stochasticity in population parameters creates a high level of variability in the population response.

In this example, the productivity parameter α in the spawner-recruit relationship was selected so that F_{MSY} was equal to the $F_{40\%}$. A less productive population would have a lower α, and consequently, $F_{40\%}$ would be higher than F_{MSY}, and *vice versa*. Therefore, the degree of conservatism in the $F_{40\%}$ strategy is dependent on the spawner-recruit relationship. The reason that $F_{40\%}$ emerged as a harvest strategy is that it tends to be lower than F_{MSY} (and thus more conservative) across a variety of spawner-recruit relationships. For the North Pacific, it appears that most populations are in the productivity range considered in the development of this strategy. In contrast, the Pacific Fishery Management Council (with responsibility for marine waters off Washington, Oregon, and California) found that their populations were less productive and consequently altered their strategy to be more conservative by using $F_{45\%}$ to $F_{60\%}$. Therefore, it is important to evaluate the harvest strategy in relationship to the actual productivity of the populations, rather than use a one-size-fits-all approach (see Section 3.1).

The underlying Beverton-Holt spawner-recruit relationship used in this example has the feature that recruitment increases as a function of spawning biomass to an asymptotic level. In contrast, other spawner-recruit relationships, such as the Ricker [$N_1 = \text{Eggs} \times \alpha \exp(-\beta \text{Eggs})$], have a dome-shaped pattern in which recruitment peaks at an intermediate level. The biological mechanism(s) for this peak include cannibalism of adults on small fish and crowding effects due to overescapement. Consequently, it need not be true that equilibrium recruitment is highest at the pristine carrying capacity. Therefore, the result in this example that equilibrium spawning biomass was lower than 40% of pristine biomass is not general.

Table 2.1. Parameters used in the prototype of an age-structured fish population. Notation is as in Quinn and Deriso (1999).

Age	1	2	3	4	5	6	7	8	9	10
Natural mortality *M*	1	0.5	0.3	0.2	0.2	0.2	0.2	0.3	0.5	1.5
Average weight *W*	0.060	0.358	0.918	1.670	2.526	3.412	4.276	5.084	5.816	6.465
Selectivity *s*	0.000	0.269	0.500	0.731	0.881	0.953	0.982	0.993	0.998	0.999
Fecundity *f*	0	0	0	89818	252580	498944	753335	968515	1142196	1284271
Maturity *m*	0%	0%	0%	27%	50%	73%	88%	95%	98%	99%

Beverton-Holt parameters α: 2.70E-05 β: 1.20E-08

Table 2.2. Equilibrium values of catch, yield, spawning stock biomass (SSB), egg production, recruitment (R), and abundance (N) for ages 1–10, for three values of full-recruitment fishing mortality: 0 (pristine), 0.26 ($F_{40\%}$, F_{MSY}), and heavy (0.74). The last row shows the underestimation of projected pristine quantities if density-dependence is ignored and the equilibrium recruitment under F40 fishing is used.

	Equilibrium					
Full-recruitment *F*	Catch	Yield	SSB	Egg Prod.	R	N
0 (Pristine)	0	0	3524	7.0E+08	2012	4539
0.26 ($Fmsy_{MSY}$, $F_{40\%}$)	179	372	1173	2.3E+08	1660	3230
0.74 (heavy fishing)	25	33	28	5.7E+06	143	246
Ignoring density-dependence						
0 (using R = 1660)	0	0	2908	5.8E+08	1660	3746

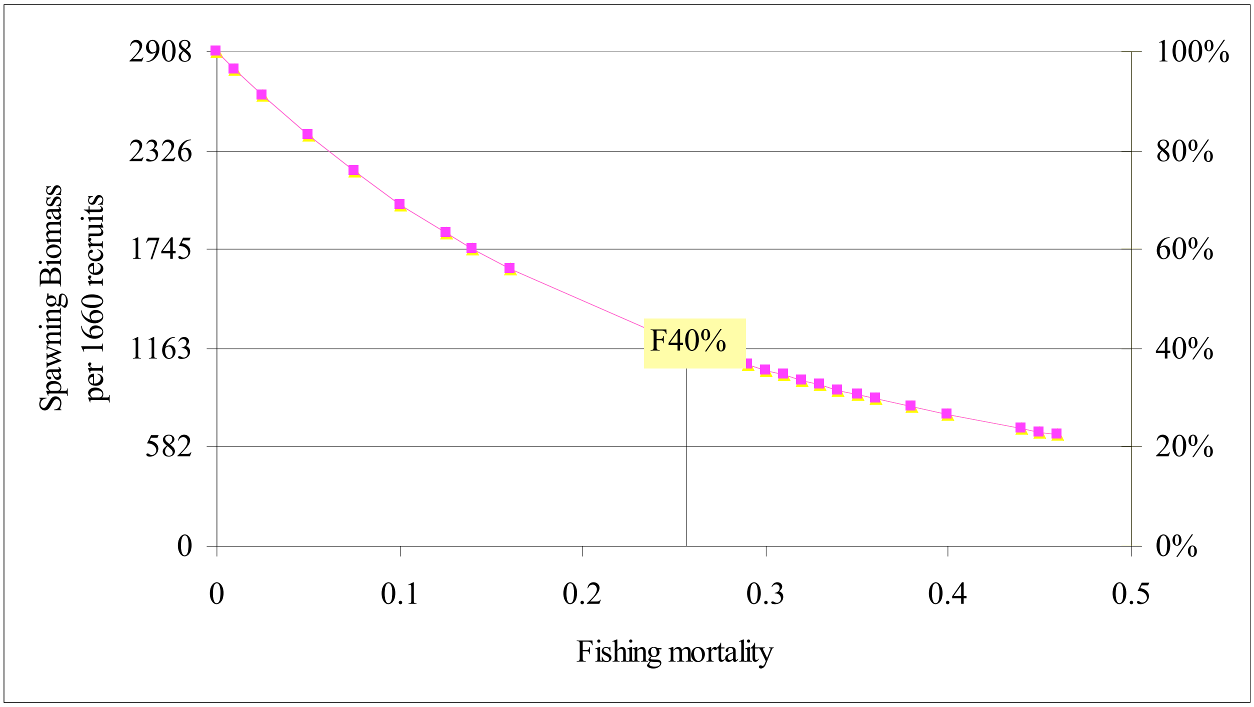


Figure 2.1. Determination of the $F_{40\%}$ full-recruitment fishing mortality value. A starting recruitment of 1660 fish at age 1 experiences natural and fishing mortality, grows, and eventually contributes to reproduction. If fishing mortality is increased, then fewer fish will survive to become part of the spawning biomass, as shown in the figure. The $F_{40\%}$ value (0.26) is the fishing mortality at which spawning biomass per recruit drops to 40% (1163 biomass units) of the unfished (or pristine) value (2903 biomass units). The starting recruitment used is immaterial to the determination of $F_{40\%}$; the value 1660 was used, because that is the equilibrium value of recruitment for the hypothetical population.

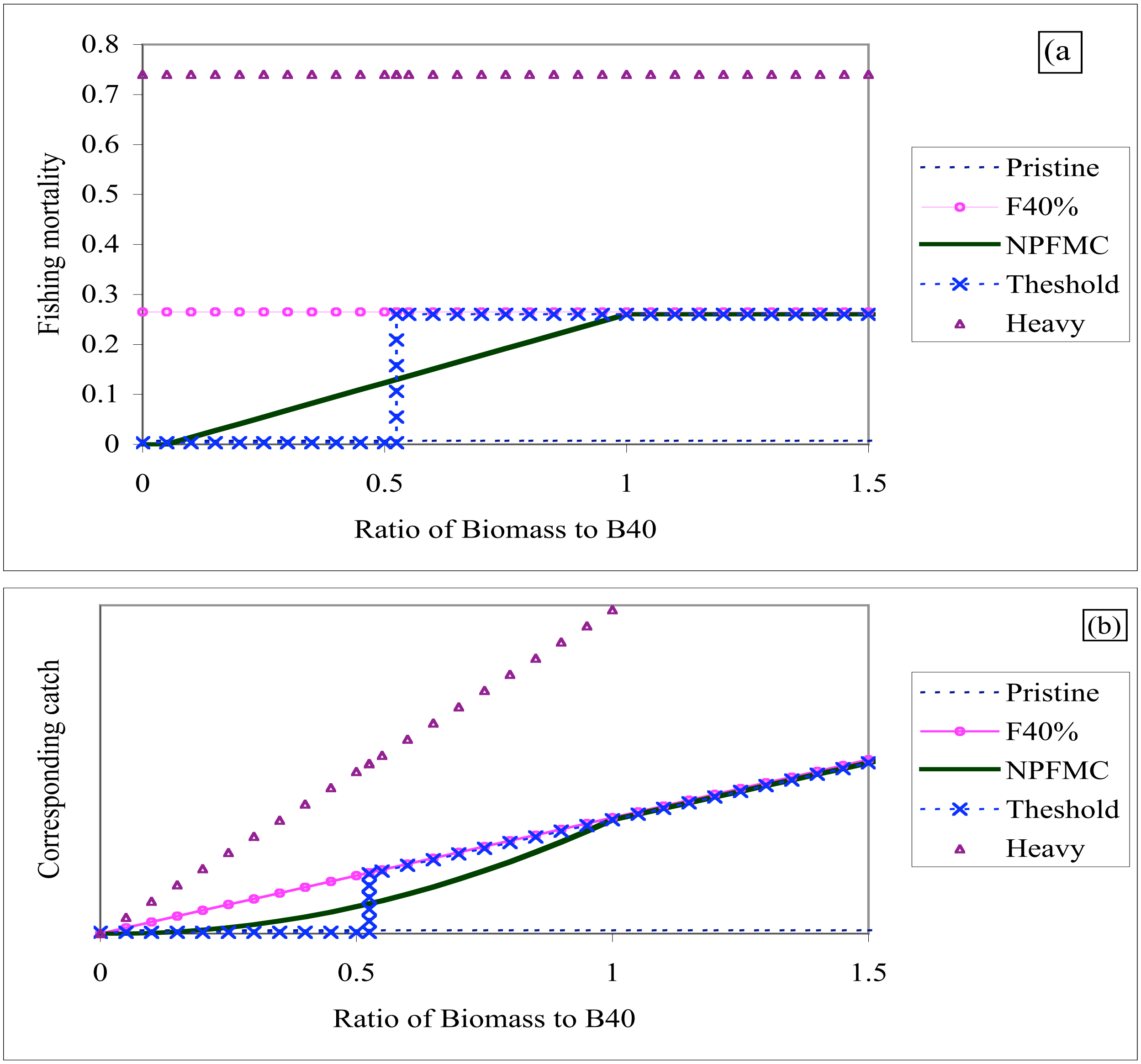


Figure 2.2. Depiction of various harvesting policies with values used in the example given in parentheses: (1) No fishing, or pristine conditions (F=0), (2) Constant fishing mortality with an $F_{40\%}$ strategy (F=0.26), (3) the NPFMC biomass-based strategy with lower fishing mortality when biomass drops below the target $B_{40\%}$ and a threshold (0.05 $B_{40\%}$), (4) threshold strategy with no fishing below the threshold [chosen as $B_{40\%}$ (1+0.05)/2], (5) heavy fishing (F=0.74). Values shown are fishing mortalities (F) and approximately corresponding values of catch or yield obtained by multiplying F by biomass B.

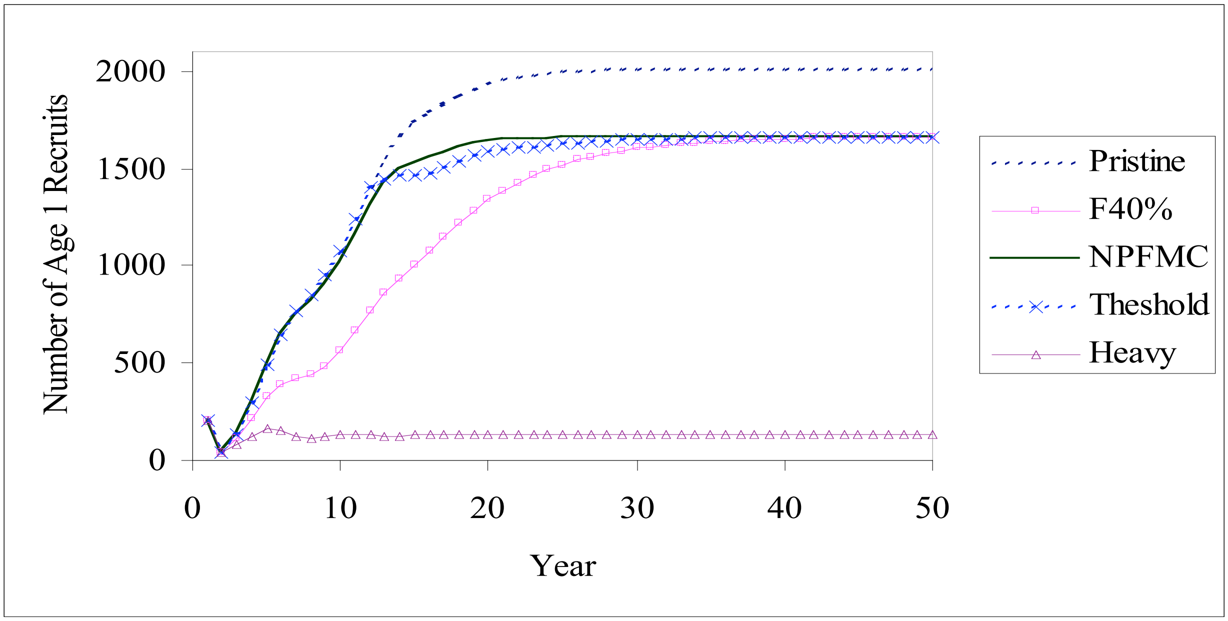


Figure 2.3. Number of age 1 recruits over time for the hypothetical population fished according to five alternative strategies, described in Figure 1.

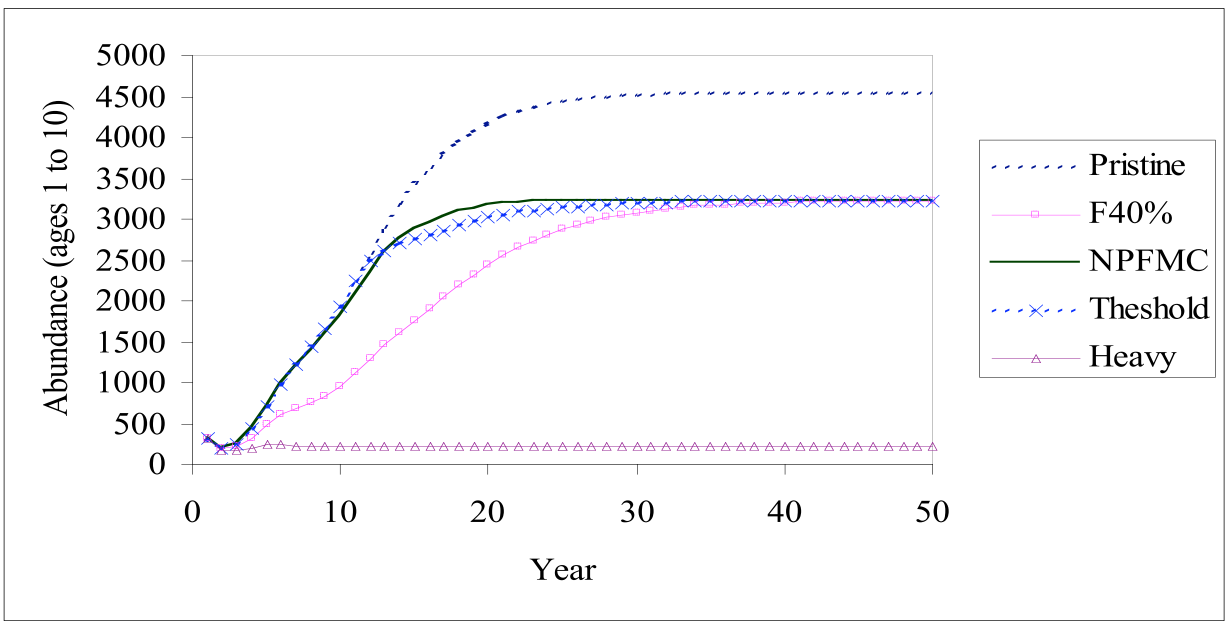


Figure 2.4. Abundance in numbers of fish for ages 1–10 over time for the hypothetical population fished according to five alternative strategies, described in Figure 1.

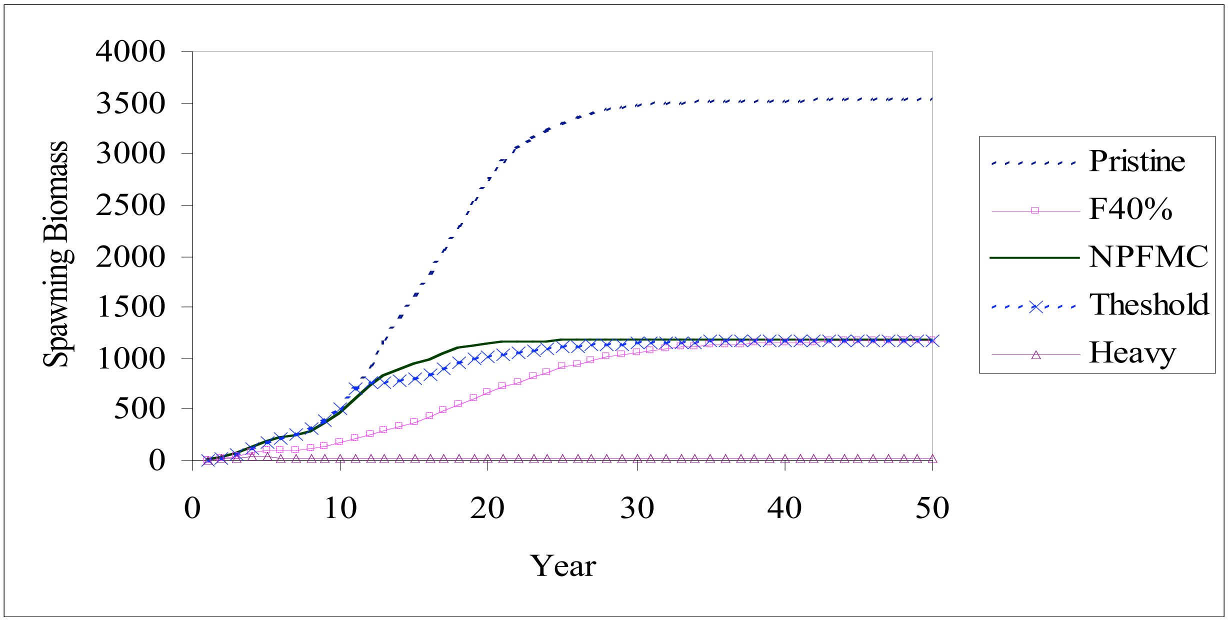


Figure 2.5. Spawning biomass over time for the hypothetical population fished according to five alternative strategies, described in Figure 1.

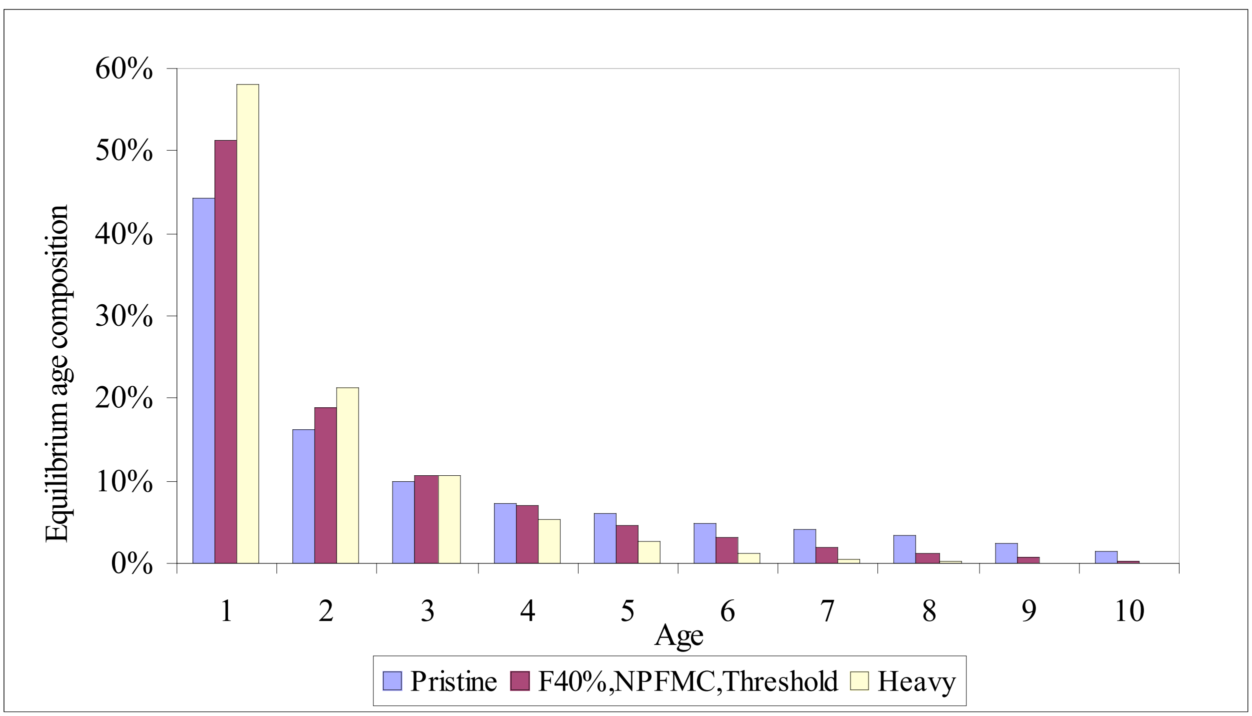


Figure 2.6. Stable age composition at equilibrium for the five alternative strategies.

3 SINGLE-SPECIES ISSUES

This chapter contains a review of how well the current NPFMC harvest strategy meets the requirements and goals pertaining to management and conservation of the target stocks, from a single-species perspective, as set forth in the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA), and as interpreted and expanded in the National Standard Guidelines (NSGs) developed by National Marine Fisheries Service (NMFS). Thus this chapter seeks to address charge b) of the terms of reference for this review.

3.1 Use of $F_{35\%}$ and $F_{40\%}$ as F_{MSY} Proxies

This section contains a brief review of the development of F_{MSY} proxies with a view to address the adequacy of such rules of thumb.

3.1.1 F_{MSY} as a Target or a Limit

F_{MSY} has a long history as a target level of fishing mortality at which stocks could be managed in order to maximize yields. However, the experience accumulated over past decades shows that F_{MSY} is not necessarily a good target from a conservation perspective, or from an economic one or even from the perspective of sustainable yields (Ludwig 1995). Due to natural fluctuations, for example, the long-term average yield that can be obtained from an F_{MSY} policy will be lower than the MSY level that would be estimated assuming constancy. Similarly, if a constant catch level, equal to the calculated MSY assuming constancy, is taken annually from a fluctuating population, the stock will decline. An added problem is that F_{MSY} is difficult to estimate and, therefore, it is difficult to implement F_{MSY} policies accurately without exceeding the intended target with an unsatisfactory frequency.

The practical realization that many stocks in fact were overfished, despite being hypothetically managed at F_{MSY}, was influential in the negotiation of several international instruments during the mid-1990s and in the reauthorization of the MSFCMA in the U.S. An end result was a policy change to treat F_{MSY} as a *limit* rather than a *target*. Annex II of the 1995 Agreement of the Implementation of the United Nations Convention on Law of the Sea (UNFA) stipulates:

> "*Fishery management strategies shall ensure that the risk of exceeding limit reference points is very low*," and "*The fishing mortality rate which generates maximum sustainable yield should be regarded as a minimum standard for limit reference points*."

The MSFCMA states the following as the first National Standard that FMPs shall be consistent with:

> "*Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry*."

The MSFCMA also provides the following definitions for "overfishing" and "optimum yield" to be read in conjunction with the above National Standard:

> "*The terms 'overfishing' and 'overfished' mean a rate or level of fishing mortality that jeopardizes the capacity of a fishery to produce the maximum sustainable yield on a continuing basis*."
>
> "*The term 'optimum', with respect to the yield from a fishery, means the amount of fish which--*
>
> *(A) will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems;*
> *(B) is prescribed as such on the basis of the maximum sustainable yield from the fishery, as reduced by any relevant economic, social, or ecological factor; and*
> *(C) in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the maximum sustainable yield in such fishery*."

Thus, the Act is another example of the general policy shift in the treatment of F_{MSY} from being a target that should be achieved on average, to being an upper limit that should rarely be exceeded.

3.1.2 F_{MSY} Proxies

F_{MSY} proxies are necessary in situations where there is insufficient knowledge, either due to lack of data or to other sources of uncertainty, that make the estimates of F_{MSY} too unreliable to be applied in management. This concept is perfectly identified in the Tier system of the Status Determination Criteria for the GOA and BSAI FMPs in which F_{MSY} estimates are only used in the information-rich tier, and a series of proxies are used in the more data-poor tiers.

Restrepo *et al.* (1998) and Gabriel and Mace (1999) review a series of F_{MSY} proxies that have been advocated by various authors in the past, primarily based upon simulation studies. Some of the proxies used in the past include F_{max}, $F_{0.1}$ and F_{med}. But the class of reference points based on spawning potential ratios ($F_{\%SPR}$) has gained more prominence recently, first as reference points for recruitment overfishing and later as proxies for F_{MSY}. Values in the range $F_{20\%}$ to $F_{30\%}$ have been proposed as recruitment overfishing thresholds (Goodyear 1993; Rosenberg *et al.* 1994) while values in the range $F_{35\%}$ to $F_{40\%}$ have been proposed as F_{MSY} proxies (Clark 1991; Clark 1993; Mace 1994).

On the question of what value of $F_{\%SPR}$ should be used as an F_{MSY} proxy, Clark (1991) simulated a variety of life history types and concluded that $F_{35\%}$ was a reasonable proxy, unless recruitment presented strong serial correlation, in which case $F_{40\%}$ would be more appropriate (Clark 1993). However, a recent study by MacCall (2002) suggests that harvest policies that used $F_{35\%}$ to $F_{40\%}$ as targets may have been "too aggressive" for several groundfish stocks off the west coast of the U.S. Furthermore, Clark (2002) suggested that it may be necessary to have targets of $F_{50\%}$ to $F_{60\%}$ for stocks with low resilience in order to maintain a proper balance between average yields and average abundance. Here, "resilience" refers to a stock's capability to recover from overfishing. Long-lived stocks that are characterized by an old age at first maturity—such as many rockfish—have low resilience.

There is also the question of what F_{MSY} proxies should be used for other non-groundfish species in the groundfish FMPs such as squid or octopus. However, we are not aware of any studies that recommend alternatives for these species.

It is difficult to evaluate the appropriateness of a specific F_{MSY} proxy for a specific stock because such evaluation requires the analyst to make assumptions about key population parameters (e.g., the stock-recruitment relationship) that will determine the outcome of the evaluation. For the most part, the guidance that has been provided has been generic and based on simulating hypothetical life history types. Nevertheless, the current scientific reasoning can be summarized by the advice on default F_{MSY} proxies provided by Restrepo *et al.* (1998):

- $F_{30\%}$ for stocks with high resilience
- $F_{35\%}$ for stocks with "average" resilience
- $F_{40\%}$ for stocks with moderate to low resilience
- $F_{50\%}$ to $F_{60\%}$ for stocks with very low productivity (such as rockfish and most elasmobranches).

In cases where there is so little information about a stock's population parameters that it is not possible to estimate spawning potential ratios, the options for using proxies are very few. The natural mortality rate (M) or a fraction of M, have been advocated as proxies for F_{MSY}. Thompson (1993) suggested that F=0.8M could provide reasonable protection against overfishing, and Deriso (1987) showed that M was approximately equal to $F_{0.1}$, a reference point that is advocated as an F_{MSY} proxy when selectivity and maturity schedules coincide.

Collie and Gislason (2001) showed in a multispecies context that commonly used biological reference points, including F_{MSY}, $F_{0.1}$, $F_{40\%}$, B_{MSY}, and $B_{40\%}$, are much more sensitive to changes in natural mortality (i.e., predators) than to growth changes (i.e., prey). They recommend for a species that is primarily a prey item, that conservative BRPs must be conditioned on the level of predation. For a species that is primarily a predator, the usual reference points are amenable to conservation needs.

3.1.3 $F_{\%SPR}$ and other Proxies in the BSAI and GOA Fishery Management Plans

Six Tiers are used to determine the overfishing level (OFL) and the maximum Allowable Biological Catch (ABC) for North Pacific groundfish stocks (as explained in chapter 2). These Tiers are harvest control rules in which the OFL definitions set the absolute maximum harvest levels, while the maximum ABC definitions (maxABC) set maximum intended harvest levels. At least for Tier 1, the difference between the maxABC and OFL levels is a function of uncertainty. This within-tier link to uncertainty is not explicit in other Tiers, but the concept of a safety buffer between OFL and ABC remains. Since the Tiers themselves are arranged in order of uncertainty (higher numbered Tiers have less information available), there should ideally be an increased safety buffer between OFL and ABC in moving from one Tier to the next higher numbered one. Whether this between-tier link to uncertainty results in increased conservatism for higher numbered Tiers has not been analyzed.

Tiers 3 and 4 (in which the majority of the assessed stocks have been categorized--see Table 3.2) make use of $F_{35\%}$ and $F_{40\%}$ to determine upper limit and default target fishing mortality rates, respectively. A simplistic interpretation of this system is that $F_{35\%}$ is being used as the default proxy for F_{MSY}, while $F_{40\%}$ is used as an estimator of a target F that is safely below F_{MSY}.

For the most part, the $F_{35\%}$ level as a proxy for F_{MSY} is in line with the values suggested in the literature (see the previous section). However, it should be noted that direct comparisons with literature studies are difficult to make for Tier 3 because the OFL and ABC control rules are not constant-F strategies. In these control rules, fishing mortality decreases linearly with stock size if the biomass falls below a threshold equal to $B_{40\%}$ (the B_{MSY} proxy). In contrast, the simulation studies mentioned in the previous section evaluated harvest rates that were kept constant, even when the simulated populations reached a low size. While average long-term yields may be similar in simulations using both shapes of control rules, it is likely that the average biomasses will differ. All else being equal, the control rules in Tier 3 are more conservative than the strategies analyzed by Clark (1993) and others and labeled as $F_{35\%}$ or $F_{40\%}$. For a more complete evaluation of the performance of Tier 3, it is recommended that the simulation study of Clark (1993) be carried out applying the F_{OFL} and F_{ABC} harvest rates of Tier 3.

The tier system in the groundfish FMPs is a blanket system that covers all stocks in the two Plans without making allowances for the diversity in life-history types present. As suggested by Clark (2002), $F_{35\%}$ harvest rates may not be sufficiently conservative for stocks with very low productivity, such as rarely-recruiting and long-lived rockfish species. Lower rates, on the order of $F_{50\%}$ to $F_{60\%}$, may be more appropriate to balance yield and conservation objectives for such species. Another potential problem has to do with stock complexes. Because productivity of each species in the complex is likely to be different, a single $F_{\%SPR}$ proxy will not perform equally well for all stocks in the complex.

The OFL values that are set according to Tiers 5 and 6 seem reasonable as conservative estimates of F_{MSY} levels in data-poor situations. While it may be possible to set up simple simulation studies to evaluate the performance of Tier 5 and 6 proxies, it is better to improve the general knowledge about these stocks in order to facilitate their classification into more data-rich tiers.

3.2 Conclusions about the MSY Proxies

The $F_{35\%}$ and $F_{40\%}$ proxies for MSY used in the groundfish FMPs are defensible, for this purpose, in that these values are supported by a body of scientific literature as being reasonable F_{MSY} proxies for "typical groundfish" species. However, the Council should be aware that harvests taken at these levels may be too high for species that have very low productivity and that are characterized by highly episodic recruitment. The Tier system could improve if allowances were made for the different life history types covered by the FMPs.

3.3 Consistency with MSFCMA

The MSFCMA requires FMPs to contain conservation and management measures that are consistent with the Act's National Standards. This section deals primarily with National Standard 1, which requires the avoidance of overfishing while achieving the optimum yield (see Section 3.1). The MSFCMA also requires the Secretary of Commerce to establish advisory guidelines based on the National Standards to assist in the development of FMPs (these are known as the National Standard Guidelines, NSGs). Because the NSGs contain far more details and guidance than the Act does, the comments on this section address the issue of consistency with the MSFCMA primarily from the NSGs' perspective. Therefore, noting that the NSGs "do not have the force and effect of law", the conclusions herein should not be taken as judgment on the legal compliance of the groundfish FMPs with the MSFCMA.

3.4 Status Determination Criteria

In referring to National Standard 1 of the MSFCMA, the NSGs require FMPs to define Status Determination Criteria (SDC) in order to determine if stocks are being subjected to overfishing or if stocks are in an overfished state:

> "*Each FMP must specify, to the extent possible, objective and measurable status determination criteria for each stock or stock complex covered by that FMP and provide an analysis of how the status determination criteria were chosen and how they relate to reproductive potential. Status determination criteria must be expressed in a way that enables the Council and the Secretary to monitor the stock or stock complex and determine annually whether overfishing is occurring and whether the stock or stock complex is overfished. In all cases, status determination criteria must specify both of the following:*
>
> "*(i) A maximum fishing mortality threshold* (MFMT) *or reasonable proxy thereof. The fishing mortality threshold may be expressed either as a single number or as a function of spawning biomass or other measure of productive capacity. The fishing mortality threshold must not exceed the fishing mortality rate or level*

associated with the relevant MSY control rule. Exceeding the fishing mortality threshold for a period of 1 year or more constitutes overfishing.
"(ii) A minimum stock size threshold (MSST) *or reasonable proxy thereof. The stock size threshold should be expressed in terms of spawning biomass or other measure of productive capacity. To the extent possible, the stock size threshold should equal whichever of the following is greater: One-half the MSY stock size, or the minimum stock size at which rebuilding to the MSY level would be expected to occur within 10 years if the stock or stock complex were exploited at the maximum fishing mortality threshold specified under paragraph (d)(2)(i) of this section. Should the actual size of the stock or stock complex in a given year fall below this threshold, the stock or stock complex is considered overfished.*"

3.5 The Maximum Fishing Mortality Threshold

The OFL definitions in the North Pacific Groundfish Tier system equate to the Maximum Fishing Mortality Threshold (MFMT) definitions required by the NSGs. The following observations can be made with respect to the specific requirements stated in the NSGs:

- The OFL levels are objective and measurable in all Tiers.
- The OFL levels are expressed in a way that enables the Council and NMFS to decide if overfishing takes place in any given year.
- The OFL is expressed as a single number in Tiers 4, 5 and 6, and as a function of biomass in Tiers 1, 2 and 3.
- The OFL level in Tiers 1 and 2 does not exceed F_{MSY}. According to the current scientific thinking, the OFL level in Tiers 3 to 6 probably does not exceed F_{MSY}, with the possible exception of stocks with very low productivity (see Section 3.1).

Therefore, the MFMT definitions given by the OFL values in the Tier system are consistent with the NSGs. In addition, it should be reiterated that the shape of the control rules in Tiers 1 to 3 require that the limit fishing mortality be reduced when stocks fall below B_{MSY} (or its proxy). This is a conservative feature.

3.5.1 The Minimum Stock Size Threshold

The Tier system used by the groundfish FMPs has no explicit definition of Minimum Stock Size Threshold (MSST) and, therefore, one would conclude that the Plans are inconsistent with this aspect of the NSGs. But this conclusion has to be examined in a larger context in order to understand its relevance.

The reasons for not including an explicit definition of MSST in the FMP were explained in a May 10, 2000, memorandum from the Council to NMFS. In it, the Council argues that the NSGs' requirement for an MSST definition is more of a suggestion from NMFS than a requirement of the law (MSFCMA). The memorandum also highlights some of the scientific and logistical difficulties that the Council has in defining an MSST, some of which are rephrased below. The Council argues that:

a. If a stock falls below an absolute biomass limit, it is not necessarily due to fishing pressure. Thus, there are difficulties in relying solely on the position of biomass with respect to an MSST for the purpose of classifying stocks as being in an overfished condition.
b. The requirement MSST $\geq B_{MSY}/2$ is arbitrary and based on outdated scientific concepts.
c. If projections are made to determine whether a stock can recover to B_{MSY}, the projections should be made using the target harvest strategy. Because the target strategy (ABC) is more conservative than the limit one (MFMT) in the groundfish FMPs, it may be unfair to make projections based on the limit and to then conclude that the stock cannot recover under the existing policy.
d. The MSST and target control rules for Tiers 1 to 3 already include an implicit accelerated rebuilding program because F is reduced as a stock falls below B_{MSY}. The fact that the NSGs also require that an FMP be amended to include a rebuilding plan when a stock falls below MSST would create unnecessary complications in a situation where rebuilding plans are already in place.

All of the issues raised by the Council are important and largely valid from a single-species perspective, but with respect to (a) we would point out that regardless of why a stock has fallen below a limit, its current stock size will affect the level of fishing mortality that it can sustain. Further, one should not leave aside ecosystem considerations completely. For example, with regards to the Council's difficulty labeled (b), above, it is true that B_{MSY} may be very low for stocks that have little or no relationship between recruitment and parental stock size. Perhaps this does not matter from a single-species perspective but very low levels of stock could have quite significant impacts on the ecosystem.

Considering that the Council's FMPs are conservative in establishing some buffer between the intended target (ABC) and the maximum limit (OFL or MFMT) and that they also provide for further reductions of the fishing mortality rate in Tiers 1-3, in case biomass falls below B_{MSY}, the absolute biomass limit defined by an MSST is probably not essential for the protection of the stocks. On the other hand, the NSGs' requirement that FMPs include both an MFMT and an MSST seems to be a useful nation-wide policy intended achieve some level of conservatism by avoiding potential problems in FMPs that do not distinguish between the target and the limit, and avoiding problems in FMPs that do not incorporate an automatic rebuilding program (neither of which is the case with the North Pacific groundfish FMPs).

Thus, from a single-species perspective, the lack of an explicit MSST definition in the North Pacific groundfish FMPs is more of a bureaucratic problem than a real conservation danger.

It should be noted that the SAFE reports for stocks that have a quantitative age-structured or length-structured assessment do include a set of projections made to address the question of whether or not the stock is below the MSST as it would be defined by a literal reading of the NSGs. The projections are made based on assumptions that the analysts deem to be appropriate and include the options implied by the NSGs to determine status relative to MSST. However, these assumptions do not generally contemplate pessimistic scenarios such as depensatory recruitment at low stock sizes or negative serial correlation between successive recruitment events. None of the 6 stocks examined in the 2001 GOA SAFE reports or the 10 stocks examined in the 2001 BSAI SAFE reports were estimated to be below MSST.

3.6 Optimum Yield

OY is defined in the BSAI FMP as being 85% of the overall MSY for the entire species complex, plus incidental catches of non-specified species (see chapter 2). The last overall MSY estimate of 1.7-2.4 million t was made on the basis of average 1968-1977 catches. Thus, OY is defined as 1.4-2.0 million t. The FMP attributes the 15% deviation from MSY to the influence of various biological and socioeconomic factors. Among the biological factors, it is argued that estimates of exploitable biomass for the complex are in the order of 9.0 million tons, which might support catches greater than 2.0 million tons which is then a conservative limit that would allow for multi-species interactions.

In the GOA FMP, OY is defined as 116,000-800,000 t (see chapter 2). The lower end corresponds to the lowest groundfish catches observed during the period 1965-1985 (116,053 t were caught in 1971, a year with low catches of pollock, Pacific cod and Atka mackerel), while the upper end corresponds approximately to the mean MSY estimate for the five-year period 1983-1987. The FMP states that "the OY range was selected in consideration of the volatility in pollock and flounder ABC, and the potential for harvesting at MSY."

The MSFCMA requires that OY be equal to or lower than MSY taking into consideration relevant social, economic and ecological factors. In the case of the BSAI FMP, there is an explicit reduction from the overall MSY, but such a relationship is not obvious or explicit for the GOA FMP.

A comparison is made below between the combined 1999 F_{OFL} catches, OY and the realized catches:

Table 3.1. 1999 F_{OFL} equivalent catch, OY and actual catch

(in tons)	**OY**	**OFL**	**Yield**
BSAI	1,400,000-2,000,000	3,719,391	1,424,765
GOA	116,000-800,000	778,890	227,614

The OFL catches, which are theoretically the catches that would be taken under the F_{MSY} limit control rule defined by the Council are considerably larger than OY for the BSAI. However, for the GOA, the OFL catch level is within the OY range. This comparison, albeit a crude one, suggests that the upper end of the OY range is close to MSY for GOA. In contrast, the BSAI yield is near the low end of the range and well below OFL, suggesting that management actions have been keeping catches well below the typical MSY level. A comparison between realized yields and OY in the above table indicates that the optimum yield, as defined by the Council, was achieved, at least in 1999.

The MSFCMA also requires that the OY definition take into account the protection of marine ecosystems. The BSAI FMP asserts that ecosystem considerations have been taken into account qualitatively, although the 15% deviation from MSY appears to be arbitrary. Any linkage to ecosystem considerations in the GOA FMP is even less obvious.

Thompson (1998) reviewed the OY definitions in the context of the MSFCMA. In addition to the language in the FMPs, he also examined several working documents that led to the current definitions such as Plan Amendments and Environmental Impact Statements. He concluded that the GOA FMP specification failed to address explicitly the protection of marine ecosystems and that it may fail to ensure that OY ≤ MSY. For the BSAI FMP, Thompson (1998) concluded that the OY specification addressed ecosystem considerations to an unknown degree and that it most likely ensured that OY ≤ MSY. On the basis of his analyses and the fact that the OY definitions date back to the late 1980s, he concluded that both BSAI and GOA definitions of OY should be reanalyzed.

In conclusion, the Council should consider a review of the OY definitions for both FMPs so that they are consistent with the MSFCMA in a more explicit way. A possible reduction in the upper range of the GOA definition of OY should receive priority, although it may be cost-effective and advisable from the point of view of internal consistency to address both FMPs together.

3.6.1 Other Conservation Aspects and TACs in Practice

There are other aspects to the FMPs and decision-making process that should be taken into consideration when evaluating the degree of conservatism inherent in the management of North Pacific groundfish. The Plans include:

- Potential closures of areas to directed fishing for a species whose remaining TAC is needed as bycatch in other fisheries.
- Potential for in season adjustments to allowable gears or TACs.
- A limited entry program for the groundfish fishery as a whole and Individual Fishing Quota arrangements for some species.
- Prohibition of some fishing practices such as roe-stripping of pollock.
- Bycatch measures to reduce potential adverse impacts on depleted crab resources.
- Measures to limit the bycatch of halibut, herring and salmon.
- Measures to reduce fishery interactions with marine mammals.

In addition to these measures, there are strict reporting requirements and an observer program (100% coverage for vessels over 125 ft, and 30% coverage for vessels 60-124 ft) which make it possible to implement the measures within a season.

In evaluating a management policy it is important to see how it is implemented in practice, not only in theory. Table 3.2 presents an example of the results of the decision-making process by stock. The table shows 2001 catches under the OFL and maxABC definitions of the 6-Tier system, ABC levels after adjustment by the Plan Teams or the Council's SSC, and the TAC set by the Council. In addition, the table shows the average 1997-2000 ratio between realized catches and TACs.

Assuming that 2001 was a typical year for the current management system, the median (and range) results in Table 3.2 indicate that:

- By design, maximum target catches (maxABC) are about 80% (60-90%) of the overfishing specification.
- The maxABC upper targets are often modified downwards before being submitted to the Council for decision, with the resulting ABCs being about 75% (10-85%) of the overfishing specification.
- The TACs decided upon by the Council are such that recommended ABCs are often subject to more downwards revisions, with the resulting TACs being about 65% (0-85%) of the overfishing specification.
- In addition, the last column in Table 3.2 shows that catches exceed TACs very rarely. The median ratio of realized catch to TAC is about 60% for both GOA and BSAI. The range in the ratios for both areas is 21-112%, with the higher value coming from the GOA sablefish fishery where catches exceeded TACs in three of the four years examined.

Overall, target catches, as measured by TACs, are set very conservatively, from a single-species/target-stock standpoint, and they are implemented conservatively from this same standpoint.

3.7 Conclusions about Single-species Aspects of the OY levels

The management system contained in the groundfish FMPs is generally consistent with the single-species/target-stock components of the MSFCMA in a conservative way. While the FMPs specify only one of the two status determination criteria that are required by NMFS' National Standard Guidelines, the FMPs are sufficiently conservative, with respect to the target stocks evaluated from a single-species perspective, and incorporate automatic rebuilding plans to such a degree, that this lack of conformity with the Guidelines should not pose a conservation danger from a single-species viewpoint.

In terms of Optimum Yield, there is uncertainty about the conformity of the FMP definitions with the MSFCMA. The Council should review and revise its OY specifications in order to make more explicit links with environmental considerations and to more directly specify the relationship between OY and MSY for GOA groundfish.

In a single-species/target-stock context, the TAC-setting process employed by the Council is a very conservative one and the in-season monitoring and management system seems adequate for implementing the TACs with little risk of exceeding them.

3.8 Robustness of the Current Harvest Strategy

In Section 3.2 we presented an overview of the consistency of the NPFMC management system with MSFCMA. In particular, we looked at whether the current Tier system for setting ABCs complies with National Standard 1 of the MSFCMA. In this section we consider the harvest strategy from a somewhat different perspective. We ask how robust the current harvest strategy is to uncertainties in data, models, assumptions, and application of the Tier rules. It also proposes a way in which the robustness of the current harvest strategy can be more formally evaluated, using the simulation testing approach introduced in chapter 2.

3.9 Previous Evaluations of the Harvest Strategy

For each species for which ABCs are recommended there is a (generally) annual process of collecting data, undertaking some form of quantitative assessment of stock status, and then feeding that information into the Tier rules (the harvest control rules) to determine an upper limit for an ABC. This collection of activities (monitoring, assessment, application of control rule) is called a harvest strategy. The harvest control rules in the NPFMC harvest strategy are well defined, but the details of monitoring and assessment vary between stocks.

There are several ways in which a harvest strategy can be evaluated. Given that a harvest strategy has been in place for some time, an empirical approach might be taken – has the harvest strategy achieved its goals (e.g., no/few stocks overfished)? Although there have been changes in the detail of NPFMC harvest strategies over time (Section 2.12 of this report, and Witherell *et al.* 2000), it can be argued that the basic approach has delivered good outcomes with no groundfish stocks currently classified as overfished according to NMFS' Guidelines. As Section 3.2.3 shows, actual catches can be considerably below ABCs.

It is also possible to evaluate individual components of a harvest strategy. External reviews of monitoring strategies and stock assessments (e.g., Stokes review of EBS pollock) are examples of this approach. Thompson and others have presented reasoned arguments as to why the current (Tier) control rules should achieve the goals of MSFCMA with regard to stock protection (see references in Section 3.1 above). These arguments have been subject to external scrutiny, for example by NOAA Regional Office under EPA requirements, and by external consultants as part of the Programmatic Supplemental Environmental Impact Statement process (Deriso 2000).

While individual parts of a harvest strategy can be evaluated, as just indicated, it is also important to evaluate how all the components work together, and particularly whether the overall harvest strategy is robust to a range of uncertainties that may not be dealt with explicitly, for example in the stock assessment models used to draw inferences from the data. Despite the strength in the empirical argument mentioned above, many fisheries have sought to formally evaluate the robustness of their harvest strategies using the "management strategy evaluation" or "management procedure" approach (Cooke 1999). This approach can be used for several purposes: 1) to help identify prospective harvest strategies, 2) to select among alternative harvest strategies, and 3) to evaluate the extent to which an existing strategy is robust to various sorts of uncertainty. This third use is outlined in section 3.10, while the second use is discussed in Section 3.12.

3.10 Proposal for Management Strategy Evaluation

In its most general use, management strategy evaluation (MSE) involves assessing the performance of a range of (possibly adaptive) management strategies, and evaluating the tradeoffs across a range of management objectives (Smith *et al.* 1999). The approach involves explicitly testing the robustness of each strategy to a range of uncertainties (such as those listed in chapter 2 of this report). The method tests the performance of each strategy against a simulated "real" world, called an *operating model*. In the case where a single species harvest strategy is being evaluated, the operating model will include a model of the stock dynamics and the fishery as well as an "observation" model that simulates the monitoring process in the fishery. The operating model will usually seek to incorporate more of the perversities of the world than are generally included in stock assessment models. The data generated by the operating model are fed into the assessment model, which in turn feeds into the harvest control rule. The TAC that results from applying the harvest control rule to these inputs drives the fishery management decisions for the next year, and the application of the harvest strategy continues for a specified number of years. The performance is evaluated against the outcomes in the operating model. The MSE approach is illustrated in Figure 3.1. This approach captures (albeit in a simulation) all aspects of the application of a harvest strategy (monitoring, assessment, control rule and implementation), and differs from the types of projections that are often undertaken in a stock assessment, which assume some fixed sequence of catches or fishing mortality rates into the future, but which do not capture the feedback nature of the decision making process.

The MSE approach to evaluate single species harvest strategies has been used in two modes to date. The first has been to develop or evaluate harvest strategies for particular stocks (e.g., Punt and Smith 1999; Butterworth and Punt 1999). The approach has also been used to evaluate generic harvest strategies for a generic set of species (e.g., Punt 1995). The latter approach is the appropriate one to use in this instance, and (fortunately) has fewer requirements for model "conditioning". Indeed the aim here is to test the robustness of a harvest strategy that uses an assessment model that is quite different from the representation of the real dynamics in the operating model, which the latter should include many more of the complexities and error sources that function in the world. The potential benefit of an MSE approach to evaluating the NPFMC harvest strategy is to provide further confidence to decision makers and the public that the strategy is a robust one. Hilborn and Walters (1992) have argued that all harvest strategies should be tested in this way. Hilborn *et al*. (2000) in fact carried out a rudimentary MSE of this sort to analyze the performance of a management strategy that, like the BSAI/GOA FMP, is based on fishing mortality rate reference points, and concluded that there may be better alternatives to this family of control rules.

There are several steps and choices that have to be made in undertaking an MSE, and these are outlined in the rest of this section, which is concerned only with evaluating the current NPFMC harvest strategy.

3.10.1 Objectives and Performance Measures

Although a comparative MSE would be concerned with all management objectives (utilization as well as conservation), in this instance we are mainly concerned with how the current harvest strategy performs from a target-stock protection point of view. The focus of performance will, therefore, be on stock size and exploitation rate.

3.10.1.1 Spawning Stock Biomass Indicators:

These could include $B/MSST$, $B/B_{40\%}$, B/B_{MSY} and B/B_{init} where B is spawning stock biomass, and B/B_{init} is biomass relative to pre-fishing biomass. The reason for including the latter indicator is that both MSST and B_{MSY} potentially change as biological parameters of the stock change (e.g., under regime shift) and so B/B_{init} provides a "fixed baseline" indicator. This indicator is illustrated in section 3.12.

3.10.1.2 Exploitation Rate Indicators:

These could include F/OFL, $F/F_{40\%}$, and F/F_{TARG} where F_{TARG} is the F corresponding to maxABC. Both of these indicators will be dependent on stock size for Tiers 1-3 control rules. This indicator is illustrated in Section 3.12.

3.10.1.3 Performance Measures:

All of the above indicators are potentially defined for all years in the projection period and also need to be summarized across multiple projections. Turning time series of indicators into performance measures typically takes three forms: 1) evaluate only at the end point of the projections; 2) identify the worst case (minimum/maximum in the time series); 3) integrate across time (e.g., frequency that $B/MSST$ <1 or F/OFL >1, or averages (probably not relevant here)). It would also be interesting to examine trajectories of indicators over periods of change (e.g., regime shifts or changing selectivity).

3.10.1.4 Robustness Tests:

Selection of robustness tests is the key issue in evaluating an existing harvest strategy. The selection should reflect the major known or potentially consequential uncertainties in the system. For NPFMC groundfish, these would seem to include:

- Regime shift: how well does the harvest strategy cope with major (and potentially rapid) changes in underlying productivity of a species. Issues to consider here include which (life history) parameters are affected by regime shift (e.g., stock recruitment steepness, carrying capacity, natural mortality–note also possible interaction with spatial structure and selectivity);

- Spatial structure: most populations have internal spatial structure that may not be well captured in conventional stock assessment models (implying for example possible loss of local spawning units). Also, stock boundaries may not correspond to management boundaries (e.g., Russian catches of EBS pollock).
- Changing selectivity: this may result from changing fish behavior (e.g., due to regime shifts and "ecosystem" effects), as well as through changes in targeting practices or regulation induced changes in fishing. Some stock assessment models try to capture such effects by modeling selectivity as a random walk (e.g., pollock)–How well do they do it? What are the consequences of ignoring changing selectivity?
- Depensation: recruitment may decline at low stock levels more quickly than conventional models assume. This may reflect underlying changes in predation or productivity. Meta-analyses have provided some empirical evidence for such effects (Liermann and Hilborn 1997, but see also Myers *et al.* 1995). An alternative to modeling a depensatory stock recruitment relationship is to model time changing mortality M (e.g., Fu and Quinn 2000), which provides a useful middle ground between modeling single and multi-species effects.
- Survey catchabilities: application of the Tier system to many species requires absolute estimates of biomass, generally derived from most recent surveys. Absolute estimates apparently assume survey catchability (q) of one. This may be a precautionary assumption for many species but then again maybe it isn't. Harley and Myers (2001) provide some empirical evidence for survey q's. Estimates of survey q's from stock assessments where surveys are used as relative indices also provide "local" empirical evidence for some species. What is the distribution of possible q values (by species type?) and what are the implications of assuming q=1?
- Life history categories: we suggest that at least three species types be considered – cod-like (e.g., pollock), flatfish, and rockfish. The Council might also wish to see how some other species such as octopus, squid and mackerel fare under the current harvest strategy.

Other robustness tests could be considered (e.g., levels of observation and process error). However, we believe that the major "qualitative" effects listed above are likely to be more important than statistical details of sampling and incorporation of "stochastic" effects (such as recruitment variability), although the latter should not be ignored.

3.10.2 Choice of Operating Models

Operating models need to be sufficiently detailed to capture the effects in the range of robustness tests selected. Suitable operating models almost certainly exist within the suite of models developed by staff at AFSC. Existing single-species models could easily be modified to incorporate most of the robustness tests listed above (e.g., Thompson regime shift model?). Potentially the most difficult (or time consuming) modification is to incorporate various hypotheses about spatial structure, so perhaps these are best dealt with initially by indirect means such as changes in selectivity or assumptions about "unknown" catches. Although there might be some advantages in including a "fleet dynamic" (spatial effort allocation) sub-model, we do not see a need for this in the first instance (unless existing models can easily be adopted). For the "observation" model, some thought will be required to include the range of data types typically used in assessments, and various levels of bias and variance in such data. Assuming that AFSC staff undertake the MSE, they will be in the best position to judge the appropriate assumptions to make in this regard.

Although the initial focus should be on single species operating models, we were impressed by the range of models in use or under development at AFSC, a number of which could potentially be used as operating models in a second phase MSE. These models include a model dealing with by-catch and technical interactions (Ianelli pers comm.), and several models dealing with broader ecological interactions (Livingston, Hollowed, Geitches). Although a lot of work has evidently gone into the development of a range of modeling approaches in this area, there appears to be little or no use of these models in framing management advice for the BSAI/GOA FMP, at least with regard to ABCs. Their potential use as operating models is much less problematic than their direct use as assessment models, and they could play a useful role in extending the evaluation of harvest strategies beyond single-species considerations.

3.10.3 Choice of Assessment Models

Testing harvest strategies at Tier levels 1-4 implies the use of some sort of quantitative stock assessment model. The models used for some of the AFSC assessments are highly sophisticated "state of the art" Bayesian assessments, and several have been independently reviewed, as noted above. However it may not be computationally feasible to include very sophisticated assessment models in MSE analyses, and so simpler approaches are required. We suggest that as a first approximation, a Cagean-like assessment model, potentially incorporating changing selectivity, be utilized (Deriso *et al.* 1985), but other suitable methods could be considered. Tiers 5 and 6, which do not use an explicit assessment model, also need to be tested for performance.

3.10.4 Other Issues

Several other options/issues need to be considered for MSE evaluations. These include:

- Catch history and current depletion: We suggest a 20-year catch history (perhaps longer for regime shift tests) and a range of current depletion levels (10-70% in steps of 20).

- Projection period: This might have to vary with life history type (longer for rockfish) but we suggest at least 30 years.
- Parameters of operating model: Choose parameters appropriate to the three life history types (pollock, flatfish, rockfish). Parameters can be drawn from meta-analyses (e.g., steepness, survey q) and/or existing assessments (selectivities etc).
- Implementation of the harvest control rule: Although this review has noted that TACs (and catches) are often below ABCs, sometimes substantially so, the MSE in the first instatnce should assume that catch=ABC under the appropriate current Tier level. This is equivalent to a worst case scenario, since inadvertent overfishing due to uncertainties or errors in stock assessment are already dealt with in the MSE approach. Mimicking the way in which real TACs (and catches) are influenced by other factors such as by-catch limits and quotas on other species would require a much more complicated model.

3.10.5 Recommendations

We recommend that an MSE analysis, along the lines indicated above, be undertaken to provide additional assurance that the current NPFMC ABC harvest strategy is a robust one and is likely to continue to meet the objectives of MSFCMA and of NPFMC itself (noting that the actual harvest strategy is difficult to define except to say that it is ≤ABC). We recognize that an MSE analysis can be potentially a time-consuming and technically difficult undertaking. Sufficient resources in time and people would need to be allocated to undertake the work. The skills and expertise to undertake the work already reside within AFSC.

Many judgments of detail are required in undertaking an MSE analysis, and we are not in a position to be prescriptive about those details. However, we offer the following suggestions about priorities and sequence.

1. Undertake a "Tier 1" evaluation in the first instance, using a single species operating model, across all three species groups.
   - Priorities for robustness tests are: regime shift, the "simple" approach to spatial structure, survey catchability.
2. Compare performance across Tiers, noting particularly whether the intention of "increasing precaution" is achieved in practice.
   - Note that an "empirical" test across Tiers could be done very quickly by taking an existing Tier 1 assessment (EBS pollock) and seeing what ABC would be recommended under each lower Tier level.
3. Expand the operating model to include more detailed and realistic technical interactions between species. Repeat Tier 1 analysis.
4. Evaluate the potential to use some of the multi-species models developed at AFSC and NEFSC as operating models to test performance of current harvest strategies. These could include models such as Ecopath, MSVPA, and Bormicon.

3.11 Possible Modifications to the Current Harvest Strategy

Section 3.10 discussed how the current NPFMC harvest strategy could be tested for robustness using MSE methods. This section will briefly discuss alternatives to the current strategy and how they might be evaluated.

3.11.1 Alternative Harvest Strategies

- Alternatives to $F_{40\%}$: Section 3.1 noted that $F_{40\%}$ may be too high a harvest rate for some species or groups of species. Alternative values should be evaluated for these groups.
- Form of harvest control rule: The location of thresholds in the current harvest control rules could be altered (e.g., value of biomass threshold α at which zero ABCs are set; use of B_{MSY} as a breakpoint in Tiers 1-3). Note that to speed up the "search" for improved values, the utility function approach suggested and previously used by Thompson (ref) might be used to identify candidate control rules. These should then be further evaluated using the MSE approach.
- Inputs to the harvest control rules: Different Tiers use different inputs to the control rules (estimates of biomass, reference values of F, use of various moments of distributions, use of absolute estimates of biomass direct from surveys, etc). These inputs may be thought of as "indicators" that derive from the stock assessment process. The indicators appear to be sensible at the higher Tier levels (1-4), but alternatives could be explored at the lower Tier levels, making more use of information that is almost certainly available and often used in a "qualitative" way by stock assessment groups (changes in spatial distribution, trends in catch rates and length or age composition, etc). Control rules based on a suite of indicators could be evaluated.
- Constraints on inter-annual changes in ABC: Some management procedures (e.g., in South Africa) incorporate constraints to changes in TAC from year to year, as well as upper and lower caps to TACs. The NPFMC may or may not want to consider such approaches, which can have some benefits to the fishing industry by stabilizing catches.
- Multi-annual catch limits: MSE methods have been used to evaluate the costs and benefits of annual versus multi-annual TAC setting (e.g., Punt *et al.* 2001). Some work along these lines has already been done in the NPFMC setting, because NPFMC is considering a Plan Amendment to change the TAC-setting process. Such modeling might be easily adaptable to an operating model.

There is obviously a wide range of alternative harvest strategies that might be considered, and MSE methods are a useful way to design and evaluate alternatives. If this "comparative" approach is used, a wider set of performance measures, including utilization as well as conservation objectives, should be evaluated and the tradeoffs across objectives highlighted. We suggest that wider stakeholder discussion (via NPFMC?) on alternative approaches be held before embarking on a major exercise to evaluate alternatives.

3.11.2 *Species-specific Harvest Strategies (priority species, low data species)*

Apart from exploring and evaluating generic harvest strategies, several of the target species in the BSAI/GOA groundfish fishery are of sufficient value (and importance) to warrant the effort to formally evaluate species-specific harvest strategies (e.g., for pollock). This would allow more of the detailed knowledge and understanding about these species and associated fishery to be incorporated in the operating models, and potentially lead to better performance of harvest strategies for those species. It would also allow changes to harvest strategies that occur for other reasons to be more formally evaluated. An example is the recent change to the pollock harvest control rule to set zero ABCs if the stock falls below the MSST. This change was brought in because of concerns about food chain impacts of the fishery on Steller sea lions.

3.12 Conclusions about Single-species Requirements

Overall, the current NPFMC approach to advising on ABCs appears to meet the requirements of MSFMCA, from a single-species/target-stock management perspective for most of the target stocks (the exceptions are primarily the rockfish). Evidence for conservatism in the strategy can be seen in graphs of the fishing mortality and biomass indicators in Figure 3.1 (for BSAI pollock from 1964 to 2001) and Figure 3.2 (for species in the GOA and BSAI in 2001). Ideally, one would like to see values in the lower right quadrant of the graph. For BSAI pollock (Figure 3.1), the ideal situation has occurred in most years since 1983, after the strong 1978 yearclass recruited to the fishery. Periodic strong recruitments since then have occurred, and the groundfish fishery has been managed within its targets (F/F40 < 1). For species in 2001 (Figure 3.2), only Pacific cod in the BSAI and sablefish, pollock, and POP in the GOA were to the left of the B/B40 = 1 line and none are very far away (i.e., more than 20%). GOA POP in 2002 are to the right of the line. Fishing mortality for all species is at or below the F/F40 = 1 line, indicating that the various fisheries are being managed at or below the $F_{40\%}$ level. We recommend that analysts provide similar graphs in the SAFE documents for each species over its history, and Summarized by all species for the most recent times.

Other conservative elements in the current NPFMC approach derive from the additional constraints in the overall management system that often result in catches well below ABCs. Nevertheless, we recommend that additional work be undertaken to more formally test the robustness of the current NPFMC harvest strategy to various uncertainties, and to explore alternative harvest strategies that may be more appropriate for some groups of species or individual species. Existing staff at AFSC have the expertise and a range of suitable models to undertake the MSE approach suggested, but time and resources will need to be allocated for such a task.

Table 3.2. Example of the Decision-making Process for North Pacific Groundfish

GOA FMP	Tier	OFL [1,3]	maxABC [1,4]	ABC [1,5]	TAC [1,6]		maxABC / OFL	ABC / OFL	TAC / OFL	Catch [2,7] / TAC
Pollock	3	84,090	64,110	58,250	58,250		0.76	0.69	0.69	0.94
Pacific cod	3	77,100	65,200	57,600	44,230		0.85	0.75	0.57	0.98
Deepwater flatfish	5-6	6,430	4,880	4,880	4,880		0.76	0.76	0.76	0.35
Rex sole	5	12,320	9,470	9,470	9,470		0.77	0.77	0.77	0.36
Flathead sole	5	29,530	22,690	22,690	9,280		0.77	0.77	0.31	0.21
Shallowwater flatfish	4-5	61,810	49,550	49,550	20,420		0.80	0.80	0.33	0.25
Arrowtooth flounder	3	171,060	146,260	146,260	38,000		0.86	0.86	0.22	0.49
Sablefish	3	19,350	15,760	12,820	12,820		0.81	0.66	0.66	1.12
Pacific Ocean perch	3	15,670	13,190	13,190	13,190		0.84	0.84	0.84	0.87
Shortraker/rougheye	4-5	2,340	1,910	1,620	1,620		0.82	0.69	0.69	0.97
Other rockfish	4-5	6,610	5,160	5,040	990		0.78	0.76	0.15	0.31
Northern rockfish	3	5,910	4,980	4,980	4,980		0.84	0.84	0.84	0.74
Pelagic shelf rockfish	4-5	8,220	6,620	5,490	5,490		0.81	0.67	0.67	0.63
Thornyhead	3	2,330	1,990	1,990	1,990		0.85	0.85	0.85	0.66
Demersal shelf rockfish	4	480	430	350	350		0.90	0.73	0.73	0.59
Atka mackerel	6	6,200	4,700	600	600		0.76	0.10	0.10	0.40
						min.	***0.76***	***0.10***	***0.10***	***0.21***
						med.	***0.81***	***0.76***	***0.68***	***0.61***
						max.	***0.90***	***0.86***	***0.85***	***1.12***
BSAI FMP										
Pollock BS	1	3,530,000	2,110,000	2,110,000	1,485,000		0.60	0.60	0.42	0.99
Pollock AI	5	31,700	23,800	23,800	1,000		0.75	0.75	0.03	2.89[8]
Pollock Bogoslof	5	46,400	34,800	4,310	100		0.75	0.09	0.00	0.10
Pacific cod	3	294,000	253,000	223,000	200,000		0.86	0.76	0.68	0.92
Sablefish BS	3	2,900	2,370	1,930	1,930		0.82	0.67	0.67	0.39
Sablefish AI	3	3,850	3,140	2,550	2,550		0.82	0.66	0.66	0.39
Atka mackerel BSAI	3	82,300	71,300	49,000	49,000		0.87	0.60	0.60	0.82
Yellowfin sole	3	136,000	115,000	115,000	86,000		0.85	0.85	0.63	0.54
Rock sole	3	268,000	225,000	225,000	54,000		0.84	0.84	0.20	0.43
Greenland turbot	3	36,500	32,400	8,100	8,000		0.89	0.22	0.22	0.79
Arrowtooth flounder	3	137,000	113,000	113,000	16,000		0.82	0.82	0.12	0.40
Flathead sole	3	101,000	82,600	82,600	25,000		0.82	0.82	0.25	0.33
Other flatfish	3	21,800	18,100	18,100	3,000		0.83	0.83	0.14	0.23
Alaska plaice	3	172,000	143,000	143,000	12,000		0.83	0.83	0.07	0.23[9]
Pacific Ocean perch	3	17,500	14,800	14,800	14,800		0.85	0.85	0.85	0.83
Northern rockfish	5	9,020	6,760	6,760	6,760		0.75	0.75	0.75	0.92[10]
Shortraker/rougheye	5	1,369	1,028	1,028	1,028		0.75	0.75	0.75	0.92[10]
Other rockfish BS	5	482	361	361	361		0.75	0.75	0.75	0.61
Other rockfish AI	5	901	676	676	676		0.75	0.75	0.75	0.61
Squid	6	2,620	1,970	1,970	1,970		0.75	0.75	0.75	0.40
Other species	5-6	78,900	59,200	39,100	30,825		0.75	0.50	0.39	0.78
						min.	***0.60***	***0.09***	***0.00***	***0.10***
						med.	***0.82***	***0.75***	***0.60***	***0.61***
						max.	***0.89***	***0.85***	***0.85***	***2.89***

[1] For year 2002.
[2] Average for the period 1997-2000.
[3] Projected catch at the limit F_{OFL} in the Tier system.
[4] Projected catch at FABC in the Tier system.
[5] ABC after adjustment by Plan Team or by the Scientific and Statistical Committee.
[6] Total allowable catch set by the Council.
[7] Average ratio of realized catch relative to TAC.
[8] In 1999, the extremely low TAC of 109 t was exceeded by 956 t. If this point is excluded, the average is 0.90.
[9] Alaska plaice was managed as part of the “other flatfish” in 1997-2000 (i.e., it did not have a separate TAC).
[10] Northern, shortraker, and rougheye rockfish shared a common BS TAC in 1997-2000, so these have been pooled.

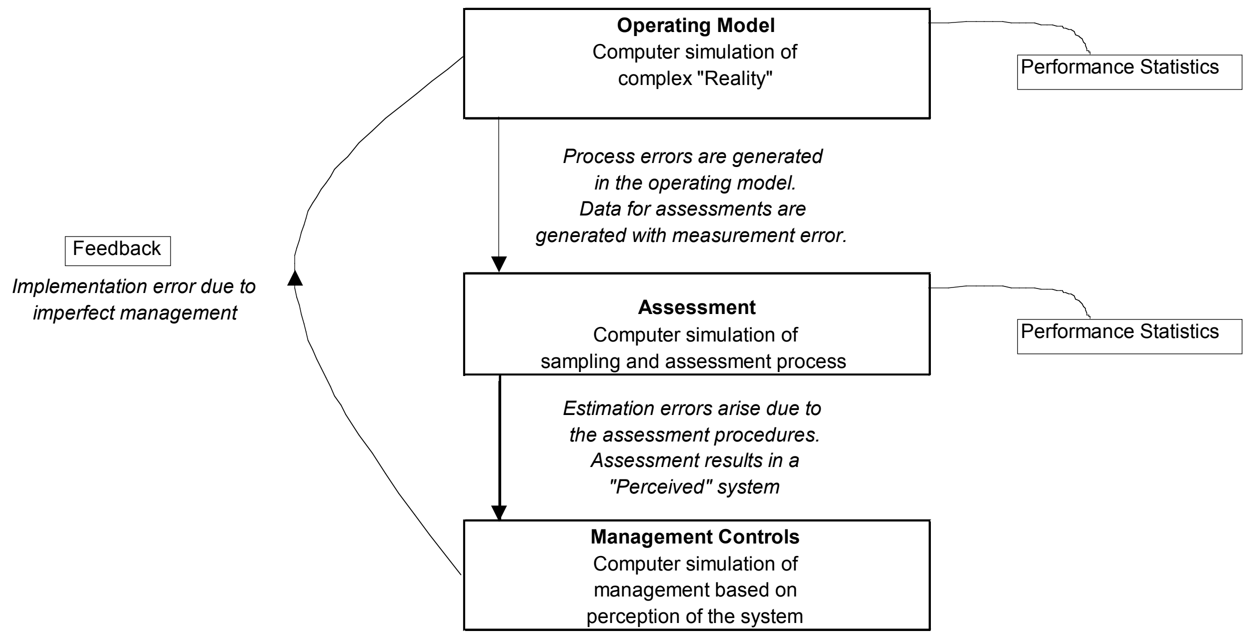


Figure 3.1. Outline of the MSE approach.

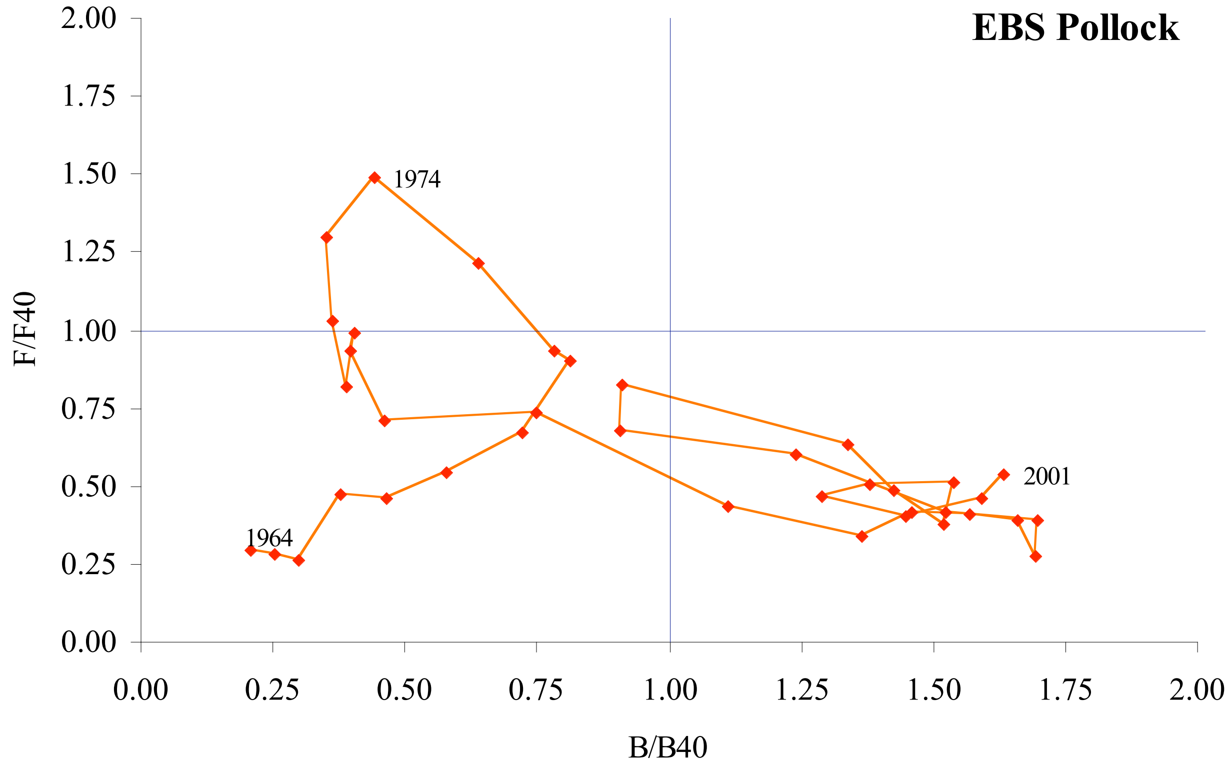
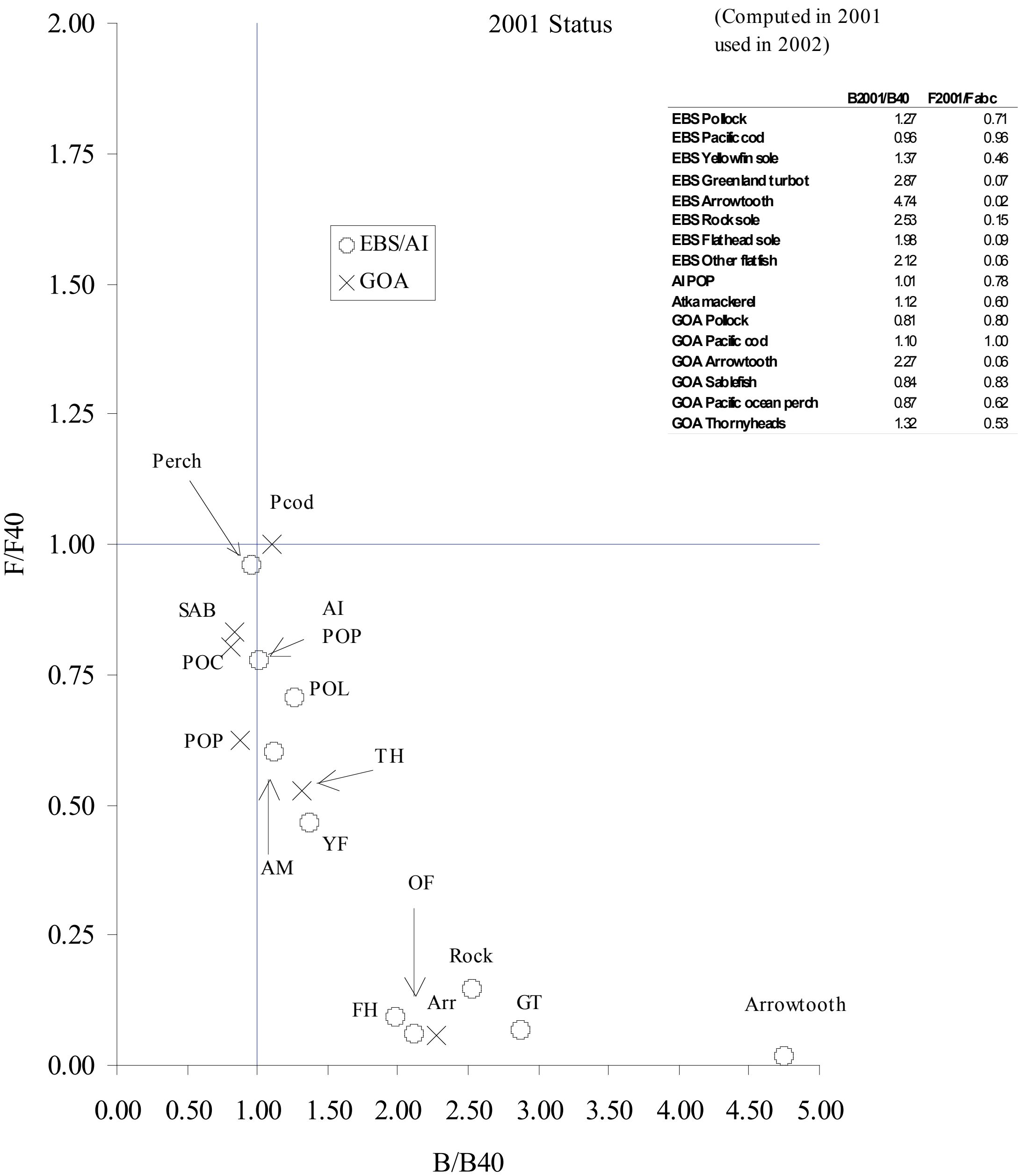


Figure 3.2. $F/F_{40\%}$ versus $B/B_{40\%}$ for BSAI walleye pollock, 1964–2001. Values in the lower right quadrant are considered ideal (population above the target, fishing mortality below the target).



	B2001/B40	F2001/Fabc
EBS Pollock	1.27	0.71
EBS Pacific cod	0.96	0.96
EBS Yellowfin sole	1.37	0.46
EBS Greenland turbot	2.87	0.07
EBS Arrowtooth	4.74	0.02
EBS Rock sole	2.53	0.15
EBS Flathead sole	1.98	0.09
EBS Other flatfish	2.12	0.06
AI POP	1.01	0.78
Atka mackerel	1.12	0.60
GOA Pollock	0.81	0.80
GOA Pacific cod	1.10	1.00
GOA Arrowtooth	2.27	0.06
GOA Sablefish	0.84	0.83
GOA Pacific ocean perch	0.87	0.62
GOA Thornyheads	1.32	0.53

Figure 3.3. $F/F_{40\%}$ versus $B/B_{40\%}$ for species in the GOA and BSAI in 2001. Values in the lower right quadrant are considered ideal (population above the target, fishing mortality below the target).

4 MULTISPECIES AND ECOSYSTEM ISSUES

The third component of our charge involved a general consideration of ecosystem effects. We begin with an overview of ecosystem approaches and considerations, including a discussion of the interpretation of the charge. We then discuss the importance of spatial scale and uncertainty, which lead to the potential for *implicit* and *explicit* consideration of ecosystem aspects in management. We further subdivide explicit ecosystem approaches according to *tractable* problems and *less tractable* problems associated with the effects of fishing. We then emphasize the importance of modeling as a tool for understanding ecosystem effects of fishing. With this background, we are able to evaluate current management in the Eastern Bering Sea and Gulf of Alaska in the context of ecological and ecosystem considerations. We conclude by offering a "road map" towards explicit inclusion of ecological and ecosystem considerations in fishery management in the Eastern Bering Sea and Gulf of Alaska.

4.1 An Overview of Ecosystem Approaches

4.1.1 Interpreting the Charge

The third point of the charge in the terms of reference is

> "Is the approach considerate of ecosystem needs in the BSAI and GOA?"
>
> - If not, how should it be changed?
> - Are sufficient data available to allow implementation of the alternative approach?
> - How would the transition from the current approach to the proposed revised one be handled?"

We had considerable discussion about the phrase "ecosystem needs", which suffers, to borrow a phrase from Churchill, from terminological inexactitude. That is, although the meaning seems to be apparent on the surface, the meaning of the phrase "ecosystem needs" must reflect both the social construction of what conditions we want to prevail in an ecosystem and the scientific judgment of how management can ensure or encourage that these conditions will materialize. While there are opinions circulating on both counts, there is not a well developed consensus.

Regarding "needs", we found it more helpful to consider the needs of the species that are part of the ecosystem rather than the needs of the ecosystem itself. The "needs of a species" are those attributes of the biological and physical systems that allow the species to persist in a condition not drastically changed in abundance and structure (e.g., in age and size distributions) from the "natural" (unfished) state. Some examples of species needs are described below, related to predation, competition, habitat, and environment.

According to Margalef (in Smith 1994, pg 8) "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]". Without human intervention, species exist in the ecosystem in their "natural" state and the "needs" of species are met to a greater or lesser extent. Human intervention, such as fishing, modifies the properties of the ecosystem in a variety of ways, such that it may no longer meet the needs of the species that exist within it in the same way as it did without intervention. The question is, then, does the fishery management plan limit modification of ecosystem properties caused by actions taken under the plan in such a way that the ecosystem continues to support the needs of the species it contains in the way that it did prior to modification?

This formulation immediately and clearly demonstrates the complexity of the issue. In order to show how human intervention has modified the ecosystem, and the extent to which this modification compromises ecosystem function, it is necessary to have some way of measuring and evaluating ecosystem function under various states of nature, both with and without human intervention. The need for measures of effectiveness is discussed further in Section 4.5.2.

With the above in mind, it becomes clear that what is being managed in natural resource contexts is human intervention in ecosystems, not the species or the ecosystems themselves. This has consequences for the terminology that is generally used to describe the process of managing fisheries while being considerate of the needs of non-target species in the ecosystem. Frequently the term "ecosystem management" is used, but since nobody can profess to manage regime shifts, changes in food webs, or climate change, this is patently inappropriate.

Modification of the marine ecosystem is an inevitable consequence of the scale of human activity in areas such as fishing and coastal development. It is possible and valuable, however, to conduct fishery management while recognizing ecosystem effects and taking ecological considerations into account. The concept of "rational use" of living marine resources is now widely accepted and enshrined in international agreements, such as the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) Convention, which aims to take an "ecosystem approach" while allowing fisheries to proceed on a rational basis. Conservation is, therefore, concerned with how we sustain renewable resources in ecosystems so that future options are maintained. This might be called an "ecosystem-based approach to management", "fishery management with an ecosystem perspective" or "fishery management taking account of ecological and ecosystem considerations." Although each of these has merit in different circumstances, overall we prefer the latter.

Irrespective of how the approach is described, what it attempts to do is to bring more of the lesson from ecological sciences (such as thresholds, uncertainty, and surprise) into scenarios used to inform management decisions. It considers not only the target stock, but also 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), that changes in the physical components of the ecosystem and flips in biomass availabilities may occur, independent of fishing activity (Sherman 1991) and that the physical abiotic environment may have profound effects on biological interactions.

For a variety of reasons, which are discussed in detail later in this section, conventional stock assessment tends to focus only on the effects of fishing on target species (the "conventional stock assessment world view" diagrammed in Figure 4.1) and does not take explicit account of ecological and ecosystem considerations. Broadly speaking, this is the case in the North Pacific groundfish fisheries. However, we note that the answer to the question in the third point of the charge is complex. In this section we elaborate on the differences between an essentially single-species approach and an explicit ecosystem approach and describe several intermediate steps in between. While we use a number of discrete stages to describe the range of options available to managers, in fact there is a continuum of modifications and adjustments to current thinking and practice that can move the management process towards the more desirable goal of fishery management taking account of ecological and ecosystem considerations. These modifications and adjustments may require substantial time and resources to achieve, not least because our current state of knowledge, and particularly our ability to predict future states of nature and the effects of fishing on them, is limited.

Moving from the conventional assessment view towards an ecosystem view involves a shift in the components of fundamental underlying ecological science that is relied upon. In essence, for current fishery management, population ecology is the fundamental ecological science, but for an approach that takes ecological and ecosystem considerations into account, community ecology is the fundamental ecological science. For example, when one thinks about single species, there can be "excess production" from a stock, but when one thinks about the "needs" of all the other species in an ecosystem, the notion of excess production from a single member of the community becomes far more complicated.

All fishery management regimes are at some point along this continuum and are addressing their management goals with varying degrees of success. It is important, however, not to consider current approaches as necessarily wrong simply because they do not take ecosystem considerations explicitly into account. There may be perfectly good reasons why this is either not possible or not necessary, in which case the implicit approach, based on incorporation of uncertainty into the process, is likely to be the best way forward.

In the following sections we describe a range of management goals described previously for an ecosystem approach, and discuss these in the context of the management of the North Pacific groundfish fishery. We then attempt to show where the management of the North Pacific groundfish fishery currently sits in relation to these goals and what might be done differently to achieve them more effectively and efficiently.

4.1.2 *Implications of Different Approaches to Fishery Management*

4.1.2.1 The Conventional Assessment World View

Currently, almost all fishery management is based on a "conventional assessment world view" (Figure 4.1).[1] This recognizes the biophysical world in which the stock exists, the socio-economic world of the fishing community that takes the stock, and the management world in which catch limits and other controls on fishing activity are determined and implemented.
Use of single-species management makes the assumptions that:

- stocks can be assessed and managed outside of the context of their role in the ecosystem,
- density dependence is the main regulating factor in population dynamics, and
- if one simply knows enough about the vital information of the stock, then it is possible to fully control the trajectory of the stock.

These assumptions are relied upon whether one uses surplus production models, dynamic pool models, stock-recruitment models, delay-difference models, or complex age- and length-based assessment models such as Stock Synthesis, AD Model Builder, and ADAPT. Additionally there are usually a number of further assumptions with respect to:

- Stationarity. That fluctuations are 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.
- Linkage. 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 also, but for many species instead of just one.
- Time and space scales. That only one temporal and spatial scale is sufficiently important to be included in the model (also see the next section).
- Genetics. That the population is composed of genetically identical subunits.

Within the resource management framework, Charles (1992) identifies three main paradigms:

- *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."

[1] Figures 4.1 to 4.5 are a representation of putative stages in the transition from a conventional stock assessment and management approach to one that fully embraces ecosystem interactions. The structure depicted in these figures is based on an original diagram drawn by Bill de la Mare and developed by Andrew Constable (Australian Antarctic Division).

None of these paradigms explicitly focuses on the ecosystem, although the ecosystem itself is a component of each paradigm and thus connects the otherwise conflicting notions. These three paradigms can be identified with the major components diagrammed in Figure 1.

The conventional assessment world view does recognize that there is a natural mortality rate, M, operating on the target stock, and this natural mortality is assumed largely to be the result of consumption in the food web. Generally M is assumed to be constant in the conventional assessment models, but it must be understood that this does not assume (or assign) a constant total consumption by higher trophic levels. The constant natural mortality rate, M, is in units *per capita* of the target stock. Thus the total consumption by higher trophic levels, when M is assumed to be constant, will vary in proportion to the target stock size (or biomass). A harvest management strategy, such as $F_{40\%}$, that by design reduces the biomass of the target stock biomass by a large fraction, will, all other things being equal, reduce the total consumption by higher trophic levels by a similar large fraction, and we would expect the predator populations to be reduced accordingly. This may or may not be deemed a desirable, or acceptable, outcome from the standpoint of policy. And, in fact, all other things often are not equal, especially in ecosystems, and there are a variety of mechanisms whereby the reduction in target stock biomass by a harvest strategy such as $F_{40\%}$ could cause a more than proportional reduction in the populations of predators dependent on those same stocks for prey, as is recognized in the ecosystem-effects world view.

4.1.2.2 The Ecosystem Effects World View

A management plan that recognizes ecological and ecosystem effects must be broader and deeper than the conventional single-species world view, as it attempts to deal with three interlocking goals (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.

Furthermore, this approach tends to embrace a greater range of temporal 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. Lluch-Cota *et al.* (1997) suggest that decade-scale regime shifts may be a global phenomenon in small pelagic stocks. During the course of the 20th century they identified periods of alternating global dominance of sardines and anchovies on an approximately 20 to 25 year cycle: sardine dominance from 1925 to 1950, anchovy dominance from the early 1950s to the late 1970s, returning to sardine dominance again in the late 1970s and 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. There is evidence of these same patterns in other commercial stocks (Klyashtorin 2001).

Other examples of ecosystem analysis suggest cascading and perverse food web effects that are interpreted as being triggered by fishing. Noteworthy among these is the phenomenon of intense fishing largely eliminating higher trophic levels among the suite of target stocks, but with a declining overall productivity of harvested stocks, as unharvested species replace the overfished higher trophic level target stocks (Pauly *et al.* 1998) in a kind of predator-prey "triangle" (Pimm 1982).

The ecosystem is 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 many practical questions, the boundaries will be vague and determined to some extent by the kinds of questions being asked. This definition leads to two crucial questions: what is an ecological 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). Here 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 given mixture can vary over time or space, but there is a consistent pattern to the mixture, even if it can only be described in terms of probabilities (Fager 1957; Fager 1963; Hubalek 1982).

Mangel and Hofman (1999) review concepts concerning ecosystems. In doing so, they identify a series of conditions of ecosystems that may influence management of human interventions:

- Patchiness and variability in space and time are characteristics of most ecosystems.
- Ecosystems are characterized by multiple cause-effect relationships among biotic and abiotic ecosystem components.
- The consequences of events at one trophic level often will be manifested across many other trophic levels.
- Organisms do not recognize political boundaries and management should plan accordingly.
- Ecosystems should be viewed as the current state of an ongoing process of selective extinction and differential speciation (Fowler and MacMahon, 1982 elaborate this idea).
- Change is the rule, not the exception, in ecosystems.
- Interactions between components of ecosystems may be both one way and two way.
- Marine food chains are complex and in many species the trophic level varies with life stage.
- Competition and predation both contribute to the structuring of food webs, but their relative importance varies.
- 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.

Arising from these conditions and other published work, we have developed the three basic expectations, which, when accepted, have implications for the way in which human intervention in ecosystems should be managed (Table 4.1).

Table 4.1. Expectations of the behaviour and characteristics of ecosystems and the associated implications for managing human intervention

Expectation	**Implication for Management**
Ecosystems change over time and changes 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.
Human intervention will have an effect on parts of the ecosystem other than the target stock .	Monitoring is essential and stock assessment of a targeted species is not enough. One must monitor its prey, competitors and predators
No amount of monitoring or sampling will fully reduce the uncertainty that one faces	Management must be cognizant of the levels of ignorance in which it is working. Discussion must focus on the balance between 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. Similarly, the decision to completely close a fishery, while it might appear necessary in some circumstances, may not lead to recovery of depleted species over moderate time scales (see Hutchings (2001)).

4.1.3 Previously Published Ecosystem Principles and Management Goals

There is a rapidly growing body of published ecosystem principles and management goals to guide management of human activities in the natural environment in a way that recognizes ecological and ecosystem considerations (e.g., Charles (2001)). These can be conveniently organized within a management framework that comprises four levels (see Table 4.2):

- Ecosystem principles;
- Management goals;
- Management policy required to achieve management goals; and
- Management and scientific activities (including monitoring) in support of implementing management policy

The conclusions of the Ecosystem Advisory Panel (Fluharty *et al*. 1999) cover much of the same ground as the publications referred to in Table 4.2. The Ecosystem Advisory Panel was created by Congress during the 1996 reauthorization of the Magnuson Fisheries Conservation and Management Act, and charged with assessing the extent that ecosystem principles are used in fishery management and research, and to recommend how such principles can be further implemented to improve management of living marine resources. Below we use the same four-level framework described above to summarize the conclusions of this panel:

Ecosystem Principles

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

Management goals

- Maintenance of ecosystem health and sustainability

Management policies to achieve the goal

- 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

Management and scientific activities

Develop a Fisheries Ecosystem Plan (FEP) that mimics the Fishery Management Plan with the following 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 organism plays different roles in the food web at different times in its life. Although these ideas are very old, going back at least to Hardy and Elton, they are difficult to implement. 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, 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. <u>Describe the habitat needs of different life history stages of the organisms in the "significant food web" and document how they are considered in conservation and management measures</u>.

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

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

6. <u>Develop indices of ecosystem health as targets for management</u>.

7. <u>Describe available long-term monitoring data and how they are used</u>. At this stage, an evaluation of habitat condition, oceanographic variability, potential confounding influences (e.g., terrestrial, freshwater, waste disposal) and scales of interactions among these factors need to be described and the overall status of the system related to the targets for management.

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

The NPFMC has independently moved in these directions. The Council's Ecosystem Committee, established in 1996, has developed a draft policy for ecosystem-based management of North Pacific fisheries, based on principles and elements of ecosystem management from the scientific literature (e.g., Grumbine 1994; Mangel *et al.* 1995; Christiansen *et al.* 1996). This draft was reported in Witherell *et al*. (2000), and to date has not changed (Witherell pers. comm.):

Definition:

Ecosystem-based management, as defined by the NPFMC, is a strategy to regulate human activity towards maintaining long-term system sustainability (within the range of natural variability as we understand it) of the North Pacific, covering the Gulf of Alaska, the Eastern and Western Bering Sea, and the Aleutian Islands region.

Objective:

Provide future generations the opportunities and resources we enjoy today.

Goals:

1. Maintain biodiversity consistent with natural evolutionary and ecological processes, including dynamic change and variability.
2. Maintain and restore habitats essential for fish and their prey.
3. Maintain system sustainability and sustainable yields of resources for human consumption and non-extractive uses.
4. Maintain the concept that humans are components of the ecosystem.

Guidelines:

1. Integrate ecosystem-based management through interactive partnerships with other agencies, stakeholders, and public.
2. Utilize sound ecological models as an aid in understanding the structure, function, and dynamics of the ecosystem.
3. Utilize research and monitoring to test ecosystem approaches.
4. Use precaution when faced with uncertainties to minimize risk; management decisions should err on the side of resource conservation.

Understanding:

1. Uncontrolled human population growth and consequent demand for resources are inconsistent with resource sustainability.
2. Ecosystem-based management requires time scales that transcend human lifetimes.
3. Ecosystems are open, interconnected, complex, and dynamic; they transcend management boundaries.

Table 4.2. Ecosystem principles, management goals, ecosystem policy, and implementation support

Author:	**Holt and Talbot 1978**	**May *et al.* 1979**	**Grumbine 1994**	**Olver *et al.* 1995**
Ecosystem Principles				Aquatic ecosystems should be managed to ensure long-term sustainability of native fish stocks.
Goals and Objectives	The ecosystem should be maintained in a desirable state such that: consumptive and non-consumptive values could be maximized on a continuing basis; present and future options are ensured; and the risk of irreversible change or long-term adverse effects as a result of use is minimized.	Populations [other than those at the top of the trophic ladder] should not be depleted to such a level that their productivity or that of other species dependent upon them is significantly reduced. The intersections of these [ecosystem] considerations with economic and political factors imply consequences and management implications that "defy crisp summary".	Maintain evolutionary and ecological processes Maintain viable populations of all native species *in situ* Accommodate human use and occupancy within these constraints.	Vulnerable, threatened, and endangered species must be rigidly protected from all anthropogenic stresses. Harvest must not exceed the regeneration rate of a population or its individual stocks
Management Policy	Management decisions should include a safety factor to allow for the fact that knowledge is limited and institutions are imperfect. Measures to conserve a wild living resource should be formulated and applied so as to avoid wasteful use of other resources.	Harvesting levels should be set conservatively to safeguard against the combined effects of environmental variation and harvesting.		Exploitation of populations or stock undergoing rehabilitation will delay, and may preclude, full rehabilitation Direct exploitation of spawning aggregations increases the risk to sustainability of fish stocks. The sustainability of a fish stock requires protection of specific physical and chemical habitats utilized by the individual members of that stock. The sustainability of a fish stock requires maintenance of its supporting native community.
Management and Scientific activities	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.	For populations at the top of the trophic ladder, the concept of maximum sustained yield (MSY) will often remain useful. Monitoring should be set to the slowest population process time scale.	Manage over periods of time of sufficient duration to maintain evolutionary potential of species and ecosystems Represent, within protected areas, all native ecosystem types across their natural range	

Author:	Mangel *et al.* 1996	Harwell 1997	Pitcher and Pauly 1998; Pitcher 2000
Ecosystem Principles	Maintenance of healthy populations of wild living resources in perpetuity is inconsistent with unlimited growth of human consumption of and demand for those resources	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.	
Goals and Objectives	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.	Develop a shared vision of desired conditions for societal systems and ecological systems. Provide for ecosystem governance at appropriate ecological and institutional scales.	Rebuilding ecosystems, not sustainability, is the appropriate goal for fishery management
Management Policy	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. 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.	Implement ecosystem management principles through coordinated government and non-government plans and activities. Adopt a management approach that recognizes that ecosystems and institutions are characteristically heterogeneous in time and space. Integrate sustained economic and community activity into the management of ecosystems.	Scientific evaluations should be separated from management decisions and enforcement issues.
Management and Scientific activities	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. Effective conservation requires communication that is interactive, reciprocal, and continuous. The full range of knowledge and skills from the natural and social sciences must be brought to bear on conservation problems.	Use adaptive management as the mechanisms for achieving both desired outcomes and new understandings regarding ecosystem conditions. Use an ecological approach that recovers and maintains the biological diversity, ecological function, and defining characteristics of natural ecosystems. Integrate the best science available into the decision-making process, while continuing scientific research to reduce uncertainties.	Data should be openly available, so that any individual with the right skills can perform and confirm analyses A wide variety of data - much broader than for single species stock assessment - needs to be collected, in a fully geo-referenced manner. Data must include information about the target stock and prey and predators of the target stock; Peer-reviewed publications, rather than gray literature or other un-refereed formats such as the Internet, remain the means for communicating information; Use Bayesian and Monte Carlo methods as means for incorporating uncertainty in the models that are used to describe ecosystems

4.1.4 The Meaning of Measuring Ecosystem Health: Reference Points for Management

Maintenance of ecosystem "health" is often quoted as the goal of an ecosystem approach to fisheries management (e.g., Fluharty *et al.* 1999), frequently without a clear explanation how this should be interpreted. Larkin (1996) noted the difficulty in developing this concept into an operational definition that is sufficiently objective and measurable for it to be used in resource management. The conclusions of the Ecosystem Advisory Panel included the need to develop indices of ecosystem health as targets for management, but there is little in the published literature that addresses this need directly and quantitatively. Harwell (1997) described the need to "use an ecological approach that recovers and maintains the biological diversity, ecological function, and defining characteristics of natural ecosystems." The Council's draft policy for ecosystem-based management of North Pacific fisheries (Section 4.1.3) provides qualitative goals in terms of biodiversity, habitats, system sustainability and sustainable yield. Nevertheless, what is needed for management is quantitative indicators and reference points akin to those used in conventional fisheries management to identify overfishing and stocks in an overfished condition (as explained in chapters 2 and 3 of the present report)

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, 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 noted 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 prevent their interaction with the fishing gear, but for this to be complete 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 or depleted stocks of marine mammals the question needs to be asked as to what fishing controls are necessary to reduce to zero the effective threat to timely recovery of continued mortality through fishing. In the context of the MMPA, the "negligible" impact goal is addressed through the PBR system for permitting incidental mortality. In the context of ESA, the question of "significant" adverse impact is evaluated in "biological opinions" which must render determinations about "jeopardy."

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 the history of development of regulatory standards for implementation of some legislated policies which were drafted in ways that did not clearly define the objectives and were 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. As defined in the original language of the ESA, these terms were not sufficiently defined to be operational, and since they were policy-laden, there was not an automatic unambiguous scientific definition either. 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. Basically, there is a tendency for legislation to mandate "safety" without defining "how safe is safe enough."

Considerable effort, largely on the part of NMFS, has been expended on developing interpretations of these terms, and similar terms from the MMPA, in order to arrive at more definite policy statements, that define a coherent and quantified burden of proof, that respond in a precautionary manner to uncertainty, and that can be used for consistent, data-driven decisions (Angliss *et al*. 2002; Wade and Angliss 1997; Taylor *et al.* 2000). A comparable degree of formality in translating broad policy goals for ecosystem considerations into operational statements remains to be accomplished.

Notwithstanding the need for operational definitions of policy goals, it is not currently possible, in many cases, to determine quantitatively (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" are value-laden terms which do not have unambiguous scientific meaning, unless the policy constraints are made explicit. Their meaning is interpreted instead through legal, regulatory and policy usage and through precedent.

Murawski (2000) provides an important contribution in the quest for operational indices of ecosystem condition by considering the quantitative basis for defining what he terms "ecosystem overfishing". He points out that there is no specific ecosystem analogue to single-species definitions of overfishing–no single utilitarian metric of ecosystem condition, and hence ecosystem overfishing. However, he proposes the development of explicit ecosystem overfishing criteria that may be used to establish multiple tiers of measures to address issues inadequately covered by conventional single-species oriented management. He concludes that ecosystems can be considered to be overfished when cumulative impacts of catches (including discards), non-harvest mortality and habitat degradation result in one or more of the following conditions:

- Biomasses of one or more important species assemblages or components fall below minimum biologically acceptable limits, such that (1) recruitment prospects are significantly impaired, (2) rebuilding times to levels allowing catches near MSY are extended, (3) prospects for recovery are jeopardized because of species interactions, or (4) any species is threatened with local or biological extinction;
- Diversity of communities or populations declines significantly as a result of sequential "fishing-down" of stocks, selective harvesting of ecosystem components, or other factors associated with harvest rates or species selection;
- The pattern of species selection and harvest rates leads to greater year-to-year variation in populations or catches than would result from lower cumulative harvest rates;
- Changes in species composition or population demographics as a result of fishing significantly decrease the resilience or resistance of the ecosystem to perturbations arising from non-biological factors;
- The pattern of harvest rates among interacting species results in lower cumulative net economic or social benefits than would result from a less intense overall fishing pattern or alternative species selection;
- Harvests of prey species or direct mortalities resulting from fishing operations impair the long-term viability of ecologically important, non-resource species (e.g., marine mammals, turtles, seabirds).

These conditions could, therefore, be regarded as a selection of metrics of ecosystem status that provide the basis of thresholds that should be avoided in an attempt to prevent ecosystems from becoming "unhealthy". What is perhaps harder to do is fulfill the need described by the Ecosystem Advisory Panel for management targets that can be aimed at, in the sense of restoration and maintenance of ecosystem function, as opposed to thresholds that should be avoided.

4.1.5 The Decision-making Environment

Regardless of the particular synthesis of principles and indicators that one achieves, it is important to consider how they can be used in a charged decision-making 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).

4.2 Achieving Management Goals with Respect to the Ecosystem

In this section, we describe the options available to fisheries managers when attempting to achieve management goals with respect to the ecosystem. In essence, policy and actions under the fishery management plan may succeed in maintaining ecosystem health and sustainability, while allowing fishing to proceed, through either *implicit* or *explicit* consideration of ecosystem effects. We note also that the FMP may fail to meet the needs of non-target species even though management actions aim to take explicit account of ecological and ecosystem considerations

4.2.1 An Implicit Approach

4.2.1.1 The Importance of Scale

An implicit recognition of ecosystem effects arises in its most basic form from an appreciation of the importance of scale. As described in Section 4.1, patchiness is a characteristic of marine ecosystems. An alternative formulation would be that pattern and scale (both in time and space) are essential factors in developing a management approach that takes ecological and ecosystem considerations into account. Levin (1992) argues that the problem of pattern and scale is the central problem in ecology since it unifies population biology and ecosystem science and marries basic and applied ecology. Most importantly, Levin argues that there is no single correct scale for viewing a natural system but that the appropriate description of variability and scale depends upon the question being asked. The National Standard 1, in which a "unit stock" is discussed, recognizes the potential effects of spatial scale. As currently employed, the harvest control rule is generally applied at an aggregate level of the Bering Sea/Aleutian Islands or Gulf of Alaska (see Section 4.4). However, to learn about ecosystem effects of fishing and means of mitigating potential adverse effects, one may need to consider issues at a finer spatial scale.

The most rudimentary implication of scale is that the fishery can have effects spatially and temporally removed from the fishery itself. Most fisheries stock assessment models recognize effects other than fisheries on the population. For example, in standard age structured models, the dynamics that relate the numbers of individuals from one year to the next assume that when fish disappear, a fraction $F/(F+M)$, of the fish are taken by the fishery, and the remaining fraction $M/(F+M)$ of the fish that disappear go to "natural" predators (Hilborn and Walters 1992; Quinn and Deriso 1999). The choice of $F/(F+M)$ makes an assumption that the effect of the fishery is the same as the effect of all other predators combined, in the sense of "sharing the take;" whether this is true or not is generally unknown.

Depending on the relative levels of F and M and how the model uses them, this approach may contain an implicit allowance for the predators of the target population, even though there is no explicit consideration of predator needs when catch limits are determined (Figure 4.2). Without additional calculation it is presently impossible to say the extent to which a harvest strategy including a target F of $F_{x\%}$ (x=40, 50, 60, 70) takes account of predator needs on a gross scale in the BS/AI and GOA. However, one can easily surmise that the larger x is, the more likely it is to do so. On the other hand, the effect of regime shifts and other environmental fluctuations may mean that even x=100 (i.e., no fishing) may not take care of the needs of the predator populations at certain times. For example, a regime shift might cause a first tophic level to decline, which could cause the fished species to decline even in the absence of fishing, which could cause predators to decline due to prey shortage. Nevertheless, one might consider a harvest strategy that uses a target of $F_{50\%}$ or $F_{60\%}$ to be inherently more precautionary than $F_{40\%}$ with respect to predator requirements. But this needs to be balanced against the possible economic losses due to fishing at a less than optimum level. As mentioned in chapter 3, the trophic level occupied by a species may be very important in whether single-species reference points are applicable Collie and Gislason 2001).

As a first proxy, one might adopt the approach used by CCAMLR to manage fishing on krill, a fundamentally important forage species in the Southern Ocean. The CCAMLR approach is to try to list all the per capita consumption rates of predators of the fished species, multiply by the estimated number of predators and determine x such that the "excess productivity" when $F_{x\%}$ is applied exceeds this predator total. Of course, this assumes that it is possible to accurately measure and characterize all the consumption rates for different predators and prey. In this respect it is worth noting that when managing fishing on krill, CCAMLR is dealing with somewhat less complicated foodweb interactions than those encountered in the BS/AI and GOA groundfish fisheries. Krill has been clearly identified as an important forage species that makes up a substantial proportion of predator diets in the Antarctic ecosystem. Even with the limited monitoring that is feasible in the Antarcti (compared to more accessible places), it has been possible to detect clear effects of poor krill availability on predators (e.g., Reid and Arnould 1996; Iverson *et al*. 1997; Everson *et al*. 1999). Added to this, the "precautionary" catch limits imposed on the krill fishery arising from this approach are still considerably higher than the harvest actually taken annually by the commercial fishery. The aggregate yield limit for the Atlantic Sector of the Southern Ocean is 4 million tonnes. The limit for the area around South Georgia (Subaera 48.3) is 1.056 million tonnes, whilst the annual catch in this area is presently of the order of 100 thousand tonnes. Nevertheless, it should be noted that application of this rule to fishing regulations for the Antarctic leads to acceptable fishing rates which allow far less fishing mortality, or biomass reduction of the target stocks, than would be tolerated under an $F_{40\%}$ rule.

An ecosystem rule based on evolutionary arguments leads to even lower recommended harvest rates (Fowler 1999).

4.2.1.2 Incorporating Uncertainty

In recent years, a major feature of many single-species assessments has been an investigation of uncertainty in various components of the analysis (as explained in chapter 2). Much of the uncertainty in assessments and the subsequent prediction of the effects of alternative fishing strategies arises due to the effects of environmental variability on biological processes, such as stock-recruitment relationships and stock movements. Measurement error, if not taken into account, will introduce biases in the estimate of a stock-recruitment curve (Walters and Ludwig 1981). Real variation in recruits per spawner requires further adjustment of the control rule in order to optimize yield (Engen *et al*. 1997). Simulation models that explore the potential effects of alternative management strategies can include substantial uncertainty. Catch limits and other measures based on these models can be set in a more or less precautionary way depending on how the uncertainty is used. Setting limits in a precautionary way can, therefore, provide an implicit buffer against the effects of fishing on ecosystem properties, even though these effects are not fully understood and have not been explicitly included in the analytical process. The concern is that without better understanding of the effects of fishing on an appropriate spatial and temporal scale, we cannot be sure that the size of the buffer is sufficient to mitigate all adverse impacts on the ecosystem. On the other hand, it may be that in fact the precautionary buffer is too large and with a better understanding of ecosystem effects it would be possible to set higher catch limits without causing adverse impacts. The amount of buffer that should be established in conditions of uncertainty is an expression of the level of strictness with which the burden of proof is shifted in the direction of requiring evidence that there will not be harm, before allowing the proposed activity to proceed. There is a wide spectrum of responsible opinion on how strict this so-called “reversal” of burden of proof should be (Dayton 1998).

A possible example of an implicit approach to taking ecosystems considerations into account is the overall cap on the annual North Pacific groundfish harvest of 2 million mt. Since 1981, the total annual allowable catch of groundfish for this region has been required to fall within an optimum yield range of 1.4 to 2.0 million mt. Apparently, the upper limit of 2 million mt was set on the basis of indications from previous years that when the aggregate catch exceeded this level, there was evidence of stress in the ecosystem (Loh Lee Low pers. comm.). This has limited the sum of TAC’s for all species to 2 million mt per year, which has been considerably less than the sum of all allowable biological catches (ABCs). In some years, ABC’s have totaled more than 2.8 million mt (Witherell *et al.* 2000). Uncertainty is also used to adjust TACs downwards compared to ABCs in the tier system of management (as explained in chapter 2). As a result, the Council considers that many groundfish stocks, particularly flatfish stocks, have been exploited well below sustainable levels.

An implicit ecosystem approach, therefore, recognizes the existence of ecosystem interactions, but does not make any specific attempt to quantify the surplus production that must be reserved to satisfy ecosystem needs, nor does it attempt to modify fishing behavior to specifically mitigate adverse impacts other than those on the target species (Figure 4.1).

It seems intuitive that an explicit consideration of ecosystem effects is more likely to be successful than an implicit approach. But there may be several possible reasons which an explicit approach is not employed. First and foremost, there may not be sufficient information to take an explicit approach either because ecosystem data have not been collected, or because ecosystem based goals have not been developed. Under these circumstances one would expect to see an implicit approach, relying on the safety margins created by application of a precautionary approach, without really knowing whether these margins are appropriate for the purpose. As experimentation and enhanced monitoring contribution more information on the ecosystem, and its response to management, it may be possible to shift to a more explicit approach.

The effectiveness of an implicit approach based on "safety margins" will depend on how carefully these rules are crafted. The safety margin should be stated in terms of satisfying an upper confidence limit on the estimate of the effect *on the ecosystem*. Then the width of the margin will generally increase with our uncertainty about the effect on ecosystem. By contrast, if the safety margin intended for ecosystem considerations is a statistical confidence built into the decision rule for single-species management (such as the present Tier system for setting ABC in The NMPFC FMP for groundfish), it will have the ironic result that as the quality and quantity of information available for single-species management calculations increases, the decision will become *less* protective of ecosystem considerations.

Second, a more explicit approach may not actually be necessary for every fishery. In some fisheries, fishing activity may simply not modify the ecosystem to the extent that non-target species and/or habitats are affected, even though such indirect effects were not considered explicitly in the FMP. For example, if the actual level of fishing activity is low, consequent catches are very small compared to the MSY level and the fishing gear does not interact in any substantial way with non-target species, and/or habitat structure.

4.2.2 An Explicit Approach

Taking ecological and ecosystem considerations into account in fishery management calls for an explicit view of ecosystem effects. This is an important step beyond the implicit approach, but the analytical process has a long way to develop before it can truly, if ever, be regarded as representing a unified ecosystem-based approach. Full implementation will require models capable of representing reliably the dynamics of the interacting components of interest. Experience, to date, with such large complex models of marine ecosystems indicates that the behavior of the model is very sensitive to the values of parameters which are poorly known and which are difficult to measure. In particular, food web models are highly sensitive to the representation of the functional response of predators (Magnusson and Palsson 1991; Mohn and Bowen 1996; Gao *et al*. 2000; Tett and Wilson 2000; Fu *et al*. 2001; Steele and Henderson 1992) and the predominance of top-down versus bottom-up control as determined by the quantitative representation of prey vulnerability to predation (Walters *et al.* 1997; Stevens *et al.* 2000; Shannon *et al*. 2000; Vasconcellos *et al*. 1997; Aydin and Friday 2001). The details of a predator's functional response may depend on complicated and subtle spatial behavior, as, for example, when predators aggregate in areas of high prey density, and, therefore, switch away from feeding on prey species when these are sparse (Hassell and May 1974; Anderson 2001).

The first step beyond the implicit approach is illustrated in Figure 4.3. Here the status of predators of the target species, which may compete for resources with the fishery, is assessed using quantitative methods. The results of this analysis are fed into the management procedure, but are not integrated with the analysis that focuses on the target species. Similarly, there may be some environmental information that influences decisions at the management level, but again this is outside of the analytical process. The essential characteristic of this stage is that there is no link made between the fishery and its effects on ecosystem properties other than the direct effects on the target population.

The first stage at which the assessment and management process really begins to embrace explicitly the ecosystem approach is illustrated in Figure 4.4. Here, information from the environment, including non-target species is fed directly into the assessment process, and influences the scientific advice that is provided to the managers. The fundamental difference between this level and that described in Figure 4.3 is the difference between the population ecology and community ecology views of management. That is, putting the predators in the lowest box in Figure 4.4 is a more explicit treatment of the community issues.

At this stage, the assessment starts to address explicitly the more tractable problems associated with the effects of the fishery on ecosystem properties. These are problems that are relatively straightforward to define, although their solutions may take substantial time, resources and innovation to implement. They include issues such as reduction of discards and discard mortality, avoidance of incidental mortality of endangered or threatened species such as seabirds and marine mammals, and reduction of adverse impacts on habitat.
To date, much of the effort applied to incorporating ecosystem considerations into fisheries management has been applied to addressing these more tractable problems. This is, in part, because they are, relatively, easier to identify, and usually easier to mitigate. However, fisheries management in a truly ecosystem context involves substantially more than just, for example, modifying the operation of fishing gear to reduce undesirable interactions. In its fullest sense, managing for ecosystem considerations must address both more tractable and less tractable problems in a fully integrated sense within the analytical process that generates scientific advice for managers (Figure 4.5). The less tractable problems are those for which the cause and effect are much more difficult to demonstrate. These include the effects of human intervention (of which fishing may be only part) on complex species interactions that propagate through the food web with unpredictable results, and the influence of regime shifts (both short and long term) on factors that affect the way in which we look at population dynamics, such as natural mortality (for example, due to changes in species interactions), carrying capacity and stock-recruitment processes.

In the following sections we review in more detail these tractable and less tractable problems and how these might be addressed, with examples drawn from the North Pacific groundfish fishery.

4.2.2.1 More Tractable Ecosystem Problems

The more tractable ecosystem problems generally comprise the direct effects of fishing activity, other than those on the target species, such as bycatch and incidental mortality, and some direct effects on habitat. These direct effects are relatively easy to detect and can often be mitigated through some modification in the way fishing vessels operate or the configuration of the fishing gear. Well known examples include the use of streamer lines to reduce the capture of seabirds in longline fisheries, the use of turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) in shrimp trawls and a variety of gear modifications and approaches to area management that are part of the Atlantic Large Whale Take Reduction Plan.

The term “more tractable” is not meant to imply that these types of problems and their solutions are straightforward issues. Many of the mitigation techniques now being used have taken several years to develop and are still evolving. What makes these problems more tractable is that the relationship between cause and effect is relatively clear, i.e., it is clear that the fishing activity is the cause of the problem (for example when seabirds are caught on longlines). Although fishery managers have been generally aware of these types of problems for some time, it is only more recently, through the use of enhanced monitoring techniques (e.g., observers), that it has been possible to quantify them and monitor the implementation of viable solutions.

The North Pacific groundfish fisheries have begun to address a number of more tractable ecosystem problems, which are outlined in Table 4.3. In addition, since 1995, the Council’s Groundfish Plan Teams have prepared a separate Ecosystem Considerations section to the annual Stock Assessment and Fishery Evaluation (SAFE) report. The intent of the Ecosystems Considerations section is to provide the Council with information about the effects of fishing from an ecosystem perspective, and the effects of environmental change on fish stocks. The most recent of these (Ecosystem Considerations for 2002) contains substantial information on a range of ecosystem status indicators. These indicators include the physical environment, habitat, zooplankton, chlorophyll and nutrients, forage fish, groundfish biomass and recruitment trends, historical abundance trends from bottom trawl data, benthic communities, non-target fish species, marine mammals, seabirds and other ecosystem or community properties. This information is considered in relation to the four ecosystem goals developed under the Council’s draft ecosystem based management policy (Witherell *et al.* 2000). Although the management measures described in Table 4.3 result in changes to the way in which the fishery operates, and may limit the catch of target species (in the case of by-catch limits for example), and the Ecosystems Considerations document provides substantial scientific information on status indicators, the stock assessment and estimation of yield of the target species is still undertaken essentially in isolation of ecosystem considerations. One of the stated purposes of the Ecosystems Considerations document is to address the need to promote stronger links between ecosystem research and fishery management. The management measures described in Table 4.3 do address at least one ecosystem problem that is less tractable: competition for prey. This is an indirect effect that is more difficult to diagnose, and more costly to solve. In fact, competition for prey is a suspected, but unproven, problem associated with interactions of the North Pacific groundfish fishery with marine mammals. And temporal and spatial redistribution of fishing effort has been adopted as a management measure in the hope that this will limit local depletion of prey species, but the efficacy of this management measure also remains unproven.

4.2.2.2 Less Tractable Ecosystem Problems

The common thread that identifies the less tractable problems is that they involve indirect effects of fishing, where cause and effect may be several steps removed from each other. This tends to introduce complications into the picture, because the fishery may not be the only, and perhaps not even the major cause of the problem. There is, therefore, a much higher level of uncertainty regarding the role played by the fishery in affecting the ecosystem properties in question. Finding ways to mitigate these problems, beyond the implicit approach discussed in Section 4.2.1, is, therefore, very difficult.

We can demonstrate the difference between these less tractable problems and the more tractable problems by looking at a single example of an endangered species (e.g., a marine mammal or bird) with declining population size. The first response in such a situation would normally be to address more obvious issues such as direct mortality. As indicated previously, with modern monitoring tools it is relatively easy to determine if a fishery is causing any direct mortality. Once shown, methods can usually be found to mitigate against it; gear modifications, seasonal closures, closed areas etc. This can be regarded as a tractable problem. However, consider the situation where the direct mortality problem has been solved (and has been shown to be, through monitoring) but the endangered population continues to decline, or at least not recover. It then becomes necessary to look for other explanations, including the possibility that the fishery is having a different, indirect effect that is contributing to the failure of the endangered species to recover. This is clearly a much less tractable problem.

Table 4.3. Some of the more tractable problems associated with interactions between fisheries and the ecosystem being addressed by the management of North Pacific groundfish fisheries (based on Witherell *et al*. 2000)

Interaction	Problem	Management Measures (summary)
Interactions with seabirds	Direct mortality	Catch deterrent devices (streamer lines) required on longline vessels; Incidental catch limit for short tailed albatross, *Diomedea albatrus* (four over a two year period).
Interactions with marine mammals	Direct mortality; disruption at rookeries and haulouts	Observer monitoring; Incidental catch limits for Steller sea lions; Area closures.
Bycatch and discards	258,000 mt of groundfish discarded in the 1997 in the BS/AI fisheries, equating to a rate of about 15%.	Bycatch limits set for specific species in specific groundfish fisheries: King crab, *Paralithodes* and *Lithodes* spp.; Tanner crab, *Chionoecetes* spp.; Pacific herring, *Clupea harengus pallasi*; Pacific halibut, *Hippoglossus stenolepis*; and Pacific salmon and steelhead trout, *Oncorhynchus* spp. Gear restrictions: Biodegradeable panels and limited opening size in pot gear; Prohibition on gillnets; 100% retention of pollock (*Theragra chalcogramma*), Pacific cod (*Gadus macrocephalus*), demersal shelf rockfish species, rock sole* and yellowfin sole*.
Habitat degradation	Effects of bottom trawling, dredging and other gears	Marine Protected Areas; Trawl closures; No bottom trawling for pollock; Closed areas; Essential Fish Habitat (EFH) designations.
Competition for prey	Fishing on forage species	Prohibition on fishing for forage fish species, including capelin, *Mallotus villosus*, sand lance, *Ammodytes hexapterus* and krill, euphausia spp. (See also measures under marine mammal interactions)

*due 2003

Because our only possible course of action when something goes wrong with an ecosystem is to modify human intervention, there is a tendency to try to lay blame (Taylor 1999), rather than to seek explanations via a series of multiple causes. The common outcome is to blame fishing pressure, because it is often the only visible human activity. However, even when fishing pressure is reduced, stocks may not recover on time scales of relevance to human socio-economic systems (Hutchings 2001).

To consider what might be causing the continued decline of the endangered species, one needs to study the food web with which the fishery is interacting, consider possible second and third order effects that the reduction of the population of the target species might have, and one needs to consider possible spatial, temporal and life-history details which may be critical to the possible mechanisms of interaction between the fishery activity and the endangered population. The range of possible mechanisms that needs to be investigated is daunting, as is the depth to which investigations would have to be pursued in order to be conclusive. The fishery, by design, will substantially reduce the biomass of the target stock; this will inevitably change the age distribution and size distribution in the target stock. The fishing effort could be very unevenly distributed in time and space. Table 4.4 shows some of the possible outcomes that would need to be considered, and their implications for management. The possibility of these outcomes, and their implications for management, might motivate precautionary adoption of mitigating management measures or major research efforts to determine if the feared possible outcome really is manifesting itself. All this having been considered, it could also turn out to be the case that foraging was not a proximate factor in the decline. In hindsight, then, the mitigative measures would have been unnecessary, and the main management value of the research effort would then have been its role in showing that the mitigation was unnecessary.

Assuming that one can delineate the geographical boundaries of the ecosystem of interest, the simplest model for the interactions between marine mammals or birds and a fishery is shown in Figure 4.6. In this case, the target fish are assumed to be prey for both the fishery and marine mammals or birds. The interactions can be either direct, in which there is incidental mortality of mammals or birds during fishing operations, or indirect, in which the fishery removes prey that the mammals or birds would otherwise take. This food web is one used implicitly by investigators when changes in marine mammals or birds are assumed to be caused by fishery activities. One possibility in the above example, therefore, is that the fishery and the endangered species are competing for the same food resource. Practice has shown that this is very hard to demonstrate clearly, even when the fishery and the predator are known to take the same species (e.g., the Steller sea lion and pollock fisheries in the North Pacific). Nevertheless, action can be taken, including seasonal and area closures that will mitigate the potential for such interactions. An important component of such a strategy, however is a comprehensive monitoring program and the use of open and closed areas according to an experimental design to reveal, eventually, whether the intended mitigative measures are over-protective, under-protective, irrelevant, or just right for the purpose (see Section 4.3).

Table 4.4. Possible fishery effects, outcomes, and management implications for predators

Effect of the Fishery	Possible Outcome	Implications for management
substantial reduction of the biomass of the target stock	if this literally reduces the density of this stock, so that their distribution is more sparse, could this push a population that preys upon them over an energetic cost-benefit threshold, where the predator now needs to expend more energy searching for its prey than it gains from consuming the prey	this effect could be disastrous to the predator, even though the brute inventory of remaining prey biomass might superficially be judged to be enough to supply the food demands of the predator
change in the age distribution, and, therefore, the size distribution in the fully recruited size classes of the target stock, disproportionately reducing the abundance of the larger size classes	could this disproportionately impact the population of a predator with a life history stage that is specially dependent on those depleted size classes for its growth and survival?	this effect could operate at a level that was disastrous to the predator, even though the total prey biomass, summed over all prey size classes superficially was judged to be enough for the aggregate food demands of the predator population
local and temporal depletion of prey species	if this occurs at places and times, such as breeding, or rearing or wintering, areas of the predator, could it critically interfere with the predator's ability to successfully complete their life cycle	simply computing the average prey availability, averaged over space and time, might entirely miss a critical situation for the predator.

Regarding competition, 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; Clark 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).

While Figure 4.6 may be a useful conceptual tool for framing interactions, it is an overly simplified management tool for deciding on the level of human intervention in an ecosystem. Potential complexities of the primary interactions can be divided into three broad categories:

- age-specific factors in the interactions between major species;
- temporal and spatial components to the interactions; and
- availability of target species.

These complexities are included in the food web in Figure 4.7. Although this enlarged food web is more complex, it ignores environmental factors that affect the production of fish stocks. These may be biotic (e.g., the level of zooplankton, or primary production) or abiotic (e.g., different temperature regimes) factors that affect the fish stocks. Figure 4.8 captures these ideas.

Even though there may be considerable uncertainty regarding the propagation of effects through the food web, it may again be possible to use a precautionary approach, particularly when dealing with species of particular importance to a large number of predators, i.e., forage fish. In 1997, the Council adopted a regulation that prohibits directed fishing for forage fish, which are prey for groundfish, seabirds, and marine mammals (Witherell *et al.* 2000). Under this amendment, protection is provided for forage fish species such as capelin (*Mallotus villosus*) and other forage species including euphausiids (krill). Nevertheless, the generally high rates of turnover of such species mean that it may be possible to prosecute fisheries for them without limiting prey availability. For example, there has been an important fishery for krill in the Southern Ocean for decades with no apparent adverse effects on krill predators such as fur seals and penguins. Here again, however, scale may play an important role, since some predators (e.g., land based fur seals), are limited in their foraging range and season of peak demand, meaning that effects localized in time and space may be missed by the measures that we are using.

Fish habitat can also be used to demonstrate the difference between the more tractable and less tractable problems. Since the re-authorization of the Magnuson act in 1996, there has been substantial emphasis on the need to identify and mitigate adverse impacts on essential fish habitat. Whilst it may be possible to identify and map areas of important habitat, and even demonstrate and mitigate fishing activities that damage those habitats (a more tractable problem), there is currently very little research that can demonstrate the effect of different habitat impacts on the productivity of the species that use those habitats (a less tractable problem). Since just about all fishing gears that come into contact with marine substrata cause some modification of the habitat, this begs the question of how much of which habitats is it necessary to protect. Research is continuing on this topic, but it will probably be many years before results with widespread application become available (NRC 2002).

One of the least tractable problems in managing North Pacific fisheries in an ecosystem context is the effect of regime shifts. It is now generally recognized that the North Pacific Ocean exists in two macroscopic states with very different characteristics and low frequency of change of once per several decades. The simplest way to characterize the two regimes is that one state corresponds to northern waters (the Bering Sea and Gulf of Alaska) being highly productive and southern waters (Washington, Oregon and California) being less productive; the other state corresponds to northern waters being less productive and the southern waters more productive. The regimes exist for a period of about 25-30 years (so that the entire cycle from more productive to less productive and back to more productive takes about 50-60 years). It is generally agreed that there was a regime shift in1975-77, in which northern waters became more productive, and that there may have been most likely there was a regime shift toward cooler conditions around 1998/99.

The regime shift itself happens over a relatively short time scale (perhaps a year or two), but our ability to detect it requires a longer time scale (perhaps up to 5 years) due to the inherent lags in the biophysical system (see Mangel and Hofman 1999 for an example involving marine mammals). Thus, at any point in time, one needs to assess the probability that the regime in northern waters is more productive. Historically the detection lag was up to 20 years; possibly it would be less now that we know more about what to look for, and perhaps have better monitoring programs in place.

A further probabilistic element is that the responses of individual species groups to a regime shift may be quite variable. For example, walleye pollock in the Bering Sea have never experienced strong year-classes in two consecutive years, whereas GOA pollock had several years of good recruitments in the 1980s. Furthermore, pollock in the Bering Sea have had periodic strong year-classes throughout their observed history since 1964. Year-class strength for these stocks does seem to be related to environmental conditions, but it seems that inter-annual variability in environment is more important than regime changes.

Taking such changes into account in fishery management involves, among other things an appreciation and consideration of risk; there are risks involved in continuing to assess and manage on the basis of one regime when in fact it has shifted. Conversely there is also the risk of managing on the basis that a regime shift has occurred when in fact it has not (Section 4.3.4).

To address these less tractable issues in an explicit way, it is necessary to develop an analytical process that is fully integrated between the fishery, the biology of the target stock, its predators and prey, and the environment. A more integrated approach generates comprehensive scientific advice that incorporates all the potential consequences of the fishery to enable managers to take decisions that will meet the ecosystem goals of the management process. It must be recognized, however, that the practical predictive power of this kind of integrated ecosystem model will be severely constrained by limitations of availability of requisite data, limitations of our quantitative knowledge of the pertinent ecosystem processes, and limitations of our understanding of the mechanisms whereby fishing activities affect non-target species. As we explain in Section 4.3, this emphasizes the need for alternative techniques such as risk analysis and adaptive management, so that action can be taken with an acceptable chance of success in the face of irreducible uncertainty.

4.2.2.3 Prevention, correction, and the precautionary approach

We note two general situations in which managers may find themselves attempting to better account for ecosystem effects of fishing. Firstly, explicit management measures may be established in advance in order to mitigate *potential* likely adverse ecosystem effects of fishing (*preventive* action). Secondly, and more commonly, management action may be required to promote recovery from adverse impacts that have already occurred, but either were not considered when the FMP was formulated, or were not thought to be likely outcomes of the activity sanctioned under the FMP (*corrective* action). The latter situation is probably the more common.

Garcia (1996), cited in Auster (2001), has outlined these basic types of environmental management approaches, showing how their applicability may be driven by uncertainty in information and potential costs of errors (Figure 4.9). Preventive action is taken in advance of implementation of a management plan to avoid undesirable consequences that can be predicted only with a low level of uncertainty. Although the potential cost of errors can range from low to high, there is a high probability of making correct decisions. Corrective action allows uncertain processes to unfold, with a plan that unintended consequences will be mitigated if and when they occur. This is appropriate when the potential costs of the unintended consequences are low enough, and when the effectiveness of the planned mitigation can be counted on. This allows trial-and-error types of decision making in an adaptive framework. When both uncertainty and the potential cost of errors are more than low, it becomes necessary to adopt a *precautionary approach*. This weights decisions in a conservative direction, which is appropriate when the potential costs of errors are too high for the wait-and-see corrective approach, or when the uncertainty is too great to justify a fully preventive approach.. The fourth approach applies the more extreme *precautionary principle* which adopts a strong presumption in the direction of acting to prevent or mitigate highly uncertain threats when the potential cost of errors is also high, as, for example, when errors can cause irreversible or almost irreversible damage. Then, the high costs justify action even when the uncertainty is high.

The diagram in Figure 4.9 is illustrative, but something of a simplification. The most sophisticated and rigorous approach to dealing with uncertainty is the fully quantitative statistical decision theory (Berger 1985), which takes account of costs of errors of omission and errors of commission, and is very formal about the quantification of uncertainty. The fully developed theory is based on a predominantly economic model, which is not always easy to apply in a natural resources arena where non-economic values may play a large role along with economic considerations, and where some controversy may be as much over the distribution of costs and benefits among constituencies rather than just about the the aggregate balance of costs and benefits.

While it is clearly more desirable to establish measures that avoid adverse impacts before they take place (i.e., take preventive action), this is often problematic, in part because there is usually very little information available prior to the onset of fishing. Hence the level of uncertainty is worse than low. With regard to the corrective approach, fisheries may develop faster than the acquisition of data necessary to ensure that management can address mitigation of adverse impacts that subsequently arise. A fisheries development framework such as that being elaborated by CCAMLR is a useful tool in such situations. This framework incorporates a number of regulatory requirements including advance notification of an intent to participate in a fishery, research and fishery operations plans, and data collection plans for all fisheries commensurate with their current status.

4.3 A Roadmap for Ecosystem Based Management in the BS/AI and GOA: Confronting the Less Tractable Problems

4.3.1 Introduction

The approach to fishery management taking ecological and ecosystem considerations into account recognizes that stocks are distributed 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 variable abiotic environment is part of the milieu in which organisms live and fishing occurs.

To be sure, what lies ahead is a difficult task: development of management that takes ecological and ecosystem effects into account 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 (Langton and Haedrich 1997). Furthermore, this approach to management is more difficult to model because of the often nonlinear interactions in biological systems and the needs of data for estimation of many influential parameters. For example, a two-species ecosystem has 48 quantities to be studied or modeled, but a 5 species ecosystem has up to 300. Because of our limited predictive ability for complex systems, we may have to think of management options as experiments, in the sense of adaptive management (Parma *et al*. 1998). 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. With regard to developing knowledge about the ecosystem and the value of experimental approaches, there are lessons that can be learned from operations research. Solandt (1955) describes three stages in the analysis of operations and approaches to improving their effectiveness:

> "The <u>first</u> stage, and one which is sometimes missed although it is a very important one, is to discover the purpose of the operations...too often operations research fails because of the failure to do this. The <u>second</u> stage, once you have decided accurately what the system or organization is set up to do, is to try to find some measure of its effectiveness. Obviously, you cannot start to improve its performance unless you know how well it is doing now... Very often one will find the effectiveness of an organization or process has, in fact, never been measured; it has never been measured because measurement was difficult, and no on had had the time or ingenuity or urge or means to devise a method of measurement. Therefore, the operations-research worker very often has to begin his task by devising means of measuring things that have never been measured before. The <u>third</u> stage, once you have measured the total effectiveness of the process of organization, is to start on the task of trying to improve effectiveness...most operations-research workers take a good look at the process, decide which links seem to them to be the weakest, and then set out either to measure the performance of these links, or in many cases, set out to vary the factors involved in the supposedly wink link in order to see if, in fact, variations in these factors produce the expected end result. Here, you get the introduction of experiment into

> operations research; and at this stage, there is again a field for a good deal of mathematical analysis of results."

If the goal is transition from an implicit to an explicit treatment of ecological and ecosystem effects of fishing within the management framework, one may proceed in stages of increasing complexity. These are illustrated in Figures 4.3-4.5**.**

As outlined in Section 4.2, in the first stage (Figure 4.3), one takes account of both the status of the target stock and its predators and prey, but does not integrate these in an holistic management plan. In some sense, the status of prey and predators thus constrain the catch limit from the management procedure. In the second stage (Figure 4.4), one takes into account environmental effects in a more direct fashion in consideration of the status of the target stock and incorporates measures for the tractable problems described in Section 4.2.2.1. In the third stage (Figure 4.5), the environment, target stock, and its predators and prey are integrated in the assessment before the management procedure is used to determine catch limits and other management measures. At the same time, the less tractable problems identified in Section 4.2.2.2 are included. While this third stage is a goal (perhaps far off), one can proceed towards it.

4.3.2 <u>Data and Models Redux</u>

As we have explained, management that takes ecological and ecosystem aspects into account is limited by the data. Scarcity of data and fluctuating environments mean there are commonly substantial uncertainties in analyzing and predicting the effects of fishing on the ecosystem. Models can play various roles here, and the development of more complicated models does not necessarily mean improvement.

For example, data concerning incidental take are often so sparse that one 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," will be drawn unless the statistical power is taken into account (Mangel 1993). That is, incidental take data are often evaluated, misleadingly, by testing and not rejecting a hypothesis that "the incidental take had no effect on the state of the stock". ". Even in the North Pacific, estimates of bycatch from the Observer Program can be quite variable, especially for prohibited species.

The most common error in statistical 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; also see Ellner *et al.* 2002). 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 Anderson 1999; Osenberg *et al.* 1999). Management based on hypothesis testing without consideration of the power of the test is inherently biased, and 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; Ellison 1996; Hilborn and Mangel 1997; Press 1997; Malacoff 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 to Bayesian neglect of methods such as randomization, examination of residuals, and design of sample surveys. Indeed, there is psychological evidence that people find it difficult to reason about probabilities attached to hypotheses. 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.

To some extent, both data and models need to be case-specific, designed to inform managers and the public about characteristics of the ecological system that are of interest to them. Such ecological attributes need to be summarized in a collection of ecological indicators, which are quantities that can be computed from the data and which provide information about the status of the ecological attributes. (e.g., Charles 2001). Examples of such indicators (Smith *et al*. 2001) include

- Biomass / stock size
- Total mortality (catch divided by a catch limit)
- Size / age-structure
- Catch-rate
- Discard rate
- Size-spectra (using log size-classes)a
- k-dominance curves
- Coefficient of variance for total biomass
- Average trophic level
- Diversity index (e.g., Reyni or Shannon-Weiner)
- Species composition (MDS plots)
- Rate of damage
- Benthic habitat complexity
- Biomass of cover-defining species / species groups
- Reproductive success
- Ratios of piscivores to planktivores and/or demersal fishes
- Chlorophyll-a
- Redfield ratio
- Throughput
- Production / biomass
- System ominvory index
- Dominance of detritus
- Relative ascendancy
- Residence time (= biomass/(respiration+export))
- Index of Biological Integrity (IBI)

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

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

The source of these paradoxes lies in statistical issues and the relationship between models and data. Their implication 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 stock-recruitment 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-fisheries management that takes ecosystem considerations into account (Mangel *et al.* 2001):

- statistical;
- theoretical; and
- logical.

Statistical models 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 a lack of relationship between the ecological variables, but no formal allowance is made for the possibility that the model can be wrong, unless the exercise is one of model selection, where statistical constructs such as the Akaike Information Criterion can be used to select the "best" model from among a specified list of alternatives. 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 in a systematic way, so that statistical relationships based on simple random error structures 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 where the observation was obtained. 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 behavior 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 Lotka (1925) using mathematical arguments. As a logical statement it is not open to experimental verification or questioning, and it is true within the realm 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 has developed (e.g., Tuljapurkar Tuljapurkar 1990; Tuljapurkar and 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. For example, harvest can exert a selection pressure that eventually provokes an evolutionary response on the part of the target species, particularly with respect to size, growth rates, and size at maturity (Conover 2002). A thorough consideration of ecosystem effects will take this into account, and will consider further how this might affect other species that interact with the target stock.

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, as in Werner and& 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 the real world, they only cover parts of the whole.

We offer advice for dealing 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 an 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 are 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 too quickly abandoned just because of discrepancies 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 problematic 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; Bali–o 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.

Given this caveat, one should work to clearly identify clearly the factors that could be included in models. These include factors such as: the system type (e.g bay or coastal; shelf; slope/ pelagic; the nature of oceanographic forcing; the level of the ecological model (numbers of species; functional group aggregation; different ecological scenarios); anthropogenic impacts (e.g., different scenarios regarding nutrient influx); the level of fishing mortality (including gear types, and the temporal sequence of fishing); and management options.

- ***Avoid the trap of false precision***

Results or analyses that are presented with a high level of precision but which have large potential ranges (as summarized by error bounds or confidence intervals) or have great variability when they are put into operation have the potential to be misleading through a false representation of precision. That is, they suggest we know much more about the system than we really do know. For example, specifying F=0.152 in one year and F=0.148 in another suggests a level of operational precision that can rarely be met.

New directions

Embracing uncertainty and avoiding false precision in fishery management may require us to forgo the hope of precise and finely tuned management plans, opting instead for a series of indicators that can be broadly categorized and manipulated by Boolean logic. For example, Caddy (2002) describes a "basket" of indicators, each of one of which is associated with a yes/no question:

a) is total mortality in excess of the optimal mortality for the stock,
b) is spawning stock biomass less than 20% of the estimated value in the unfished case,
c) is fishing mortality larger than a specified multiple of natural mortality,
d) is recruitment much less than average recruitment, and
e) is fishing mortality more than 2/3 of F_{MSY} (or $F_{0.1}$)?

Each answer that is "yes" produces a "red" traffic light. The decision rule is that, with 5 red lights leading to closure of the fishery, and 1-4 lights lead to an open fishery with decreasing levels of fishing effort (75%, 60%, 40% and 20% of F_{MSY}). What is noteworthy here is that although the standard quantitative measures are evaluated, they are used in a non-standard way. Such a very unconventional decision rule can still be evaluated for its performance by the methods of embedding in trials of a conventional operating model as described in the section on Management Strategy Evaluation, in chapter 2.

The traffic light method has been extended to summarize the inputs and outputs of single-species assessments, along with other relevant biological, environmental, and ecological information (Halliday *et al*. 2001). It includes a fuzzy logic algorithm for making decisions. Our view is that this method is a novel visualization tool for synthesizing complex inputs and outputs. In the single-species setting, it can focus attention on a variety of information, rather than the conventional approach of looking only at the most recent estimates of fishing mortality and abundance. In a multi-species setting, the results of several assessments could be summarized for a good overview of how the management system as a whole is dealing with the individual species.

Another novel visualization tool is the AMOEBA plot, first derived in the environmental quality control literature (Collie *et al.* 2001). In this method, species and fisheries are analyzed in a principal components approach, and plotted on a single graph, indexed by the biological reference points for each species. At a glance one can see which fish species are below or above the target, and which fisheries are having the impact.
The use of biomass spectra (Duplisea *et al*. 1997) is another approach for looking at temporal variability in multiple species. As biomass is directly related to energy in the system, this approach can be used to make inferences on relationships among species.

Finally, the collection of gut contents over long time periods is resulting in data series for which multi-species models may be more amenable than at any time in the past. Advances using MSVPA (Tsou and Collie 2001) are now being made in Georges Bank, the North Sea, and the North Pacific. The Ecopath/Ecosystem suite of programs is being used around the world, including the North Pacific. It will be interesting to see if these models will develop a predictive ability to match or exceed that found in single-species assessment models.

4.3.3 Prospective Evaluation of Management Procedures

The evaluation of management procedures by extensive computer simulation (management strategy evaluation, MSE) 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 various kinds of problems. 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). Prospective evaluation via simulation in a staged approach 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.

Prospective evaluation of management procedures is especially important if one wants to conduct adaptive management, in which harvest rules are set to produce both fish and information that allows one to reduce uncertainty. In the context of the BSAI or GOA, adaptive management will involve explicit spatial experimentation with harvest rates. It is likely that such experiments will involve short term costs (e.g., reduced catch rates or a decline in population size of some predators) and the justification of those costs must be given in terms of the information that will be gained and how that information will be used to provide benefit (e.g., enhanced catch or increased size of the predator population). Consequently, prospective evaluation of consequences via modeling will provide the most convincing evidence of the value of adaptive management, short of carrying out a well-planned and long-term experiment.

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; and
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).

4.3.4 The Role of Risk Analysis

Although surveys and other various kinds of information can reduce uncertainty, it is almost guaranteed that decisions will have to be made in the context of some level of irreducible uncertainty (Mangel 2000; Jonzen *et al.* 2002). When decisions are made and the scientific evidence involves uncertainty, risk analysis provides a natural framework for structuring reasoning (Anand 2002). For fishery applications see MacCall (1998) and McAllister (1996).

Risk analysis allows decision-makers to see the range of possible situations and possible consequences of decisions, rather than just the consequence of the average. It thus makes more explicit the consequences of choices. The following steps are involved in a risk analysis:

1. Collection of information about the "state of nature": In the simplest cases, this would be estimates of the probability distribution of abundance of the target stock. With increasing levels of sophistication concerning ecosystem aspects, the state of nature might include environment (regime type), prey population sizes, and predator population sizes.
2. Decide on potential actions: These would be harvest mortality levels in the simplest (aggregate) case, and harvest mortality distributed over time and/or space in the more complicated situations.

3. Assess the outcome of the different combination of actions and states of nature: This is often best presented in the context of a matrix in which the different states of nature are columns, potential actions are rows, and the matrix entries represent the value of different outcomes.
4. Compute the expected value of different actions: in terms of the average over the states of nature of the value associated with a particular action.

Risk analysis makes explicit the uncertainty about the world, the available choices, and their consequences. It may lead to decisions that are consonant with current practice, in which case support for current practice is provided. Alternatively, it may suggest that a different approach is more appropriate.

Risk analysis becomes especially important because of regime shifts (See Section 4.2.2.2). Clearly, there are dangers if one continues to assess and manage on the basis of a productive regime when in fact it has shifted to less productive. It is also clear that one cannot assess and manage on the basis of an hypothetical 'average' environment since that is only a mathematical summary and does not manifest itself to be operated upon directly by our interventions. Similarly, there are dangers if one asserts that a regime shift has occurred (either way: if the assertion is from more productive to less productive the risk is loss of yield to the fishing community; if the assertion is from less productive to more productive, the risk is overexploitation of the target stock, with a series ecosystem effects). Risk analysis provides the framework for making these dangers clear and accountable.

Bayesian networks (Jensen 2001) provide a methodology which could be used as a management tool to make predictions and to explore the consequences of alternative scenarios for a particular fishery interacting with a particular ecosystem. This approach has its roots in expert systems rather than statistical modeling. In statistical modeling it is customary to warn mangers that a significant correlation (or regression model) between variables does not necessarily imply any causal relationship. In contrast, Bayesian networks deliberately set out to model patterns of causality. Figure 4.10 shows an influence diagram between variables in which the arrows represent causal links (Halls *et al.* 2002). In a Bayesian network, the causation does not have to be deterministic and can incorporate a degree of uncertainty. In fact, the variables are modeled as random variables and the links are probabilistic. In Figure 4.9, therefore, a link from *A* to *C* would be interpreted as meaning that the value of *A* affects the value of *C* by means of influencing the probability distribution of *C*. Due to the variety of possible linkages, quite complex patterns of association can develop, rather like in a foodweb. The roles of "response" and "explanatory" variables become blurred, with variables taking on each role in turn. In Figure 4.10, variables *E* and *D* could be regarded as "responses", and *A* and *B* as "explanatory". But *C* seems to play both roles. It looks like a response with *A* and *B* acting as explanatory variables, and it is an "explanatory" variable for *E*.

The use of Bayesian networks in resource management is relatively new, but the approach seems to show great potential, particularly for modeling the indirect effects of interventions as they propagate through complex systems, such as foodwebs and benthic communities. Historically, these models evolved largely in the artificial intelligence (AI) community, and have formed the basis of expert systems. Generally they are not tools for statistical inference, but rather they are mechanisms for encoding probabilistic causal relationships and making predictions from them (Halls *et al*. 2002). The aim of this approach would be to develop a model as a tool for guiding decision making in a variety of areas, including the conduct of the fishery, and the targeted collection of information to improve understanding of the system and its response to change.

4.3.5 <u>The Role of Marine Reserves</u>

There is currently great interest in no-take marine reserves as a tool in the management of fisheries taking account of ecological and ecosystem considerations. A number of calculations (Dugan and Davis 1993; Auster and Schakell 1997; Gunderson 1997; Hart 1997; Lauck *et al.* 1998; Mangel 1998; Horwood *et al.* 1998; Hastings and Botsford 1999; Sladek Nowliss and Roberts *et al.* 1999; NRC 1999; Mangel 2000a; Mangel 2000b; NRC 2002) have shown that the yield from fisheries that include reserves in their management can be as great as the yield from fisheries that don't. It is less clear that reserves will increase the yield of fisheries (Mangel 2000a; Mangel 2000b; Lockwood *et al*. 2002), although proponents of reserves tend to assert this to be true. However, reserves can help increase the likelihood of sustainability of the stock, and thus of the fishery (Mangel 1998; Mangel 2000a; Mangel 2000b).

In the context of fishery management that takes ecological and ecosystem considerations into account, reserves play two other extremely important roles. First, a no-take marine reserve of sufficient size will allow one to maintain a source of baseline data for stocks in the ecosystem. This is important because we should expect change to occur in ecosystems. Without having a source of baseline data in which there is no (or at least limited) human intervention, it will often be difficult to ascertain whether changes are due to fishing or other factors. Second, for stocks that have complicated social structure (e.g., sex-changing fish or harem or lek breeding marine mammals or birds), a no-take marine reserve will allow a full representation of the social structure of that stock; such social structures might otherwise be truncated by either direct or indirect effects of fishing. . The effectiveness of a reserve for conservation purposes will depend on the relationship between the reserve size, and the natural spatial structure and dispersal rates of the populations. If these spatial scales coincide, the results could be counter productive: then closed areas may result in protection within the area but an *increased* chance of depression outside.

4.3.6 <u>The Role of Monitoring</u>

Monitoring plays a crucial role in making less tractable problems more tractable. Monitoring of catch, by-catch and fishing effort is of course criticentral to the data gathering that supports the assessments of status of the target stocks. Thus we recommend that the Observer Program be maintained and improved to provide even more precise and accurate information about directed catches and bycatch of all species. Systematic and well-designed monitoring is also essential for determining the magnitudes and timing of real environmental variation, such as regime shift, and it is at the heart of all experimental approaches to ecosystem management which hopefully will increase our knowledge about the ecosystem and reveal which management strategies work and which do not.

Until recently the intensity and extent of the regular surveys conducted by ADFG and NMFS in the BSAI/GOA ecosystem were as listed in Table 4.5.

Table 4.5. Surveys delivering data that were used in the Steller Sea Lion Biological Opinion

1. Bottom trawl survey
   a. summer bottom trawl surveys of eastern Bering Sea, annual since 1972 (standardized starting 1979)
   b. summer bottom trawl surveys for Aleutian Islands, roughly triennial, but with some gaps, since 1980
   c. summer bottom trawl surveys for Gulf of Alaska, triennial since 1984
2. Hydroacoustic (echo integration-trawl)
   a. winter echo integration-trawl for pollock spawning in Shelikof Strait, annual since 1981
   b. winter echo integration-trawl for pollock biomass near Bogoslof Island, annual since 1988
3. Longline survey
   a. summer longline survey for sablefish over Gulf of Alaska upper continental slope, initiated by Japan 1979, taken over by U.S.
   b. summer longline survey for sablefish in the Aleutian Islands, biennial
   c. summer longline survey for sablefish over the eastern Bering Sea slope, biennial

More recently, motivated by the ecosystem issues connected with the Steller sea lion ESA jeopardy questions, the survey efforts have been extended and now consist of the programs listed in Table 4.6.

It is important that the surveys be continued, and perhaps extended even further to provide adequate information for addressing the ecosystem questions.

Table 4.6. Current major survey efforts in the BSAI/GOA

Survey/Season	Purpose	Eastern Bering Sea	Aleutian Islands	Gulf of Alaska
NMFS Bottom Trawl Summer	Groundfish	Annual	Biennial	Biennial
NMFS Hydroacoustic Summer	Pollock	Biennial		
NMFS Hydroacoustic Winter (Spawning Aggregations)	Pollock	Annual (Bogoslof and EBS shelf)		Annual (Shelikof Strait; expanded to Shumagins and E. Kodiak recently)
NMFS Ichthyoplankton Spring	Pollock (but others collected and recorded)	Annual		Annual Shelikof gully

Survey/Season	Purpose	Eastern Bering Sea	Aleutian Islands	Gulf of Alaska
NMFS Larval fish Summer	Pollock (but others collected and recorded)	(mid-1990s)		Annual Shelikof and Shumagins
NMFS Pelagic	Salmon			Annual
ADFG/NMFS Nearshore Small Mesh Trawl	Originally Shrimp; now general nearshore community structure			Central and Western
ADFG Hydroacoustic Winter	Pollock			Prince William Sound

4.4 Conclusions with Respect to Ecosystem Considerations

The panel was asked to consider two basic questions about the ecosystem aspects of the present NPFMC groundfish fishery management plan and the role of $F_{40\%}$ in it. These are (1) Is the approach "considerate" of ecosystem needs in the BSAI and GOA? and (2) Are data available to implement an alternative approach for satisfying ecosystem needs? Our brief response is that the MSY based approach in the setting of F_{ABC} in the current NPFMC system for groundfish management, which is consistent with the explicit OY goals of the MSFCMA, makes only a slight adjustment for *possible* ecosystem needs; while the TAC setting adjustment downward from ABC allows for considerably reduction in harvest, but the procedure for doing so is ad hoc. The available data could be used for a more ambitious, and more formalized, decision system that might be more protective of ecosystem considerations. However, the available data have not, to date, proven sufficient to demonstrate conclusively that more protection is or is not needed. Present legislative policy mandates in the MSFCMA are not explicit enough about the burden of proof in deciding between utilization and protection goals to determine how much protection of ecosystem considerations is legally required when the uncertainty about the needs for such protection is great. Other legislation, notably MMPA and ESA, is much clearer about the burden of proof and the required standards of protection for special species, and actual FMPs have been modified to conform when those regulatory frameworks have come into play. Resolution of this question for other non-target species, and for the ecosystem as a whole, will require the articulation of more specific policy.

These comments are not peculiar to the $F_{40\%}$ driven aspects of the FMP. They would apply to any single-species MSY-based, or MSY-surrogate, approach, as indeed they apply to the management of Tier 1, Tier 5 and Tier 6 stocks in the BSAI/GOA FMP where $F_{40\%}$ does not play a role. Regardless of the use of $F_{40\%}$ as a F_{MSY} surrogate, fishing so as to achieve near MSY will inevitably reduce the equilibrium biomass very substantially from the unfished condition, and will inevitably shift considerably the age and size structure of the target stock. These changes to the target stock *could* propagate through the food web, and effect large changes in the populations of other species, but the theoretical models for predicting such effects in practice have low predictive power, and the intensity of monitoring required to document such changes for particular species, and to attribute causation convincingly, require a major undertaking. Furthermore, with the exception of species listed under the ESA, there are no general policy standards for whether effects of this kind, or of any particular magnitude, are acceptable consequences of management.

4.4.1 *Adjusting the $F_{40\%}$ Role for Ecosystem Needs*

The $F_{40\%}$ approach to estimating the ABC, by itself, is inherently a single- species approach. It is thought that, for most of the target species in the FMP, a fishing mortality rate of $F_{35\%}$ would be appropriate for achieving near MSY, under conditions of unchanged oceanographic regime. The main exceptions among the target species are the rockfish, which apparently need a considerably lower fishing mortality rate to avoid overfishing. That the actual target fishing rate is $F_{40\%}$ rather that $F_{35\%}$ creates some additional margin of safety, from a single- species perspective, for target species excluding rockfish. The decision to use $F_{40\%}$ rather than $F_{35\%}$ was deliberately protective, and was intended to function as a buffer against several sources of uncertainty, including the concern that theoretical models have shown that managing each species for its single -species MSY will not achieve MSY for the aggregate. Nevertheless, it is not clear *how much* of the margin between $F_{35\%}$ and $F_{40\%}$ was "allocated" to ecosystem considerations. Nor was a calculation carried out to demonstrate what amount of escapement is needed for ecosystem purposes, or to assess whether the margin between fishing at $F_{35\%}$ and $F_{40\%}$ supplies this amount.

The TAC setting process has provisions for adjusting the allowed catch downward from the ABC, and in practice the TAC is adjusted downward. Such adjustments are made for considerations of by-catch, protected species, and general concern about the ecosystem. Again, except for the adjustments in response to the very specific requirements of ESA, it is not clear how the magnitude of this downward adjustment of the TAC from a $F_{40\%}$-based ABC is chosen, how much of it is attributed specifically to ecosystem considerations, and whether there are specific grounds for believing the magnitude is enough for those purposes.

4.4.2 *Alternative Approaches to Accommodating Ecosystem Needs*

It is easy enough to say that a management system could be made more protective of ecosystem properties by building additional margins of safety into a fishing mortality rate rule (such as shifting to $F_{50\%}$ or $F_{60\%}$ for example) or stipulating a more stringent threshold on the total allowed depression of equilibrium biomass (such as the CCAMLR limit). But current knowledge does not allow precise scientific specification of what margin or threshold would be appropriate to achieve what level of protection of various ecosystem properties.

Modeling can offer up hypothetical scenarios to illustrate various possible outcomes, but multispecies ecosystem modeling has not yet developed to the point where it has documented predictive power in real applications. Nevertheless, this modeling is very interesting on several grounds, and continued investment in developing and testing such models is warranted.

At present, we essentially face a sliding scale of possible ecosystem protective measures, where the alternatives are largely policy choices. Current policy guidance is insufficiently specific, and the available science is insufficiently conclusive about the precise magnitudes of expected effects. Given the scientific uncertainty, there is merit in approaching ecosystem management in the spirit of cautious experimentation supported by a large investment in carefully-designed monitoring.

In this section we explored a variety of frameworks for expressing ecosystem goals, and a spectrum of management approaches that might be conducive to achieving those goals. The large uncertainties, and the overt appeal to experimental management puts a high premium on continuing and expanding the regular monitoring in this ecosystem, along with surveys of the fishery resources, and oceanographic survey programs. Marine protected areas (reserves) could serve two very valuable functions in an adaptive management approach. They would provide a "control" area in the experimental design, and they provide a refuge that would serve an insurance function in the event that any treatments develop unintended consequences. There is further some indication, which deserves closer investigation, that a system of such reserves can be put in place at little or no cost to total harvest.

Currently available data might well be adequate for implementing imaginable ecosystem control rules. But currently available data almost certainly are not sufficient for specifying the quantitative details of such general ecosystem control rules in the absence of more explicit policy formulations. We can hope that continued research and monitoring will improve our general understanding of the BSAI/GOA ecosystems. There is reason to expect that the present increases in research directed specifically at population dynamics of the Steller sea lion will bring more satisfactory resolution to the vexing outstanding questions about causes of the decline of *that* population and its possible relation to the fishery. Elucidation of broader aspects of the ecosystem, and their relationship to the fishery, may prove to be an even greater challenge.

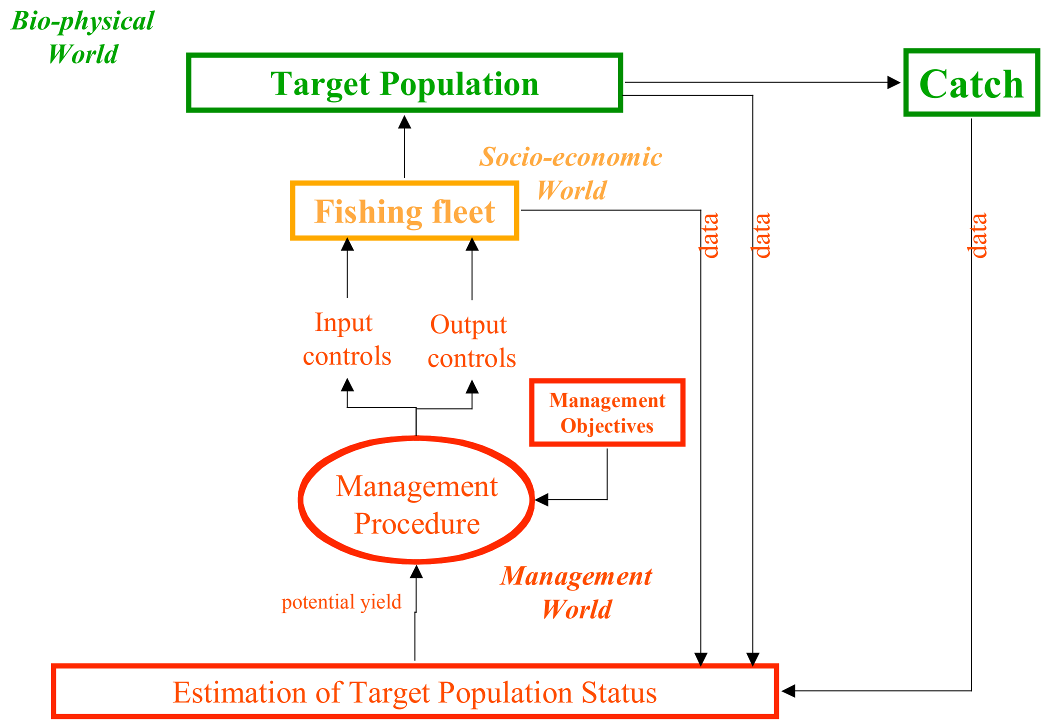


Figure 4.1. The conventional assessment world view, in which nearly all fishery management is currently done, recognizes the biophysical world in which the stock exists, the socio-economic world of the fishing community that takes the stock, and the management world in which catch limits are determined.

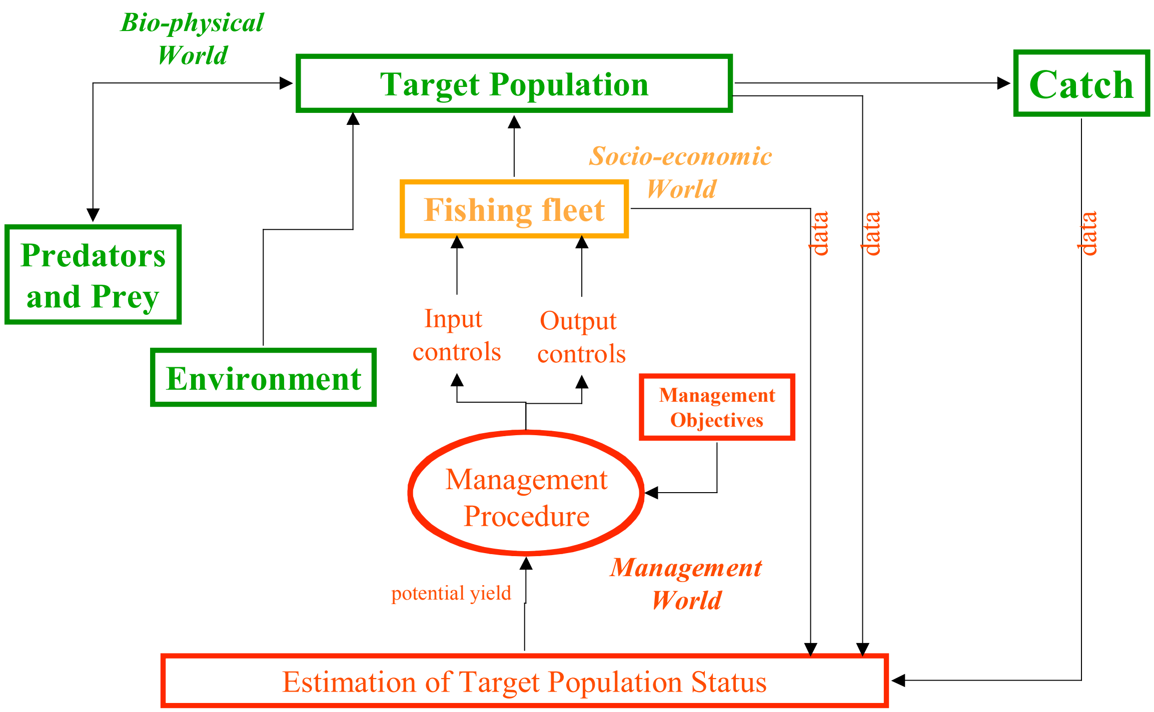


Figure 4.2. In the implicit ecosystem effects world view, we recognize that target species in fisheries are generally prey for other components of the ecosystem. While management objectives only take such predator needs into account in a very general way, the implicit view is cognizant of those needs.

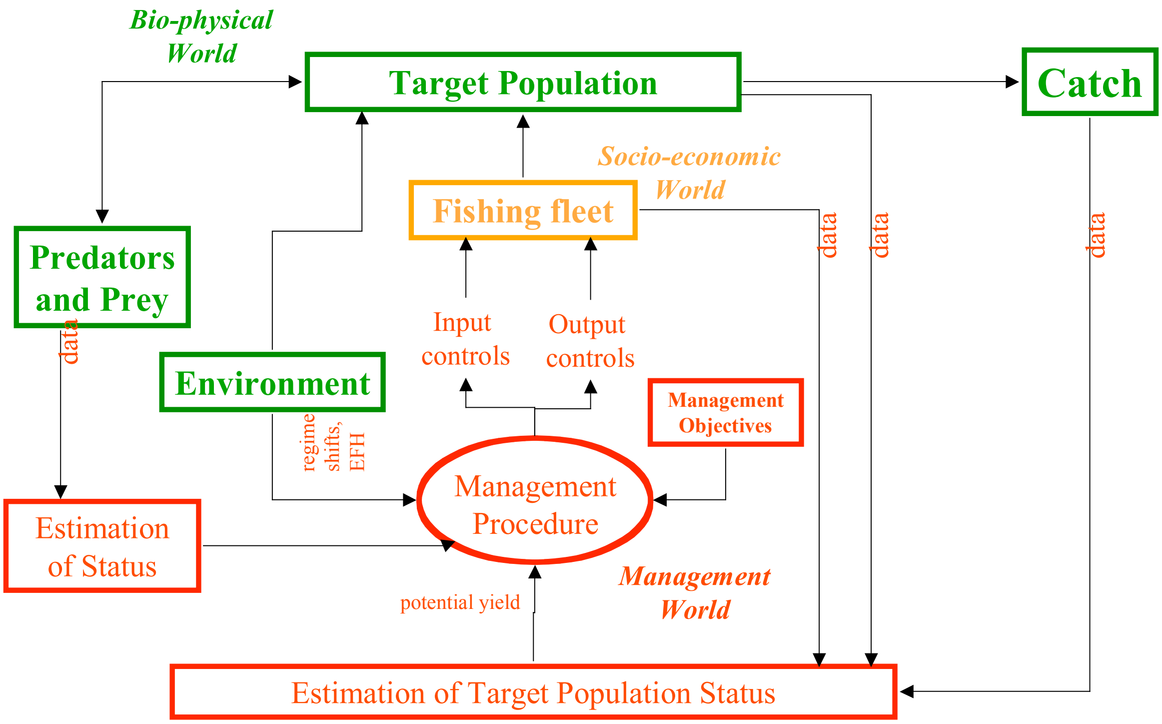


Figure 4.3. In the first stage of management that takes ecological and ecosystem considerations into account in an explicit manner, both the status of the target stock and its predators and prey are considered, but these are not integrated in a holistic management play. In some sense, then status of prey and predators thus constrain the catch limit from the management procedure.

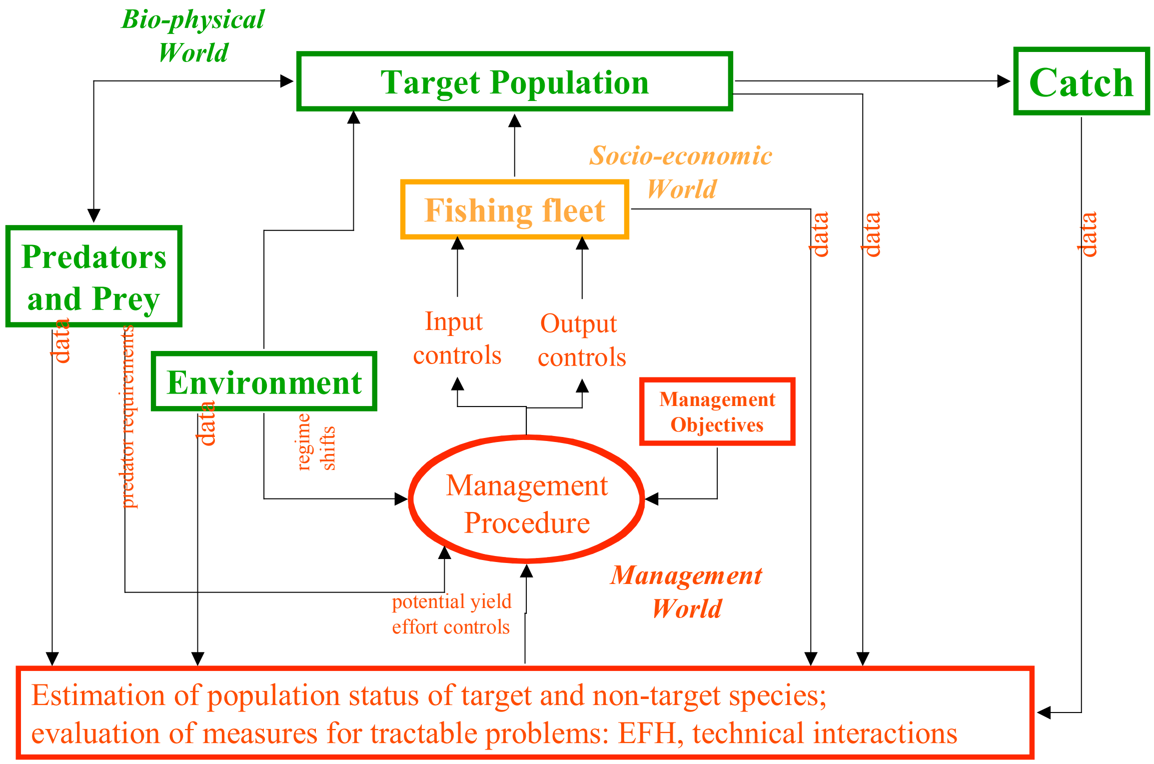


Figure 4.4. In the second stage of explicit consideration of ecological and ecosystem effects, one takes into account environmental effects in a more direct fashion in consideration of the status of the target stock and incorporates measures for the tractable problems described in Section 4.2.2.1.

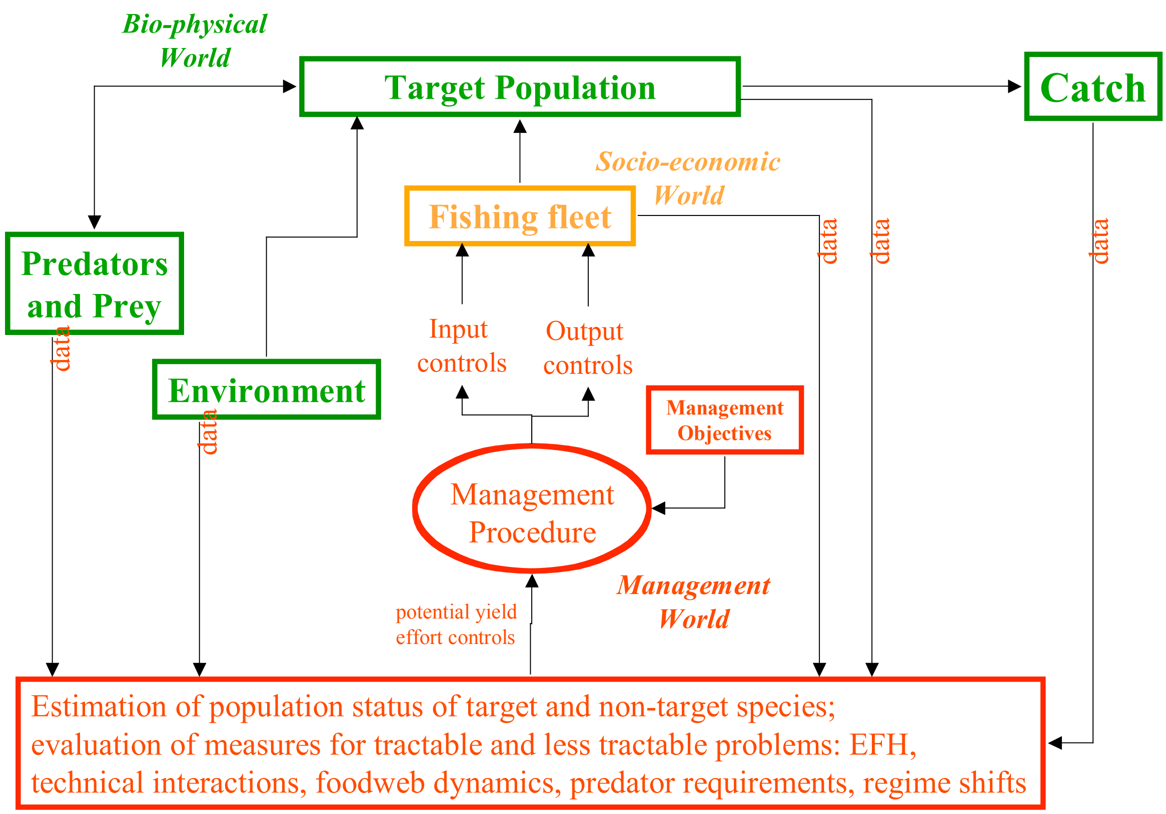


Figure 4.5. In the third stage, the environment, target stock, and its predators and prey are integrated in the assessment before the management procedure is used to determine catch limits. At the same time, the less tractable problems identified in Section 4.2.2.2 are included

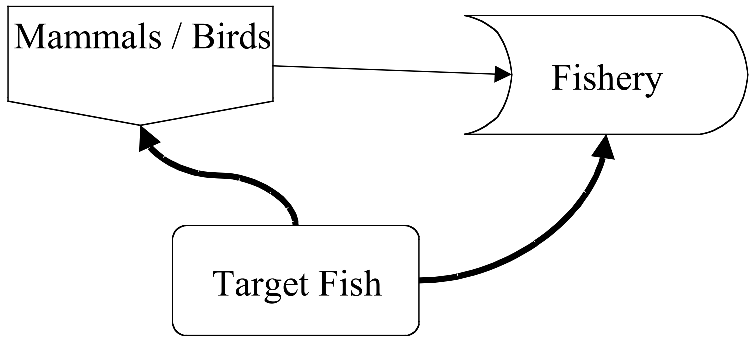


Figure 4.6. The simplest model for the interaction between marine mammals and birds and a fishery. In this case, the target fish are assumed to be prey for both the fishery and marine mammals or birds. The interactions can be either indirect, in which the fishery removes prey that the mammals or birds would otherwise take, or direct, in which there is incidental mortality of mammals or birds during fishing operations.

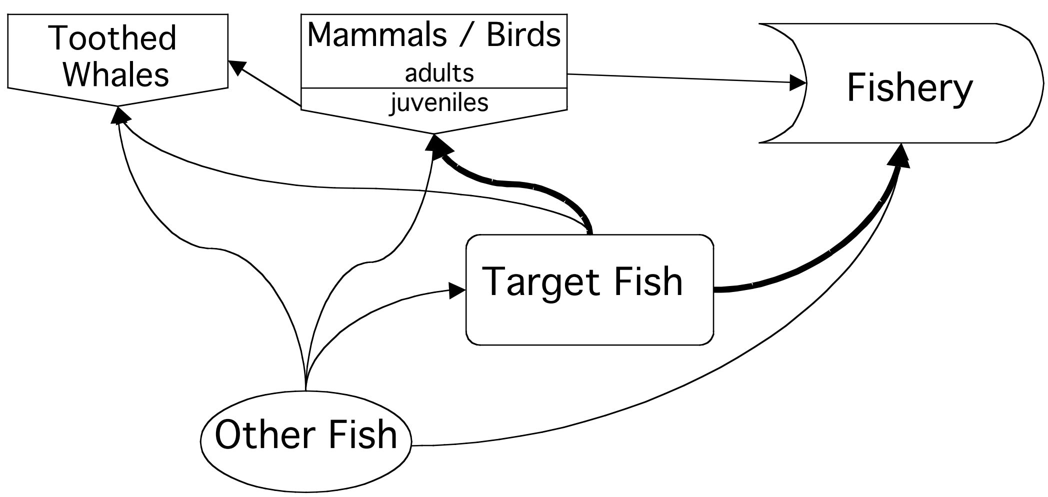


Figure 4.7. An elaboration of the simplest food web to account for age-specific factors in the interactions between major species, temporal and spatial aspects of the interactions, and availability of target species.

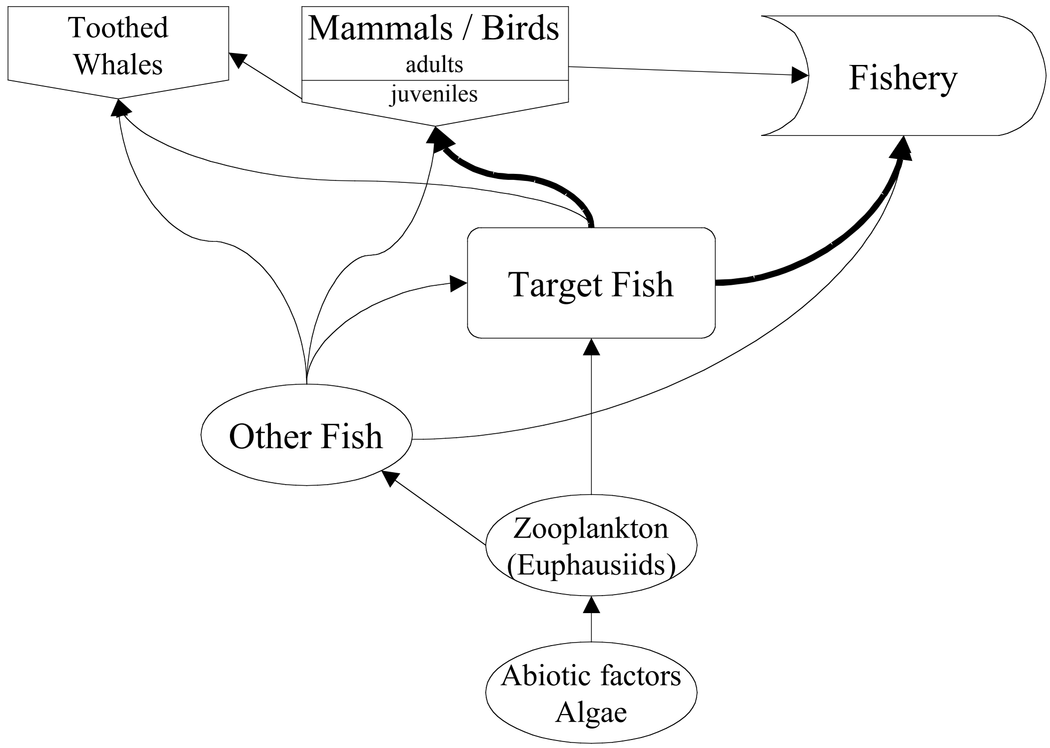


Figure 4.8. A food web that includes environmental factors.

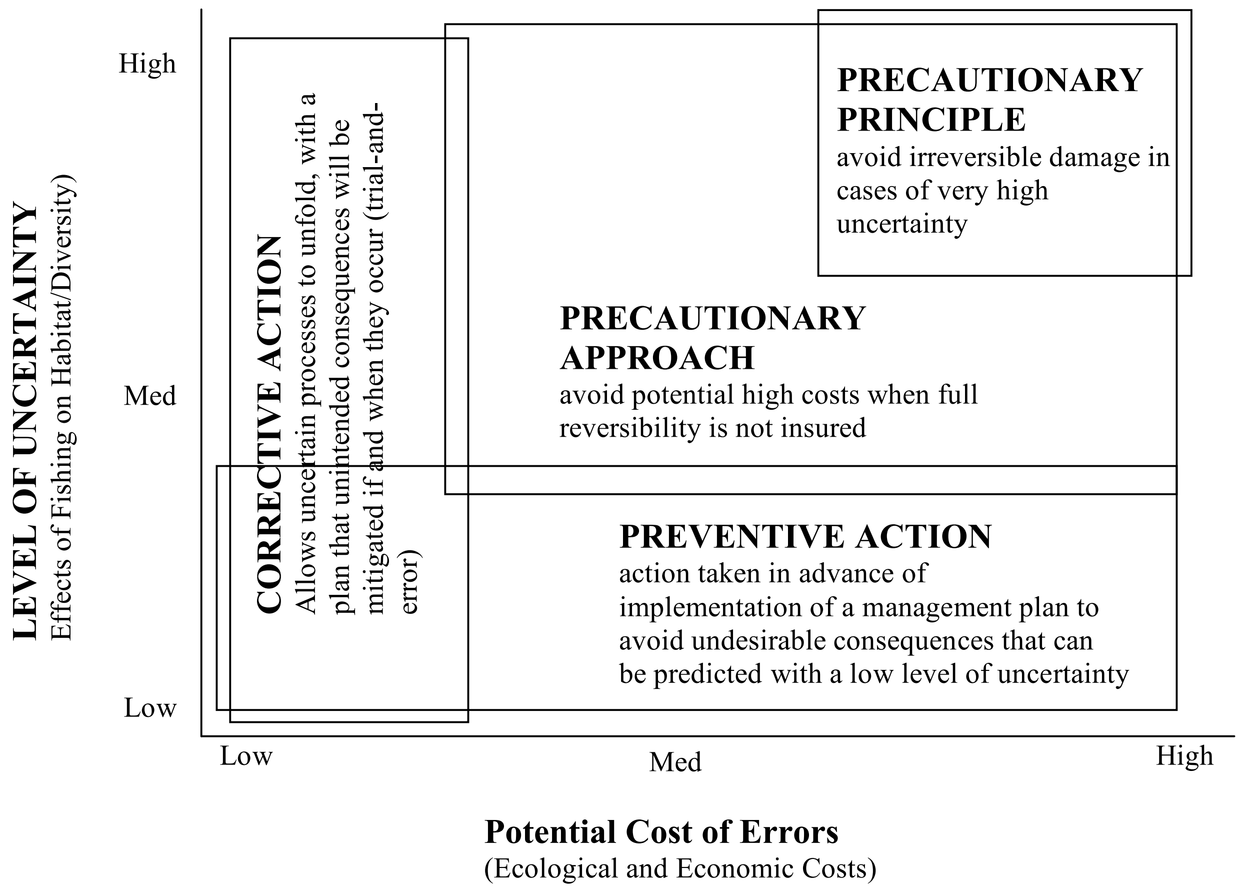


Figure 4.9. A comparison of management approaches that are based on levels of uncertainty and the potential cost of errors (Auster 2001, modified from Garcia (1996)). Costs should be considered in both ecological and economic terms (e.g., lost economic opportunities when the ecological functions of habitat are damaged or regulatory actions limit fishing due to the endangerment of particular taxa). The level of uncertainty is based on our understanding of the effects of fishing practices on both habitat metrics and biological diversity.

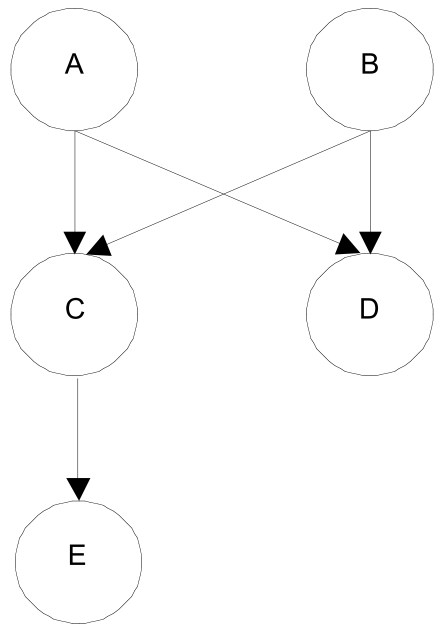


Figure 4.10. Bayesian networks: indirect mediation of effects of explanatory variables

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