A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management

Pamela M. Mace

2001

SEDAR63-RD05

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Pamela M Mace

National Marine Fisheries Service, Woods Hole, MA, USA

Correspondence:
Pamela Mace,
National Marine Fisheries Service, 166 Water Street, Woods Hole, MA, USA
E-mail: pamela.mace@noaa.gov
Tel.: +1508 495 2357
Fax: +1508 495 2393

Received 4 Oct 2000
Accepted 18 Dec 2000

Abstract

In 1977, Peter Larkin published his now-famous paper, 'An epitaph for the concept of maximum sustained yield'. Larkin criticized the concept of single-species maximum sustained yield (MSY) for many reasons, including the possibility that it may not guard against recruitment failure, and the impossibility of maximising sustainable yields for all species simultaneously. However, in recent years, there has been a fundamental change in the perception of the fishing mortality associated with MSY (F_{MSY}) as a limit to be avoided rather than a target that can routinely be exceeded. The concept of F_{MSY} as a limit is embodied in several United Nations Food and Agriculture Organization (FAO) agreements and guidelines, and has now been incorporated into the US Magnuson–Stevens Fishery Conservation and Management Act. As a result, the United States now requires the development of overfishing definitions based on biological reference points that treat the F_{MSY} as a limit reference point and must also define a lower limit on biomass below which rebuilding plans with strict time horizons must be developed. This represents a major paradigm shift from the previously mandated (but often unachieved) objective to simply maintain fishing mortalities at levels below those associated with recruitment overfishing. In many cases, it requires substantial reductions in current fishing mortality levels. Therefore, the necessity of the new paradigm is continually questioned. This paper draws on examples from several fisheries, but specifically focuses on the recent US experience illustrating the practical difficulties of reducing fishing mortality to levels below those corresponding to MSY. However, several studies suggest that even more substantial reductions in fishing mortality may be necessary if ecosystem considerations, such as multispecies interactions, maintenance of biodiversity and genetic diversity, and reduction of bycatch and waste, are taken into account. The pros and cons of moving beyond single-species assessment and management are discussed. A US plan for improving stock assessments indicates that even a 'basic' objective such as 'adequate baseline monitoring of all managed species' may be extremely costly. Thus, the suggestion of Larkin (1983, 1997) that the costs of research and management should not exceed 10–20% of the landed value of the catch may preclude comprehensive ecosystem management. More importantly, neither single-species nor ecosystem-based fisheries management is likely to improve appreciably unless levels of fishing capacity are aligned with resource productivity, as is currently being promoted by FAO and several individual nations.

Key words biological reference points, ecosystem approaches, fisheries management, fish stock assessments, maximum sustainable yield, precautionary approaches

Introduction

The previous three Larkin lectures (Beverton 1998; Caddy 1999; and Cochrane 2000) have focused on global syntheses of the current problems and solutions in marine fisheries. While I have included some elements of global synthesis in this, the 4th Larkin lecture (initially entitled 'MSY reborn, but with a new identity: is it necessary, is it sufficient?'), I wanted to do something a little different and use some of Larkin's own work as a point of departure. In particular, one of Larkin's better-known papers is his epitaph to maximum sustained yield (equivalently, maximum sustainable yield, MSY), in which he stated that it was time to lay the concept to rest and try something different (Larkin 1977). However, although approaches to fisheries science and management have evolved considerably since the late 1970s, MSY has remained an integral part of most approaches, at least in concept, if not in actuality. Recently, there have been several important developments in the fields of fisheries science and management. Two of these are the generation of the precautionary approach to fisheries management, and emerging awareness of the ecosystem effects of fishing which has resulted in calls to define and conduct ecosystem-based fisheries management. Ecosystem-based management (EBM) was another topic about which Larkin produced some memorable papers in which he recognised the difficulties of defining operational ecosystem objectives and producing practical guidance for management (e.g. Larkin 1996).

One of the primary messages in this paper is that evolution of the precautionary approach has resulted in a new interpretation of MSY, that also is more consistent with EBM. However, the paper focuses more on single-species approaches to stock assessment and management, as single-species approaches are still applied more often than ecosystem approaches. (Also, this paper was originally presented at the University of British Columbia to an audience including members of the University of British Columbia Fisheries Centre who are intensively developing and applying mass-balance ecosystem models such as EcoPath, EcoSim, and EcoSpace to marine ecosystems throughout the world (e.g. Pauly *et al.* 2000). It would have been like preaching to the choir to focus on the merits of ecosystem approaches).

The first part of this paper is primarily concerned with the evolution of the concept of MSY in the context of single-species models and the recent development of the precautionary approach to fisheries management. This evolution has resulted in the fishing mortality associated with MSY (F_{MSY}) now being thought of as a limit to be avoided rather than a target that can routinely be exceeded. I then address the question 'Is it necessary to treat F_{MSY} as a limit rather than a target in order to ensure responsible management of marine resources?'. The next question is 'Is it sufficient to simply harvest all marine resources at levels below their single-species F_{MSY} in order to satisfy broader ecosystem objectives?'. This leads into a discussion of single-species vs. ecosystem approaches to fish stock assessments and fisheries management. Finally, I speculate about the future of marine fisheries science and management, and outline what I feel to be three basic prerequisites for successful ecosystem approaches to assessing and managing marine resources.

MSY in brief

The foundation for the concept of MSY was laid down in the 1930s by Russel (1931), Hjort *et al.* (1933), Graham (1935), and others (see Smith, T. 1994). Since then, there have been many comprehensive accounts written of its use and misuse. Its popularity increased considerably in the 1950s with the advent of surplus-production models which explicitly estimate MSY (e.g. Schaefer 1954). Production models have continued to be applied ever since although to different degrees in different forums, and in many cases they have been replaced by more complex age-structured models that may also provide estimates of MSY, albeit with more data. Because of its apparent simplicity and logic, and the availability of estimation techniques requiring relatively limited data, and in the absence of other management goals with similar qualities, MSY was adopted as the primary management goal by several intergovernmental organisations (e.g. IWC, IATTC, ICCAT, ICNAF) and individual countries. Even while the scientific community was beginning to question the general utility of MSY (e.g. Larkin 1977; Sissenwine 1978), it was integrated into the important 1982 United Nations Convention on the Law of the Sea, thus paving the way for integration into national fisheries acts and laws.

Based on work by Larkin, Sissenwine and others, Punt and Smith (2001) divided the criticisms of MSY into three categories: (i) estimation problems; (ii) the appropriateness of MSY as a management goal; and (iii) the ability to effectively implement harvest strategies based on MSY. Estimation problems arise due to poor assumptions in some models, lack of contrast in data, and lack of reliability of the data. As a management goal, the static interpretation of MSY (i.e. MSY as a fixed catch that can be taken year after year) is generally not appropriate because it ignores the fact that fish populations undergo natural fluctuations in abundance and will usually ultimately become severely depleted under a constant-catch strategy. Thus, most fisheries scientists now interpret MSY in a more dynamic sense as the maximum average yield (MAY) obtained by applying a specific harvesting strategy to a fluctuating resource. Such harvesting strategies include constant escapement strategies, constant catch strategies, constant fishing mortality strategies, and variable fishing mortality strategies. Some of these harvesting strategies may be inferior to others in theory, in the sense that they are only approximations to the 'global' MAY, but they may have compensatory practical advantages. Numerous theoretical modelling studies have demonstrated that, of the first three strategies, constant escapement tends to give the largest MAY, but at the expense of high interannual variability in yields (Getz *et al.* 1987). While constant fishing mortality strategies may not achieve the 'global' MAY, modelling results suggest that they often come close to this level, that they result in relatively low interannual variability in yields, and that they are relatively robust to estimation errors (Walters and Parma 1996).

Nowadays, constant escapement strategies are most commonly recommended for Pacific salmon, and constant fishing mortality strategies for other finfish. Thus, management goals for nonsalmonids are commonly framed in terms of F_{MSY} (which generates an approximation to MAY, contingent on the adoption of a constant fishing mortality harvest strategy), rather than MSY itself. Despite refinement of the concept, neither MSY nor F_{MSY} may be considered appropriate management goals for several reasons. For example, F_{MSY} can be close to $F_{extinction}$, the fishing mortality that will lead to stock extinction (e.g. Cook *et al.* 1997), in which case, management based on MSY is unlikely to guard against recruitment failure. In addition, it is impossible to attain MSY for all species in an assemblage simultaneously and it ignores economic and social considerations (Larkin 1977).

In terms of the implementation problems outlined by Punt and Smith (2001), the criticisms that have been levelled at MSY can also be levelled at many other management strategies where it is common for management targets to be overshot.

Ensuing developments

Other biological reference points

While MSY was being promoted as a fishing target, other biological reference points were being developed as fishing targets or limits, and as indicators of overfishing. Most of these fall into two categories: proxies for MSY (or F_{MSY} or B_{MSY}, where B_{MSY} is the average biomass associated with an F_{MSY} harvest strategy), and indicators of recruitment overfishing. Proxies for F_{MSY} include natural mortality (M), $F_{0.1}$ and F_{max} from yield-per-recruit analysis, and $F_{30\%}$ and $F_{40\%}$ from spawning-per-recruit analysis. Common indicators of recruitment overfishing include $F_{10\%}$, $F_{20\%}$ and $F_{30\%}$ from spawning-per-recruit analysis, and the fishing mortality associated with the slope at the origin of a stock–recruitment relationship, $F_{extinction}$ (called F_τ by Mace 1994 and F_{crash} by ICES 1997a). The formulation, utility and limitations of these reference points have been reviewed in numerous publications including those of Caddy and Mahon (1995) and Gabriel and Mace (1999).

I believe the use of biological reference points as indicators of overfishing has been taken far more seriously and has achieved more success in ensuring rational exploitation of marine resources than the use of MSY or its proxies as fishing targets. Part of the reason is that the consequences of overfishing (exceeding biological limits) can be readily portrayed and appreciated in terms of risk to the resource, while the consequences of somewhat exceeding annual fishing targets are generally considered inconsequential and the cumulative effects of annual overshoots are underappreciated. Even though many of the biological reference points used as indicators of overfishing suffer from much the same limitations as MSY-based reference points (e.g. data requirements, estimation difficulties, and single-species focus), the notion of avoiding the biological limits of marine resources is difficult to argue against.

Definitions of overfishing

In 1989, the US National Marine Fisheries Service (NMFS) published its first set of guidelines on interpreting 'National Standard 1' of the US Magnuson Fishery Conservation and Management Act (MFCMA). National Standard 1 states that '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 NMFS' initial interpretation of the requirement to prevent overfishing was that it was recruitment overfishing that was to be avoided. Although there was considerable flexibility allowed in defining recruitment overfishing, by far the majority of Fishery Management

Plans (FMPs) used one of two reference points from spawning-per-recruit analysis: either $F_{20\%}$ or $F_{30\%}$. These were set as limits on allowable fishing mortality rates, although they were often treated as (initial) targets in fisheries where realised fishing mortality rates were well in excess of the new 'limits'. It did not happen overnight, but eventually fishery management councils began to take serious action to reduce fishing mortality rates in several fisheries that had been exploited at extremely high rates for many years. Thus, the requirement to define and prevent recruitment overfishing often signalled the beginning of responsible management in cases where it could be argued that little existed before. This is particularly true in some regions of the United States.

The precautionary approach

While US fishery management councils were grappling with the requirement to prevent recruitment overfishing, and beginning to achieve success in reducing excessive fishing mortality rates, the United Nations was negotiating the Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas, the United Nations Implementing Agreement on the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks (the Straddling Stocks Agreement; United Nations 1995), and the United Nations Food and Agriculture Organisation (FAO) International Code of Conduct for Fisheries (FAO 1995). The first two of these are intended to be binding agreements, whereas the third is voluntary. A precautionary approach to the management of fisheries is considered to be an integral part of all three agreements. The precautionary approach is difficult to define succinctly because it has so many facets. The FAO (1996) compiled 137 paragraphs describing the features of the precautionary approach as it relates to capture fisheries and species introductions. Restrepo *et al*. (1999) made the following attempt at a concise yet all-encompassing definition 'In fisheries, the precautionary approach is about applying judicious and responsible fisheries management practices, based on sound scientific research and analysis, proactively (to avoid or reverse overexploitation) rather than reactively (once all doubt has been removed and the resource is severely overexploited), to ensure the sustainability of fishery resources and associated ecosystems for the benefit of future as well as current generations'.

For the purposes of the current paper, the most salient feature of the precautionary approach is contained in Annex II of the Straddling Stocks Agreement, which specifies the need for target and limit reference points in fisheries management, the need to ensure that the risk of exceeding limit reference points is very low, and the suggestion that (paragraph 7) 'the fishing-mortality rate which generates maximum sustainable yield should be regarded as a minimum standard for limit reference points'. As will be shown below, in some arenas this paragraph has come to be identified as a central feature of the precautionary approach to fisheries management.

In the United States, the MFCMA was reauthorised in 1996 as the Magnuson–Stevens Fishery Conservation and Management Act (MSFCMA), with many significant revisions. Although the MSFCMA never explicitly refers to the precautionary approach, one of the seemingly more simple revisions has been perceived as being in conformance with paragraph 7 of Annex II of the Straddling Stocks Agreement. The previous definition of optimum yield '. . . the maximum sustainable yield from the fishery, as modified by any relevant economic, social, or ecological factor' was revised by changing a single word; the word 'modified' was replaced by 'reduced'. Whether the US Congress ever intended that this one-word change would be interpreted to mean that fishing mortality rates should never exceed F_{MSY} (or, more generally, the fishing mortality rate associated with an MSY control rule; see NMFS 1998; Restrepo *et al*. 1998), or whether they understood the consequences, is currently a matter of debate as the Act is again up for reauthorisation. Certainly, the fishery management councils and the fishing industry dislike the reduced flexibility afforded by this change in the law and NMFS' interpretation of it.

Harvest control rules

The call to develop precautionary approaches to fisheries management has been answered by many intergovernmental organisations and individual countries. The International Council for the Exploration of the Sea (ICES) and the Northwest Atlantic Fisheries Organisation (NAFO) were amongst the first intergovernmental groups to set up scientific working groups to investigate and provide recommendations on interpretation and implementation of the precautionary approach (ICES 1997a, 1998; Serchuk *et al.* 1997). In both cases, a key recommendation concerned the development of operational harvest control rules–rules which specify a pre-agreed course of management actions depending on the status of the resource (Gabriel and Mace 1999). The recommended ICES and NAFO harvest control rules have many features in common (Mace and Gabriel 1999): emphasis on single-species or species groups, reference points based on both fishing mortality (F) and biomass (B), and inclusion of both limit and target reference points with the (precautionary) target fishing mortality being set below the limit fishing mortality as a function of the amount of uncertainty in stock status. The generic control rule recommended for NAFO by Serchuk *et al.* (1997) is reproduced in Fig. 1. The primary difference between the NAFO and ICES approaches is that the NAFO working group appears to have taken a literal interpretation of Annex II of the Straddling Stocks Agreement and accepted F_{MSY} as a limit reference point, whereas the ICES working group considered a fishing mortality rate akin to $F_{extinction}$ to be a more appropriate limit (Mace and Gabriel 1999).

Obviously, these two approaches could be made equivalent if different probabilities of avoiding or exceeding the two limits are employed. Little guidance on appropriate probability levels has been given in the published literature, but it makes sense that there should be only a very small probability of exceeding $F_{extinction}$ in any given year (e.g. no more than 0–5%). More work is needed to provide guidance on the appropriate probabilities with which F_{MSY} should be avoided. As F_{MSY} and $F_{extinction}$ are confounded, it is important to ensure that a given probability level assigned to avoidance of F_{MSY} does not result in an unacceptably high probability of exceeding $F_{extinction}$. Modelling efforts by Mace (1994) indicated that (deterministic) F_{MSY} never exceeded 47% $F_{extinction}$ for a wide range of combinations of life history characteristics. However, situations not covered by the analysis (e.g. fisheries where the age of recruitment is considerably less than the age of maturity, or stock-recruitment relationships other than the Beverton–Holt and Ricker types) may result in the two reference points being much closer. Indeed, Cook *et al.* (1997) estimated an F_{MSY} of 0.77 and an $F_{extinction}$ (their F_{crash}) of 0.91 for North Sea cod based on the Shepherd stock-recruitment relationship. (Estimates from the same data using Beverton–Holt and Ricker stock-recruitment functions were much more widely separated).

The approach adopted in the United States in response to the need for precautionary approaches to management and the 1996 reauthorisation of the MSFCMA has also included development of target and limit reference points and harvest control rules (NMFS 1998; Restrepo *et al.* 1998). Revised National Standard Guidelines published by NMFS in May 1998 now require the specification of objective and measurable 'status determination criteria' for each managed stock or stock complex that include a maximum fishing mortality limit that cannot exceed the level associated with an MSY control rule, and a minimum stock size threshold that must be in the range 0.5–1.0 B_{MSY}, depending on the rate at which

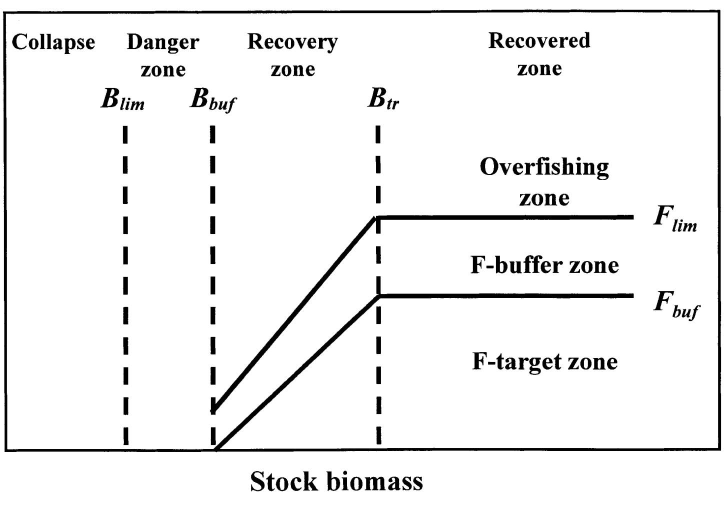

Figure 1 Generic control rule devised for the North-west Atlantic Fisheries Organisation (NAFO) illustrating potential biomass targets, thresholds and limits, and fishing mortality targets and limits (*lim*, limit; *buf*, buffer; *tr*, target for recovery). Redrawn from Serchuk *et al.* (1997).

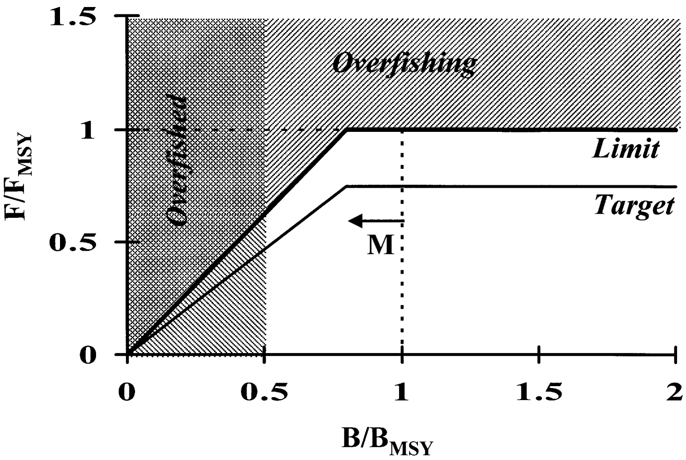

Figure 2 Default harvest control rule suggested by Restrepo *et al.* (1998). The limit and target fishing-mortality rates start to decline at $(1-M)(B/B_{MSY})$, reflecting the observation that fisheries resources tend to fluctuate in relation to their rate of natural mortality (M). Forward hatching indicates the area where overfishing is occurring, back hatching indicates the area where the stock is considered to be overfished, and cross hatching indicates the area where both are occurring simultaneously.

a stock is capable of rebuilding. Exceeding the fishing mortality limit constitutes overfishing, while falling below the stock size threshold means that the stock or stock complex will be considered overfished. In the former case, fishing mortality must be reduced below the limit as quickly as possible; in the latter case, a rebuilding plan is required to restore the stock to at least the B_{MSY} level. In addition, targets should be below limits. A generic US harvest control rule adapted from Restrepo *et al.* (1998) is shown in Fig. 2. Note that falling below the biomass threshold does not necessarily imply that fishing must cease. The control rule shown in Fig. 2 would never require a complete fishery closure, although fishing mortality targets and limits become very low as the stock size approaches zero. Variations on this control rule have the left limb of the control rule crossing the biomass axis above zero, implying the need for a complete fishery closure at some minimum biomass level, a potentially very contentious issue if any stock were to approach such a limit.

The National Standard Guidelines (NMFS 1998) are less explicit about fishing targets, except to state that targets should be below limits. The default suggested by Restrepo *et al.* (1998) is that the fishing mortality target should be 75% of the limit, based on analysis of an age-structured deterministic model that demonstrated that a fishing mortality of 75% F_{MSY} would result in yields around 94–98% MSY and biomass levels of about 125–131% B_{MSY} over a wide range of life history characteristics. However, it was intended that the default limit and target control rules be superseded by fishery-specific modelling studies. In fact, a number of innovative approaches have subsequently been devised. Two will be mentioned here. The first addresses the need to set biomass limits at levels from which a stock can rebuild within a certain period of time, generally less than 10 years. Cadrin (1999) developed a procedure demonstrating the trade-offs between the minimum biomass threshold and the maximum fishing mortality that would allow rebuilding from this threshold over periods of 1–10 years (Fig. 3). This is a

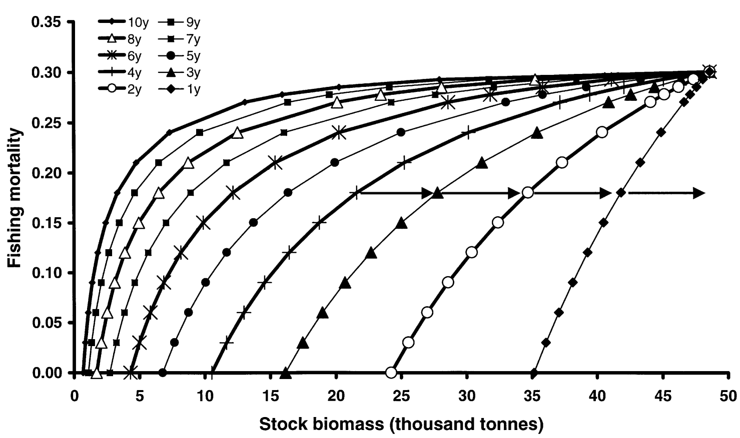

Figure 3 Rebuilding isopleths of maximum fishing mortality and minimum stock biomass that allow rebuilding to B_{MSY} over several time horizons (1–10 years) for Georges Bank yellowtail flounder. Arrows indicate a 4-year rebuilding scenario from the current biomass level. From Cadrin (1999).

useful way of illustrating alternatives for parameterising the control rule.

The second example of an improved basis for harvest control rules is that which has been developed in Amendment 8 to the (Pacific) Coastal Pelagic Species FMP based on work by Jacobson and MacCall (1995) and Jacobson and Parrish (see PFMC 1999). Coastal pelagics such as sardines and anchovies undergo extremely large fluctuations in biomass that are known to be linked to environmental and climatic factors. Thus, Jacobson and MacCall (1995) developed a relationship between F_{MSY} and average sea surface temperatures for use in control rules for these species. Dynamic interpretations of MSY are essential for such highly variable species in order to accommodate changes in important environmental variables, particularly those leading to regime shifts.

Throughout the rest of the world, a number of other fisheries organisations are also in the process of developing precautionary approaches and harvest control rules that will probably involve MSY-based reference points; these include NASCO, ICCAT, MHLC, SEAFO and many national governments (Mace and Gabriel 1999).

The MSY transition

To summarise, the interpretation of MSY has proceeded through three stages: from (i) a fixed amount that could be taken each year indefinitely (the maximum constant yield), to (ii) the maximum average yield (or an approximation to it) that can be taken by varying catches in response to fluctuations in stock size (e.g. by fishing at a constant rate of F_{MSY}), to (iii) F_{MSY} being a limit to be avoided, rather than a target that can routinely be exceeded. F_{MSY} and related proxies are now an integral part of harvest control rules designed to foster rational exploitation of marine resources.

It was when MSY was widely considered as a viable fishing target that Peter Larkin coined his now-famous 'Epitaph to MSY' (Fig. 4). My humble response, which reflects the new perception of MSY as a limit rather than a target, is given in Fig. 5.

MSY
1930s–1970s
by Peter A. Larkin

Here lies the concept MSY.
It advocated yields too high,
And didn't spell out how to slice the pie.
We bury it with the best of wishes,
Especially on behalf of fishes.
We don't know yet what will take its place,
But hope it's as good for the human race.

Figure 4 Larkin's (1977) epitaph to the concept of MSY.

MSY reborn
1990s
by Pamela M. Mace

Up springs MSY.
No, it didn't die.
It just metamorphosed
To be better-for-those
Who wrote of its woes.

Ahead of his time was Larkin,
To him we would always harken.
But MSY has a cause,
It's the focus of many laws.

It finally discovered the key:
"Please, don't take all of me.
Just take a quarter,
Leave more in the water.
(For my son and my daughter)
You know that you ought'a."

Now MSY has found a new niche,
And all that's needed is to make the pitch
That production from the oceans is finite,
And civilized people should *not* have a pie-fight.

Figure 5 Response to Larkin's (1997) epitaph to the concept of MSY, reflecting the recent evolution of the concept in the context of the precautionary approach, international agreements and the 1996 reauthorisation of the US Magnuson-Stevens Fisheries Conservation and Management Act (NMFS 1996).

Implementation reality check

The message that 'MSY has found a new niche' as a limit rather than a target (Fig. 5) is meant to be interpreted as a progressive step towards more risk-averse fisheries management. But are we really making progress?

Resistance to changing the status quo

The scientific community has been working on formulating harvest control rules, incorporating MSY in its new role, and interpreting the precautionary approach for the last 4–6 years. One problem is that much of their work to date has been in isolation from fishery managers and the fishing industry. Thus, it is only in the last year or so that managers and the industry have come to realise the consequences of the precautionary approach and/or the scientists' interpretation thereof. In some cases, managers have (rightly) perceived harvest control rules as taking away all or much of their discretionary power. Control rules that require fishery closures at some predefined low biomass level are particularly controversial. Apparently, a negative response by the NAFO Commission to the report of their precautionary approach working group is responsible for relatively little progress being made in adopting and implementing the approach in the NAFO management arena since the report was delivered in May 1999 (NAFO 1999), although the working group has continued to develop methods for implementing the approach (NAFO 2000). Neither in the NAFO nor the ICES arena have control rules based on the precautionary approach been endorsed as a whole by the appropriate management authorities. In fact, to date, no comprehensive formulations of the precautionary approach framework have been accepted by international fisheries organisations. However, several elements of the precautionary approach have been implemented by various management authorities (NAFO 2000) (and some existing management measures have now been termed 'precautionary' after the fact). In these and other cases, it may be necessary and desirable to go through several iterations in the development of the control rules in order to develop rules that all affected parties can buy into. If resulting harvest control rules were perceived as belonging to the managers and the industry, then perhaps the 'loss of discretionary power' would be more than compensated by the reduced scope for interference by politicians or demands from special interest groups. In theory, harvest control rules should absolve managers of the blame for needing to make 'conservative' decisions, remove the temptation to make risk-prone decisions, and reduce the uncertainty about future management actions, thereby facilitating strategic planning by the industry.

The scientific community may have erred in developing harvest control rules in relative isolation from fishery managers and the fishing industry. In fact, biologists probably thought that they were not going beyond their proper role because they were primarily simply specifying the biological limits within which fisheries must operate. Provided managers and the industry stay within such limits, management decisions can be based on whatever economic, social or other considerations are deemed to be relevant. The scientific community also believed it was appropriate that managers and users determine the degree of precaution, or level of risk to the resource, that should be adopted. In fact, the scientific community has emphasised that precaution should be exercised only in fisheries management, not fisheries science; i.e. that scientific stock assessments should always produce best estimates, not conservative estimates of assessment outputs (ICES 1998; Restrepo *et al*. 1998). Also, inclusion of uncertainty and a risk analysis framework for presenting the consequences of management actions should allow managers to balance risks and benefits (Mace and Sissenwine 2001). As it happens, managers have apparently been quite willing to choose from the upper, risk-prone percentiles of probability distributions, as the following two US examples illustrate.

Summer flounder example

The NMFS (1998) official guidelines and technical guidance currently require that there be a limit fishing mortality (which may or may not vary with biomass but should be expected to achieve MSY on average) and a target fishing mortality. By definition, the limit fishing mortality should have a low probability of being exceeded while the target should be achieved on average, meaning that the target fishing mortality should be lower than the limit. Proxies for MSY-based reference points are allowed. The Mid-Atlantic Fishery Management Council (MAFMC) chose to use F_{max} (from yield-per-recruit analysis) as a proxy for F_{MSY}. Except in the case of stocks with super-compensation (strongly dome-shaped stock–recruitment relationships), F_{max} will always exceed F_{MSY}. Therefore, this choice is risk-prone. In addition, the MAFMC chose to set F_{max} to be the target as well; i.e. the target was set equal to the limit, again a risk-prone choice. To make matters worse, the MAFMC recommended a quota for the

1999 fishing season that had a 3% chance of not exceeding the target; i.e. a 97% change of exceeding both the target and the limit. The NMFS' counter-proposal, which was implemented, was to impose a quota that had an 82% chance of exceeding the target and the limit.

Subsequently, several environmental groups took NMFS to court and the Court of Appeals ultimately ruled that 'the 1999 quota is unreasonable, plain and simple...' '...to assure achievement of the target F, to prevent overfishing, and to be consistent with the fishery management plan, the TAL [total allowable landings] must have had at least a 50% chance of attaining an F of 0.24 [the F_{max} for summer flounder]...' '...only in Superman Comics' Bizarro world, where reality is turned upside down, could the Service reasonably conclude that a measure that is at least four times as likely to fail as to succeed offers a 'fairly high level of confidence'...' (United States Court of Appeals for the District of Columbia Circuit 2000). Even though NMFS lost the case, the ruling sets a legal precedence that targets should have a 50% chance of being attained.

King mackerel example

King mackerel assessment results are presented to managers as 16th to 84th percentiles of bootstrap distributions of the Allowable Biological Catch (ABC). The reasons for truncating the distributions at the 16th and 84th percentiles are because these bounds correspond to roughly one standard deviation from the mean in a normal distribution, and because it was felt that the tails of the distribution are not well-estimated (Legault *et al.* 2001). The agreed management target for king mackerel is the ABC. As targets should be achieved on average, it would make sense to choose an ABC level near the 50th percentile of this distribution to set the total allowable catch (TAC), as was ruled by the court in the summer flounder case. However, king mackerel managers have frequently chosen the 84th percentile to set the TAC because this was the closest level to the status quo TAC. In other words, they have been willing to accept an 84% probability of exceeding the agreed target. In addition, TAC allocations have routinely been overrun. Legault (1999) showed that if such risk-prone management decisions had not been taken, the stock would have already recovered from an overfished state by 1998.

The role of uncertainty

Another example where it is likely that reality will be at odds with theory is the logical assumption that more and better data should lead to reduced uncertainty, which essentially means smaller confidence intervals and higher allowable catches for the same level of risk. This does not seem to work in practice for a very simple reason. More and better data means more and better options for modelling the system, and more and better ways of characterising or estimating the myriad sources of uncertainty. Thus, confidence intervals may well become wider with additional data. Similarly, the maxim that high uncertainty should result in greater caution is unlikely to be easy to implement in practice in existing fisheries. At the extreme where there are insufficient data even to indicate whether or not a stock is being overfished, it would be difficult to reduce landings below recent levels as a 'precautionary measure'.

Socio-economic considerations

A final example of resistance to the FAO formulation of the precautionary approach is the suggestion from some members of the US fishing industry to develop a precautionary approach to social and economic considerations. The underlying premise is that management measures should attempt to preserve the character of coastal communities and in particular to avoid changing them irrevocably, while at the same time ensuring that the fishing fleets associated with these communities are financially viable. Unfortunately, the objectives of high employment, high profits, and high stock size are often in conflict with one another. In addition, although the precautionary approach acknowledges the importance of taking socio-economic considerations into account, it definitely gives highest priority to conservation of the resource–without which there will be no social or economic benefits to consider. One of the main tenets of the precautionary approach is that the status quo is not acceptable if that status quo is resulting in overexploitation of fisheries resources. I believe that the precautionary approach will be rendered meaningless if it is generalised to the point where it is necessary to simultaneously avoid disturbing the status quo in terms of fishing fleets and fishing communities, rebuild overfished stocks and avoid depleting those not currently overfished, and enhance fishers' incomes.

Is it necessary?

The examples above illustrate the resistance by managers and the industry to the concept of reducing fishing mortality rates below F_{MSY}. Is it necessary to treat traditional single-species management targets such as F_{MSY} as limits rather than targets? In my opinion, yes. A brief examination of the record on fisheries production and fisheries management is offered in support of this opinion.

Report card on fisheries management

At the global level, there is strong evidence that most fish stocks are fully or overexploited, and many are substantially overexploited. According to FAO statistics (located in direcories on http://www.fao.org/fi/default.asp), total world fish production (fish and invertebrates but not seaweeds) reached 122.4 million tonnes in 1997, continuing a long-term increasing trend. However, most of the recent increase has been made up by aquaculture (Fig. 6). In 1994, the total production of 112.3 million tonnes was made up of 84.7 million tonnes of marine landings, 6.7 million tonnes of landings from inland waters, 12.1 million tonnes from aquaculture in inland waters, and 8.7 million tonnes from mariculture. The corresponding figures for 1997 are 86.1 million tonnes, 7.5 million tonnes, 17.6 million tonnes, and 11.2 million tonnes, respectively. Preliminary estimates for 1998 are 78.3 million tonnes, 8.0 million tonnes, 18.7 million tonnes, and 12.1 million tonnes, respectively, for a total of 117.2 million tonnes. Thus, globally, the landings of marine fish are levelling off. In fact, the global picture masks trends such as the gradual disappearance of high-valued species and replacement with species lower down the food chain (the 'fishing down the food chain phenomenon', Pauly *et al.* 1998). In addition, among the major fish stocks for which information is available, an estimated 44% are fully exploited and 25% are overexploited or depleted (16% overfished, 6% depleted, 3% slowly recovering) (FAO 1999a).

Moving from the global to the specific, the collapse of northern cod off Newfoundland and Labrador in 1992 (Fig. 7), and the lack of significant recovery since, has certainly brought home the potentially devastating social and economic repercussions of management failures. Although a number of factors have been implicated in the northern cod collapse, it is clear that the stock was exploited at unsustainable levels for many years. Hutchings and Myers (1994) estimated the sustainable exploitation rate at 17% per annum, close to the estimate of $F_{0.1}$ of 18%, which was equal to or above the stated management targets during the 1970s and 1980s. Realised harvest rates greatly exceeded the stated management targets through most of the two decades prior to collapse, due to overestimation of stock size and other factors. Had the management target been attained on average, the collapse might not have happened.

What might have been

It is easy to simulate what might have been if risk-averse or optimal management targets had actually

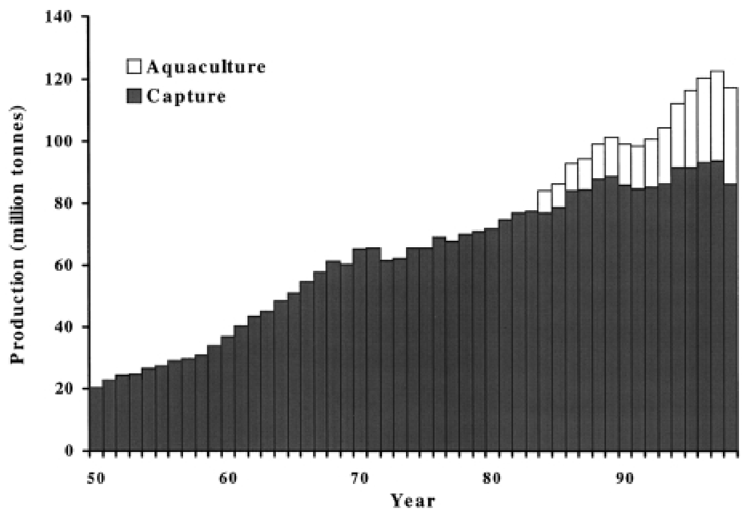

Figure 6 Global production from marine fisheries including both capture fisheries and aquaculture, based on FAO statistics.

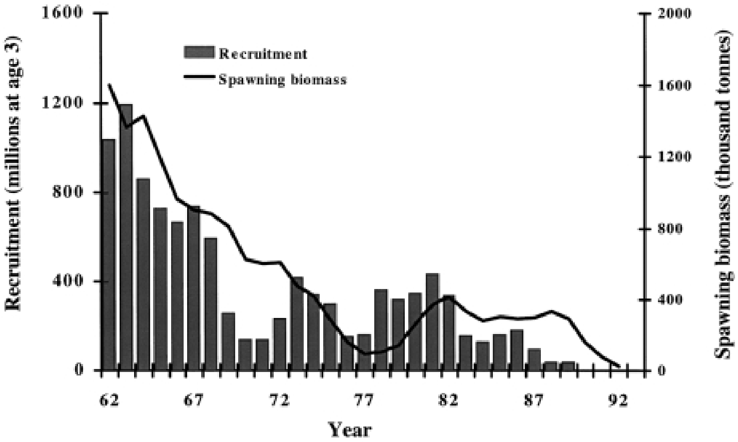

Figure 7 Historical trends in spawning stock biomass and recruitment for northern cod. Based on data from Bishop *et al.* (1993) and R.A. Myers, Dalhousie University, Halifax, Canada (personal communication).

been set and achieved. The basic approach is to start with the observed population numbers at age at some point in history and simulate forward using either observed recruitment, recruitments based on observed survival ratios (R/S), or a stock–recruitment relationship, applying a risk-averse or optimal fishing mortality rate, and then compare results with those obtained using the actual fishing mortality time series. In fact, examples of such 'retrospective projections' assuming specific stock–recruitment relationships were presented in the second Larkin lecture given by Beverton (1998) who examined 'losses in catches foregone as a result of the failure of rational management', using North Sea cod and Icelandic herring as examples. Here, I provide examples for Georges Bank cod (Fig. 8), Georges Bank haddock (Fig. 9), and Georges Bank yellowtail flounder (Fig. 10) using (A) observed recruitment and (B) observed survival ratios derived from stock assessments conducted by NMFS (2001). In all cases, if a fishing mortality rate equal to F_{MSY} or a related proxy had been achieved each year, the spawning biomass in 1993 would have been higher, often considerably higher, and cumulated landings would have been similar or higher. (The reason the time series are truncated in 1993 is because 1994 marks the year when effective management measures were finally implemented, as will become evident in a later section).

In cases where realised fishing mortality rates have generally been in excess of optimal levels (and assuming the stock–recruitment relationship is not domed over the range of simulated stock sizes), one would expect that use of observed recruitment would result in underestimates in the retrospective projections of both stock size and catch because higher stock sizes should have resulted in higher recruitment levels on average. Similarly, the use of observed survival ratios to estimate recruitment under these assumptions is likely to overestimate recruitment (and therefore subsequent stock sizes and catches) because survival ratios generally decrease with increasing stock size. Examination of Figs 8–10 suggests that while it may be true that use of observed recruitments underestimates the projected stock sizes and catches, it seems that use of observed survival ratios to calculate recruitment may substantially overestimate the growth in stock sizes and catches. Nevertheless, the benefits of never exceeding, rather than routinely exceeding, target fishing mortality rates are evident, even if somewhat overstated in the cases where survival ratios are used.

The Georges Bank examples illustrate what might have been if fishing targets approximating F_{MSY} had been estimated, set and implemented exactly. In fact, there were no specific fishing mortality targets during most of this period and, even if there had been, it is highly unlikely that realised fishing mortality rates would have been as low as F_{MSY}. Even if F_{MSY} had been explicitly adopted as a target, it almost certainly would have been exceeded in most if not all years. In recent years, F_{MSY} has been explicitly adopted as a limit for fisheries in the north-eastern United States, but for most species the managers are still in the process of bringing fishing mortalities down towards the limit as a first step to setting fishing mortality targets below the limit. In these cases, full implementation of control rules similar to those illustrated in Figs 1 and 2 will probably need to wait until fishing mortality rates can be brought under control and stocks are rebuilt; nevertheless, considerable progress has been made in developing such rules (e.g. Cadrin 1999; NMFS 2000, 2001).

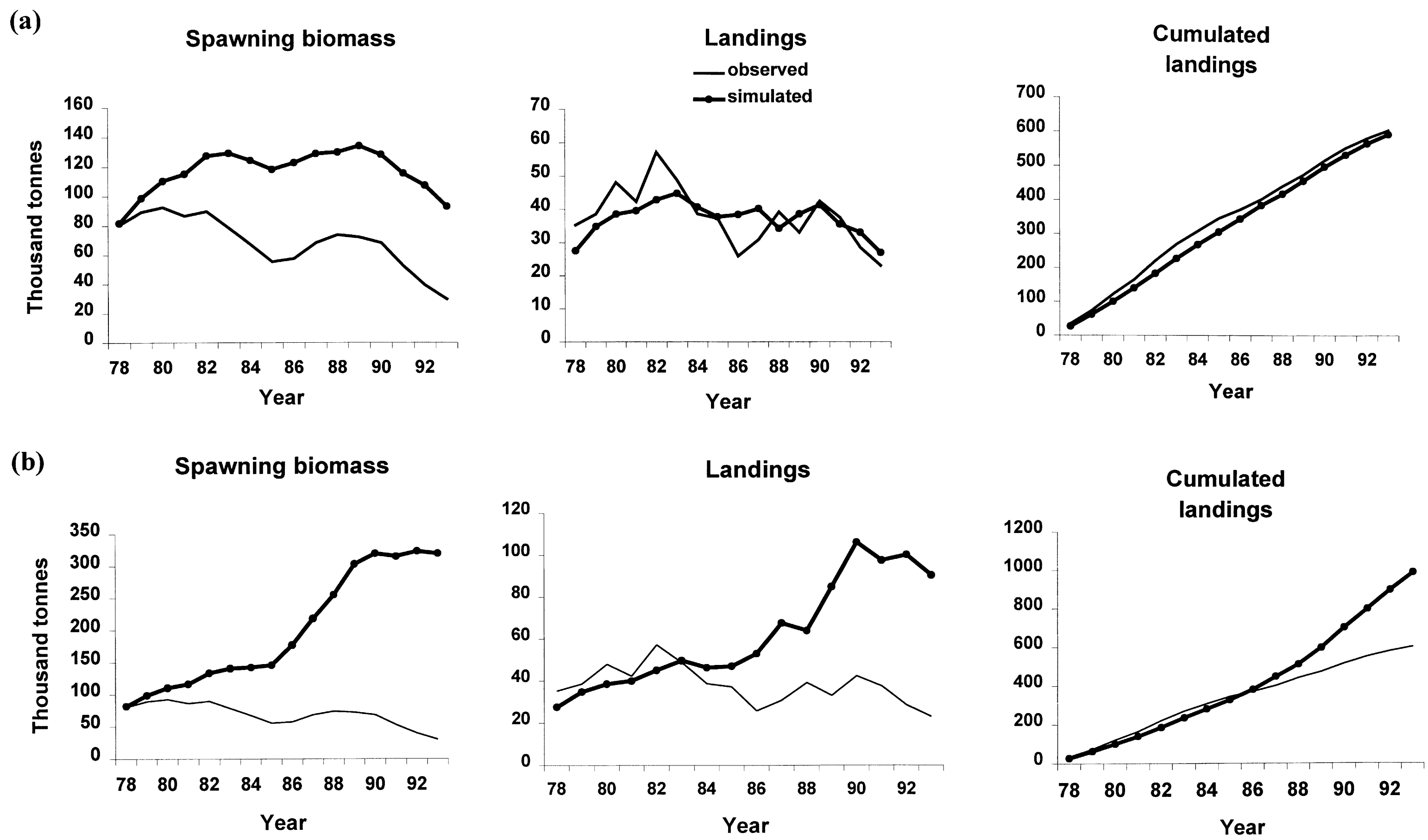

Figure 8 Georges Bank cod. Observed and simulated trends in spawning stock biomass, landings and cumulated landings, where simulated exploitation rates reflect what might have been if historical fishing mortality rates had never exceeded F_{MSY}, assuming (a) observed recruitment and (b) observed survival ratios (R/S, where R is recruitment and S is spawning stock biomass).

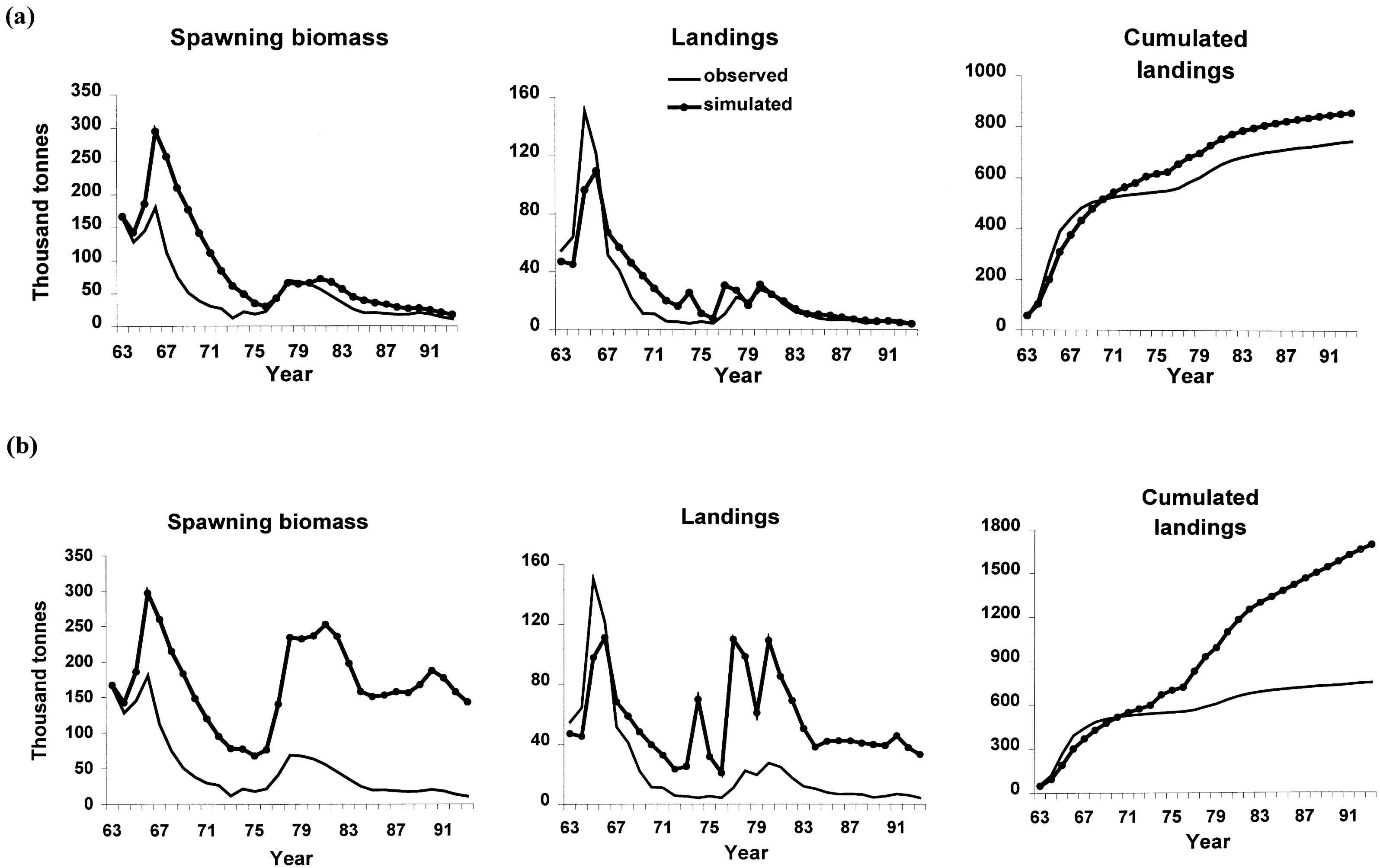

Figure 9 Georges Bank haddock. Observed and simulated trends in spawning stock biomass, landings and cumulated landings, where simulated exploitation rates reflect what might have been if historical fishing mortality rates had never exceeded F_{MSY}, assuming (a) observed recruitment and (b) observed survival ratios (R/S, where R is recruitment and S is spawning stock biomass).

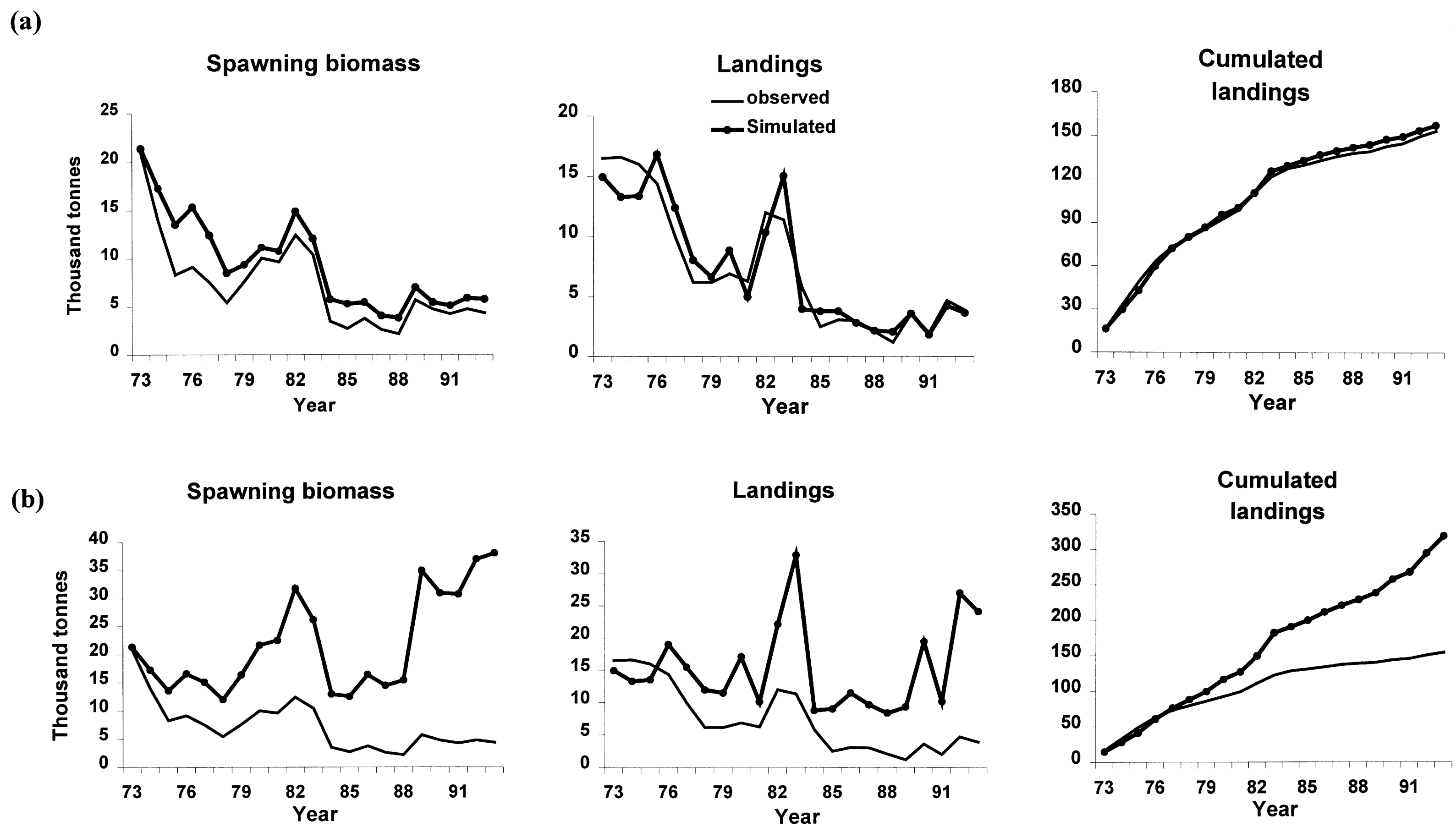

Figure 10 Georges Bank yellowtail flounder. Observed and simulated trends in spawning stock biomass, landings and cumulated landings, where simulated exploitation rates reflect what might have been if historical fishing mortality rates had never exceeded F_{MSY} (here approximated by F_{max}), assuming (a) observed recruitment and (b) observed survival ratios (R/S, where R is recruitment and S is spawning stock biomass).

Five reasons for the MSY transition

This section summarises and adds to arguments previously presented above. There are at least five reasons why it makes sense to treat F_{MSY} as a limit rather than a target.

- Experience suggests that there is a far greater chance of overshooting a fishing target than there is of falling short of it. For a target to work (i.e. to be achieved on average), overshoots and undershoots must balance out, but this has rarely been the case in fisheries purportedly managed with F_{MSY} or a similar reference point as a target. The cumulative effect of overshooting F_{MSY} (due to assessment errors, implementation errors, or other factors) is that eventually yields will fall below the associated maximum average yield and the stock will become depleted relative to B_{MSY}. The end result is that there will soon be a need to implement economically and socially disruptive restrictive management measures in order to rebuild the stock. On the other hand, the cumulative effect of habitually undershooting F_{MSY} (e.g. by treating it as a limit to be avoided with high probability), is that the risk of future economic and social disruption is minimised and there may even be future opportunities for temporary increases in catches. In addition, reference points that are defined as limits to be avoided with high probability are usually taken more seriously than targets that are to be achieved on average.
- Fishing somewhat below F_{MSY} generally results in a relatively small loss in average catch for a relatively large gain in average stock biomass. For example, by parameterising the deterministic age-structured model described by Mace (1994) with 600 different combinations of natural mortality levels, growth rates, ages of maturity, stock–recruitment relationships and stock recruitment parameters (i.e. a wide range of combinations of life history parameters), it was shown that the long-run consequences of fishing at 75% of F_{MSY} were that yields would average 0.949–0.983 MSY and biomass levels would average 1.265–1.311 B_{MSY} (Restrepo *et al*. 1998). The reason for such relatively small losses in yield is that the fishing mortality rate is being applied to a larger stock size. Thus, it is possible to take almost the same yield with far less risk of stock depletion and potentially lower harvesting costs.
- Traditional fisheries economics models indicate that the fishing mortality associated with maximum economic yield (F_{MEY}) is less (often considerably less) than F_{MSY}, and consequently that the equilibrium maximum economic yield (MEY) is also less than MSY.
- Treating F_{MSY} as a limit rather than a target is in conformance with the precautionary approach as presented in the United Nations Implementing Agreement relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks (United Nations 1995) and referred to in several other United Nations agreements and guidelines.
- Treating the single-species F_{MSY} as a limit may well be a good interim or first step towards EBM. Ecosystem-based fisheries management is a developing science that has not yet advanced to the point where it can provide operational guidelines for defining and attaining ecosystem-based objectives. Across the board reductions in fishing mortality may represent a good start towards preserving biodiversity, genetic diversity, community structure and function, trophic relationships, and future options.

The question then becomes, why choose F_{MSY} instead of some other limit? The primary answer is that F_{MSY} has no operational rival. By definition, F_{MSY} is sustainable, so any fishing mortality below it must also be sustainable. On the other hand, use of $F_{extinction}$ as a limit is problematic. In a deterministic, error-free world, any fishing mortality below $F_{extinction}$ is sustainable, but in a stochastic world with measurement, assessment and implementation errors, $F_{extinction}$ should not even be approached. How far below $F_{extinction}$ is it necessary to go to achieve a high probability of sustainability?

Other candidates for limits include $F_{30\%}$, $F_{35\%}$ and $F_{40\%}$ from spawning-per-recruit analysis, but these are already used as proxies for F_{MSY}, with the actual $F_{\%}$ level varying, depending on the productivity of a stock. Another choice is F_{MEY}, which is usually calculated to be below F_{MSY}, but this has not yet been advocated as a limit.

Is it sufficient?

If it were possible to reduce fishing mortalities on all, or most, harvested species to below the single-species F_{MSY}, would this be sufficient to ensure sustainability of the structure and function of marine ecosystems? In other words, are single-species approaches adequate? The short answer is, 'probably not', but they are certainly likely to be a step in the right direction, given the excessive fishing mortalities being exerted on many stocks. In fact, very recently I was engaged in a dialogue that has been repeated many times with slight variation. It goes something like this.

'It is essential that we move towards ecosystem-based management because single-species approaches have not worked. Our fishing mortality rates are almost all in excess of 1.0.'

''So, pray tell, should the main priority be to devote resources to developing ecosystem-based management, or to as quickly as possible reduce fishing mortalities below 1.0? How are 'ecosystem approaches' going to be an improvement over 'single-species approaches' in this instance?''

The fact of the matter is that the fisheries management record to date cannot be used as evidence that single-species approaches do not work, because even when operational management targets are specified on paper, they have rarely been achieved on average. Achieving management targets on average means that if they are exceeded in one year they should be underachieved by a similar magnitude in some subsequent year. However, the usual 'best case scenario' is that an honest attempt is made to not exceed a management target in a given year, but not to compensate the following year in the (likely) event that the management target is exceeded. For similar reasons, MSY-related reference points should not be used as scapegoats.

Therefore, the conclusion that we need a new system because the current (single-species) one has failed cannot be substantiated because even when management objectives have been clearly defined, they have rarely been seriously pursued or implemented. Instead, unstated goals such as maintaining the status quo, maintaining short-term financial viability, and maximising employment opportunities for coastal communities have generally taken precedence and undermined the stated objectives. In the following sections, I attempt to compare the merits and shortcomings of single-species and ecosystem-based approaches.

Single-species approaches vs. ecosystem approaches

Fisheries management can be approached in two fundamentally different manners: (i) a holistic approach that uses the entire ecosystem as the starting point for consideration, and (ii) a single-species approach that assesses species of interest in isolation and provides management advice as though the rest of the ecosystem was a black box. Of course, these two approaches represent the extremes of a continuum, and most management strategies will fall somewhere along the continuum rather than at the extremes. For example, while detailed single-species assessments still form the core of management advice in most cases, they are more and more frequently embedded in an ecosystem context, at least in a qualitative sense.

What's wrong with single-species approaches?

The obvious limitation of single-species fisheries management is that it is not holistic. It generally does not explicitly consider species interactions, changes in ecosystem structure or function, biodiversity, nonharvest ecosystem services, the needs of protected or rare species, other nontarget species, the ecosystem effects of discarding large quantities of unwanted bycatch (that provide energy subsidies to scavengers), or gear impacts on habitat. As Pauly *et al.* (1998) state, single-species analyses may mislead researchers and managers into neglecting the gear and trophic interactions which determine long-term yields and ecosystem health. A major criticism of single-species models is that they cannot predict changes in community structure (Trites *et al.* 1998).

What's wrong with ecosystem approaches?

Due to the complexity of most marine ecosystems, approaches that attempt to model the system as a whole generally do not consider age or size-based variation in demographic parameters; density-dependent effects; stock–recruitment relationships; effects of fishing on genetic diversity; uncertainties in the form of measurement errors, estimation errors, process errors and implementation errors; standards and reference points; or interactions and feedback between the harvested species, the stock assessments and the management system. A major criticism of ecosystem models is that they may not be able to predict changes in community structure (Trites *et al.* 1998).

Another major problem with ecosystem approaches is the difficulty of defining operational objectives and performance measures. Pitcher *et al.* (1998) outline what they coin the Back to the Future (BTF) policy process in which the objective is to restore the ecosystem that maximises net benefits to society. Pitcher *et al*. (1998) and Pitcher and Pauly (1998) argue that the goal should be to 'rebuild ecosystems', rather than 'species-by-species sustainability'. However, precise definition of the desired target configuration of a rebuilt ecosystem would be difficult. Obviously, if the overall objective is to maximise net benefits to society, the goal would generally not be to rebuild the ecosystem back to its pristine state. In between the current (over-exploited) state and the pristine state, there are numerous alternative states, some of which may be more desirable then others, but many of which are viable. Even if we could agree on the optimal state, it is doubtful that we have the capability to manipulate the system to achieve that state. Some authors even argue that it may be impossible to restore overexploited marine ecosystems to some previous state, particularly given environmental regime shifts and habitat changes (Link 2000).

Ecosystem models have not yet proved themselves as management tools, particularly in terms of making realistic predictions about the future. As Hall (1999) states, given current levels of understanding about marine ecosystems, it is difficult to identify key interactions between species, let alone manage for them.

The pros and cons of moving beyond single-species assessment and management

Examination of the costs and benefits of moving beyond single-species management may seem like a moot consideration because there are many instances where this evolution has already occurred and it is inevitable that many more, if not all, will ultimately follow. However, it is important to consider the potential increased information requirements for EBM. We need to ask the question: what do we gain from an ecosystem approach that we cannot already get from single-species approaches? Indeed, this was the trigger question for the most recent NMFS National Stock Assessment Workshop on 'Incorporating ecosystem considerations into stock assessments and management advice' (Mace 2000). At a very general level, two potential answers are: (i) a better appreciation of the effects of fishing on ecosystem structure and function, and (ii) a better appreciation of the need to consider the value of marine ecosystems for functions other than harvesting fish and invertebrates for food and livelihoods. Conversely, the impediments to EBM include: greater information requirements (e.g. more data on a greater variety of species), greater complexity (and no 'stopping rule' for determining the optimal level of complexity to include in models), numerous alternative hypotheses about ecosystem structure and function, lack of operational objectives (due in part to the difficulty of valuating alternative ecosystem states), lack of widely applicable performance measures (such as maintaining species diversity at or above some specified threshold), and possibly reduced predictability about future stock sizes of key species.

Integrating the two approaches: the best of both worlds

The previous paragraphs have presented single-species and ecosystem approaches as though they were a dichotomy whereas, of course, most management procedures will contain elements of each. The extent to which single-species approaches should be augmented with ecosystem considerations, or ecosystem models should focus on key species of interest, or the two should be more fully integrated, depends on the question being asked. Models and assessments should be tactical; that is, the level of complexity should be commensurate with the type of question being asked.

In recent years, single-species stock assessments and management advice have become increasingly sophisticated, incorporating increasingly more sources of uncertainty and complexity (Mace and Sissenwine 2001). One example is the continued evolution of the study of feedback control policies over the past 2–3 decades. Recent applications include the management-procedures simulation model approach, adopted by the International Whaling Commission as their 'revised management procedure' (IWC 1993, 1994) and adapted and further developed by several other groups, including ICES (1994) and ICCAT (2000). These models generally consist of an operating model that provides a simulation of a 'true' population, a procedure for sampling the true population, an assessment model that uses the sampled data to produce a 'perceived' population, a management model that implements specific harvest rules, and performance statistics and feedback associated with each of these components (Fig. 11). They are particularly useful for evaluating potential management actions for their robustness to a wide spectrum of uncertainties. Although this type of model tends to be orientated to single-species, it

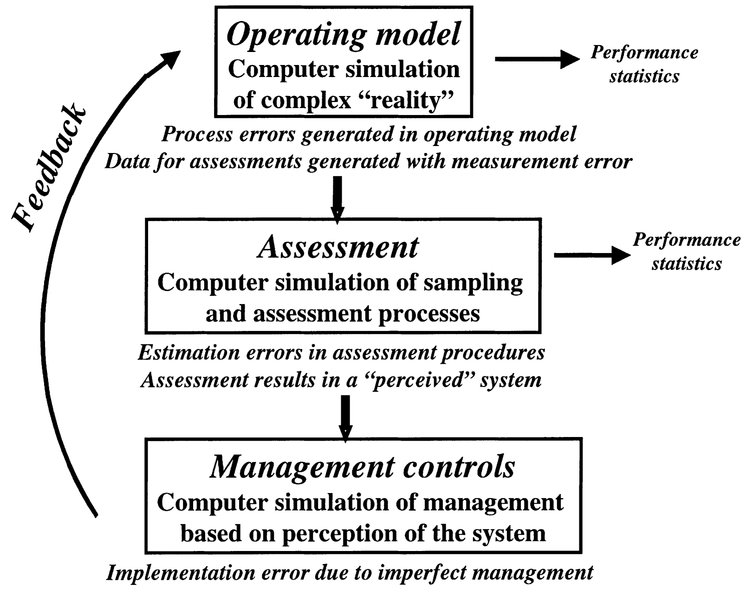

Figure 11 Schematic diagram of a management-procedures simulation model. From ICCAT (2000).

goes beyond uncertainties in stock assessments alone by also considering uncertainties in other aspects of the data collection–assessment–management system. Other forms of complexity are also being added to single-species models; for example, empirical studies and models demonstrating that fishing reduces genetic diversity are being developed (e.g. Smith, P.J. 1994).

Inclusion of multispecies interactions and environmental effects (i.e. ecosystem approaches) results in different dimensions of added complexity. In many marine ecosystems, the large number of species involved and the potential interactions between them are virtually impossible to represent in a coherent way. Even when there is some degree of aggregation of species, the number of links between species and species groups may be immense. For example, Link (1999) determined that there are more than 2500 links between 75 species or species groups on Georges Bank (Fig. 12). Although food web analysis and other types of multispecies models are gradually leading to a better understanding of the structure and dynamics of marine ecosystems, they are still far from providing quantitative predictions that can be used for management advice (Larkin 1996; Trites *et al.* 1998). Simpler models demonstrating relationships between species of interest and environmental variables have had greater success in terms of providing quantitative management advice, but the relationships on which they are based have frequently failed to hold up in the long term.

There is a need to integrate comprehensive models of key species of interest with comprehensive models of the biological, physical and chemical environment in which they live, while at the same time reducing the dimensions of complexity to manageable levels (i.e. do not try to combine Figs 11 and 12!).

To date, the main way of integrating single-species and ecosystem approaches has been to continue to focus on single-species assessments and management targets but to modify these targets or other aspects of the management strategy based on qualitative or semiquantitative analyses of species interactions or habitat effects. For example, a single-species quota might be set relatively more conservatively for a forage species than a predator. Or the allowable catch may be set according to some benchmark, but fishing may be restricted further in certain areas or seasons to avoid local depletion of fish species consumed by protected or endangered species, or to reduce bycatch of nontarget species, or to minimise impacts on fragile bottom types. In fact, gear impacts on habitat, fishing impacts on marine mammals, seabirds and endangered species, and other factors not normally included explicitly in single-species models are nevertheless receiving increasing attention in FMPs. The NMFS (1999) recommended that Fishery Ecosystem Plans (FEPs) should be developed as umbrella documents for existing FMPs, and that such FEPs should include comprehensive descriptions of current knowledge about the structure and function of the ecosystems impacted by FMPs.

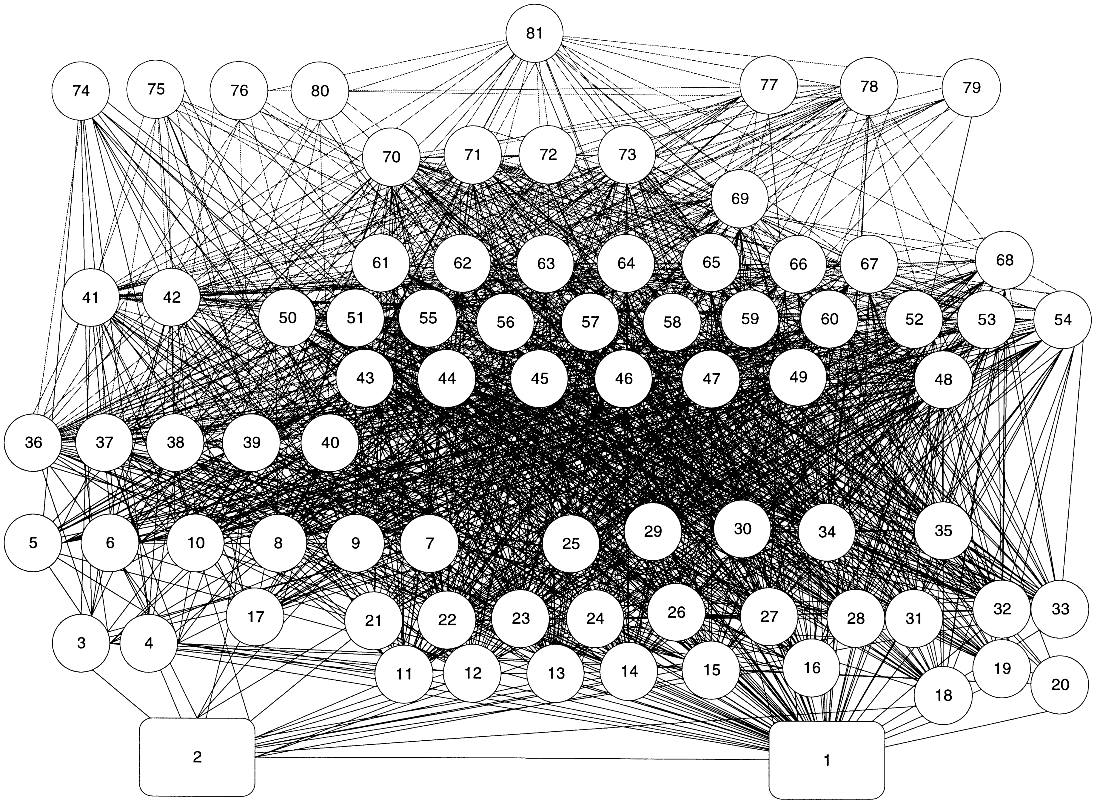

Figure 12 Species and links for a north-west Atlantic food web. The boxes are (1) detritus and (2) phytoplankton, and the circles represent higher trophic levels, either species or species groups, with 75 being humans. The left side of the web generally contains pelagic organisms, and the right and middle represents more benthic or demersal organisms. Reproduced from Link (1999).

Another approach to integrating single-species and ecosystem models is to explicitly embed single-species population dynamics models (that may include demographic parameters as functions of size, age or stage and density; stock–recruitment relationships; and other common features of single-species models) for a small number of key species of interest into ecosystem models (that may include associated species or species groups and environmental variables). Such models could potentially be very useful for more fully investigating the impacts of harvesting species of interest and discarding unwanted associated bycatch species. In particular, it may be informative to make long-term projections of future stock size in a multispecies context. Perhaps single-species projections that suggest rapid rebuilding rates with reduced fishing mortality rates are overly optimistic, particularly for depleted stocks, because they do not account for species interactions and other effects that may influence rates of recovery. This is one of the points emphasised by Hollowed *et al.* (2000) in their comparison of single-species and multispecies models. In addition, the ICES (1997b) showed that relative to multispecies models, single-species models tended to underestimate recovery times in rebuilding programs.

A pioneering approach to embedding single-species dynamics in an ecosystem framework has recently been developed by Walters *et al.* (2000). EcoSim II is an extension of the previous Ecopath and Ecosim models that incorporates key species within an ecosystem using delay-difference population models that explicitly represent growth, mortality and recruitment processes as functions of age, size and density. This is exactly the type of development that I believe will move us forward towards providing ecosystem-based quantitative management advice and towards addressing interesting and important questions pertinent to fisheries science and management, and to marine ecology in general.

For example, there are several 'axioms' that have resulted from single-species approaches that need to be re-examined in an ecosystem context.

- Single-species models demonstrate that stock productivity is usually maximised somewhere in the range of 30–60% of the unexploited level, depending on life-history characteristics; does this result hold for multispecies models?
- Yield-per-recruit arguments suggest that for most species, harvest should be curtailed until some minimum size or age corresponding to maximum cohort biomass has been reached; what is the optimum age or size of harvest in an ecosystem context?
- There is a general belief that more selective fishing gears are needed and that gear types such as trawls, gillnets and longlines which tend to catch large numbers of unwanted species are destructive to ecosystems. However, is it 'better' to selectively remove top predators or high biomass pelagics, or to harvest a range of species more or less in proportion to their relative abundance? Which of these options is more conducive to maintaining a relative balance of species or to maintaining 'ecosystem integrity' or 'species diversity'? What are the ecosystem effects of discarding large quantities of unwanted bycatch? What might be the effects of requiring all bycatch to be landed, as is currently the case in Norway?
- Are (self-regenerating) yield curves really flat near and beyond the single-species MSY?

It will also be important to compare the results of single-species and multispecies models in terms of alternative management strategies. How much complexity should multispecies models have? For example, how many species should be modelled and how should the optimal number (the 'stopping rule') be determined? In a single-predator–single-prey model, there is very strong coupling between the two components, but as alternative prey are added, along with more predator species (as well as additional trophic levels), the coupling becomes much weaker and the dynamics of any one species are less dependent on the dynamics of another.

In addition to providing quantitative predictions that can be used in management advice, integrated single-species/ecosystem approaches need to provide operational objectives, reference points, and performance standards (FAO 1999b). Here, the best way forward may be to build on existing successful single-species approaches and extend them into the ecosystem arena. For example, despite many obstacles, the requirement in the United States to create and implement definitions of overfishing for each managed species has been successful in substantially reducing overfishing and promoting rebuilding of depleted stocks. If ecosystem-based definitions of overfishing are to be developed, it would make sense to begin formulating them as extensions to, or analogues of, single-species approaches, as has recently been suggested by Murawski (2000).

Institutional constraints

Contemplation of the application of EBM within the context of present-day institutions provides another type of reality check. In the United States and elsewhere, it is difficult to imagine how a comprehensive EBM plan could be devised or implemented. Getting all players to agree on the overall objectives and the ultimate 'optimal' ecosystem configuration would be the first obstacle. Even if a sound, comprehensive plan could be devised and agreed upon by the majority of stakeholders, fisheries management is so controversial and politically charged that it is likely that certain 'politically palatable' elements of the 'comprehensive plan' would be implemented, while others elements would be resisted. Thus, in a comprehensive EBM plan that involved cropping down one species while only lightly exploiting another species, it may happen that the first part is politically acceptable while the second part is not, thus compromising the overall objectives including the ultimate desired ecosystem configuration. In fact, the concept of full top-down ecosystem management in the context of current fisheries management institutions may be more foreboding than attempts to achieve species-by-species sustainability.

Secondly, demands on the science may well be unrealistic. Is it better to provide weakly supported 'scientific' advice based on inadequate or insufficient data or, at some point, to admit that existing data cannot provide an acceptable foundation for meaningful analyses? Some people argue that managers need to make decisions anyway and even no action is a decision; therefore, it is best that science contribute whatever it can. However, there comes a point–which may have already been reached in some cases–when the ability of fisheries science institutions to absorb progressively more complex questions that are being asked with ever greater frequency, in the absence of budget and personnel increases, results in the quality and integrity of the science becoming diluted. By trying to do too much with too little, fisheries scientists may be undermining their own discipline and their own credibility.

Looking to the future

Terrestrial ecosystems

While Earth may have permanently lost a few species and probably many genotypes due to overfishing and insufficient attention paid to ecosystem considerations, marine ecosystems are not nearly as despoiled as terrestrial ecosystems. Noss *et al.* (1995) summarised estimates from a large number of studies, primarily conducted in the United States, indicating for example that in the 50 US states, there has been a 90% loss of old-growth forests; while in the 48 contiguous states, 95–98% of virgin forests had been destroyed by 1990, 99% of the virgin eastern deciduous forests have been eliminated, 53% of wetlands were lost between the 1780s and 1980s, and 98% of an estimated 5.2 million km of streams are degraded enough to be unworthy of federal designation as wild or scenic rivers. In the north-east, there has been a 95% loss of natural barrier island beaches in Maryland, and 97% of Connecticut's coastline is developed. In the south, 100% of intact bluegrass savanna-woodland and more than 99.99% of native prairies have disappeared from Kentucky, virtually all of the dry prairies of Florida have been converted to cattle pasture and agriculture, and 95% of native habitat has been lost from the lower delta of the Rio Grande River in Texas. In the Midwest and Great Plains, 90% of the original 58 million ha of tallgrass prairies has been destroyed, including 99% of the tallgrass prairie east of the Missouri River, more than 99% of the original tallgrass prairie in Illinois and Indiana, and more than 97% of the tallgrass prairie that once covered the eastern third of Nebraska. In the west, 99% of California's native grassland is gone, more than 90% of the native shrub-steppe grassland in Oregon and south-western Washington is gone, and 96% of the original coastal temperate rainforests in Oregon have been logged. These are just a few selected examples of the extensive compilation provided in Noss *et al.* (1995) (see references therein for sources of the above statistics), who also compiled some global statistics on these types of ecosystems.

What could still be

Depletion levels of the magnitude documented for terrestrial ecosystems only apply to very small parts of the marine environment–notably, some coastal areas. Even in areas that have been affected, it may be possible to reverse, or partially reverse, the adverse impacts of fishing and other anthropogenic factors. For example, while some ecologists speculate that it may not be possible to ever recoup the former Georges Bank ecosystem, which was characterised by large stocks of high-valued gadids and flounders, there are currently strong indications that effective management actions are resulting in a return of many of the more economically valuable stocks. Beginning in late 1994, the Northeast Fisheries Management Council implemented a number of restrictive management measures including permanent year-round closures of 17 000 km^2 of fishing grounds to all bottom gears, limited entry licensing, and restricted days at sea. This has resulted in dramatic reductions in fishing mortality, an increase in biomass due to an increase in average size, and recent increases in numbers for some stocks due to increased recruitment. Based on assessments conducted in the year 2000 (NMFS 2001), for Georges Bank cod, the mean fishing mortality over ages 4–8 years has decreased from 1.42 in 1994 to 0.22 in 1999, and spawning biomass has begun to rebuild (Fig. 13). For Georges Bank haddock, the mean fishing mortality over ages 4–7 years decreased from 0.43 in 1993 to 0.16 in 1999, and spawning biomass has increased from about 11 000 to 48 500 tonnes over the same period (Fig. 14), with strong year classes recently recruiting to the fishery. For Georges Bank yellowtail flounder, the mean fishing mortality over ages 4–5 years has decreased from 2.45 in 1994 to 0.13 in 1999, and spawning biomass appears to have responded rapidly (Fig. 15). Thus, instead of being cited as an example of management failure, as it has for the last 20 years or so, it may soon be time to tout the Georges Bank groundfish fishery as a success story. In addition, although the restrictions were put in place primarily to reduce fishing mortalities on groundfish, exclusion of all bottom gears from the closed areas had the secondary effect of producing a 14-fold increase in scallop biomass over a 4-year period (Murawski *et al.* 2000).

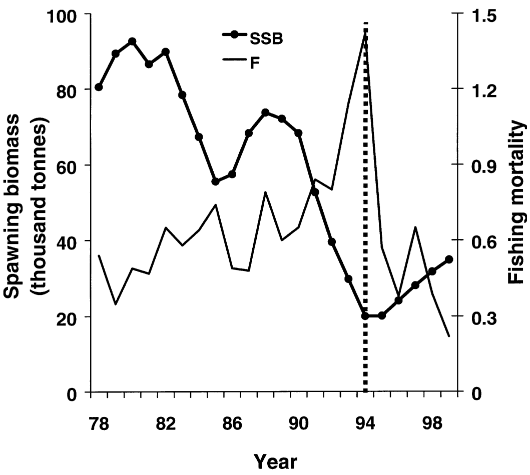

Figure 13 Historical trends in spawning stock biomass (SSB) and fishing mortality (F) for Georges Bank cod. The vertical dotted line denotes the first year in which effective implementation of restrictive management measures began to take effect.

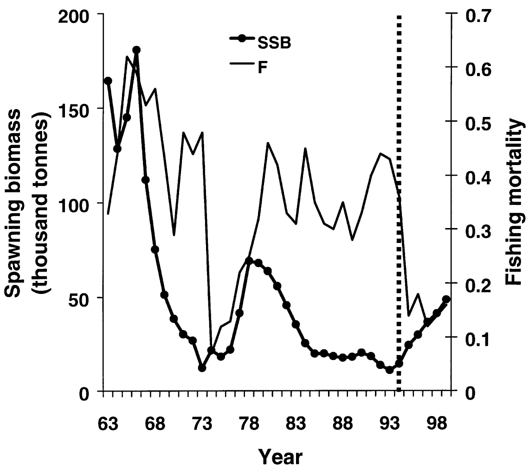

Figure 14 Historical trends in spawning stock biomass (SSB) and fishing mortality (F) for Georges Bank haddock. The vertical dotted line denotes the first year in which effective implementation of restrictive management measures began to take effect.

Successes are also being recorded in adjacent areas. Based on assessments conducted in 1999 (NMFS 2000), for southern New England yellowtail flounder, the mean fishing mortality over ages 3–6 years has decreased from 2.48 in 1990 to 0.20 in 1998. For Gulf of Maine cod, the mean fishing mortality over ages 4–5 years has decreased from 2.04 in 1994 to 0.64 in 1998, although it is still well above the management target.

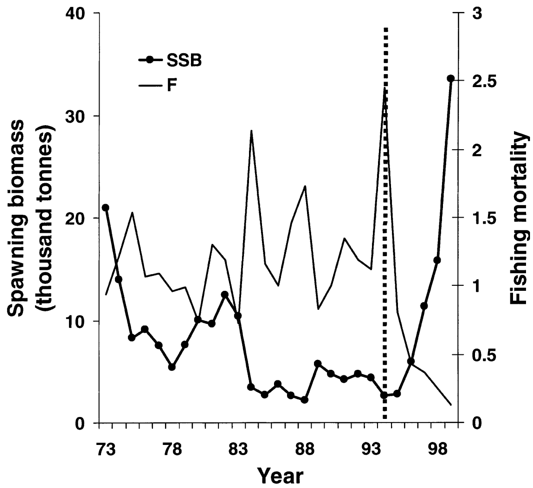

Figure 15 Historical trends in spawning stock biomass (SSB) and fishing mortality (F) for Georges Bank yellowtail flounder. The vertical dotted line denotes the first year in which effective implementation of restrictive management measures began to take effect.

Prerequisites for successful ecosystem approaches

At the risk of oversimplification, I believe there are three fundamental prerequisites for successful EBM.

(i) Reduce fishing mortality overall

I contend that eliminating overfishing on individual target species is a crucial first step towards ensuring the persistence of marine ecosystems. There are many fisheries throughout the world where exploitation rates are excessive; examples from the North Atlantic include North Sea cod (Cook *et al.* 1997), North Sea haddock, Gulf of Maine cod (Mayo *et al.* 1998), Baltic cod, North-east Arctic cod, Georges Bank yellowtail flounder (until recently, see Fig. 15), and plaice in the Skagerrak-Kattegat, all of which have been subjected to fishing mortality rates near or beyond 1.0 in recent years. The global list of stocks with excessive exploitation rates would probably be very long if exploitation rates were more widely estimated and if realised fishing mortality was compared to some reasonable target fishing mortality.

How far should exploitation rates be reduced? For many fisheries, reducing fishing mortality on the primary target species to below the single-species F_{MSY} would be an extremely good start to any kind of responsible management. It is almost a truism that more conservative single-species fishing mortality rates are more likely to satisfy common ecosystem objectives than are excessive rates. An operational objective such as ‘reduce single-species fishing mortalities below F_{MSY}’ may even be more likely to achieve positive results than vague objectives such as ‘maintain ecosystem integrity’ or ‘consider ecosystem objectives’. Note also that reducing fishing mortalities on target species or species groups will almost certainly reduce fishing mortalities on bycatch species as well. Even prestigious Committees set up to make recommendations about the implementation of ecosystem considerations have suggested significant overall reductions in fishing mortality as perhaps the best strategy for the immediate future (e.g. NRC 1999). By the time we have succeeded in bringing fishing mortality to levels below F_{MSY} for most target species, perhaps we will have developed equally operational ecosystem objectives.

(ii) Eliminate overcapacity

Some fisheries scientists and managers believe that overcapacity (including both amounts of gear and numbers of participants) is the single most important factor threatening the long-term viability of exploited fish stocks and the fisheries that depend on them (Mace 1997). Ward *et al*. (2001) list 23 potential negative consequences of overcapacity encompassing socio-economic, management and assessment issues. Socio-economic consequences may include decreased economic performance, increased enforcement costs, more intrusive regulations, decreased stability of the industry and dependent communities, decreased fishing safety, increased processing and product storage costs, and decreased product quality and prices. However, there are other consequences that have repercussions for stock assessments and management, and ultimately for resource sustainability. For example, overcapacity exacerbates: challenges to the validity of the science; pressure on managers and politicians to make risk-prone decisions; discard rates; mortality of discards due to lack of time for careful handling of discards; cryptic mortality from encounters with unnecessarily large amounts of fishing gear; ghost fishing from lost or abandoned fishing gear; reduction in the quality of mandatory catch and effort data submitted by fishers due to lack of time for careful recording; increased probability of exceeding the quota or target fishing mortality as a result of the preceding items (i.e. actual removals including reported landings, unreported landings, at-sea discards, cryptic mortality from encounters with fishing gear, and ghost fishing that may greatly exceed the catch level needed to achieve conservation); and decreased probability of correctly specifying target harvest levels as a result of errors in inputs.

It is essential that the size of fishing fleets be controlled, either explicitly by limiting participation levels by regulation, or implicitly by implementing a management system such as Individual Transferable Quotas (ITQs) that encourages self-regulation of capacity, so that capacity is commensurate with the levels of fishing mortality implied by prerequisite (i). The FAO has been conducting important work on this topic over the last few years, including development of an International Plan of Action for the management of fishing capacity (FAO 1999c).

(iii) Conduct adequate baseline monitoring of marine species and their environment

NMFS is currently developing a stock-assessment improvement plan in which one of the objectives is to move towards ecosystem approaches to fisheries assessments. Among other things, the plan calls for improved baseline monitoring of all managed marine species. Originally, the team preparing the plan had intended to recommend improved monitoring of all marine species in order to facilitate a comprehensive ecosystem approach, but it quickly became evident that this would entail resources well beyond those currently available or likely to be available in the near future. Even countries as wealthy as the United States and Canada do not monitor all target species, let alone associated marine mammals, fish, and benthos. In addition, there is seldom routine monitoring of habitat or all potentially relevant physical and chemical oceanographic features of the environment. Nevertheless, it is virtually impossible to provide meaningful scientific advice and conduct meaningful management without some form of inventory of the oceans’ assets and some understanding of their dynamics (NMFS 1999).

Although I have suggested these three issues as prerequisites for successful EBM, it could be said that they are all prerequisites for any kind of successful fisheries management, regardless of the underlying philosophy or approach. To achieve an overall reduction in exploitation rates and capacity, it does not matter whether this is accomplished using single-species approaches, ecosystem approaches, or magic; it just matters that it does get done.

Unfortunately, there are many obstacles along the way. One of the most fundamental is the rapidly growing human population. In some parts of the world, the dependence on fish for food and livelihood is so great that it is virtually impossible to reduce fishing mortality for sufficient time to enable stocks to rebuild. As stated by Cochrane (2000), it is essential to somehow reduce dependency on fisheries by finding alternative sources of protein and alternative employment. In countries where the dependence need not be as critical for survival, a reorientation of underlying value systems is required. People need to stop fighting over the pie and to think beyond the short-term. Expectations of what natural resources, fisheries science, and fisheries management are capable of providing need to be aligned with reality. Yet as people increase and fish decrease, developed countries have tended to devise progressively more complex management systems to divide a shrinking pie between numerous interest groups with varying perceptions about the fishery and varying objectives. One of the most extreme examples of this must be the US western Atlantic bluefin tuna fishery,in which about 25 000 permitted vessels compete for a quota of about 1400 tonnes (thus averaging less than one fish per vessel). Competing interests include commercial purse seiners, harpooners, and rod-and-reel gear types, charter vessels and private recreational vessels, and environmentalists. The fishery is regulated by size limits, bag limits, and frequent openings and closings as the tuna travel along a semipredictable migration path through waters adjacent to about 20 coastal states. Managers strive to ensure that everyone gets their 'share', but rarely is any segment of the fishery satisfied. Each faction is constantly in conflict with the others, the validity of the science is hotly debated, and the monitoring, management and enforcement systems are extremely complex and costly.

The costs of doing it better

The structure and function of fisheries science and management institutions would change dramatically if the above three prerequisites were satisfied. Across-the-board reductions in fishing mortality together with elimination of overcapacity would probably reduce the frequency at which stock assessments need to be conducted and free up resources for basic scientific research, including ecosystem research. The costs of management and enforcement would also be reduced because it would no longer be necessary to 'manage at the edge' where a few extra tonnes of catch can make the difference between sustainability and collapse. Allocation conflicts would be reduced, as would the need for micromanagement that requires frequent adjustment of regulations in order to spread the allowable harvest through the season, and between users and geographical areas. However, the costs of the transition towards eliminating overexploitation and overcapacity could be extremely high, in both monetary and social terms.

In contrast to the reduced burden on assessment science and management if fishing mortality and capacity could be brought under control, the costs of adequate baseline monitoring would be continuous and could be immense, depending on the definition of 'adequate', particularly if expanded fleets of research vessels are required. 'Adequate' baseline monitoring may well violate Larkin's (1983 and 1997) rule of thumb that expenditures on fisheries research and management should generally not exceed 10–20% of the landed value of the fisheries. Larkin (1997) suggested that Canada already operates well above this range, but this assumes that fisheries management is the only reason for conducting ocean research and monitoring. Obviously, fisheries are not the only reason that we need to understand the ocean environment and preserve marine ecosystems. Fisheries institutions need to pool resources and form partnerships with other users of marine data from government agencies, private foundations and academia.

One may also argue that EBM should not cost any more than effective single-species management; we are fooling ourselves if we think we can remain ignorant about associated species and still effectively manage target species. However, there is a limit to how much the public is, and should be, willing to contribute to fisheries research and management. Even with greatly increased resources, there will always be uncertainty associated with assessments and predictions about marine ecosystems (Hall 1999). There are also many other priorities for national budgets. Expectations of the science need to be aligned with the reality of what the public is willing to pay (don't ask for the moon when you're only willing to pay for a chunk of green cheese) and also what the science is likely to be able to provide, given the complexity of the systems we are attempting to understand.

Alternative institutions

The move towards EBM provides an opportunity for examination of alternative, more workable institutional structures. One of the major issues is the dichotomy between greater separation of science and management vs. greater integration of the two. On the one hand, it is argued that the objectivity of the science may be compromised if it is too closely aligned with the management system. On the other hand, it is argued that science should be more responsive to the needs of fisheries management. In many fisheries, the fishing industry also believes that it should have greater influence over both the science and the management.

The evolution of the relationship between fisheries science and management and industry involvement in New Zealand warrants further examination as a case study. In the 1980s, the evolution followed a similar path to the norm in most other industrialised countries, but it has since diverged considerably. Prior to the introduction of a comprehensive fisheries management system (ITQs) in 1986, much of fisheries science was focused on basic biological research. Introduction of ITQs required estimation of sustainable catches for management purposes and a new emphasis on stock assessments. Thus, government fisheries science programs were radically altered with the focus changing to the development and analysis of consistent time series of data to determine the status of harvested stocks. Since the early 1990s, and particularly since the requirement that it fund scientific research, the fishing industry has had progressively more influence over the types and frequency of research conducted in support of stock assessment and management, and has questioned the necessity of data-intensive scientific stock assessments. In the author's opinion, there may be at least three reasons: (i) the obvious one is that it reduces costs to the industry; (ii) it reduces the danger of damaging the industry's public image (as most surveys of marine resources are unlikely to conclude that fishing has a beneficial effect on marine ecosystems); and (iii) there is a widespread belief in the theory that management by ITQs should be self-regulating because ITQs promote economic efficiency and resource stewardship, and simple equilibrium production models indicate that the maximum economic yield is less than the biological MSY. The industry has tended to argue against the baseline monitoring of resources that several other countries consider to be fundamental to estimation of stock status and sustainable harvest rates, and appears to have curtailed most of the previous time series of fishery-independent data. One reason this approach warrants further investigation is to provide insights into the question 'How much science is really needed to support effective management?'. Does more and better science increase the probability of sustainability? It is interesting that the fishing industries in most other industrialised countries are calling for more and better scientific research in support of stock assessments, rather than less.

A related question is 'Are there robust management systems that have reduced reliance on accurate or precise science?' It is certainly easy to envision more robust management systems than those that exist at present. For example, fisheries management would be considerably easier and less controversial if fishing capacity were more commensurate with resource productivity. Greater use of marine-protected areas as mechanisms for hedging against uncertainty, particularly in ecosystems that are vulnerable (e.g. coral reefs), or have low productivity (e.g. deepwater systems), or contain rare or protected species, may also increase robustness. Further restrictions on the use of gear with large by-catches of sensitive species or suspected large physical impacts on habitat, and accelerated research on more environmentally friendly gear types may also be helpful. The utility of these approaches is largely independent of whether one is considering management based on species-by-species assessments or integrated systems approaches. At present, the demands on stock assessments are unrealistically high because we are 'managing at the edge'. More robust systems with lower fishing mortalities would reduce the burden to produce stock assessments with ever-increasing accuracy, precision, complexity and frequency.

Other alternative institutions include various forms of comanagement with some defined 'optimal' distribution of fisheries science and management responsibilities between government, the industry and other stakeholders. Closer cooperation between scientists, managers and fishers may also facilitate the design and implementation of adaptive management experiments (Walters 1986) and other avenues for promoting scientific research.

The paradigm transition

We have made the transition from viewing the resources of the oceans as essentially infinite (the inexhaustibility paradigm) to proving that they are not, at least in selected cases. As a result of undeniable evidence of overfishing, and increased public awareness of the consequences of overfishing, there has been a rapid evolution of paradigms primarily in the last decade–from 'it is not possible to overexploit natural marine resources' to 'it is not acceptable to overexploit natural marine resources' to 'it is both possible and acceptable to adopt a precautionary approach to the exploitation of natural marine ecosystems to ensure that they are preserved in perpetuity while still contributing substantially to the food, recreational and livelihood requirements of the world's current human population' (modified from Mace 1999). It is time to cease attempting to extract the maximum possible from each and every harvested species, and to adopt more conservative management approaches so as not to foreclose future options.

Concluding remarks

Which is best, simple or complex approaches? Is it better to have complex multispecies or ecosystem models or conservative single-species management? The previous three Larkin lectures all addressed this issue to some extent. Beverton (1998) stated 'In some ways, we may have been trying to be too clever. A simple management system based on careful monitoring of fishing effort, biological targets such as F_{95} [the fishing mortality where yield is 95% of the maximum derived from a yield curve that includes the processes of stock and recruitment and other compensatory process, referred to by Beverton as a self-regenerating yield model] and exploitation of a diversity of fish resources may suffice to avert further disaster and hedge against uncertainty'. Caddy (1999) emphasised the practical difficulties of implementing EBM without greatly improved management frameworks, and Cochrane (2000) called for 'a new emphasis on simplicity' in which eight simple principles of fisheries management would be recognised.

In the last five years, there has been increasing criticism of management decisions that are based on single-species approaches and a call for the implementation of ecosystem approaches. However, it is certainly not yet time to abandon single-species approaches to fish stock assessments and fisheries management in favour of multispecies or ecosystem approaches. First, before it can be comprehensively adopted as a management tool, EBM needs to be much further along in terms of evaluating alternative ecosystem states, defining operational ecosystem objectives, and specifying ecosystem management standards and performance measures analogous to those that currently exist for single-species management. Secondly, at least in the temperate waters of industrialised countries, fisheries issues seem to be dominated by a few 'politically hot' species, including protected or endangered species. For example, in the United States, such species currently include Atlantic bluefin tuna, red snapper in the Gulf of Mexico and surrounding areas, each species of Pacific salmon, Atlantic right whales and Stellar sea lions off Alaska. Because of the high sensitivity of a relatively small number of species, modelling efforts will probably continue to be focused on such species or species-groups. Thirdly, according to FAO statistics (www.fao.org/fi/default.asp), in 1997 six species (anchoveta, Alaska pollock, Chilean jack mackerel, Atlantic herring, chub mackerel, and Japanese anchovy) accounted for 24% of the global capture fisheries production.

Whether or not ecosystem approaches to fisheries management will provide practical management advice in the near future, it is certainly true that ecosystem approaches are needed to move marine science forwards. Conducting single-species assessments and applying single-species management targets may or may not be conserving ecosystems but it is doing little to advance our understanding of ecosystem dynamics.

In my opinion, treating F_{MSY} – or some related reference point – as a limit is a necessary first step towards EBM because it will result in an overall reduction in excessive fishing mortality rates. However, species-by-species reductions in fishing mortality rates in cases where these are in excess of F_{MSY} are almost certainly not sufficient, by themselves, to attain ecosystem objectives. Protection of bottom habitats, special measures for invasive species, reductions in nutrient loadings and contaminants, special considerations for rare or vulnerable or protected species, and greater use of marine protected areas as mechanisms for hedging against uncertainty may also be needed. In addition to across-the-board reductions in fishing mortality, successful implementation of ecosystem approaches to fisheries management will require commensurate reductions in fishing capacity and alignment of people's expectations from marine resources with the reality of what they are capable of producing – processes that have already begun but still have far to go.

Acknowledgements

My sincere thanks to the Fisheries Centre at the University of British Columbia for the honour of being invited to present the 4th Larkin Lecture at the University of British Columbia, and to the Larkin family and friends for funding my visit. Thanks also to Steve Murawski for input to the construction of Figs 8–10, to two anonymous referees for constructive comments that helped to strengthen some of the arguments made in the paper, and to Chuck Hollingworth for detailed technical comments.

Abbreviations

ABC, Allowable Biological Catch

B, biomass

B_{MSY}, average biomass associated with an F_{MSY} fishing strategy

EBM, ecosystem-based management

F, fishing mortality

$F_{0.1}$ and F_{max}, proxies for F_{MSY} from yield-per-recruit analysis

$F_{30\%}$ and $F_{40\%}$, proxies for F_{MSY} from spawning-per-recruit analysis

FAO, (United Nations) Food and Agriculture Organization

$F_{extinction}$, the fishing mortality that will lead to stock extinction (based on the slope at the origin of a stock–recruitment relationship)

FEPs, Fishery Ecosystem Plans

FMPs, Fishery Management Plans

F_{MSY}, the fishing mortality that yields MSY on average

IATTC, Inter-American Tropical Tuna Commission

ICCAT, International Commission for the Conservation of Atlantic Tunas

ICES, International Council for the Exploration of the Sea

ICNAF, International Commission for Northwest Atlantic Fisheries

ITQs, Individual Transferable Quotas

IWC, International Whaling Commission

M, natural mortality

MAY, Maximum Average Yield

MAFMC, Mid-Atlantic Fishery Management Council

MEY, Maximum Economic Yield

MFCMA, (US) Magnuson Fishery Conservation and Management Act

MHLC, Multi-lateral High Level Conference

MSFCMA, (US) Magnuson–Stevens Fishery Conservation and Management Act

MSY, Maximum Sustainable Yield or, equivalently, Maximum Sustained Yield

NAFO, Northwest Atlantic Fisheries Organisation

NMFS, (US) National Marine Fisheries Service

NASCO, North Atlantic Salmon Conservation Organisation

R/S, Recruitment/Spawning biomass

SEAFO, Southeast Atlantic Fishery Organisation

SSB, Spawning Stock Biomass

TAC, Total Allowable Catch

TAL, Total Allowable Landings

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