3

The gospel of maximum sustainable yield in fisheries management: birth, crucifixion and reincarnation

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A traditional key objective for fisheries management is that removals should be as large as possible but nevertheless sustainable in the long term, i.e. that the fishery catch should be equal to the maximum sustainable yield, MSY. MSY has therefore played a very significant role in fisheries management. However, its general use and applicability have been challenged. This chapter provides an overview of some of the arguments against the use of MSY, and of how many of these have been overcome in recent years so that MSY is once again a key concept in fisheries management advice.

MSYに対する反論とそれが近年どのくらい克服されてきたか ↓ もういちど、かんりにおけるkey conceptになっている

Larkin (1977) provided a tongue-in-cheek description of the 'Gospel of MSY' as it is implemented in fisheries as 'any species each year produces a harvestable surplus, and if you take that much, and no more, you can go on getting it forever and ever'. Larkin also noted that MSY-based management involves setting the harvest rate (in fisheries, defined in terms of fishing effort) to that level which produces a catch of MSY 'no more and no less'.

MSY has been, and to some extent remains, a key paradigm in fisheries management science. Fisheries management science itself consists of a wide range of scientific disciplines including biology, ecology, mathematics and even criminology (Stephenson & Lane, 1995). However, the primary purpose of the fisheries scientist is to provide advice (usually to decision-makers) regarding the relative merits of alternative management actions (Punt & Hilborn, 1997). The traditional questions addressed by fisheries scientists have related to levels of 'sustainable' catch and fishing effort, and appropriate selection of fishing gear. That role has expanded over time to include issues such as evaluating marine protected areas and the 'ecosystem impacts' of fishing. Nevertheless, the fundamental role of the fisheries scientist has changed little, and scientists today still provide advice on the implications of different levels of catch and fishing effort.

fisheries sciencesの目的は、さまざまなかんりに対する ~~対策~~ optionを示すことだ

The concept of MSY has played a central role in fisheries science for over four decades. This is because MSY is simple to explain to non-mathematicians, it is a purely physical measure (i.e. not tied to any economic or social and hence political doctrine), fishing the biomass to below that at which MSY is achieved has been a traditional definition of biological overexploitation, and MSY can (in principle at least) be estimated using limited data and computing resources. Cunningham (1981) argued that, of the traditional objectives of fisheries management, aiming at MSY results in the least decline in effort (and hence the least decline in employment).

The examples used in this chapter are based on the Cape hakes *Merluccius capensis* and *M. paradoxus* off northern Namibia and the west coast of South Africa. The quantitative results presented should, however, be considered to be illustrative only, because the data used are not the most recent. Nevertheless, the qualitative results should be insensitive to including the most recent data.

PROVIDING MANAGEMENT ADVICE IN AN UNCERTAIN WORLD – THE BIRTH OF MSY

The concept of a sustainable yield and hence of the MSY in fisheries appears to have originated in the 1930s (see e.g. Russel, 1931; Hjort *et al.*, 1933; Graham, 1935), a time when mathematical approaches to fisheries management were gaining increasing attention (Smith, 1994). However, in constructing their theory of fishing, these early pioneers of quantitative fisheries science were not unaware of the limitations of their simple models. For example, Russel (1931) noted that 'It appears that the ideal of a stabilised fishery yielding a constant maximum value is impractical', although he also stated that 'the aim of rational exploitation is to get the maximum yield annually, compatible with maintaining stocks at a steady level'.

The 1950s saw the development of two mathematical approaches that could be used to estimate the relationship between fishing mortality and catch under the assumption that the population is in equilibrium. Both approaches attempted to assess the impact of different levels of fishing intensity on changes in population biomass. The first approach, initially proposed by Schaefer (1954), is based on the concept of 'surplus production' – the difference between the increase in biomass due to growth and recruitment to the fishable population, and the loss in biomass due to natural mortality. A surplus arises because density dependence means that growth will be faster, survival will be greater or maturation earlier at stock

かんこく。
サワラの
成熟体長
↓
78cm
中国では？

levels below the unfished biomass (K). The surplus production approach does not, however, attempt to model the various processes (growth, mortality, etc.) that determine surplus production explicitly. Instead, Schaefer's arguments lead to a simple relationship between surplus production (which is the same as sustainable yield) and biomass, with MSY occurring at a biomass B_{MSY} that is half the unfished biomass K (Figure 3.1a). The relationship between biomass and surplus production in Figure 3.1a is based on the assumption that population growth follows a logistic function in the absence of exploitation. Other forms for the surplus production function (see e.g. Fox, 1970) arise from different assumptions in this regard.

Production model.

A variety of methods exist for estimating MSY using the data traditionally collected from a fishery. Figure 3.1b illustrates the simplest of these. The catch rate (the annual catch C divided by the corresponding fishing effort E) is regressed on the annual fishing effort:

MSYのあれこれ方法 CPUEに対する回帰.

$$(C/E)_y = \alpha - \beta E_y + \varepsilon_y$$

where $(C/E)_y$ is the catch rate for year y, E_y is the fishing effort for year y, α, β are regression coefficients, and ε_y is the residual for year y.

If catch and biomass are related according to the simple (quadratic) relationship in Figure 3.1a and catch rate changes linearly with abundance, it is straightforward to show that MSY is $\alpha^2/4\beta$. The values of the parameters of the curve that relates biomass to catch in Figure 3.1a are determined using an alternative algorithm, based on different assumptions (for details see Butterworth & Andrew, 1984).

The second approach, yield-per-recruit analysis, as outlined in particular by Beverton and Holt in their seminal volume (Beverton & Holt, 1957), is based on consideration of the dynamics of a single year class, where account is taken of changes with age in relative probability of capture, abundance and mass. In simple terms, the lower the fishing mortality, the longer an animal will live and hence the larger it will be if caught. Yield-per-recruit analysis thus allows an examination of the trade-off between allowing fish to grow larger, and the probability that they will die of natural causes before capture. It allows the relationship between fishing intensity and the yield per individual to be determined (Figure 3.2). The fishing mortality rate corresponding to MSY (F_{MSY}) can be determined from this relationship, assuming that recruitment (the number of individuals born) is not affected by changes in stock level. The level of fishing effort corresponding to MSY can then be determined from a relationship between fishing effort and fishing mortality, which is usually assumed to be one of linear proportionality.

YPR

かり入がSに えいきょうしないなら $F_{MSY} = F_{max}$

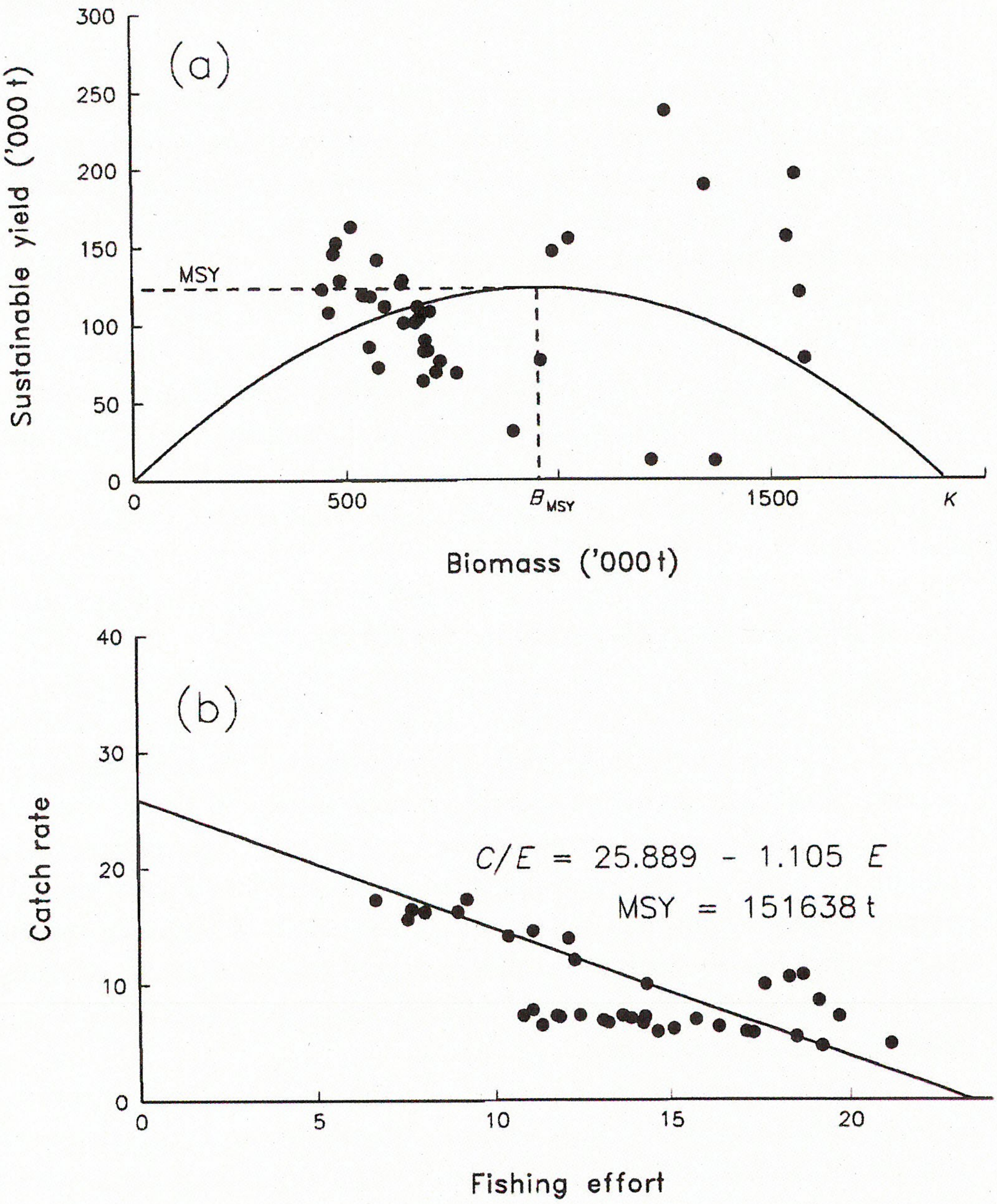

Figure 3.1. Production model assessments for Cape hake off the west coast of South Africa. (a) Equilibrium sustainable yield against population biomass based on the Butterworth–Andrew method. B_{MSY} is the biomass at which MSY is achieved and K is the environmental carrying capacity (unfished equilibrium biomass). The solid dots represent realised sustainable yield in each year (derived using the method of Schaefer (1954)). (b) Application of the catch rate regression method to estimate MSY (see equation 3.1).

The Food and Agriculture Organization (FAO) of the United Nations gave considerable support and emphasis to management based on MSY. For example Sparre & Venema (1992) noted in an FAO manual that 'fish stock assessment may be described as the search for the exploitation level

実線よりも10倍速く成長する

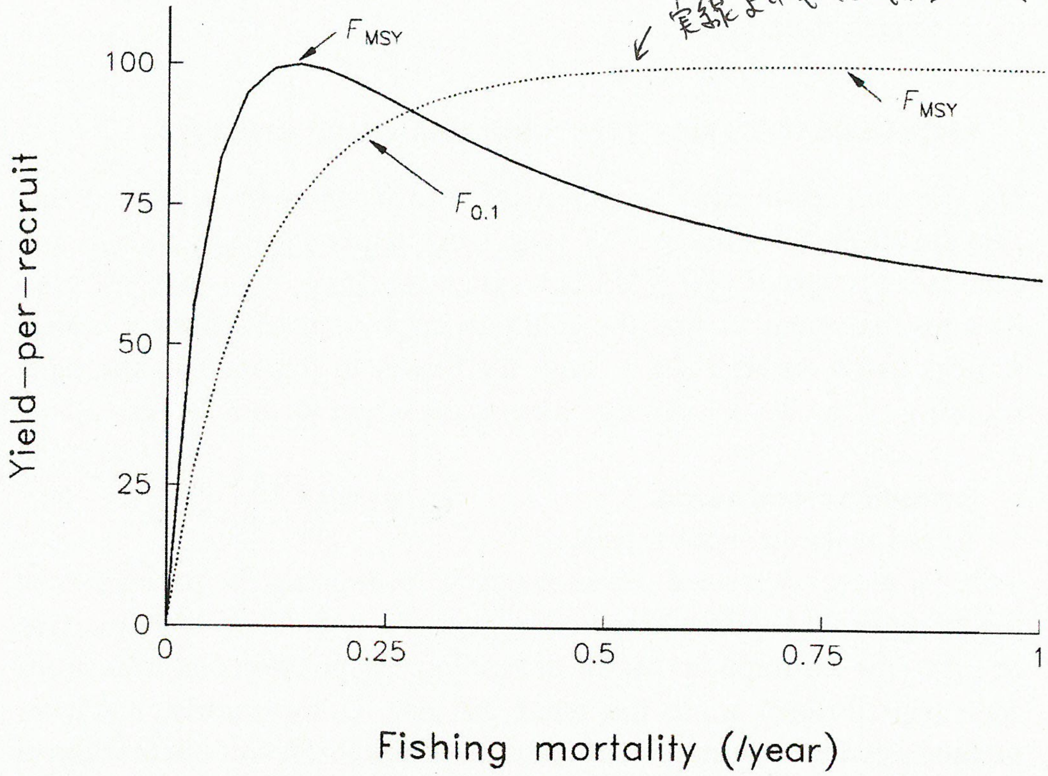

Figure 3.2. Yield per recruit against the instantaneous rate of fishing mortality. F_{MSY} is the level of fishing mortality at which the yield-per-recruit curve is maximised and $F_{0.1}$ is the level of fishing mortality at which the rate of change of yield per recruit is 10% of that at the origin. The dotted line shows a species that grows 10 times as quickly as the species with the solid line.

which in the long run gives the maximum yield in weight from the fishery'. The development of quantitative methods for determining the level of fishing mortality or fishing effort corresponding to MSY, and the support from the FAO, led to the wide adoption of MSY as a management goal (either explicitly or implicitly) in the 1950s, 1960s and even 1970s. Several international fisheries commissions (e.g. the Inter-American Tropical Tuna Commission, the International Commission for the Northwest Atlantic Fisheries and the North Pacific Fur Seal Commission) had adopted MSY as a management goal by the mid 1960s (FAO, 1966). The International Whaling Commission's 'New Management Procedure' (NMP), adopted in 1974, was based explicitly on MSY (Allen & Kirkwood, 1988). MSY was the only reference point explicitly referred to in the 1982 Law of the Sea Convention (Caddy, 1999).

1950〜1970のあいだMSY=targetだった。

By the mid 1970s, however, MSY as a management goal had moved beyond the intentions of its originators. For example, Holt & Talbot (1978) (cited by Smith, 1994) noted that 'like some other simplified concepts,

1970年代もともとの意図以上のものになった

maximum sustainable yield has become institutionalised in a more absolute and precise role than intended by the biologists who were responsible for its original formulation'.

PROBLEMS WITH MSY – THE CRUCIFIXION OF THE 1970s

① MSYすいていの不確実性

MSY批判

The criticisms of the MSY paradigm can be divided into three main categories: the ability to estimate MSY given uncertainty regarding models and data; the appropriateness of MSY as a management goal given other objectives for management; and the ability to implement effectively a harvest strategy based on MSY (i.e. to limit the fishery to actually take the catch according to the harvest strategy rather some larger (or smaller) catch).

③ 実行可能性

② 他の目的もある中でMSYをgoalとすることの妥当性

Estimation considerations

Surplus production model approach

Schaefer (1954) presented two methods for estimating the parameters of the relationship between sustainable yield and population biomass. One relied on the assumption that the data reflect a population that is in steady state (equilibrium) while the other did not. Unfortunately, the latter method required information on the catchability coefficient (the fraction of the biomass taken per unit effort), while the steady state (equilibrium) method did not. The equilibrium method involves regressing catch per unit effort (CPUE) on fishing effort (Figure 3.1b). Since such data are usually available, this method was applied widely. However, it has two major faults. First, the dependent and independent variables of the regression both involve fishing effort, which leads to some correlation between these variables irrespective of whether the data actually contain any information on the shape of the surplus production function (Sissenwine, 1978; Uhler, 1980). Secondly, the assumption that the resource is in steady state is seldom, if ever, valid.

Figure 3.3 shows relative error distributions for estimates of MSY for Cape hake off northern Namibia, based on the Schaefer (1954) equilibrium method and the method developed by Butterworth & Andrew (1984), which does not assume the population to be in steady state. The distributions were obtained from the results of 10 000 simulations, each of which involved generating an artificial dataset based on a fit of the discrete logistic model to the catch and data for northern Namibian hake (for details, see Punt, 1994). The results of the simulations show that the Schaefer (1954) method can be biased by close to 50%, while the non-equilibrium method is close to unbiased. The quantitative level of bias in Figure 3.3 may too low for most fish

33~84

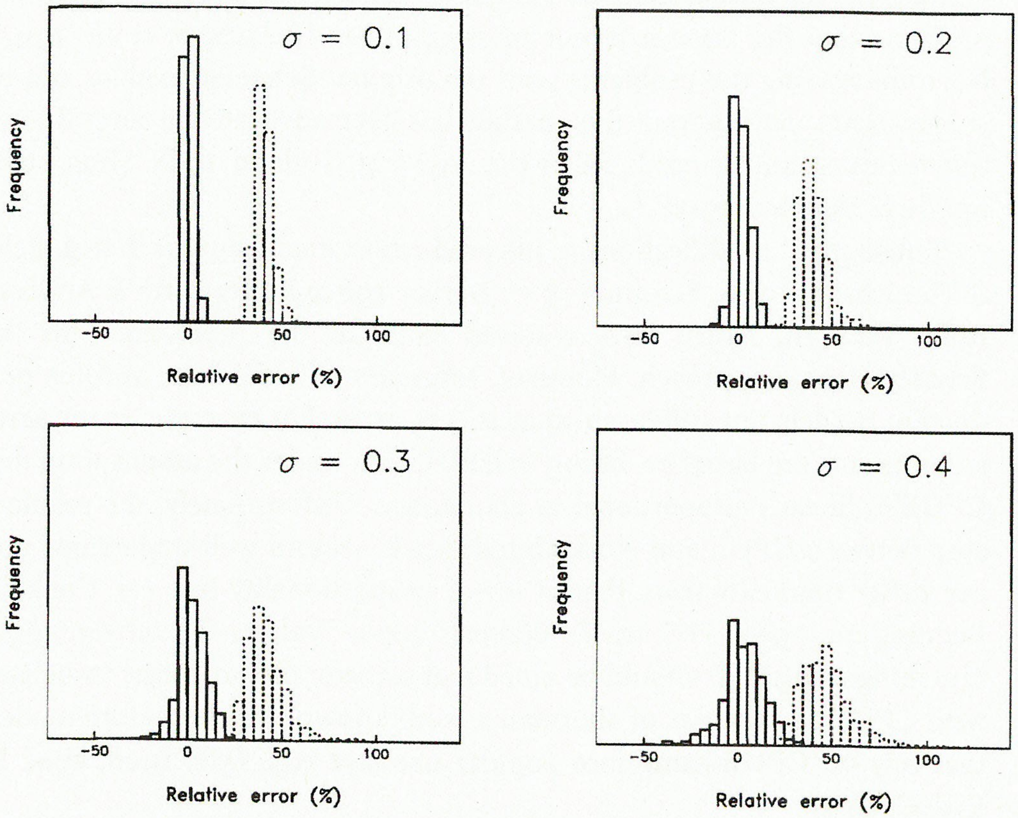

Figure 3.3. Distributions for the relative error of estimates of MSY for Cape hake off northern Namibia. Results are shown for the Schaefer (1954) equilibrium approach (dotted lines) and the Butterworth–Andrew (1984) non-equilibrium approach (solid lines), for four levels of assumed variability, σ, about the relationship between catch per unit effort and abundance.

stocks, as the dataset for northern Namibian hake is very informative. This is because it started with the commencement of substantial harvesting in 1964 and because the stock experienced large changes in both fishing effort and biomass over the period to 1987. Greater levels of bias and variability in estimates of MSY are to be expected for developing fisheries (Smith, 1993), and when effort and biomass do not vary substantially over the period for which data are available (Hilborn, 1979).

Gulland (1961) provided a modification to the Schaefer (1954) method that involved regressing the catch rate for each year on a 'moving average' fishing effort, in an attempt to deal indirectly with the equilibrium assumption. For a given year the moving average is defined by taking a weighted average of the effort for the year concerned and for some previous years. This method has been applied widely (e.g. Fox, 1975) and sadly continues to be. However, the averaging procedure does not remove the problem that the dependent and independent variables are implicitly correlated, a problem which is compounded because fishing effort is itself generally temporally correlated (Roff & Fairbairn, 1980), nor does it resolve the problem that the system is not in steady state (Polacheck *et al.*, 1993). Notwithstanding the problems with the original Schaefer method and its subsequent modifications, this method has been advocated in several stock assessment manuals produced by the FAO (e.g. Gulland, 1965; Sims, 1985; Sparre & Venema, 1992).[1]

Subsequent modifications to the production model approach (e.g. Pella & Tomlinson, 1969; Schnute, 1977; Deriso, 1980; Butterworth & Andrew, 1984; Hilborn, 1990) have resolved some of the problems with the Schaefer (1954) approach. However, estimates of MSY from surplus production models can still be substantially in error. For example, many stock assessments are based on fitting to CPUE data under the assumption that CPUE is linearly proportional to abundance. Unfortunately, the relationship between CPUE and stock abundance is seldom well understood and can differ markedly from that of linear proportionality (see e.g. Cooke & Beddington, 1984; Hilborn & Walters, 1992; Walters & Ludwig, 1994; Hutchings, 1996). It should be noted that some of the problems associated with CPUE as an index of abundance were known decades before models that rely on CPUE came into popular use (see e.g. Kyle, 1928, cited by Russel, 1931).

These problems can be overcome by fitting surplus production models to catch data and estimates of fishing mortality, rather than to catch and effort data (Garcia *et al.*, 1989; Caddy & Defeo, 1996). Estimates of fishing mortality can be obtained using information on the age structure of the catch (see Ricker, 1975; Hilborn & Walters, 1992) or from tagging studies. However, there remain many other sources of error in estimating MSY from surplus production models, including biases in fishing effort and the absence of reliable estimates of catches (owing, for example, to misreporting and discarding).

Yield-per-recruit approach

The yield-per-recruit approach has not suffered quite the same amount of criticism as the surplus production model approach even though the assumptions underlying most yield-per-recruit analyses are unrealistically simple. For example, the common practice of linking the fishing mortality

[1] At this point, the first author should confess that he was a co-author of a publication (Butterworth *et al.*, 1989) in which the Schaefer (1954) method was advocated (admittedly in a qualified way).

at which yield per recruit is maximised, F_{max}, to that at which MSY is achieved, F_{MSY}, is based on the assumption that recruitment is independent of spawner stock size for fishing mortalities between 0 and F_{max}. However, analysis of stock and recruitment datasets for many species (e.g. Myers *et al.*, 1994; Myers & Barrowman, 1996) reveals that this assumption is probably invalid for many, if not most, stocks by some amount. The relationship between F_{MSY}, F_{max} and other commonly applied fishing mortality reference points such as $F_{0.1}$, the level of fishing mortality at which the rate of change of yield per recruit is 10% of that at the origin (Gulland & Boerema, 1973; Figure 3.2), has been examined by several authors (e.g. Deriso, 1982, 1987; Clark, 1991, 1993; Mace, 1994; Die & Caddy, 1997; Punt, 2000). F_{max} is almost always the largest (and hence most risky) of the commonly used fishing mortality reference points (Punt, 2000).

F_{MSY} and other commonly used fisheries reference points such as $F_{0.1}$ are usually much larger than F_{crash}, the lowest fishing mortality, which, if fishing continued at that level, would eventually render the resource extinct. For example, based on a simple density-dependent demographic model, Punt (2000) showed that F_{crash} for Cape hake is three times larger than F_{MSY} and $F_{0.1}$. However, F_{MSY} can be quite similar to F_{crash} for stocks for which the stock–recruitment relationship exhibits depensation (Allee effect) or (more generally) for stocks for which recruitment drops off rapidly with reductions in stock size (Cook *et al.*, 1997; Punt, 2000). For example, Punt (2000) shows that F_{max} can exceed F_{crash} substantially for slow-growing, long-lived and unproductive species such as sharks. These species have low values for F_{crash} but are species for which F_{max} can be very large (or even infinite). Punt (2000) shows that F_{max} for school shark *Galeorhinus galeus* is infinite and $F_{0.1}$ is 0.169/year but F_{crash} ranges from 0.088/year to 0.385/year, depending on the assumed level of productivity.

The problem that F_{crash} may be similar to F_{MSY} for some species is exacerbated by uncertainty regarding the estimation of F_{MSY} and current fishing mortality from actual fisheries data. Imprecision in these estimates could lead to the estimate of F_{MSY} greatly exceeding F_{crash} for stocks for which F_{MSY} is really similar to F_{crash}. Unfortunately, F_{MSY} (and F_{crash}) is often poorly estimated using fisheries data because to estimate F_{MSY} accurately requires good information not only on growth rates but also on the shape of the stock–recruitment relationship. The latter is, however, seldom well determined because of uncertainty regarding estimates of spawner stock size and recruitment, and lack of contrast in spawner stock size.

Other problems with yield-per-recruit analysis are that the results can be sensitive to the assumed age-specific selectivity pattern (Goodyear, 1996), the level of natural mortality (Punt, 1994) and any age dependence in natural mortality. Furthermore, the yield-per-recruit curve is often flat near its maximum, so virtually the same yield per recruit can be achieved by fishing intensities markedly lower than F_{max} (e.g. see for example the dotted line in Figure 3.2).

Other considerations

A number of other problems stem from the fact that stock assessments, no matter how complicated, are nevertheless based on simple models of the system being managed. Such problems are not necessarily associated only with MSY-based management, but may be exacerbated if attempts are made to extract the maximum possible sustainable harvest. For example, spatial structure and biological interactions are generally ignored when researchers are conducting stock assessments, owing to lack of data. However, it is well understood that if the population consists of substocks rather than a single homogeneous resource, harvesting may target (and perhaps even extirpate) the more accessible substocks. Unfortunately, few attempts have been made to estimate MSY by taking account of the spatial dynamics of the population (a notable exception being Die *et al.*, 1990).

One implication of model uncertainty is that reality is often (substantially) outside the modelled estimates of uncertainty. This can be seen from the results of retrospective analyses (Sinclair *et al.*, 1991; Mohn, 1993, 1999). Retrospective analysis examines how well a model predicts events that have already been observed. For example, Figure 3.4a shows applications of the Butterworth & Andrew (1984) method to CPUE data for Cape hake off the west coast of South Africa. Results are shown for a range of final years of the assessment from 1978 to 1996. For the years beyond the end of the assessment, the curves correspond to projections to 1996 under the actual catches for the years from the end of the assessment to 1996. This is a clear example of 'retrospective bias', where the estimates of historical and future population sizes change systematically as additional data are included in the assessment. In this case, estimates of resource productivity reduce as the length of the data series is increased (Figure 3.4b). The reasons for retrospective bias will differ among stocks. For Figure 3.4, a key reason appears to be that, as the resource recovered, the frequency with which net liners were used (illegally) to capture small fish declined (D. S. Butterworth, University of Cape Town, personal communication). However, the general cause is that the model is too simple or the data are misleading. Sensitivity tests, although conducted routinely, cannot easily represent uncertainty due to model structure (Sainsbury, 1998).

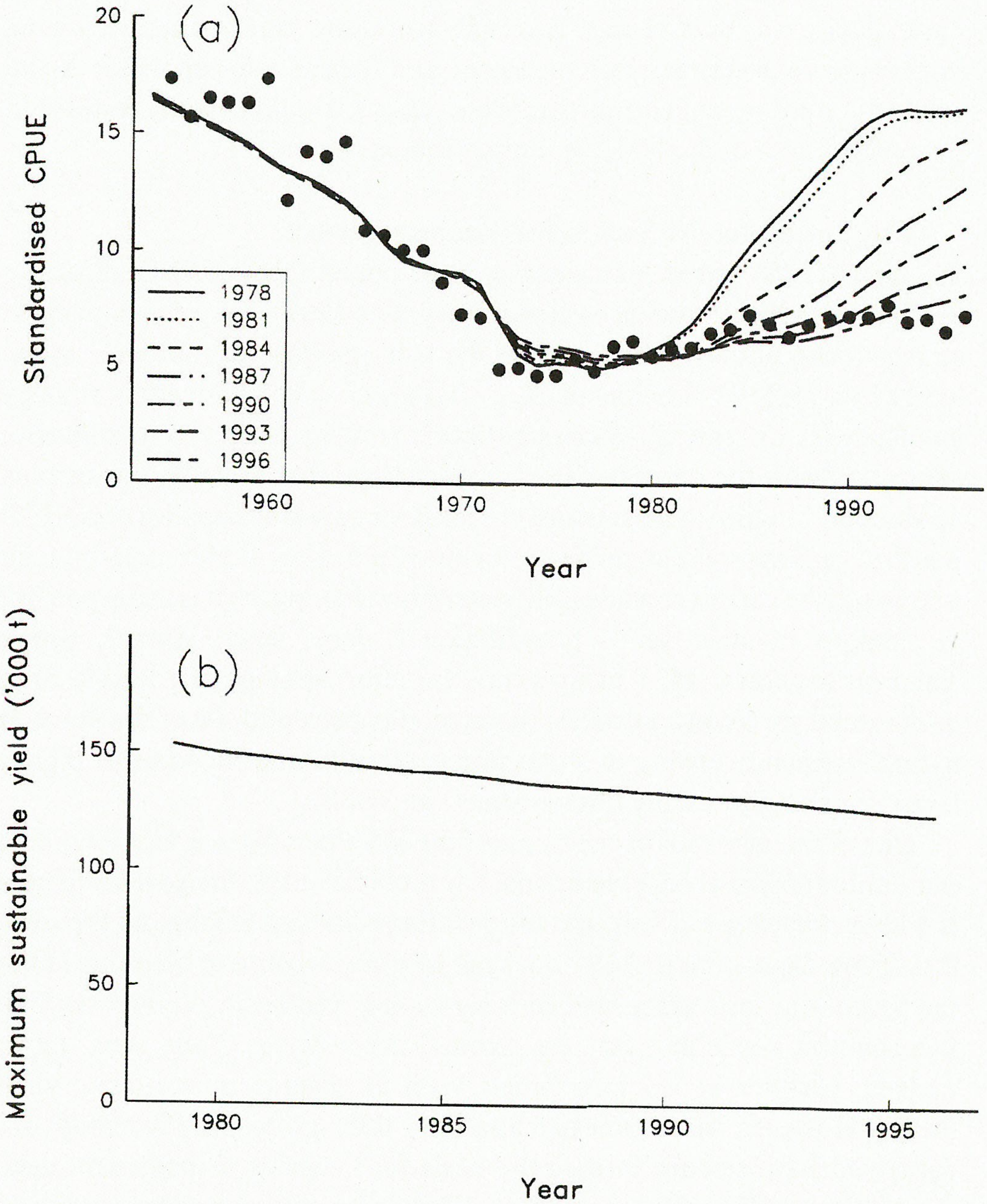

Figure 3.4. (a) Observed (solid dots) and model predicted catch rates for Cape hake off the west coast of South Africa for various choices for the final year of the assessment. The population is projected to 1996 for each assessment. CPUE, catch per unit effort. (b) MSY estimates as a function of the year in which the assessment was conducted.

Attempts to use more complicated (and realistic) models can lead to estimation problems, and worse performance than use of simple models (Ludwig & Walters, 1985).

The International Whaling Commission's management regime based on MSY apparently worked adequately for a few years but, by the late 1970s, problems emerged. These related mainly to the inability to estimate MSY and the current and pre-exploitation stock sizes reliably. The estimates of these quantities could change markedly from one year to another (owing to changes in assessment methodology and dataset choice), which led to wildly fluctuating catch limits (Kirkwood, 1997). This eventually resulted in the replacement of the NMP by a more robust regime.

Is MSY an appropriate goal for fisheries management?

The goal of MSY-based management in its purest form involves adjusting the fishery so that the catch each year is equal to MSY – i.e. a constant catch strategy. Alternatively, one could set the fishing effort (or mortality) to the level at which MSY is achieved, E_{MSY} (F_{MSY}), i.e. a constant effort strategy (see Ludwig, Chapter 2). A constant catch strategy is easy to understand, consistent with the deterministic considerations that underlie the surplus production function, and satisfies the need for constant supplies of catch to markets, and approximately steady income for fishers. Unfortunately, such a strategy also carries considerable risks of overfishing, as it is unresponsive to changes in stock levels (Beddington & May, 1977; Murray, 1993). Attempts to achieve MSY using a constant effort strategy may lead to large interannual variation in catches, because the susceptibility of the stock to natural variability (owing to, for example, recruitment) increases at higher levels of fishing mortality (Sissenwine *et al.*, 1988).

One of the other main criticisms of the MSY paradigm is that it ignores economic and social considerations. For most fisheries, the goal for fishers is dollars not tonnes of fish, and the goal for politicians is jobs and the votes they bring. Managing for MSY does not take any account of the value of the catch or of the costs associated with harvesting, processing and marketing. Bioeconomic modelling (see e.g. Scott Gordon, 1954; Clark, 1973, 1976; Ludwig, Chapter 2) led to concepts such as maximum economic yield (MEY). However, these concepts also have their problems (Cunningham, 1981) and have seldom formed the basis for fisheries management decisions. Discounting of future catches (Clark, 1973) places greater weight on current rather than future catches and hence maximisation of net present value (the sum of discounted future profits) is the 'correct economic objective' (Cunningham, 1981; Ludwig, Chapter 2). For a risky activity such as fishing, discount rates are likely to be high relative to the social discount rate. Unfortunately, future discounting can lead to an inability to achieve conservation-related objectives if the discount rate is high relative to the intrinsic productivity of the stock. For example, some bioeconomic theories that pay little attention to the costs of fishing and the response of price to supply suggest that it is 'optimal' to extirpate resources (such as baleen whales) if the intrinsic growth rate is smaller than the discount rate (Clark, 1973). A more common problem is that fishers often wish to defer reductions in catch (e.g. to achieve recovery from overexploitation) because they can see an immediate loss in profits for what is only the possibility for increased profits in the future. They rate the loss in profit now as not being as valuable as increased profits in the future. This fear may well have a sound basis because once recovery is achieved, it is not uncommon for the decision-makers to allow additional operators to enter the fishery with the net effect that the original participants are no better off than they were earlier.

The biological reference point $F_{0.1}$ (Gulland & Boerema, 1973; see Figure 3.2) is argued to correct for economic factors to some extent. However, this correction, while in the appropriate direction for most fisheries, makes arbitrary assumptions regarding the dynamics of prices and costs (Kaufmann *et al.*, 1999). In the 1970s, the concept of 'optimum yield' was introduced (Roedel, 1975) but unfortunately this meant something different to everyone who ever considered its use (Larkin, 1977; Cunningham, 1981).

The impact of social factors on the ability to implement fisheries management regulations is well known. Many of the risks faced by the fishing industry are social and political rather than biological (or even economic) and governments often look to fisheries to provide a source of income for coastal communities. Unfortunately, attempts to integrate economic and (particularly) social models with those used to provide biological advice on the status of fish populations have been (and continue to be) rather unsuccessful. At the time of writing, only frameworks (e.g. Catanzano & Mesnil, 1995) have been developed for this integration.

Can a management policy based on MSY be implemented effectively?

MSY (and its economic counterparts) arose from theoretical models of fish populations and fisheries. Unfortunately, the implementation of fisheries regulations is much easier in mathematical analyses than it is in the real world, and fisheries management is often described as being more about the management of people than that of fish (see e.g. Anderson, 1987). If MSY were known, setting the catch to MSY could be achieved by setting fishing quotas (or catch limits). However, situations in which multiple species are harvested but quotas are set for single species promote misreporting and discarding of species for which the quota is scarce or already used (Caddy, 1999). Systems based on allocating a proportion of the overall quota to each operator (an individual quota system) can be even worse because fishers who have fully used their allocation might discard excess catches even when many other fishers have hardly caught any of their allocation. In principle, allowing allocations to be transferable resolves this problem. However, for the Australian South East Trawl Fishery at least, fishers have been reluctant to sell or lease quota, so that quota-related discarding occurs even for species for which the total allowable catch (TAC) is underlanded by up to 50% (Tilzey, 1999).

The reality that fisheries management occurs in a political context cannot be ignored. Increases in TACs are far more likely to be acceptable politically (and socially) than reductions, so that the impact of the political process is often to keep catches at unsustainable levels, especially where there is substantial uncertainty. The desire not to reduce catches and fishing effort is perhaps not unexpected, given that the time needed for resources to rebuild frequently exceeds the tenure of the politician who has to make the decision to impose harsh management measures (Corten, 1996; Caddy, 1999). This problem is exacerbated by the tendency to appoint Ministers of Fisheries (or their equivalents) whose electorates include fishing communities.

Overcapacity occurs when there are more fishing vessels (fishing capacity) than are needed to harvest sustainably. It is acknowledged as a (if not the) key problem in achieving effective fisheries management (Mace, 1996, 1997; Garcia & Newton, 1997). Reduction in fishing capacity is both costly and causes social dislocation. However, for most fisheries, fishing mortality increases over time even if the number of vessels and the time they spend fishing remains constant as a result of technological advances such as the use of the Global Positioning System to identify fishing locations (e.g. Robins *et al.*, 1998). Consequently, fishing capacity has to be reduced virtually continuously if fishing mortality is to remain at desired levels. This is difficult to implement and governments themselves often encourage the increases in fishing capacity through, for example, boat-building subsidies.

All of the arguments listed above are based on the assumption that the fishery operates on a single species. In contrast, most fisheries are fundamentally multispecies (both biologically and operationally) in nature and consist of both target and non-target species. It is clear that it is impossible to achieve MSY simultaneously for all species in a multispecies fishery, because for almost any level of fishing intensity some species will be overexploited while others will be underutilised. Setting management regulations to attempt to achieve (single-species) MSY is inconsistent with 'ecosystem management'. This is because the evaluations of MSY ignore the impacts of removal of fish biomass on ecosystem dynamics and functioning, and the impacts of the use of fishing gear at different levels of intensity on fish habitat (see Kaiser & Jennings, Chapter 16).

THE 1980S AND PRECAUTIONARY MANAGEMENT – THE REINCARNATION OF MSY

Given the problems highlighted above with MSY, and how long ago most of them were identified, one would not expect the term 'MSY' to appear in the scientific literature after roughly 1980 (except, of course, to be criticised as an antiquated and rejected approach to fisheries management). In reality, a recent literature search indicated over 600 uses of MSY in abstracts since 1986. Almost all of the papers concerned aimed to estimate MSY or stock status relative to B_{MSY}, for a wide range of species from whales (Butterworth & Punt, 1992) to prawns (Wang & Die, 1996).

未だにMSYは
使われている

Why has MSY not been abandoned? The main reason is that MSY has shifted (been reincarnated – defined according to the *Oxford English Dictionary* as 'the formation of new flesh upon or in a wound or sore'!). It has changed from a management target to an 'upper limit'. The fishing mortality at which MSY is achieved, F_{MSY}, provides a logical upper bound for fishing mortality (Caddy & McGarvey, 1996). This is because both biological and economic (but not necessary social or political) arguments point to fishing mortalities in excess of F_{MSY} being undesirable.

なぜまだMSYは
使われているのか?
↓
MSYのイミが
変わってきた.
target → limit

MSY was redefined in the 1980s to reflect uncertainty more appropriately. For example, in New Zealand, where legislation dictates that TACs must be set to move the resource towards B_{MSY}, two alternative definitions for MSY are used: maximum constant yield (MCY) and maximum average yield (MAY). MCY is defined as 'the maximum constant catch that is estimated to be sustainable, with an acceptable level of risk, at all possible future levels of biomass' and MAY is defined as the maximum average catch that arises from applying a constant level of fishing mortality (Annala, 1993). The calculation of MAY and MCY is based on models that allow for variability in recruitment. MCY, being independent of the current biomass, is smaller than MAY to allow for the impact of variable recruitment. Unfortunately, adequate data are not available to estimate MCY and MAY for most of the stocks for which TACs are set in New Zealand. Therefore, simpler, more empirical, estimation methods are used for many species (for details see Annala *et al*., 1999).

MSYそのものも
不確実性を
考慮して再定義された
MSY → MCY
MAY
(加入の
変動を
考慮した
もの)

The 1980s and 1990s also saw the introduction to fisheries management of three closely linked concepts: fisheries reference points, the

80-90年代、
RP、PA、
フィードバック
control

precautionary approach and feedback-control decision rules. Each of these has had an important impact on the utility of MSY for fisheries management.

Fisheries reference points

Garcia (1996) defined a (management) reference point to be 'an estimated value derived from an agreed scientific procedure and an agreed model to which corresponds a state of the resource and of the fishery and which can be used as a guide for fisheries management'. A *limit reference point* is used to indicate 'the state of a fishery and/or a resource that is not considered desirable' while a *target reference point* is used to indicate 'the state of a fishery and/or a resource that is considered desirable'. A *threshold reference point* is used to identify 'that a fishery and/or resource is approaching a target and [or] limit reference point'. Note that if a limit reference point is triggered, this does not mean that the species has a high risk of biological extinction; an appropriate response to a limit reference point being triggered would be a reduction in fishing mortality rather than, say, a closure of the whole fishery. If appropriately set, the target probability of triggering a limit reference point should be low but clearly not zero.

Several papers have attempted to define appropriate target and limit reference points (e.g. Mace, 1994; Myers *et al.*, 1994; Caddy & McGarvey, 1996). However, although by no means the only limit reference point in current use, F_{MSY} (ideally not estimated by F_{max}), is increasingly being selected as the fishing mortality-based limit reference point. For example, the UN Fish Stock Agreement (United Nations, 1995) states that F_{MSY} should be used as the minimum standard for limit reference points, while for overexploited stocks, B_{MSY} (see Figure 3.1) can act as a (minimum) rebuilding target. Note that the use of $\boxed{F_{MSY} \text{ and } B_{MSY}}$ rather than MSY when defining reference points deals with the concerns that MSY is a deterministic concept.

MSYでなく

を使うというのが
決定論的な MSYに対する
concernに答えるもの

The precautionary approach

The FAO precautionary approach to fisheries management (FAO, 1995) involves *inter alia*:

1 Being more cautious when information is less certain.
2 Not using the absence of information as a reason to postpone or fail to implement conservation and management measures.
3 When developing management plans, taking account of uncertainties relating to the size and productivity of stocks, reference points, stock

condition relative to those reference points, levels and distribution of fishing effort, and impacts on non-target and associated or dependent species.

4 Taking a very cautious approach to the management of newly developing fisheries until sufficient data are available to assess the impact of the fishery on the long-term sustainability of the resource.
5 Defining and implementing limit and target reference points.
6 Defining, in advance, decision rules for stock management.

Feedback control decision rules

Decision rule.

Decision rules specify the management actions to take if limit or target reference points are exceeded. More generally, they specify the data to collect, how to analyse those data, and the management actions to take given the results from the analyses (Kirkwood & Smith, 1996; Butterworth *et al.*, 1997; Cochrane *et al.*, 1998). The performances of decision rules are evaluated by simulation. This involves the development of a range of (operating) models that represent the alternative possible dynamics of the system being managed, and can generate data typical of those used when conducting assessments and hence determining management actions. The operating models are then used to represent possible 'real worlds', and the decision rule is used to manage the fishery. The success (or otherwise) of the decision rule is determined by whether it is able to satisfy the (prespecified) objectives for management. The operating models are based on experience with the system being managed as well as on hypotheses related to the future behaviour of the system. The operating models can consider a wide variety of factors, including the behaviour of the fish stock, changes over time in the environment, and the reaction of fishers to management regulations. Butterworth & Punt (1999) reviewed many of the factors considered to date in operating models.

オペレーティングモデル
|
MSE

A management plan for a fishery is (generally) a legally binding agreement that describes the access rights, gear restrictions and any decision rule for the fishery. FAO (1995) and the UN Fish Stock Agreement (United Nations, 1995) both emphasise the need not to implement a management plan until it has been shown to perform effectively in terms of its ability to avoid undesirable outcomes. A management plan can be tested using the approach described above only if it involves clearly specified decision rules. The inclusion in management plans of decision rules that have been evaluated by simulation is usually supported by both industry and conservation groups (Cochrane *et al.*, 1998; Smith *et al.*, 1999). The former see decision rules as a form of security against the machinations of the managers, while the latter gain confidence that a set of rules are in place that have been shown to perform adequately (at least within the simulation framework).

Limit reference points play a direct role in the evaluation of management plans in that the performance of a management plan is evaluated *inter alia* in terms of whether the simulated stock triggers the limit reference point an undesirable number of times (Punt *et al.*, 2001). Using B_{MSY} as a limit reference point reintroduces one of the reasons for its original development, namely as a definition of overfishing (Smith, 1994). The role of B_{MSY} as a target reference point (and target reference points in general), within an evaluation of management plans, is questionable. As long as the probability of not dropping below the agreed limit reference point is low, the biological concerns about the stock should have been addressed. What to do once these concerns have been addressed then becomes an economic (and social) rather than a biological issue. However, New Zealand, for example, has legislative obligations to set TACs to move the resource towards B_{MSY}, and B_{MSY} is a natural (and convenient) target. Starr *et al.* (1997) examined the performance of a decision rule for a rock lobster population where the objective for management is to allow a high probability of rebuilding to B_{MSY}.

以前の失敗をどうこくふくするのか？

How does the reincarnation of MSY overcome its previous failings?

There were many criticisms of the use of MSY in fisheries management. It would be naive to believe that all of these have been overcome by the use of F_{MSY} and B_{MSY} as limit rather than target reference points and by developing management plans that include decision rules whose performance has been evaluated by simulation. However, many of the criticisms have been dealt with to a considerable extent. For example, methods of stock assessment based on the assumption that the stock is in equilibrium are used only rarely now. This is illustrated by the conclusion by Polacheck *et al.* (1993) on the basis of a simulation study that 'Under no circumstances should agency staff, conference organizers, reviewers, managers or journal editors accept assessments or publications that are based on effort-averaging or process-error estimators only'.

・targetでなくlimitとして使う
・シミュレーションを使ったかんりプランを使う
だけで全てをこくふくできるわけではないが多くはとり扱える。

・例えば平衡状態を仮定したモデルはほとんど使われていない

The adoption of B_{MSY} as a target for fisheries management (as is the case in New Zealand; Annala, 1993) is not equivalent to the previous practice of aiming to stabilise the biomass at B_{MSY} and take a constant catch of MSY. Rather B_{MSY} is interpreted probabilistically, and management measures are set to ensure that the biomass does not drop below B_{MSY} with a preagreed level of probability. During the 1980s and 1990s, there were substantial improvements in the methods used to conduct risk analyses for fisheries (see e.g. Smith *et al.*, 1993; Francis & Shotten, 1997; Punt & Hilborn, 1997). While problems still remain (see e.g. Cordue & Francis, 1994), considerably more confidence can be placed in the methods of risk analysis now being applied. The approach of not basing management regulations (e.g. catch limits) on the 'best' estimate but rather basing them on a lower percentage (than 50) of distributions that allow for estimation uncertainty (see e.g. Punt & Smith, 1999) means that lower harvest levels will be imposed when uncertainty is large.

The evaluation of a management plan should assess whether it is robust both to statistical (data) uncertainty, and to incomplete knowledge of factors such as stock identity and abundance. As such, the evaluation implicitly incorporates points 1 to 4 above. The most thorough evaluation of a management plan was conducted by the Scientific Committee of the International Whaling Commission, which considered the impact of uncertainty related to stock structure, historical catches, biases in abundance estimates and environmental change (Donovan, 1989; Kirkwood, 1997). It is now unusual for assessments (in Australia at least) to be based on a single hypothesis of 'how the world works' and, increasingly, very many scenarios are being considered when researchers are conducting assessments and providing management advice (e.g. Punt *et al.*, 2000). Nevertheless, problems remain in identifying the correct range of hypotheses to consider when one is evaluating a decision rule (although this range is starting to expand considerably) and in assigning weights to different hypotheses. However, it is hard to argue with the comment by Sainsbury *et al.* (2000) that 'If a strategy doesn't work on a simple [operating] model what justification is there for assuming that it will work in the real world'.

One of the major deficiencies of earlier assessments was the reliance on commercial catch rates as an index of stock abundance. While this practice remains reasonably common, fisheries assessments are now often based on 'fishery-independent' methods of indexing abundance.

The greatest challenges to fisheries management remain outside the scientific realm. These include overcapacity, the sharing of resources among different user groups, and the political and social pressure for short-term benefits at the expense of long-term environmental degradation. In essence, the last issue can be considered explicitly by quantifying expected risks. However, ultimately, the selection of the appropriate risk level is a political issue and only with appropriate consultation with all stakeholders and increased ownership and responsibility by stakeholders are major changes in 'acceptable' risk levels likely to occur.

CONCLUSIONS

MSY as a concept has been used in fisheries at least since the 1950s. It has been used variously as a management goal, a harvest strategy, and a biological reference point. The death of MSY as a key concept in fisheries management, as expected from its crucifixion in the 1970s, has clearly been overexaggerated. While achieving a constant catch of MSY is no longer regarded as an appropriate management objective, the notion remains that MSY provides a reference against which exploitation can be measured. It has been reincarnated as the fishing mortality and biomass at which MSY is achieved (F_{MSY} and B_{MSY}) and remains a key concept in fisheries management, where it has now emerged as a limit rather than as a target reference point. This reincarnation of MSY has addressed many of the concerns related to estimation problems, and lack of consideration of stochasticity. We wait with interest to see whether future studies on this subject can argue that the concerns related to the larger problems of ignoring social, economic and ecosystem issues are addressed by the next reincarnation of the ubiquitous concept of MSY!

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