Best Practice Reference Points for Australian Fisheries

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CONTENTS

Executive Summary

Reference points are the operational or measurable benchmarks that identify targets to be achieved on average, limits to be avoided, or triggers to initiate specific management responses. A fishery is expected to approach or fluctuate around a target reference point, to have a very high probability (at least 90%) of not violating a limit reference point, and to have trigger reference points and planned management responses that achieve these two outcomes.

Best practice reference points are considered here for five elements of environmental management that are central to modern fishery management – the target species; by-catch species; threatened, endangered or protected species; habitats; and food webs. These reference points are only one part of the management system, and how they are used can be as important to determining management outcome as what they are and the levels at which they are set. Consequently, a 'best practice management context' for the use of best practice reference points is also provided for each of these five elements.

The 'best practice' concept is based on the best practice that has been demonstrated through use, and recognises that views of what is 'best' will continuously improve with experience. Best practice is not an absolute or fixed entity, or a guarantee of adequacy. It is based on experience to date and it is expected to evolve over time.

In Australia, as elsewhere, there is a long history and focus of fisheries research and management on target species. The other elements of environmental management have come to prominence only in the last few decades. This means that best practice reference points are much better developed and tested for target species than for the other elements in environmental management. Greater change and evolution of what is regarded as best practice is expected in future for these other elements than for the target species.

The specification and use of reference points for key management issues and objectives is widely regarded as the desired approach to fishery management. It is recommended through recent international agreements and guidance (e.g. the UN Fish Stocks Agreement (UNFSA 1995), the FAO Code of Conduct for Responsible Fishing (FAO 1995a) and the FAO Ecosystem Approach to Fisheries (FAO 2003)). Thus the use of reference points can be regarded as the current 'best practice' approach to fisheries management. However the reference point approach has some recognised shortcomings that must be kept in mind during application. These include:

- The reference point approach can encourage an over-simplistic and often incorrect view that risks and benefits always change abruptly at certain thresholds - for example that a population is in extreme peril anywhere on one side of the limit reference point and is totally safe anywhere on the other side. Abrupt thresholds can exist in the response of populations and ecosystems to harvesting and other human activities. But more usually the change in risk is steady and smooth. The choice of where to place a reference point is a balance of risks, and there can be significant risk even on the 'safe side' of a limit reference point (especially as the limit is approached from the 'safe side'). Use of reference points should recognise that risk will not abruptly change at a

reference point. This is especially important when identifying trigger reference points and the resulting management response, and when considering the consequences of uncertainty in the estimation of where a fishery currently is in relation to its reference points.

- The use of reference points may draw and lock research and management into a narrow set of scientific issues and management responses that are strongly determined by the reference points themselves. For example it may encourage pursuit of highly precise and complex estimations for the chosen reference points, and consideration of only a few kinds of management controls and strategies that easily relate to those reference points. A fishing mortality reference point may overly focus management strategies on the control of fishing mortality alone. It may discourage consideration of simpler or more direct indicators and performance measures, and of alternative management strategies.

- The use of reference points encourages the separate examination of issues and can obscure consideration of the 'bigger picture'. Focusing on just a few specific elements can help ensure that those elements are addressed. But it can also distract from consideration of what is happening to the coupled management and ecological systems as a whole. Furthermore well-intentioned actions in response to one issue might jeopardise another. This is particularly a risk for whole-system or large sub-system properties that are not well understood and consequently have weakly developed reference points, e.g. for habitats, food webs, biodiversity (at genetic, species and community levels) and many other aspects of ecosystem structure and function that are the basis of fishery production.

The myopic application of reference points, like the myopic application of almost any management tool, is unlikely to be successful. The management intent, context and the full range of options remains crucial, and these can be expected to change through time. Nevertheless, sensibly used, reference points are a key part of modern fishery management and science.

Reference points for target or commercially retained species
For target species, best practice involves setting reference points for both biomass and fishing mortality. This is because while fishing mortality is under more direct management control, it is biomass (and related population structure) that influences key ecological processes and functions. Specifically populations with relatively large biomass and with full age/size structure are expected to e more likely to maintain their genetic diversity and natural genome, to be more resilient to recruitment overfishing, recruitment variability and environmental perturbations, and to maintain food-web structure and stability. Best practice target reference points for species high in the trophic structure of the food web are different from those for species that are low in the food web and that provide a significant prey source in the food web. These 'prey species' are dealt with in the food web element of this report, and the best practice described there applies irrespective of whether the prey species are targeted catch or by-catch. The best practice target reference points below are for species that are not regarded as key prey species in the ecosystem.

Limit reference points are set primarily on biological grounds to protect the stock from serious, slowly reversible or irreversible fishing impacts, which include

recruitment overfishing and genetic modification. The best practice limit reference point for biomass is the greatest of 3 quantities, or proxies for them:
- B_{lim}, the biomass below which average recruitment declines or stock dynamics are highly uncertain.
- $0.3 B_{unfished}$, where $B_{unfished}$ is the biomass expected to be present at a specific time in the absence of fishing. The biomass initially present when the fishery started, B_0, is commonly used as an unchanging proxy for $B_{unfished}$. But this is becoming increasingly unsatisfactory because the underlying assumption of stationarity is less tenable under the emerging understanding of natural ecosystem dynamics and the system-level effects of climate change and other anthropogenic effects. Instead a dynamic, time-varying estimate of $B_{unfished}$ should be used. This can be provided by model calculations based on the expected stock dynamics in the absence of a fishery, by reference to unfished sites, or a combination of both. For stocks that naturally exhibit large fluctuations in productivity the $0.3 B_{unfished}$ can give very low levels of absolute biomass during periods of low productivity. In these cases an additional limit reference point is required, which should be no lower than 0.2 of the median long-term unfished biomass.
- The biomass from which rebuilding to the target reference point could be achieved in a period that delivers human intergenerational equity (20-30y).

$0.2B_{unfished}$ is commonly used as a limit reference point and there is good empirical support that this avoids recruitment overfishing for productive stocks (i.e. is an appropriate B_{lim} for such stocks). But it is not regarded as the best practice limit reference point because this level of depletion (i) does not avoid recruitment overfishing in low productivity stocks, (ii) may not provide adequate protection for other fishing impacts that are likely to be slowly reversible or irreversible (e.g. genetic modification, reduced age structure with consequences to the quality of spawning, changed ecological role such as in food-web dynamics, ease of population recovery from the limit), (iii) is less robust to uncertainty in estimation and model specification, including to changes in the climate or ecosystem, and (iv) is not consistent with the precautionary reference point approach of ICES where B_{pa} was found to be about $1.4B_{lim}$ in fishery assessments based on good data sets.

The best practice limit reference point for fishing mortality is F_{MSY}, the fishing mortality giving maximum sustainable yield. Where this cannot be estimated directly, $F_{50\%}$, the fishing mortality that gives a 50% reduction in the spawning biomass per recruit is a default proxy for most species. For the stock-recruitment steepness seen in most fish (i.e. greater than about 0.3) $F_{50\%}$ provides more than 80% of the MSY and depletes the biomass to no more than about 30% of the unfished level. Use of a lower percentage in the 'per recruit' proxy value would require explicit justification as to why $F_{50\%}$ is unreasonable. Higher fishing mortality reference points (e.g. $F_{40\%}$) could be justified if there is information to suggest that the stock has high steepness in its stock-recruitment relationship. $F_{60\%}$ should be used as the default limit reference point for species suspected of having a particularly low ability to compensate for fishery removals (e.g. those with a very low natural mortality or very low 'steepness' in the stock-recruitment relationship).

For key prey species lower in the food chain the target and limit reference points for fishing mortality may be different, as they are chosen to achieve the requirements of

the target and limit reference points for biomass. But if the target biomass for key prey species is half way between the unfished and MSY biomass then a limit fishing mortality in the vicinity of $F_{60\%}$ to $F_{80\%}$ would not be unexpected for such species in many circumstances.

The best practice target reference point reflects a combination of two requirements – achieving the socioeconomic objectives of the fishery and avoiding the limit reference point.

The first requirement is the social choice of optimal yield, and there is a great deal of flexibility in this. For example the fishing mortality or biomass giving maximum economic yield can be used as a target if it is judged to appropriately meet socioeconomic objectives. Alternatively biomass targets that provide for non-economic or human community needs can be used. But in any event the best practice target reference point for fishing mortality is less than F_{MSY} and best practice fisheries maintain stock biomass at or above B_{MSY} on average.

The second requirement is that the target reference point be set at a level and in a management context that gives a low probability of breeching the limit reference point. The level of the target reference point is relevant to this, and other things being equal the chance of violating the limit reference point increases as the target is set closer to the limit. But the management context for use of the target and trigger reference points is critical in determining management performance. Best practice treatment of the target reference point is that it be set with an agreed trigger reference point, planned management responses and/or decision rules such that there is a high cumulative probability (at least 90%) that the fishery will not breech the limit reference point over an extended period (at least 2 generation times of the species) under the range of environmental and productivity conditions that could be reasonably expected to occur in that time.

The best practice specification of the recovery time if the limit reference point is breeched is no less than the time to recover without fishing and up to the time to recover without fishing plus 10y.

Reference points for by-catch species
By-catch is used here in the sense of the Australian Commonwealth Policy on Fisheries By-Catch. That is, by-catch species are landed but then discarded, or species that are affected by fishing gear even though they are not landed. But they are not target or other commercially retained species that are managed explicitly through management plans even though these species may be sometimes discarded.

By-catch is an unintended or incidental consequence of seeking the benefits from fisheries. The necessary requirement for the limit reference point is that populations of by-catch species are maintained and are not excessively depleted. The ideal, reflected in the target reference point, is to have minimal or no by-catch.

The distinction between retained and by-catch species is a result of human values and utilisation, rather than one of biology or ecology. In that limit reference points are set to so as to prevent slowly reversible or irreversible biological impacts there is no biological basis for by-catch and retained species having different limit reference

points. Unless there is a management intention to cause such impacts on by-catch species the same limit reference points should apply to populations of by-catch and retained species. Consequently the best practice limit reference points for by-catch species are the same as above for target and retained species. The target reference points for population biomass of by-catch species may differ from those for target species, and reflect the needs of achieving optimal utilization in the fishery overall. The indicator and limit reference point may not be directly measurable for all by-catch species because often there is very limited information available about historical fishery catches, population abundances or the key biological and ecological properties of by-catch species. It may not be either warranted or feasible to provide the information necessary for direct measurement of the limit reference points. In these cases, proxies for the limit reference points can be developed in a risk assessment framework that is explicit in terms of the justification for the proxies, the evidence for assessment of risk, and the use of precaution to achieve the intent of the limit reference points despite uncertainties. The Ecological Risk Assessment methodology currently being applied by the Australian Fisheries Management Authority is best practice in this context. The methods used to estimate precautionary catch limits for threatened, endangered and protected species (below) or the CCAMLR method to estimate precautionary catch limits are best practice for the setting of by-catch limits from limited data.

The best practice target reference point for the catch of by-catch species is zero, with the recognition that this is a target to be approached as far as is feasible or acceptable. Best practice also establishes interim limits to catch or fishing mortality for by-catch species that reflect what is currently regarded as being feasible or acceptable, and these are expected to change through time to reflect continuous improvement toward zero by-catch of these species. The interim limit on catch or fishing mortality must be lower than that implied by the limit reference point. In the absence of more specific information the best practice interim fishing mortality is 0.75 of the natural mortality rate, but this will be lower for species with very low productivity (e.g. for species where direct observations or analogy with similar species suggests very low natural mortality or very low steepness in the stock-recruitment relationship). The interim level is a limit not a target, as it is not desirable to achieve it. Best practice establishes trigger reference points to initiate management measures to reduce the chance of further by-catch if undesirable levels of by-catch occur, with at least one such trigger reference point being at the identified current feasible level.

These reference points relate to the species making up the by-catch, but a common management concern is the effect of by-catch on the structure and function of the ecosystem as a whole. There is considerable scientific effort going into the evaluation of possible indicators and reference points for these whole-system effects, but currently there is no agreed or demonstrated best practice in their selection or use.

Reference points for threatened, endangered or protected species
Species are usually recognised and managed as threatened, endangered or protected through a legislative process, or by international agreement (e.g. the Convention on the International Trade in Endangered Species of Wild Fauna and Flora, CITES), and these processes determine what benchmarks or requirements must be applied by a particular jurisdiction. However there are also mechanisms for identifying endangered

or threatened species that are not legislatively based, for example the IUCN (World Conservation Union) 'Red List'.

Best practice management for threatened, endangered or protected species must allow the species to recover, if it is depleted, and to remain undepleted. Consequently these reference points relate to the mortality that is imposed. The target reference point is minimal or zero fishing mortality, with the recognition that this target is to be approached as is feasible. The limit reference point is a fishing mortality that unacceptably reduces the population or unacceptably slows recovery.

The best practice limit reference point is a mortality or number of deaths calculated using the Potential Biological Removals (PBR) method with 'recovery factor' (F_r) of 0.5, or variations of that method with similar intent. This is a highly precautionary method that can be applied with limited information (life history and an estimate of population size) to calculate the number of deaths that would significantly impair recovery of depleted or severely depleted populations and/or to maintain already healthy populations. This is applied at the stock level, rather than the population level, if stocks are suspected of existing in the population.

Best practice also establishes levels of catch or fishing mortality to reflect what is currently regarded as being feasible and acceptable, and this is expected to change through time to reflect continuous improvement. The 'currently feasible' level would usually be a relatively small fraction of the limit reference point for mortality or removals, so that recovery is achieved. The 'currently feasible' levels are not targets to be achieved. Rather they are benchmarks for continuous improvement and can be triggers to initiate additional management intervention if the intended improvement is not achieved.

Reference points for habitats
A habitat is the biological and physical environment in which an organism lives. Different aged organisms, especially of fish, molluscs and crustaceans, usually occupy different habitats so that there is a 'chain of critical habitats' that is required by a species to complete its life cycle. Habitats are considered one of the basic determinants of the structure and productivity of marine ecosystems, and of the kind and amount of fishery production available.

Fisheries currently do not affect the oceanographic habitats of fish. But they can affect the seabed habitats through direct contact and removal of habitat forming organisms (e.g. sponges and corals) or some types of geological structures. Habitats determine the carrying capacity or productivity of the fishery target species, and of other species or characteristics of interest such as potential future target species, by-catch species, threatened, endangered and protected (TEP) species, and biodiversity and ecosystem processes.

It is recognised that habitats are a critical element of the ecosystems supporting fishery production. But direct management of fishery impacts of habitats is at an early stage of development and implementation, and there is no widely agreed approach to the selection or use of reference points for habitat management. Nevertheless examples of best practice are emerging and simple theoretical guidance is available about the likely limits of habitat modification for sustainable fisheries.

The best practice target reference point for habitat impacts is for no impact on relevant seabed habitats, modified as appropriate to include acceptance of minimal and temporary impacts. This is consistent with the theoretical predictions that yield from a habitat-dependent target species is reduced if the relevant habitat is reduced, and that reduction of the habitat to less than 0.6 of the unfished areal extent could result in the target species becoming excessively depleted. At this time the management of wild capture fisheries does not include the intentional modification of habitats to enhance the production of particular species and/or reduce the abundance of others, so target reference points for intentional change to habitats do not arise. But achieving and maintaining high yield from a habitat-dependent target species requires minimal loss of its habitat. The best practice context for management of habitats is to identify 'critical habitats' for species of interest, and to ensure such habitats are exposed to no more than minimal and temporary impacts. If a wide enough range of species is considered this effectively becomes a 'no net loss' requirement from the unfished habitat coverage because all habitats are likely to be critical to one species or another.

The best practice limit reference point for habitat impacts is for relevant habitats to be reduced to no more that 0.3 of the unfished areal extent. This is consistent with avoiding excessive depletion of the habitat-forming organisms themselves and of habitat-dependent species that are not subject to fishing mortality.

While this is the existing best practice limit reference point for habitat impacts there are theoretical grounds for regarding it to be inadequate for protection of habitat-dependent species that are also subject to significant fishing mortality (i.e. approximately F_{MSY} or greater). And some habitat-dependent by-catch species may have low productivity and consequently a low F_{MSY}, so that significant fishing mortality may result from relatively small catches. In cases where the species is exposed to significant fishing mortality in addition to habitat loss a more appropriate limit reference point would be reduction of the relevant habitats to no less than 0.6 of their unfished areal extent.

The full spatial range of the habitat type should be included in calculating these proportions of the unfished areal extent of habitats. This could include equivalent habitat beyond the spatial range of the fishery, in protected areas, and un-impacted habitat within the fishing grounds. In some circumstances it may be appropriate or necessary to consider habitat quality rather than simply areal extent.

Reference points for food webs
Food webs provide the direct basis of fishery production and determine many other attributes of marine ecosystems. Issues of concern in relation to the effect of fisheries on food webs include impairing the size, productivity or resilience of predators (e.g. fish, birds, marine mammals) through removal of their prey, and destabilising or switching food webs and related ecosystem structure to different 'stable states'. The fishery productivity and sustainable yield of predators can be reduced by simultaneously fishing their prey. There is a considerable body of science that describes the mechanisms and examples where food web interactions have led to significant undesired and unintended outcomes in fisheries. There is also evidence that demonstrates marine food webs can be very flexible and resilient. Marine food webs are complex systems and there is no simple and general summary of their

dynamics. However there is no doubt that fisheries can and do have effects on both caught and non-caught species through food web interactions, and consequently on ecosystem structure and function as a whole. The growing understanding of the importance of small pelagic fish in controlling both the abundance of their larger predators and of their smaller prey in productive pelagic ecosystems means that fisheries for small pelagic species are of particular concern in this regard.

Best practice in the management of food web interactions is not well developed. However a minimal requirement in the management system is explicit recognition of the potential for food web interactions and an ability to modify fishing controls in order to manage significant food web interactions that are considered likely. There are two broad approaches to relevant reference points and management responses – one concerned with the food web as a whole, and the other focused on identified key elements or connections in the food web.

Food web as a whole
Approaches that address the food web as a whole have generated a large number of potential indicators to measure change, mostly derived from food web models. While this is a very active field and advancement is likely, it has not yet provided demonstrated best practice through the use of these indicators and associated reference points in management decision-making. Current thinking is that a suite of indicators and reference points may be needed, including comparison with unfished reference sites. One potentially useful indicator (currently without a developed reference point) is a comparison of actual catches to theoretical catches as a fishery changes the trophic level it is harvesting – a departure between these two would indicate food web disruption.

Key elements of the food web
Typically this approach focuses on key prey species for predators of particular concern (e.g. dependent fishery target species and protected species). Best practice involves explicit nomination of significant prey species or forage species in fisheries management plans, and having specific management conditions and reference points for them.

In some cases the management conditions totally preclude the development of commercial fisheries on species that are designated as significant prey species, either as a permanent limitation or as a precautionary measure while better understanding is sought. In such cases best practice sets the permitted by-catch levels or trip limits (i.e. limits of catch per fishing trip) for designated prey species that are very low compared to likely species productivity and consistent with only non-commercial and incidental take. The intention is to discourage targeted commercial fishing.

When targeted commercial fishing of designated significant prey species is permitted, best practice reference points are selected so as to maintain the productivity and ecological viability of predators. In a few very well studied situations in relatively simple ecosystems it has been possible to explicitly model these interactions and estimate the appropriate reference points and management controls. While this represents best practice it will not be feasible in many situations, and it remains unclear whether the reference points derived from these cases could or can be generalised.

In the absence of appropriate trophic models the best practice target reference point for biomass of nominated key prey species is no less than the mid-point between the unfished biomass and B_{MSY}. The justification is that in an unfished state the whole unfished biomass is available to direct predators and the food web more generally, and that both experience and theory suggest that reducing the biomass of prey species to the level that gives MSY can cause undesirable impacts on direct predator species and elsewhere in the food web – including reduced yields and overfishing of these other species as a result of combined harvesting and prey reduction.. The mid-point of the unfished level and the B_{MSY} level biomasses is an arbitrary balance between meeting fishery and food web needs. It can be modified in light of more specific understanding, but it is the default approach that is currently best practice. In the absence of sufficient information to estimate B_{MSY} current best practice is to assume a logistic production model, for which B_{MSY} is at 50% of the unfished level, so that the best practice target reference point for designated key prey species is reduction to 75% of the unfished biomass.

The best practice limit reference point for key prey species is as for target species higher in the food chain, but in any event it should not be less stringent than a 0.1 probability of being at or below 20% of the median unfished biomass.

Best Practice Reference Points for Australian Fisheries

Introduction

Reference points are the operational benchmarks used in fishery assessments and decision-making for fishery management. There are three types of reference points used in fishery management – target reference points which identify an intended outcome; limit reference points which identify an undesirable outcome to be avoided with high probability; and trigger reference points which initiate a predefined management response. Target and limit reference points provide the benchmarks used for judging performance of the fishery at any point in time. Trigger reference points are an integral part of the management system for maintaining the fishery within acceptable bounds (i.e. close to the desired target and away from the undesired limits). Target and limit reference points can simultaneously function as trigger reference points, and in particular the limit reference point is commonly used to trigger new and strenuous management intervention to rebuild the stock. Trigger reference points can be a single threshold value, with a particular management response initiated if that value is reached. Alternatively there can be a set of trigger reference points or a continuous relationship between trigger reference point values and the intensity of the management response, so that for example the management response becomes steadily stronger as a limit reference point is approached.

Reference points are integral and required elements of AFMA's fishery management plans established under the Commonwealth *Fisheries Management Act (1991)*. Reference points provide the basis for the operational, measurable objectives and performance measures required in these management plans. The Australian Commonwealth Fisheries Harvest Strategy Policy and Guidelines (DAFF 2007) provides a further requirement for target and limit reference points, a requirement for decision rules, and a set of default reference points and decision rules. Reference points are also key elements in application of the Australian framework for reporting against Ecologically Sustainable Development (ESD) objectives (Fletcher 2002). The identification and use of all three types of reference points are also strongly emphasised by the *Guidelines for the Ecologically Sustainable Management of Fisheries* (Anon. 2001), against which all Commonwealth fisheries and all Australian export fisheries must be assessed to meet requirements under the *Environment Protection and Biodiversity Conservation Act (1999)*.

Reference points can be applied to any objectives of relevance to fishery management. In recent years an increasingly wide range of environmental issues, in addition to the target species of fishing, have been emphasized in fisheries management. For example ESD Reporting and the *Guidelines for the Ecologically Sustainable Management of Fisheries* require consideration of retained species, by-catch, habitats, food webs and interactions with threatened or protected species. Understanding and knowledge is very limited in relation to some of these issues, as is experience with appropriate target and limit reference points. Consequently, experience with trigger reference points for some of these issues, and the management actions they should trigger, is also limited.

There has been considerable effort put into developing reference points in many parts of the world. This has resulted in different approaches to the development and use of

reference points in different places. For example the Australian fishery assessments against the *Guidelines for the Ecologically Sustainable Management of Fisheries* show a wide range of reference points being used. At least four factors contribute to that diversity:

(1) Differences in the productivity and vulnerability of the ecological systems being fished and the manner of fishing;

(2) Differences in the objectives used for fisheries management;

(3) Differences in management attitudes to risk and uncertainty. This includes the risk that arises from imperfect knowledge of the ecological system and fishing impacts; from imperfect monitoring and management responses; and also how the risk is intentionally distributed across different and potentially competing objectives of management (e.g. use vs. conservation or short-term vs. long-term benefits); and

(4) Limited comparison of approaches and their outcomes across fisheries, particularly across different national and international fishery management jurisdictions (i.e. limited comparison and identification of what has worked, or not worked, in various circumstances).

These factors provide strong drivers for a focus on local fishery circumstances and for treating each case as if it were unique, so it is perhaps not surprising that many different approaches to the definition and use of reference points exist. However while local and specific conditions will always be an element of the approach taken to management, there are good reasons to expect benefit from increased comparison and learning from experience across fisheries. For example:

- The ecological consequences of, and human reactions to, fisheries management measures are not likely to be unique in every situation. As with other complex management systems, such as business or engineering management systems, there are likely to be general rules and best practice approaches that can be usefully identified.

- The benefits of learning any such general rules and best practices are likely to be substantial. Fisheries in Australia and elsewhere have important ecological, economic and social consequences, and so the effects of success or failure can be high. The dramatic consequences of acute over-fishing and stock collapse are obvious, and are often highlighted in public media and other reports. But the ecological, economic and social consequences of chronic overfishing can also be very high. Chronic over-fishing may be sustainable in the sense that it can persist for a long time, and may even come to be viewed as 'normal' over time (e.g. Pauly 1995). But it delivers lower socioeconomic value and higher ecological impact than is achievable. This is well demonstrated, for example, in the New England (USA) multi-species trawl fishery (Anon. 2003) where very considerable social and economic losses occurred in a fishery that was being overfished but was arguably sustainable in the sense that the situation could be continued indefinitely. Moreover recent reviews of the practical experience with stock rebuilding (Caddy and Agnew 2003, 2004) shows that recovery of overfished resources is often slow, is sometimes unsuccessful despite significant rebuilding measures, and usually requires serious restrictions on targeted fishing and by-catch for 2-3 times the generation time of the species involved. Stock rebuilding in itself has significant social and

economic cost. So there is considerable social, economic and ecological merit in preventing overfishing rather than having to recover from it.

- There is a need for efficient learning to address the increasingly complex issues confronting fisheries management and the demands by stakeholders for high quality ecological, economic and social outcomes. It is highly inefficient, and risky, to develop the management approach from 'first principles' each time. Efficient learning about best practice across fisheries, rather than funding redevelopment of similar approaches repeatedly or repeating the same mistakes, is an element of efficient management.

This study was initiated jointly by the Australian Fishery Management Authority (AFMA) and Department of the Environment and Heritage (DEH) to help develop and guide the use of reference points in Australian fisheries. Below, the concept of 'best practice' is briefly introduced and some background on the definitions and use of reference points is summarised. The reference points used in selected fisheries considered to reflect best practice are then provided along with details of the manner of their use.

The 'best practice' approach and the approach to identifying 'best practice'

The 'best practice' approach is based on a continuous improvement model of learning and feedback about what works well. 'Best practices' are those that have been demonstrated to work well in successful and highly admired examples – for example by the business showing the best performance for safety, production efficiency, accounting practice or waste minimisation. These highly admired examples need not show best practice in all facets of their activities simultaneously, but in some facets they demonstrate best practice and provide guidance and inspiration for others seeking to improve. Best practice would not normally be regarded as a minimum standard, nor is it an unchanging or 'fail safe' standard. Best practice is expected to evolve and improve over time. Best practice is intended to be achievable, with some businesses having shown that it can be done. Best practice is a positive rather than a negative message—which is probably why there is not the same level of global enthusiasm in the search for worst practice.

There are several approaches to identifying best practice. One is to develop formal metrics for desired management processes or outcomes and use them to quantitatively compare practices and identify the best. Another is to use qualitative expert judgment to select examples or practices that are regarded as best.

The latter approach has been used here. Fishery management experience and approaches were reviewed, and individual experts (Appendix 1) with wide knowledge of fisheries assessment and the use of reference points were identified and asked to choose a fishery or fisheries that in their experience illustrated best practice in the treatment of a range of issues (see Appendix 2 for the proforma used). The specific issues examined were the retained species, by-catch species, threatened or endangered species and communities, habitats, and food webs. The fisheries identified included the Pacific halibut, Alaskan groundfish, US West Coast groundfish, US northeast scallops (as an example of recovery), southern ocean icefish and krill, and Icelandic

3

cod. In addition, some specific features of the ICES management system and elements of the Australian Ecological Risk Assessment process for fisheries were incorporated. The 80 scoring benchmarks used in recent and major Marine Stewardship Council assessments – for Australian western rock lobster, New Zealand hoki, Alaskan pollock, Atlantic cod and South Georgia toothfish - were also considered because these benchmarks are intended to reflect best practice and are identified by independent expert assessment panels to reflect a well managed and sustainable fishery.

It is interesting to note that the list of fisheries identified and examined here as representing best practice is very similar to that reported more briefly by Hilborn *et al.* (2003) as demonstrating sustainability, although the two sets of fisheries were identified by entirely separate groups of experts and by separate processes that were operating independently at about the same time.

Definition, Basis and Context of Use of Reference Points

Definition of Reference Points

The history and development of reference points in fisheries is described by Caddy and Mahon (1995). Reference points began as criteria to help interpret the broad objectives of management. Both conceptual reference points and technical reference points were introduced. The conceptual reference point is a more specific articulation of a broader intent. For example the maximum sustainable yield is a conceptual reference point for the broader intent of providing a sustainable fishery. A technical reference point is a measurable value, and so is operational in the sense that it can be observed (directly through physical measurement or indirectly through modeling) and used in management decisions or performance assessment. For example the actual value of the maximum sustainable yield for a particular stock is a technical reference point. Technical reference points are usually based on statistical analysis of measurements or the implications of mathematical models, but they can be based on other sources of information and ways to quantify values (e.g. Caddy 2002). Here the focus is on technical reference points, and the term 'reference point' is used in this report solely with that meaning.

FAO (1997 and glossary) defines a fishery reference point as "a benchmark against which to assess the performance of management in achieving an operational objective". Similarly, the definition provided for use in the Australian context (Fletcher *et al.* 2002) is "the value of an indicator that can be used as a benchmark of performance against an operational objective".

In its broadest definition, a reference point is a particular value of a fisheries indicator corresponding to a situation that is important to management. A fisheries indicator is a quantity that can be measured and that is considered to reflect an operational objective of management. The situation that is of importance to management could be a desirable outcome (giving a target reference point), an undesirable outcome (giving a limit reference point) or the initiation through a decision rule of a pre-determined management response (giving a trigger reference point). There should be a low chance of the fishery violating a limit reference point, whereas it is intended that the fishery be in the vicinity of a target reference point.

Target reference points

Target reference points specify the intended state of the managed system. For example a target may be a desired fishing mortality, biomass, profitability or level of catch or by-catch. It is expected that the actual state of the fishery will approach or fluctuate somewhat about these targets. Target reference points largely reflect the societal objectives and desired outcomes of fishery management, within the more ecologically determined constraints provided by the limit reference points.

Limit reference points

Limit reference points provide operational definitions of what constitutes unacceptable outcomes, such as unacceptably high fishing mortality, unacceptably depleted fish stocks or unacceptably low profit levels. Limit reference points for stock abundance identify depletion levels that represent unacceptable overfishing. Limit reference points for the rate of fishing, or fishing mortality, identify rates that represent overfishing – that is a rate of fishing that would result in the stock becoming overfished or remaining overfished. While there remains some societal value judgment about what constitutes 'unacceptable', limit reference points are strongly determined by ecological considerations and thresholds—such as stock productivity, the chance and speed of recovery from fishing impacts, the resilience and persistence of the fished stocks and ecosystem, abrupt recruitment collapse, and impacts on dependent or associated species. Unacceptable outcomes are strongly based on avoiding irreversible, slowly reversible or long-term impacts of fishing (e.g. from UNCED 1992 and UNFSA 1995), and so there is an emphasis on avoiding recruitment overfishing, stock collapse and excessive depletion of very long-lived organisms.

Limit reference points for both overfishing and of being overfished are necessary to encompass the range of relevant fishery management situations, as the appropriate management response can be different in each of these situations. Both overfished and overfishing limit reference points operate at the same time, and so all four combinations of these two reference points are possible. That is, a stock could be experiencing overfishing but not be overfished (although continued overfishing will lead to an overfished stock in future); it could be currently overfished because of previous catches but be subjected to a fishing rate that does not constitute present overfishing (so future stabilisation or recovery of the stock would be expected); it could be both overfished and subject to ongoing overfishing; or it could be neither overfished nor subjected to overfishing.

While most approaches to fishery management incorporate uncertainty and precaution through the choice of trigger reference points and the management actions they initiate, some alter the reference points themselves as a result of such considerations. For example ICES identify a limit reference point and also a 'precautionary approach' (pa) limit reference point. The limit reference point is the value of the system property that is regarded as a limit. The pa limit reference point is the value of that same property such that, given the precision of estimation, there is a low chance of exceeding the limit reference point if the estimated value does not exceed the pa limit reference point. In this approach the limit reference point is a system property that would change only as a result of changed understanding of the system. But the pa limit reference point includes measurement uncertainty about the current status of the

fishery and so would change as the precision of measurement or monitoring changed. In particular, with increasingly precise measurement of current stock status the pa limit reference point would be set increasingly close to the limit reference point.

Trigger reference points

Trigger reference points are used to initiate a management response, usually through a predefined 'decision rule', when a measured indicator reaches the value of the trigger reference point.

A trigger reference point could be a single value that on being reached gives rise to a discrete management response through the decision rule—such as to review or close an area of the fishery if the catch rate falls below the lowest value seen in some previous period. Or there could be a series of trigger reference points that initiate different responses to the fishery indicator. For example a series of discrete trigger reference points could provide a graduated series of different management responses as a limit reference point is approached. In the extreme a continuous function or relationship could be established between the indicator and the management response. For example a decision rule could provide for continuously reducing catch as a limit reference point is approached—that is a catch control rule.

Catch control rules are a special case of decision rules in which altering the permitted catch is the only management response that is triggered. They are increasingly commonly used and a catch control rule is effectively a continuous form of a trigger reference point. For example Figure 1a shows a relationship that is used in some US fisheries between the current estimate of stock biomass and the permitted fishing mortality, and consequently the permitted catch. The decision rule is intended and designed to achieve the target reference point and avoid the limit reference point in the context of the details of that fishery, including for example the accuracy and precision of measuring the indicators and the effectiveness of the management response. At low stock sizes, the catch decision rule is intended to provide adequate rebuilding to avoid the limit reference point and return the stock to the vicinity of the target. While catch decision rules of the form shown in Figure 1a,b are common, a wide range of possible decision rules can be optimised to deliver on the objectives of management in different situations (e.g. Thompson 1999). In particular some decision rules that do not include zero catches (i.e. cessation of fishing) have very good overall properties.

Target and limit reference points can be used as trigger reference points. In particular the limit reference point is commonly used to trigger new and strenuous management intervention to rebuild the stock. However this alone is unlikely to provide an adequate trigger for maintaining the stock at desirable levels because it does not allow avoidance of the limit and provides for a response only when the limit is breached. Also use of the limit reference point as a trigger reference point in this way can result in a linkage and confusion between where the limit is set and the nature of the management response. For example if the fishery is automatically closed when a limit is breached there could be an argument for setting the limit low because the proscribed management response makes this a 'point of termination' for the fishery. Conversely if a graduated but effective rebuilding response is possible below the limit reference point then there could be an argument to set the limit relatively high. Here it is considered that the limit reference point is set largely on biological ground to avoid

irreversible or slowly reversible fishery impacts, that the trigger reference points (including decision rules) are designed to avoid the limit with high probability, and that it is not automatic that the limit reference point is a trigger for cessation of fishing.

Trigger reference points are often referred to as threshold reference points, particularly in US applications. This terminology is avoided because it can be confused with the many other kinds of thresholds that occur in fisheries and the threshold concept is not fully appropriate in the frequent situation where a continuously varying management response is triggered through a decision rule. In New Zealand applications a distinction is made between 'hard' and 'soft' trigger reference points, with a soft trigger initiating management interventions to the ongoing fishing operations (such as a rebuilding plan) and a hard trigger resulting in fishery closure.

Basis for use of reference points

Reference points are an operational reflection of the objectives of management, and so they can be used for any and all of the various objectives of management.

International agreements provide some overall guidance about the scope of objectives that may be addressed by reference points. Guidance from international agreements comes primarily through the UN Convention on the Law of the Sea (UNCLOS 1982), the UN Fish Stocks Agreement (UNFSA 1995) and the non-binding Code of Conduct for Responsible Fishing (FAO 1995).

Key guidance from UNCLOS includes:
-   "..maintenance of living resources in the Exclusive Economic Zone is not endangered by overfishing", implying a limit reference point that defines overfishing;
-   "[management] shall maintain or restore populations of harvested species at levels which can produce the maximum sustainable yield", implying a target reference point that maintains high productivity, and a trigger reference point with linked management response that is capable of restoring depleted populations.

UNFSA provides considerable guidance on the use of reference points:
-   "Two types of reference points should be used: conservation, or limit, reference points and management, or target, reference points. Limit reference points set boundaries which are intended to constrain harvesting within safe biological limits…"
-   "Management strategies shall seek to maintain or restore populations ….at levels consistent with previously agreed precautionary reference points. Such reference points shall be used to trigger pre-agreed conservation and management action."
-   "Fishery management strategies shall ensure that the risk of exceeding limit reference points is very low."
-   "Fishery management strategies shall ensure that target reference points are not exceeded on average."
-   "The fishing mortality rate which generates MSY should be regarded as a minimum standard for limit reference points."

- "…fishery management strategies shall ensure that fishing mortality does not exceed that which corresponds to MSY, and that biomass does not fall below a pre-defined threshold."
- "For overfished stocks, the biomass which would produce MSY can serve as a rebuilding target."

The FAO Code of Conduct states:
- that an aim of fisheries management is to "maintain or restore stocks at levels capable of producing maximum sustainable yield";
- that management should determine "stock specific target reference points, and, at the same time, the action to be taken if they are exceeded"; and
- that management should determine "stock specific limit reference points and, at the same time, the action to be taken if they are exceeded; when a limit reference point is approached, measures should be taken to ensure that it will not be exceeded".

Within Australia, Ecologically Sustainable Development (ESD) is the agreed basis for environmental management, as set out in the Intergovernmental Agreement on the Environment between all state governments and the federal government (IGEA 1992). The 'high level' objectives and principles in this agreement have been formally interpreted by the combined fisheries agencies in Australia to provide a number of ESD criteria for use in fishery management (see Fletcher *et al.* 2002). These criteria provide more specific guidance on the issues and objectives that are relevant to ESD in fisheries management. The three that relate to ecological wellbeing are:

> *Retained Species* (i.e. species that are captured and used, either as target or incidental catch): To manage the take of retained species within ecologically viable stock levels by avoiding overfishing and maintaining and optimising long-term yields.

> *Non-Retained Species* (i.e. species caught or directly impacted by the fishery but not used): To manage the fishery in a manner that does not threaten biodiversity and habitat via the removal of non-retained species (including protected species and ecological communities) and manage the take of non-retained species at ecologically viable stock levels.

> *General Ecosystem Impacts* (i.e. the potential indirect and more general environmental impacts of the fishery): To manage the impacts of fisheries such that only acceptable impacts occur to functional ecological relationships, habitats and processes.

The ESD Reporting Framework (Fletcher *et al.* 2002) provides a structured process for interpreting these various criteria and principles at the operational level, and identifying the issues in a fishery for which operational objectives, indicators and reference points should be developed.

In addition to the Intergovernmental Agreement, ESD is an explicit objective of management in the legislation of most fishery agencies in Australia. Specifically, the Commonwealth *Fisheries Management Act* 1991 requires that the Australian Fisheries Management Authority "ensure that the exploitation of fisheries resources

...[is] conducted in a manner consistent with the principles of ecologically sustainable development and the exercise of the precautionary principle". This requirement is made in respect of fishery resources, non-target species and the long-term sustainability of the marine environment. In addition the *Environment Protection and Biodiversity Conservation Act* 1999 (EPBC 1999) requires that native fish exported from Australia, and all Commonwealth managed fish products, come from an ecologically sustainable fishery that meets the principles of ESD. The *Guidelines for the Ecologically Sustainable Management of Fisheries* provide the principles and criteria that are used to assess the ecological sustainability of fisheries (Anon. 2001). The guidelines address target and other retained species, by-catch species (i.e. species landed on the fishing vessel but not retained or species killed by fishing but not landed on the vessel), threatened, endangered or protected species, benthic communities, food chains, and the physical environment. With respect to target and other retained species, the guidelines emphasize the maintenance of "ecologically viable stocks", defined to mean that the stock is maintained at high levels of abundance to maintain productivity, maintain yields in the long term, provide margins of safety for error and uncertainty, and conserve the stock's role and function in the ecosystem.

The combination of the Australian ESD reporting framework and the EPBC guidelines were used to identify the headings under which best practice is examined below—that is, target or commercially retained species; by-catch species; threatened, endangered or protected species; habitats; and food webs (see Appendix 2).

Context of use of reference points

Reference points are only one part of a fishery management strategy. The full management strategy includes statement of operational objectives, monitoring, analysis of monitoring data, selection of management measures (e.g. level of input and output controls, spatial or temporal access, permitted activities, and technical controls on gear), implementation of management measures, and compliance. Target and limit reference points are an expression of the operational objectives, while trigger reference points and associated decision rules are part of the management response. The performance of the whole fishery management system cannot be judged on the basis of the performance of one of its parts in isolation. Reference points that would lead to good management outcomes in one management context may not deliver desired outcomes in another. For example, a target reference point that is close to a limit reference point may give good management outcomes if there is accurate monitoring of the stock combined with quick and effective management responses as the target is exceeded or the limit is approached. But the same target reference point is likely to lead to poor management outcomes if the monitoring and management response is poorly directed, ineffective or slow.

Uncertainty in all elements of the management strategy, attitudes to risk, desire for robustness and cost can all influence the choice of reference points in a particular context. Figure 2 illustrates how the decision rule and target reference points shown in Figure 1 might be changed to reflect different levels of uncertainty in the stock biomass and fishing mortality that gives maximum sustainable yield. Excessive focus on the reference points in isolation from the broader management strategy, and on a narrow range of types of reference points, is what Hilborn (2002) refers to as the 'dark

side' of reference points. Myopic use of reference points can limit or divert research and management focus, and result in the adoption of suboptimal management strategies. For example it can result in an emphasis on estimating fishing mortality and population size, or on controlling fishing mortality directly, when other indicators, reference points and controls may be more cost-effective.

Scientific methods for the design and evaluation of the management strategy as a whole, including the reference points, are usually based on simulation testing. This has been termed Management Strategy Evaluation (e.g. Sainsbury *et al.* 2000, Sainsbury and Sumalia 2003) or Operational Management Procedures (Butterworth and Punt 2003). Both of these approaches use the concepts of Adaptive Management (e.g. Hilborn and Walters 1992) and they differ mainly in emphasis (e.g. Schnute and Haigh 2006). General and practically oriented guidance on the development of the management strategy as a whole is given in FAO (2003).

The best practice reference points here are selected because they are judged to have been successful in a 'real world' context and are likely to be successful more generally. However it is important to recognise the context of the particular fishery in which the reference points were used, and to consider whether application to another fishery is appropriate. To assist this process, a brief outline of the 'best practice context' of their use is provided. In addition, while the best practice reference points described here can be used as default options, their use in any particular fishery should still require specific justification as to why they are appropriate to the circumstances of that fishery. Ideally, the reference points chosen for a particular fishery should have been simulation tested and been shown to be robust to the uncertainties and circumstances present in that fishery.

Best Practice Reference Points

1. Target or commercially retained species

Image courtesy of AFMA

Background

Targeted and retained species provide direct benefit to people and are the reason for fisheries. It is accepted that there will be impacts on those species as a result of fishing.

The concept of Maximum Sustainable Yield (MSY) remains central to fishery management. Despite early difficulties with implementation of the concept, it is enshrined in many international and national agreements and laws. Recent summaries of the history of MSY are provided by Caddy and Mahon (1995) and Mace (2001). The essence of the concept is that the largest long-term yield will be available at an intermediate level of fishing mortality and at a population size that is less than the unfished population. The underpinning concept is that as the long-term average fishing mortality is increased, the abundance and mean age of the fished population is reduced, and the per capita productive capacity of the population is increased as a result of reduced competition or similar effects that occur in relatively large populations of old animals.

An important and lasting contribution from the concept of MSY is the distinction between a sustainable fishery and one that is providing maximum, or by some measure optimum, benefits. For example, a fished population may be reduced to the point where catches are much lower than they might otherwise be, but these undesirably low catches might be maintainable indefinitely. The catches are sustainable in this sense. And further, the same low but sustainable catch may be taken with high fishing mortality and low population size or alternatively with a low fishing mortality and high population size—providing very different economic and ecological consequences. In a fisheries management context sustainability alone, with the meaning 'can be continued indefinitely', is a weak and inadequate standard.

The practical application of MSY initially encountered difficulty and failure. This was because of its inappropriate use as a target reference point and because of inadequate appreciation of the effects of imprecision in its estimation. The obvious reference points that relate to the MSY concept are the MSY itself (i.e. the amount of yield), the fishing mortality giving MSY (F_{MSY}) and the average population biomass at MSY (B_{MSY}). Early estimations and applications treating these points as targets, and also assumed equilibrium conditions with a limited number of functional relationships between per capita population productivity and population size. Although later non-equilibrium methods used a wider range of relationships (e.g. Hilborn and Walters 1992), these approaches did not perform well from almost any perspective. The estimates of the reference points invariably had high uncertainty, and the most likely or 'best point estimate' poorly reflected the range of possibilities that were consistent with the data. Similarly the estimated current status in relation to the reference point (e.g. current F or B in relation to F_{MSY} or B_{MSY}) often had a statistical confidence interval that included both desirable and undesirable interpretations. For example, the confidence interval for the current F could include both F_{MSY} and F values expected to reduce the population to very low levels, and this frequently led to much debate about the appropriate management response. Furthermore the information available from fisheries often resulted in systematic over-estimates of MSY and F_{MSY}, and could not distinguish between interpretations of great significance to management (e.g. distinguishing an initially large but unproductive population from an initially small but productive one—see Hilborn and Walters 1992), and usually required MSY to be exceeded before it could be estimated.

In addition MSY was developed as a single species concept, and does not take account of the role of the harvested species in ecosystems. Issues flowing from a broader ecosystem perspective include that the apparent surplus production that provides MSY may be achieved by displacing natural predator populations (e.g. see review by Yodzus 1994), that the combined MSY for a group of species with direct or indirect feeding interactions is less than the sum of the MSY for each individual species considered in isolation (e.g. Pope 1979, May *et al.* 1979, Link 2002a), and that fishing all species in a food web at their individual (correct) MSY levels will result in over-depletion of many top predators (Walters *et al.* 2005). The result of all these concerns was Larkin's (1977) epitaph for MSY and widespread criticism of the MSY approach and the reference points derived from it (e.g. Sissenwine 1978).

However this was not the end of MSY. While interpretations and approaches to practical application of the MSY concept have changed greatly in the last few decades, the concept itself remains a key one (e.g. Mace 2001, Mangel *et al.* 2002). There is an average level of catch that is sustainable and that can be maximised or optimised in the long term, even though it may not be as large or as constant as previously assumed, and given the uncertainty in estimation it may not be possible or safe to extract the maximum from the resource that is theoretically available (Sainsbury 1998). Punt and Smith (2001) summarise the 'birth, crucifixion and reincarnation' of the MSY concept, pointing out that the concept is valid and that the main criticisms relate to implementation, and especially estimation problems and the inappropriateness of MSY-related measures as target reference points.

Three main developments in the practical application of MSY have led to 'reincarnation' of the concept; (i) refinement of the interpretation of MSY, (ii)

alternative approaches to use and estimation of MSY-related reference points, and (iii) taking a more holistic approach to the management system and the use of reference points. Each of these developments is examined in turn below.

(i) Interpretation of MSY: MCY and MAY

Two different interpretations of MSY are recognised—one static and the other dynamic (see Sissenwine 1978, Mace 1988a, 2001, Francis 1992a and Anon. 2003, 2007).

Maximum Constant Yield (MCY) – static MSY

The static interpretation of MSY is the Maximum Constant Yield (MCY). It is a single unchanging maximum yield that can be taken indefinitely, with an acceptable level of risk, from all probable future levels of biomass and productivity. The MCY must be set low enough to allow maintenance of the population and of the MCY catch through anticipated periods of low recruitment, productivity or ecological circumstances. So while the MCY is static in that it is a single unchanging value, the calculation of that value takes into account the fluctuations and dynamics that are expected in the stock and its environment. To some extent a precautionary or risk based consideration is intrinsic to the use of MCY, because possible future variability in productivity and risk are explicitly considered in selecting the MCY. But success of the MCY approach depends on this level of precaution being adequate to protect the resource in all reasonable future circumstances, and so the approach is vulnerable to the calculated MCY being based on overly restricted range of possible future stock productivity and dynamics. In general, constant catch strategies suffer from the problem of being potentially anti-compensatory i.e. fishing mortality increases when stock size decreases because the catch is constant, which is likely to drive stock size lower in those circumstances. Consequently there is a need for the MCY to be set with a large 'safety margin'. Also it is good practice to periodically update the calculation of MCY to ensure that the range of future productivity and dynamics used in the calculations remains appropriate.

MCY approaches are particularly useful for setting sustainable catch limits in situations where there is limited information, infrequently updated information or infrequent updating of the status of the stock—as for example in new or developing fisheries, low value fisheries, or low value by-catch or by-product catches in an otherwise valuable fishery. The Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) uses an MCY approach to set a fixed precautionary catch level for developing fisheries. In the CCAMLR methodology the MCY is the catch which, in the long term, will result in the median of the fished population each year being reduced to no more that 50% (75% for significant prey species) of the median unfished abundance <u>and</u> having no more than a 10% chance of the population being reduced below 20% of its unfished abundance (see Constable *et al.* 2000). The estimation of MCY requires some basic ecological understanding, an estimate of population size or recruitment, and a measure of the expected variability and autocorrelation in recruitment, all of which may be very approximate. This reinforces the need for a large 'safety margin' in the use of MCY and periodic review of the calculations, both of which are provided through the CCAMLR process.

New Zealand also uses MCY to set precautionary catch levels in newly developing fisheries (Annala 1993, Anon. 2007). In this case MCY is calculated from

$$MCY = 0.25\ F_{0.1}\ B_0$$

where $F_{0.1}$ is calculated from growth and natural mortality parameters and B_0 is an estimate of initial biomass. The extent to which this will deliver a catch level that is at or below the true MSY catch (i.e. is actually precautionary) will depend on whether the 'safety margin' provided by the 0.25 multiplier is sufficient to account for the potential biases in the initial estimates on $F_{0.1}$ and B_0 and for the full range of their future fluctuation. Again this reinforces the need for periodic review of the calculation of MCY.

Maximum Annual Yield (MAY) – dynamic MSY

A theoretically possible MSY can be calculated assuming perfect knowledge of the population at all times and adjusting the selectivity and intensity of harvesting through time so as to give the greatest long-term yield. But experience has shown that this is not practically implementable.

A more realistic dynamic interpretation of MSY is the Maximum Annual Yield (MAY) (Ricker 1975, Mace 1988a, 2001). This is the average long-term annual yield obtained when the yield each year results from a constant fishing mortality being applied to the available population biomass. In this dynamic interpretation of MSY the catch varies from year to year in response to changing biomass under a 'constant fishing mortality' strategy. The constant fishing mortality can be chosen to maximise the long-term average annual yield while taking account of uncertainty in the biomass estimates and/or knowledge of stock productivity and variability. The catch in any year under this approach is the Current Annual Yield (CAY). The MAY is always greater than MCY which does not permit changes in the annual catch.

In this dynamic interpretation, both the yield and the biomass fluctuate across years, and the MAY and the biomass associated with it (B_{MAY}) are long-term averages. The constant fishing mortality that gives MAY is F_{MAY}, and MAY is a proportional escapement catch decision rule (Figure 9). Constant fishing mortality strategies may not achieve the maximum long-term yield that is theoretically possible—this may require alteration of the fishing mortality and fishing selectivity applied in each year—but they usually produce close to the theoretically possible maximum. Also constant fishing mortality strategies have both lower variability in yields and greater robustness to estimation errors and natural stock fluctuations than other approaches that try to achieve the maximum long-term yield that is theoretically possible (e.g. Walters and Parma 1996). Nevertheless in some situations successful approaches to implementing dynamic MSY do adjust the fishing mortality according to the estimated stock size – for example constant escapement policies and related catch decision rules (e.g. Thompson 1999, Figure 9). While constant escapement policies generally have better performance than constant fishing mortality policies in situations where the escapement can be estimated (e.g. Mace 1988b) these situations are not usual, and so the MAY approach that utilises a constant fishing mortality has greater generality.

This dynamic interpretation of MSY, either with a constant fishing mortality or one that is changed through a decision rule, is what is usually meant by MSY in recent literature and applications. However common terminology often confuses the different interpretations of MSY, because terms such as MSY, B_{MSY} and F_{MSY} are

used for all these different interpretation and do not distinguish MCY, MAY or the various approaches that are not based on a constant fishing mortality (e.g. a catch decision rule such as those in Figure 1 or several in Figure 9). It is a case of 'user beware' of the context.

<u>MSY as a limit reference point</u>

A further important change in the development and use of MSY, especially in the context of the dynamic MAY interpretation, was recognition that it was inappropriate to treat F_{MSY} as a target reference point for fisheries management. Rather F_{MSY} should be used as a limit reference point (e.g. Caddy and Mahon 1995, UNFSA 1995, NMFS 1998, Mace 2001). This is a result of both practical experience and scientific analysis which showed that treating F_{MSY} as a target often resulted in over-depletion of fish stocks, and that recovery from over-depletion is difficult (Caddy and Agnew 2003, 2004, Mace 2004). Also MSY as a target is often sub-optimal economically – and usually would be expected to be.

In a recent review, Hilborn *et al.* (2003) conclude that one of the three factors providing hope for future sustainability of fisheries is the use of exploitation rates that deliberately do not attempt to maximise the biological yield. (The other two factors were elimination of the 'race for fish' and a combination of reduced fishing costs with increased value of products.) A consequence of fishing harder than F_{MSY} is that the population biomass will be, on average, less than B_{MSY}. In the ten world fishery management successes identified my Hilborn *et al.* (2003), all had biomasses that were held near or above B_{MSY}, and all were intentionally taking yields that were lower than the maximum that appeared to be available. The biomass and fishing mortality in these successful fisheries are being managed to have a high chance of maintaining the fishery on the high-biomass, low-fishing mortality side of MSY. Many of these fisheries experienced a period of stock depletion in their history that caused concern and a change in management approach, as was the case for the 30 cases of fishery recovery identified by Mace (2004). Most of the fisheries identified by Hilborn *et al.* (2003) as being sustainable were also identified by the participants in this study as demonstrating best practice.

(ii) Approaches to estimating reference points

With recognition of the MAY interpretation of MSY, and the many desirable properties of harvest strategies with constant fishing mortality, there followed development of methods to specify fishing mortality reference points that would achieve close to MAY. This included estimation of F_{MSY}, or a reasonable surrogate for it, as a limit reference point.

When there is sufficient information about the species and sufficient statistical contrast in the historical fishery data, a direct approach is possible. That is appropriate probabilistic population models can be developed and parameterised. These models can then be used to provide estimates of the annual catches or used to derive and test catch decision rules. Some well developed examples of this are the Pacific halibut fishery (e.g. Parma 2002) and the north Pacific pollock (e.g. Dorn *et al.* 2003). Even in this 'best case' situation there are usually still significant remaining uncertainties, often including the correct structure of the model and the estimates of current stock status. If a range of models and parameterisations are used to represent the uncertainties there are various methods to simulation test different management

strategies, including catch decision rules, and then to select the strategies that can deliver the intended outcomes despite the uncertainties (Sainsbury *et al.* 2000).

However there are many situations where there is insufficient information available to allow such analysis, and so indirect methods are needed—especially to provide surrogates for the limit reference point F_{MSY}.

One set of indirect approaches to specifying target and limit reference points for a constant fishing mortality is based on 'per recruit' methods. Another set of indirect approaches makes use of a time series of recruitment and spawning stock sizes.

<u>'Per recruit' proxies for the F_{MSY} reference point</u>
 'Per recruit' methods are based on simple life table methods (see Beverton and Holt 1959, Hilborn and Walters 1992, FAO Glossary). They calculate the lifetime expectation of a single recruit's contribution to quantities such as yield, the spawning biomass, egg production or the number of spawning seasons an individual can expect to participate in (see Glossary for Yield per Recruit and Spawners per Recruit). These 'per recruit' quantities are calculated from age specific growth, mortality, fecundity and fishing selectivity. No stock-recruit relationship is used.

The main advantage of 'per recruit' approaches is their simplicity and the fact that they do not require an historical time series of catch, stock abundance or recruitment estimates. They can be applied to fisheries with little or no historical data and so they are very popular (e.g. Rosenberg *et al.* 1994).

The main limitation and danger of the 'per recruit' approach is that it does not account for the possible reduction in the number of recruits to a population as fishing reduces the spawning stock. So the approach does not provide an internal basis for defining a level of fishing that prevents recruitment overfishing. Maximising yield per recruit gives the greatest yield from whatever number of recruits enter the population, but it does not ensure that the absolute number of recruits is maintained. The 'recruits per recruit'—that is, the expected lifetime production of future recruits per present recruit—cannot be calculated without specification of a relationship between the spawning stock size, either alone or in combination with other influences, and recruitment.

An early YPR based reference point for fishing mortality was F_{max}, the fishing mortality that gave a maximum YPR where that maximum exists—see Figure 3. For a fixed selectivity, F_{max} is the limit reference point beyond which growth overfishing occurs, although in some early (and even some present) applications this was also used as a target.

There are several problems with the use of F_{max} as either a limit or a target reference point. Besides the difficulty of it not existing for some growth and mortality combinations, F_{max} is always greater than or equal to F_{MSY} and so leads to excessive stock depletion, including depletion sufficient to cause recruitment overfishing (e.g. Clark 1991).

An alternative to F_{max} was provided by Gulland and Boerema (1973). They proposed $F_{0.1}$ as a target reference point. $F_{0.1}$ is the value of F at which the slope of the YPR vs F

curve is 0.1 of the slope at the origin (Figure 3). More conservative reference values of this type, for example $F_{0.2}$, can be defined in a similar manner but these have not been widely used or tested. For increases in fishing mortality beyond $F_{0.1}$ there are rapidly diminishing biological yield and economic returns.

The value 0.1 was initially chosen on the basis of general economic and biological arguments (Gulland and Boerema 1973). $F_{0.1}$ is expected to approximate F_{MSY} for productive populations (see Deriso 1987). $F_{0.1}$ was used extensively as a target reference point to set annual catch levels, using the MAY/CAY approach, in many fisheries—notably in the European North Sea and the northeast Atlantic during the 1970s and 1980s (Rivard and Maguire 1993, Hilden 1993). However the practical experience with use of $F_{0.1}$ as a target reference point has been that stocks have often declined, sometimes to the extent of suffering recruitment overfishing (e.g. northeastern Atlantic cod).

Some of these unexpected failures were influenced by estimation errors, misreporting of catches and difficulties with implementing intended catch limits, so that the actual fishing mortality was often higher than the $F_{0.1}$ that was intended. But nonetheless serious overfishing was the result. The $F_{0.1}$ approach as implemented did not provide a sufficient 'safety margin' to deliver sustainability in the context of the uncertainties in those fishery management systems. And it has also been found that in many situations $F_{0.1}$ exceeds F_{MSY}, particularly for relatively unproductive stocks, and so can result in a significant decline in recruitment (Mace and Sissenwine 1993, Myers *et al.* 1995). Even when $F_{0.1}$ is a reasonable proxy for F_{MSY}, it is more appropriately used as a limit rather than a target reference point (Mace 2001).

Another fishing mortality reference point based on 'per recruit' analysis is the expected spawning biomass per recruit. And similar approaches are quantities such as total biomass per recruit, eggs per recruit, and the number of spawnings per recruit. All these quantities monotonically decrease with increasing fishing mortality, as illustrated in Figure 4a for spawning biomass per recruit.

For any fishing mortality, the reduction in the spawning biomass per recruit (SPR) can be calculated, and the fishing mortality giving an x% reduction in SPR compared to the unfished level is called $F_{x\%}$. For example, $F_{35\%}$ is the fishing mortality that gives a spawning biomass per recruit equal to 35% of that at F=0. Applying $F_{x\%}$ does not result in an x% decrease in the actual biomass (i.e. does not result in x% of B_0). The exact reduction in biomass caused by $F_{x\%}$ depends on the steepness of the stock-recruitment relationship; the reduction in biomass caused by $F_{x\%}$ will be close to x% for high steepness but the reduction will be greater than x% for low steepness. But for appropriate values of x the mortality $F_{x\%}$ may be treated as a surrogate for other fishing mortality reference points of interest such as F_{MSY}. The challenge is: what is the appropriate value of x?

The reliability of 'per recruit' quantities as reference points to achieve high long-term yields and to prevent recruitment overfishing depends entirely on selecting the appropriate value of x in $F_{x\%}$. Assumptions about the stock-recruitment relationship essentially determine this selection, with lower values of x (i.e. greater reduction in per recruit output) being sustainable for stocks with steeper stock-recruitment relationships (i.e. stocks with the ability to maintain recruitment as the population is

reduced). But in most situations where 'per recruit' approaches are applied there is no direct information available on the stock-recruitment relationship for the species involved.

In early use of this approach Goodyear (1993) suggested $F_{20\%}$ as an appropriate limit reference point. Simulation testing with a range of stock recruitment relationships (Clark 1991) suggested $F_{35\%}$ as a better proxy for F_{MSY} in many circumstances. And as a result of further analysis, including consideration of stochastic recruitment, both Clark (1993) and Mace (1994) later recommended $F_{40\%}$ as a more reliable proxy for F_{MSY}.

Mace and Sissenwine (1993) introduced F_{med} as a limit reference point for recruitment overfishing (see glossary and Figure 4). They estimated F_{med} from the stock and recruitment histories of 81 stocks. For each stock they calculated the $F_{x\%}$ that was equivalent to F_{med}, and found that for 80% of stocks examined, F_{med} was less than $F_{30\%}$. Consequently they suggested that $F_{30\%}$ could be treated as a 'default' limit reference point for recruitment overfishing. But they also noted that this would not be conservative enough for 20% of the stocks they examined and that F_{med} is often associated with significantly reduced recruitment.

Their comparisons also showed other significant patterns that were counter to usual 'rules of thumb', including:
-    in about 40% of stocks, recruitment overfishing occurred at a lower fishing mortality than growth overfishing (i.e. $F_{max} > F_{med}$);
-    in about 10% of stocks, recruitment overfishing occurred at a lower fishing mortality than $F_{0.1}$; and
-    the median %SPR at $F_{0.1}$ was $F_{38\%}$, and so a higher %SPR (i.e. lower F) than this would be necessary to deliver $F_{0.1}$ for a majority of stocks (i.e. $F_{x\%}$ with x>38%).

Clark (2002) provides a very useful recent re-examination of the use of $F_{35\%}$ and $F_{40\%}$ as proxies for F_{MSY}. This analysis considered more examples of stocks with low resilience (i.e. low 'steepness' in the stock-recruitment relationship), and also placed greater emphasis on avoiding large reductions in biomass rather than just obtaining the maximum catch. The relationships between fishing mortality, the reduction in biomass and the yield for a range of steepness values in the stock-recruitment relationship are shown in Figures 5 and 6. These are deterministic relationships and in practical application sources of uncertainty and stochasticity need to be included. Nevertheless, the deterministic relationships show some important features:
-    If steepness is known, then an $F_{x\%}$ can be found that provides a reasonable approximation of F_{MSY}, or that provides a reasonable balance across the level of depletion in stock biomass or recruitment and the percentage of the theoretically maximum yield that will be achieved.
-    If steepness is not known, then experience with other fisheries or similar stocks can be used to identify a reasonable and precautionary value. For example Myers *et al.* (1998, 2002) provide a summary of the steepness parameter based on analysis of over 700 stocks. From their summaries a stock-recruitment steepness of less than 0.3 is rare. Figure 6 shows that for steepness greater than 0.3, a fishing mortality of $F_{50\%}$ delivers both a high fraction of the MSY (usually more than 85%, even for very resilient species) and maintains a

relatively high biomass. For stocks that are known or suspected to have very low resilience, values of $F_{60\%}$ or lower (i.e. greater x%) are required to reliably maintain both stock biomass and yield.

- Clark (2002) emphasises that a strategy that maintains biomass at about 40% of the unfished level (i.e. as a target reference point) is highly robust even for very non-resilient stocks and in cases where inaccurate estimates of natural mortality are used in the SPR calculations.

The fishing mortality, based on 'per recruit' proxies, that is regarded as a precautionary 'default' has substantially decreased as experience has been gained with a wider range of fisheries (e.g. Myers *et al.* 1998, Walters and Martell 2002). There are several reasons for this. An obvious contribution has come from increased empirical experience and better methods for comparison across fisheries and stocks (e.g. Meyers et al 1998, 2002 and Liermann and Hilborn 1997).

But there has also been greater recognition of the ecological undesirability of maintaining populations at low spawning biomass and with high fishing mortality. For example:
- The quality of eggs produced by inexperienced or young spawners is relatively low in many species compared to experienced and older spawners. This results in systematic over-estimation of the real breeding capacity of a stock dominated by young spawners. Murawski *et al.* 2001 show that in a recovering cod population the fishing mortality giving 20% spawning biomass per recruit is about twice the mortality giving 20% viable larvae per recruit. That is, the reference fishing mortality for $F_{20\%}$ based on biomass is twice what it should be to achieve the intended reproductive protection, and a large influence on this result was the very young age of the spawners in this depleted population.
- Irreversible modification is increasingly being recognized as a concern in fished populations (e.g. Smith 1994, Law 2000, Hauser *et al.* 2002, Conover and Much 2002, Kenchington 2003, Walsh *et al.* 2006). Genetic modification is particularly strong with high and selective fishing mortality, including the loss of sub-populations or stocks. Fish populations may be more vulnerable to genetic modification through fishing than anticipated because the genetically effective population size for fish can be a very small fraction of the census population size –even populations of many millions of individuals can have a small effective genetic population size, resulting in high genetic vulnerly (e.g. Smith *et al.* 1991, Hauser *et al.* 2002).

And further, there is now greater recognition of the high social and economic cost of chronic overfishing (e.g. Anon. 2003), and of the difficulty and cost of stock rebuilding. While there are some good examples of stocks recovering from overfishing (Mace 2004), there also many examples of populations not recovering for considerable periods even with drastic management measures in place (e.g. Caddy and Agnew 2003, 2004). And in all cases the cost of recovery is considerable, with substantial reductions in fishing mortality and changed fishing practices being required for periods that are typically 2-3 generation times of the fish. Increased recognition of the cost of overfishing, and of recovering overfished stocks, has placed a renewed emphasis on avoiding overfishing.

Old 'rules of thumb' for sustainability, such as allowing at least one spawning per cohort or setting fishing mortality about equal to natural mortality, have proved

inadequate in many cases. The fishing mortality giving high and sustainable yields is usually less than the natural mortality, and is sometimes less than half the natural mortality (e.g. Beddington and Cooke 1983, Walters and Martell 2002). The use of $F_{40\%}$ as a default proxy for F_{MSY} is well established in theory and practice. Also there is good theoretical justification for the use of $F_{50\%}$ as a default proxy for F_{MSY} in the absence of information about stock resilience, as this is appropriate for stocks with low resilience (i.e. steepness), and to require specific justification for use of higher values of F (i.e. lower values of x% SPR).

Stock-recruit based methods

If a time series of stock and recruitment data is available, then reference points for MSY and recruitment overfishing can be more directly determined. In this approach, the primary focus is on the identification of limit reference points for recruitment overfishing because recruitment overfishing has such serious consequences and is difficult to reverse (e.g. Caddy and Agnew 2003, 2004).

There are two common approaches to deriving reference points from stock and recruitment observations. The first is a non-parametric approach. This is used in conjunction with 'per recruit' methods and is particularly suitable when the observed stock-recruitment plot is a cloud of points with no trend. The second is a parametric approach in which a stock-recruitment function is formally fitted to the data.

The non-parametric approach was primarily developed by Sissenwine and Shepherd (1987) and is illustrated in Figure 4. The gradient of any straight line through the origin of a stock-recruitment plot or cloud of observations is a survival ratio (i.e. recruits per spawning biomass, Figure 4b), and the reciprocal is the spawning biomass per recruit produced under a particular fishing mortality (Figure 4a). Points along the straight line give the survival ratio necessary to support that constant fishing mortality because, for average replacement, the spawning biomass needed to produce a recruit must equal the spawning biomass per recruit.

If the observed survival ratio points are mostly higher than the straight line for a given fishing mortality through the historical stock-recruitment cloud, then for that history of recruitment the population would increase under that fishing mortality. If on the other hand they are mostly lower, then the population would decrease. A straight line drawn through the median of the observed survival ratio values can be taken to estimate the survival ratio (recruits produced per unit spawning biomass) for replacement of the observed population, and the corresponding F given by the reciprocal of the survival ratio is F_{med} (sometimes called F_{rep} or F-replacement). Similarly F_{high} and F_{low} are the fishing mortality rates giving replacement for 10% and 90% respectively of the observed survival ratios (Figure 4b).

F_{med} has been used as a limit reference point for recruitment overfishing (Sissenwine and Shepherd 1987, Mace and Sissenwine 1993, Rosenberg *et al.* 1994). This is on the basis that the median replacement line can be regarded as an estimate of the average slope of the stock-recruitment close to the origin, and so the corresponding fishing mortality is an estimate of F_{crash}. Strictly this only applies if there is no compensation across the stock sizes observed, or if the observations included are at sufficiently low stock sizes that the stock-recruitment relationship is approximately

linear (in which case the stock is already recruitment overfished, *sensu* Cooke 1982). Otherwise the observations are likely to include observations of relatively high spawning biomass with relatively low survival ratios and in these circumstances F_{med} will be lower than F_{crash} (i.e. F_{med} will be a conservative estimate of the fishing mortality for recruitment overfishing).

These 'replacement F' reference points, F_{high}, F_{med} and F_{low}, have been used both in fishery assessments (e.g. Anon. 1981, Hilden 1993, Mace and Sissenwine 1993) and as performance measures in theoretical and simulation development of fishery assessment methods (e.g. Myers and Mertz 1998, Collie and Gislason 2001). Because F_{crash} would ultimately cause population collapse, it is clearly a highly undesirable fishing mortality, even as a limit reference point with a low probability of being realised. And similarly F_{med} is an undesirable reference point if the stock and recruitment observations are such that F_{med} is likely to be similar to F_{crash} (which are the circumstances in which F_{med} is best estimated).

The parametric approach to defining reference points from observations of stock and recruitment involves fitting a stock recruitment relationship and deriving the required reference points. There is a considerable literature on fitting such relationships (e.g. Hilborn and Walters 1992 for fitting the Ricker and the Beverton and Holt functions, and ICES 2003a, b for a 'segmented regression' approach). Figure 7 illustrates a stock recruitment relationship. The Maximum Surplus Recruitment occurs at B_{MSY}, and F_{MSY} is given by the spawning biomass per recruit 'replacement line' that intercepts the stock-recruitment curve at that point.

Other reference points that have been used based on such stock recruitment functions are:
- $B_{50\%R}$, the spawning biomass at which recruitment is 50% of its maximum level. Mace (1994) and Myers *et al.* (1994) use this as a limit reference point for a recruitment overfished population. $F_{50\%R}$ reduces the population to $B_{50\%R}$ on average and is the limit reference point for recruitment overfishing for this approach;
- $20\%B_0$, the spawning biomass that is 20% of the unfished level, has been a very common limit reference point for a recruitment overfished population (Myers *et al.* 1994). The fishing mortality that reduces the population to $20\%B_0$ on average is the associated limit reference point for recruitment overfishing, and with specification of the steepness of the stock recruitment relationship the appropriate $F_{x\%}$ to give a 20% spawning biomass reduction can be calculated (x% will be close to 20% for high steepness and greater than 20% for low steepness);
- B_{lim}, the spawning stock biomass where recruitment begins to decline as identified by 'segmented regression' analysis. This is essentially the point where the stock-recruitment relationship is found to get significantly steeper. B_{lim} is used as a limit reference point for a recruitment overfished population. F_{lim} reduces the population to B_{lim} on average and is the associated limit reference point for recruitment overfishing (ICES 2003a, b).

<u>The effect of steepness and stochasticity in the stock recruitment relationship</u>

A simple modeling exercise illustrates the effects of steepness and recruitment stochasticity on common properties of interest—yield, level of depletion, and the

probability of being above different biomass thresholds (Figure 8). Two stocks are shown in Figure 8, one with high steepness ($h = 0.9$) and one with low steepness ($h = 0.4$). Otherwise the two stocks have identical population parameters, chosen to be typical of a fishery on a species with a natural mortality of 0.2. Constant fishing mortality is applied, and the CAY/MAY interpretation of MSY is used for which F_{MSY} is the F giving maximum long-term (100y) yield across multiple simulations with random variability in recruitment. Annual variability in recruitment for both stocks is represented by a lognormal distribution about mean recruitment with a coefficient of variation of 75%, which is a common level of variability (Meyers *et al.* 1995). Recruitment variability results in a probability distribution for the various properties of interest such as biomass and yield, rather than a single point outcome.

The combined effects of density dependent recruitment and of recruitment variability are illustrated in Figure 8a. For high steepness, the stock is productive and robust to fishing in the sense that a high yield and recruitment are maintained over a relatively wide range of fishing mortalities (Figure 8b). However F_{MSY} is larger for the high steepness stock and so fishing at F_{MSY} results in a greater reduction in spawning biomass for the high steepness stock than for the low steepness stock (Figure 8c). Recruitment fluctuations in the high steepness stock result in some very low stock sizes when fished at the constant F_{MSY} level. Constant fishing at F_{MSY} results in the biomass being B_{MSY} on average, but the biomass will be below B_{MSY} about 50% of the time—and sometimes considerably below it (Figure 8c and d). If such low biomass outcomes are undesirable, then a low probability of reducing the biomass below B_{MSY} or some other biomass limit must be specified in addition to specifying the maximisation of yield, and meeting these two specifications will require a fishing mortality less than F_{MSY} (Figure 8d).

Similar simulations can be conducted for the Maximum Constant Yield (MCY) interpretation of MSY. For both steepness examples the biomass of the fished population decreases steadily as the MCY is increased, and then the population collapses abruptly and with very high probability when CAY exceeds MSY. In both stocks this collapse happens at about the mean B_{MSY}, which is a relatively high level of biomass having a very low probability of collapse if fished under a constant fishing mortality strategy rather than a constant catch strategy. This illustrates the danger of inappropriately specified and tested MCY strategies.

The CCAMLR precautionary approach to MCY (e.g. Constable *et al.* 2000) requires that the median fished biomass be no less than 50% (75% for major prey species) of the median unfished biomass and that the fished biomass have no more than a 0.1 probability of being below 20% of the median unfished biomass. These criteria are effectively the reference points for this implementation of MCY, and they were chosen after extensive simulation testing.

The choice of a reference point includes a choice of risk, with uncertainty and variability in both the biological system and the observation system being important factors in determining risk. For fisheries with comprehensive information the risks and trade-offs can be examined through statistical modeling and simulation testing (e.g. Francis 1992a, Hilborn and Walters 1992, Parma 1993, Punt and Hilborn 1994). Similarly, simulation testing has been used to examine the reliability of prospective reference points in less information-rich situations. For example reference points

based on %SPR, such as $F_{40\%}$, have been simulation tested in various situations to determine whether they achieve high long-term yields and avoid recruitment overfishing for the majority of stocks examined (e.g. Mace 1994, Collie and Gislason 2001, Clark 2002). While a %SPR reference point may be used as a deterministic point estimate in an information-poor fishery, the choice of the level of %SPR can be based on testing that includes stochastic considerations.

Recruitment-based limit reference points

Limit reference points for recruitment overfishing can in theory be determined from the form and parameters of the stock-recruitment relationship. In practice few fisheries have a reliable time series of stock and recruitment data available. For many that do, the practical use of a fitted stock-recruitment relationship is limited by variability in recruitment, the short time-series available, and other methodological difficulties (e.g. Walters and Ludwig 1981). Often several different forms of curve can provide similar statistical fits to the observations (with sometimes none providing convincing fits), and each implying different reference points.

Furthermore this approach to determining a limit reference point is fundamentally anti-precautionary in that to obtain good estimates of the reference point (particularly the limit reference points for recruitment overfishing and overfished stocks), the population must be reduced to the point where recruitment overfishing has begun and can be recognised. For example Myers *et al.* (1994) emphasised the importance of demonstrating that recruitment is below a proposed recruitment overfishing limit reference point rather than above it, so as to demonstrate that the reference point is valid and need not be taken simply 'on faith'.

A requirement that limit reference points be determined and demonstrated on a stock-by-stock basis before they are accepted for use for that stock would require that recruitment overfishing is a necessary phase of fishery development. This is not consistent with international (e.g. UNFSA 1995) or national (e.g. Anon. 2001) guidance to fisheries management on the use of limit reference points, or with the common lesson that recruitment overfishing is a highly undesirable situation that is difficult to recover from (Caddy and Agnew 2002, 2003).

To avoid the need for intentional recruitment overfishing an accepted limit reference point must be identified prior to the event, and any empirical testing about its validity for a particular stock must be done in a way that has a very low risk of breaching that reference point for the stock as a whole. This is especially important in the case of newly developing fisheries.

While it is undesirable to recruitment overfish a stock, this has happened in many fisheries. Recently, considerable effort has gone into compiling stock and recruitment data sets from the world's fisheries (e.g. Myers *et al.* 1995) and conducting comparative analysis to learn from them and identify reference points for recruitment overfishing (e.g. Myers *et al.* 1994, 1998, Myers and Mertz 1998, ICES 2003b,c).

These analyses very clearly show that recruitment overfishing is a reality in many of the world's most studied fisheries. They also very clearly dispel a view, which is still commonly held, that recruitment is 'usually' independent of spawning biomass and that it is appropriate to assume independence in the absence of information to the

contrary (Myers *et al.* 1994, Gilbert 1997). The falsity of this assumption was dramatically demonstrated as an unfortunate lesson from the collapse of a major world fishery (Walters and Maguire 1996).

Recruitment is often highly variable from year to year, is correlated with environmental effects and contains multi-year auto-correlation. But virtually all stocks with a long time-series of observations over a range of stock sizes that includes low spawning stock show a threshold below which average recruitment decreases (see Myers *et al.* 1994, Myers and Barrowman 1996, ICES 2003b,c). Furthermore, in some cases they show recruitment depensation at low stock abundance rather than compensation (Liermann and Hilborn 1997). However it is also clear that species, and species groups, differ greatly in their ability to maintain high recruitment from reduced stock sizes. For example, Mace and Sissenwine (1993) found that in the 83 stocks they examined, the replacement %SPR ranged from 2% to 65%. This means that some species can provide average replacement recruitment with %SPR reduced by 98% from the unfished level while others fail to provide average replacement recruitment with %SPR reduced by only 35%.

Myers *et al.* (1994) examined 71 stocks to test different spawning biomass limit reference points to define recruitment overfishing, which they regarded as seriously reduced recruitment. They recommended setting a biomass limit reference point so that recruitment would be 50% of the maximum predicted average recruitment (R_{max}). However they noted that generalisation was difficult and no method performed well in all circumstances.

Myers *et al.* (1994) also examined $20\%B_0$ as a limit reference point. From their analysis $20\%B_0$ provides a reasonable threshold for recruitment overfishing under the definitions used by ICES (2000b,c) and Cooke (1984) because there is little (but some) reduction in recruitment below the $20\%B_0$ limit. This analysis was based mostly on productive stocks. For less productive stocks the corresponding threshold is $30\%B_0$ (Musick 1999, Mace *et al.* 2002)

The $50\%R_{max}$ approach often corresponded to very low limit spawning biomass levels (mostly less than 10% of the unfished level and many less than 5%). In the examples examined by Myers *et al.*, recruitment for spawning biomasses below the $\%R_{max}$ threshold was significantly lower than above it.

Clearly the spawning biomass giving $50\%R_{max}$ is a limit to be avoided, and it is consistent with the FAO definition of a recruitment overfished stock—that is showing a 'significantly reduced average recruitment'. However this sets the limit reference point at a level where the damage is already done, rather than where the damage can be avoided, and populations would be expected to experience significant recruitment decline as they approached the limit from the 'safe side'.

ICES (2003b, c) takes a more conservative view and sets the limit reference point for spawning biomass such that average recruitment is not impaired (rather than 50% impaired as for the $50\%R_{max}$ reference point). The ICES approach is consistent with the definition of recruitment overfishing by Cooke (1984).

Target reference points

Myers *et al.* (1998, 2002) conducted meta-analysis of over 700 stocks and provided estimates of the steepness parameter in the stock-recruitment relationship and the maximum annual reproductive rate at low stock size, which determines F_{crash}. These are good sources of guidance on selection of either of these parameters.

Myers *et al.* (1998) showed that there is relatively little variation in the maximum annual reproductive rate among most species and that these rates are lower than were generally thought. This implies that for many species, the fishing mortality rate for high and sustainable yield is also likely to be lower than previously thought (see Walters and Martell 2002).

Myers *et al.* (2002) showed that stock-recruitment steepness is greater than about 0.3 for most species and greater than about 0.5 for most species with a reproductive longevity greater than 5 years. Combining these results with those of Clark (2002) suggests that, at least deterministically, $F_{50\%}$ would provide high sustainable yields (more than about 85% of MSY) and maintain biomass above about 25%B_0 for most stocks. $F_{40\%}$ would similarly be a reasonable target for stocks with a reproductive longevity greater than 5 years.

(iii) A more holistic approach to the management system and the use of reference points

The third set of developments to improve the practical application of MSY related to viewing the fishery management system as a whole, rather than looking at its parts in isolation. Because fishery management is a system of choices made in relation to data collection, assessment analysis, reference points, selection of management measures and management implementation all interact, and the performance of the whole cannot be determined from any one part.

This more holistic approach has resulted in an increasingly widespread examination of the likely outcomes of management systems through simulation approaches such as adaptive management, management procedure testing, and management strategy evaluation (e.g. Walters and Hilborn 1976, Hilborn and Walters 1992, Kirkwood 1993, Sainsbury 2000). In some cases, analytic methods can also be used to examine the outcomes for some parts of the system (e.g. Thompson 1999).

The emphasis in these approaches is on recognition of uncertainty in model specification, estimation and implementation of management measures, and identification of management approaches that will robustly achieve the desired outcomes despite the uncertainties. Selections made in all aspects of the management strategy—including observations, assessment, selection of management measures and implementation—can be important. But, other things being equal, it is the feed-back loop caused by the use of assessment results to trigger management measures that provides much of the ability to robustly achieve desired outcomes.

Building on related work in optimal control, this approach has led to a recent focus on the definition and use of decision rules in fisheries management. Decision rules are designed to achieve target reference points and to avoid limit reference points under the circumstances of the fishery, and can include discrete or continuous trigger reference points that relate indicators to management actions.

These approaches highlight the distinction between a target or limit reference point and a trigger reference point (which includes the parameters or thresholds of decision rules). The target and limit reference points are not in themselves the trigger reference points, but rather the triggers and catch decision rules are selected so as to achieve the targets and avoid the limits.

As a result of these considerations, there are several common forms of catch decision rule used in fisheries (see Figure 9 for examples). While each can perform well in particular circumstances, the constant escapement decision rule is recognised as being close to optimal if escapement can be reasonably measured, even if recruitment varies randomly or systematically (Parma 2002, Hilborn and Walters 1992). Proportional escapement decision rules, such as the $F_{0.1}$ strategy and the constant fishing mortality-based MAY approach to MSY, can also perform very well for appropriate selections of the fishing mortality (e.g. Mace 1988b). And proportional escapement decision rules are relatively easily applied. More recent decision rules recognise that a constant fishing mortality may not provide adequate protection and opportunity for stock rebuilding at low stock sizes. So many decision rules now combine proportional escapement in the vicinity of the target biomass, with a proportional harvest rate at lower stock size. Examples are the decision rules shown in Figures 1 and 2 which combine a threshold below which fishing mortality is zero or perhaps reduced to a very low level for monitoring purposes; a central range over which fishing mortality decreases as biomass decreases so as to allow the stock to avoid the limit reference point and rebuild to the target; and an upper range within which the fishing mortality is constant. The details of the decision rule, such as the best indicators, estimators, trigger reference points and levels of fishing mortality, should be developed and tested through simulation trials that take account of the key uncertainties in estimation and stock dynamics (Sainsbury et. al 2000).

Biomass, fishing mortality and empirical reference points

Two kinds of reference points are in common use: fishing mortality reference points, and biomass reference points. Empirical reference points relate to directly measurable quantities such as catch rate or size composition of the catch. While empirical reference points have not been commonly used they provide distinct advantages in some circumstances because they are easily understood and communicated, and are often simpler and cheaper to apply. The main challenge with their use is ensuring that their use will deliver the overall fishery management outcomes that are sought.

Where recently comprehensive standards have been specified for fishery management and assessment, both fishing mortality and biomass based reference points are required (e.g. NMFS 1998a, ICES 2003a, b, DAFF 2007) relating to the concepts of overfishing and overfished respectively. In addition there are empirical reference points, based on more directly observable properties than mortality and biomass, which are not commonly used but are potentially promising.

Fishing mortality reference points include the definition of overfishing. Fishing mortality is also the parameter that is under most direct management control, for example through regulation of catch, fishing effort, fishing gear and size limits. Consequently, fishing mortality (or a proxy such as the level of fishing effort) is usually the management control variable in decision rules.

Biomass reference points include the definition of being overfished. Biomass strongly affects or determines many biological properties of a population, such as the mean recruitment, genetic diversity, and role in ecosystem energy flows. However management alone does not usually directly control biomass, and so biomass is not usually a management control variable in decision rules. Biomass reference points are often used as trigger reference points for control of fishing mortality and to initiate specific stock rebuilding measures. Both fishing mortality and biomass reference points are required because their combination determines the appropriate management action.

The use of fishing mortality and biomass reference points requires that they, or some reasonable proxies for them, be estimated so as to assess the status of the stock and determine the appropriate management response. There are no direct measures of the parameters of interest (e.g. current fishing mortality and biomass, unfished biomass, the fishing mortality giving maximum long-term yield) and so they are estimated by fitting a population model to the observational data that are available. The estimates are therefore model dependent (in that different estimates arise from use of different models), as well as data dependent.

Empirical reference points are particular values of an indicator, such as the catch, catch rate, length structure, spatial distribution or sex ratio, that can be directly measured and that usually have relatively simple statistical properties. The empirical indicators and their reference points are used to judge fishery status and trigger management actions. This is not a case of calculating an approximate or proxy value of fishing mortality or biomass for use in a decision rule that is expressed in terms of fishing mortality or biomass. Rather the empirical indicator and its reference point are used directly in the assessment and decision rule. Empirical indicators (e.g. catch rate, the size distribution of the catch or the spatial range of a stock) and the associated reference points often can be difficult to interpret uniquely. As with all decision rules, but especially decision rules based on empirical indicators and reference points, it is highly desirable to simulation-test the combination of indicator, empirical reference point and decision rule to ensure that their use will achieve the targets on average and avoid the limits that are set for the population. That is the intended performance of the fishery relates to the yield, the population and the ecosystem, and prospective empirical indicators, reference points and decision rules should be evaluated for how well they are likely to achieve that intent.

The following section examines each of these three types of reference point in turn.

1. Fishing mortality reference points

Fishing mortality is more directly under management control than biomass, through the prescription of limits to catch or fishing effort. For this reason fishing mortality is usually a major focus of management and stock assessment attention. In most cases current fishing mortality is estimated indirectly from stock assessment and applied to a biomass estimate in order to specify a catch limit (or less commonly an effort limit). But fishing mortality can be more directly measured, for example through tagging programs. Walters and Martell (2002) suggest that more attention should be given to directly measuring and managing fishing mortality.

The US National Standard Guidelines for fisheries assessment and management (NMFS 1998a) provide detailed requirements for fisheries management and assessment. These must be operationally interpreted and applied in each fishery management plan. The National Standards include requirements for:

- A Maximum Fishing Mortality Threshold (MFMT) for each stock that must not exceed F_{MSY} under the MAY interpretation of MSY. If MSY cannot be calculated directly because of lack of information, then a range of proxies can be used. In mixed stock situations one or more stocks may be used as an indicator for the whole. The MFMT can be specified by a decision rule so that it can vary with current biomass or other factors. The MFMT is a limit reference point.

- An Optimum Yield (OY) determining mechanism or rule that is based on MSY, that must prevent overfishing, and that can be reduced from the MSY limit by consideration of relevant social and economic factors. Where MSY cannot be calculated directly because of lack of information then a range of proxies can be used. The fishing mortalities provided by the OY rules are target reference points. The OY can be specified by a decision rule based on the MSY fishing mortality and/or other MSY-based reference points. The OY decision rules are designed to achieve OY on average but avoid the MSY limits for fishing mortality, biomass and catch. The OY fishing mortality for a stock below B_{MSY} must be lower than it would have been had that stock been near or above B_{MSY}. The OY is required to be risk averse so that greater uncertainty corresponds to greater caution in setting harvest rates and catch levels.

- A definition of the overfishing level (OFL) relating to conditions that jeopardise the capacity of the stock to produce MSY on an ongoing basis. The overfishing definition must not exceed the MFMT and, as for the MFMT, it can be specified by a decision rule. If the MFMT or the overfishing definition is exceeded for one year or is approached (i.e. projected to be breached within 2 years under relevant circumstances), then within 1 year a management response is required that will prevent the limit being breached or will return the fishing rate to the OY level. Under some circumstances an exception can be made that allows for overfishing of some species in a mixed species fishery.

A comprehensive approach to identifying appropriate limit and target reference points to meet these needs—with different methods used depending on information availability—has been developed for the Gulf of Alaska groundfish fisheries (see Appendix 3) and for coastal California (see Appendix 4).

The approach used by ICES for setting limit reference points for fishing mortality is described in Appendix 5. The essence of this approach is to set a precautionary limit reference point (F_{pa}) such that, for the estimation uncertainty involved, the intended fishing mortality limit (F_{lim}) has a very low chance of being reached so long as the estimated mortality at any time is below the precautionary limit. Across many Arctic cod stocks, F_{pa} was about $0.6F_{lim}$ (ICES 2003c) and B_{pa} was about $1.4B_{lim}$.

Target reference points for fishing mortality are not presently defined by ICES, other than that they should be lower than limits. There is no prescribed management response or advice for situations in which limit reference points are being approached. However methods to set the target safely below the limit, based on the uncertainty in estimation of current fishing mortality, are available (e.g. Caddy and McGarvey 1996, Gerrodette *et al.* 2002 and Figure 2). These fishing mortality based approaches are extendable to reference points based on biomass or other measures.

A summary of the fishing mortality reference points used in fisheries identified as showing very good practice in the use of reference points is provided in Table 1. Key observations from these fisheries are:
- Limit reference points for fishing mortality that define overfishing are used in most of these fisheries.
- F_{MSY} under the MAY interpretation is a frequently used limit, with a conservatively specified MCY interpretation used in some cases. The conservative specification of MCY includes consideration of uncertainty in estimation, resource dynamics, recruitment variability and the ecosystem role of the target species.
- Where F_{MSY} cannot be estimated because of data limitations, proxies based on $F_{35\%}$ to $F_{40\%}$ have been successful for many stocks, although levels of $F_{50\%}$ or lower have been found to be necessary for long-lived and low productivity stocks.
- Fishing mortality or catch decision rules are commonly used as a means of avoiding limits and maintaining the fishery in a desirable state, even if explicit targets for fishing mortality are not specified.
- The lower Tiers (i.e. 4-6) of Appendix 3 are not adequate because they do not provide a means of reducing the target fishing mortality as biomass or fishing mortality limits are approached.

2. Biomass reference points

Fishing mortality is more directly controllable by fishery management than biomass, and fishing mortality is often the focus of management actions and reference points. But it is the population biomass and its structure that determine the bulk of the ecological and fishery properties that are of interest and concern for a sustainable fishery. Specifically:
- The total productivity of the stock, and consequentially the catch available, is greatest at B_{MSY}. It is lower at both greater and smaller biomasses.
- The same catch can be taken with lower total effort when the biomass is above B_{MSY} than it can when biomass is below B_{MSY}. Economic costs and environmental impacts generally increase as total effort increases. The greatest economic benefit and the least environmental impact therefore generally occur with biomass greater than $B_{MSY.}$ (The exceptions to this economic benefits argument occur in relatively unusual circumstances such as when capture efficiency increases and capture costs decreases with population abundance, when population productivity is less than the economic discount rate, when the ratio of capture cost to catch value is very low, when the value of the catch increases substantially as the quantity caught decreases, or when there are large subsidies on the cost of capture. E.g. see Anderson 1977, Clark 1990).
- Serious and irreversible harm to species is expected to result from serious recruitment overfishing and/or loss of genetic diversity. The risk of serious

recruitment overfishing is high at low biomass. Genetic diversity usually decreases with population size and reaches unviable levels at (usually very) low population size. However the genetically effective population size (N_e) in fish populations may be considerably lower than the census population size and this can result in strong genetic selection and reduced genetic viability even though there are still large numbers of fish remaining in the harvested population. Furthermore genetic diversity is often sustained by more or less discrete stocks or sub-stocks that use different spawning times/locations, and these stocks may be more vulnerable to sequential loss at low total population size.

- The length of time required to recover overfished stocks, and often the severity of the necessary management measures needed to bring about recovery, increases as population size (i.e. the extent of depletion) decreases.
- The role of a species in many ecosystem processes (e.g. transfer of matter and energy through food webs) depends on its biomass and the age/size range. Generally the age/size range will decrease as biomass is decreased by fishing.
- The chance of population collapse due to fluctuations in the environment increases as population size decreases.
- Robust to uncertainty in estimation and model specification, including to changes in the climate or ecosystem (Ludwig et al 1993).

So is highly desirable to avoid very low levels of population biomass and to maintain relatively high levels. A combination of measurement error, assessment error and natural variability can make identifying and maintaining an appropriate level difficult. But nevertheless biomass reference points are very commonly used in successfully managed fisheries.

The US National Standard Guidelines for fisheries assessment and management (NMFS 1998a) require definition for each stock of a Minimum Stock Size Threshold (MSST), which is a limit reference point for biomass (B_{lim}). The MSST is the larger of (i) 0.5 B_{MSY} (or a proxy for it) and (ii) the minimum biomass that would rebuild to B_{MSY} in 10 years if fished at the MFMT. A stock whose biomass is lower than MSST is defined as being overfished. If biomass is lower than MSST for 1 year or if MSST is approached (i.e. projected to be breached within 2 years under relevant circumstances) then within 1 year a management plan is required that will prevent the limit being breached or will rebuild the stock biomass to a recovery target level. The US National Standard Guidelines recommend as proxies for B_{MSY} either $0.4B_0$, if a good estimate of B_0 is available, or $B_{30\%}$ - $B_{40\%}$ otherwise. The proxy $B_{35\%}$ is commonly used.

Under some circumstances an exception can be made that allows for some species in a mixed species fishery to remain overfished. If stock rebuilding is needed the rebuilding target is B_{MSY} (or a proxy for it). The lower bound to the timeframe for rebuilding is the time taken to rebuild the stock in the absence of any fishing. If this lower bound is less than 10 years then the rebuilding time can be adjusted up to a maximum of 10 years for socioeconomic reasons. If the lower bound is more than 10 years then the rebuilding time can be adjusted upwards for socioeconomic reasons to a maximum of the lower bound plus one mean generation time for the harvested species involved.

The US Guidelines do not require specification of a target for biomass. However biomass thresholds are used in fishing mortality decision rules such as those developed for the Alaskan fisheries (Appendix 3). For Tiers 1 and 2, the decision rules reduce fishing mortality when the biomass is below B_{MSY}, and for Tier 3 when biomass is slightly above B_{MSY}. There are no biomass thresholds for Tiers 4-6 other than the MSST.

The ICES approach to setting biomass limits is described in Appendix 5. This approach sets a precautionary limit reference point (B_{pa}) such that, for the estimation uncertainty involved, the intended population biomass limit (B_{lim}) has a very low chance of being breached so long as the estimated biomass is above the precautionary limit. Across many Arctic cod stocks B_{pa} was about $1.4B_{lim}$ (ICES 2003c).

There are no biomass targets specified by ICES. However, as for fishing mortality, methods are available to consider estimation uncertainty and set the target reference point safely distant from the limit reference point (e.g. Caddy and McGarvey 1996 and Gerrodette *et al.* 2002).

CCAMLR bases several of its key assessments on the specification of two biomass criteria, with the appropriate fishing mortality or catch levels being calculated to ensure that neither criterion is violated (see de la Mare 1996, Constable and de la Mare 1996, Constable *et al.* 2000). Although two criteria are identified, only one of these is usually constraining in any particular situation. These two criteria are:
(i)   The median fished biomass is no less than 50% of the median unfished biomass. In the case of designated key prey species the reduction is to be no less than 75% of the unfished median.
(ii)   There is no more than a 0.10 probability that the fished biomass is below 20% of the median unfished biomass.

The first criterion is effectively a target reference point, with the fishery expected to be above and below the target with equal frequency. The reduction of 50% in the biomass from the unfished situation is intended to be a conservative interpretation of the reduction to B_{MSY}. The use of a 75% reduction in the case of designated key prey species is to allow increased escapement from the fishery to meet the needs of predators. The second criterion is effectively a limit reference point, and specification of the 10th percentile ensures a low chance of the limit biomass being violated. The fishery is required on average to meet both criteria, and it is accepted that application of the criteria will result in the population being below the target in half of the outcomes in the real world and below the limit in 10% of the outcomes.

The CCAMLR criteria are applied in a wide variety of ways and situations. The approach was initially developed to provide a precautionary catch level for the krill fishery during its developmental phase (de la Mare 1996, Constable and de la Mare 1996). The catch level is a Maximum Constant Yield (MCY), expressed as that fraction of the unexploited biomass that simultaneously meets both criteria. The methodology for this application requires relatively little information on the stock and no prior fishery information. It requires estimates of growth and mortality, survey estimates of unfished abundance, and measures of the uncertainty in each of these.

The initial application to krill was modified to allow updating of the catch level as new information becomes available and to avoid the need for an initial biomass estimate (Constable *et al.* 2000). The modification was developed primarily for toothfish. This approach does not rely on an estimate of initial biomass but instead uses ongoing survey estimates of recruitment, together with either measured or assumed variability in recruitment. Known catches and survey estimates of abundance can be accounted for and, as before, simulation trials are used to find the catch that meets both criteria. This catch is updated periodically, typically annually, as additional recruitment and catch estimates become available.

All of these applications of reference points consider the reference points to be a fixed quantity (e.g. fishing mortality or stock size) that does not vary through time or with changed circumstances. But many populations vary greatly through time in their size and productivity as a result of natural change or fluctuations in environmental conditions, which may range in time scales from years to centuries.

Long-term (decadal or longer), large, reasonably abrupt and persistent changes of this kind are often described as 'regime shifts', and can be associated with major changes to ecosystems (e.g. Hare and Mantua 2000, Scheffer *et al.* 2001). Regime shifts can occur in the absence of fishing, and it is often debatable whether the changes are forced by external factors such as climate, caused by anthropogenic influences including fishing, or a combination of both.

It is useful to distinguish fluctuations that have the same statistical patterns and properties through time (e.g. mean, variance, amplitude, period), and so are statistically stationary, from those that are non-stationary. While a stationary system or population may vary it will have patterns and properties that are the same through time. Consequently historical observations of these patterns and properties are directly useful as guides to the future. However the patterns and properties of a non-stationary system or population change through time. As a result historical observations must be interpreted through a model or some other understanding of the cause of the non-stationarity to make useful predictions of the future—or to provide correct interpretations of observations from the past.

Population models, statistical analysis, and meta-analysis that are the basis of almost all fishery assessments make very strong assumptions of stationarity on the timescale of the historical data and the predictions made. Arguably the earth's climate system and coupled ecosystems are fundamentally non-stationary on a long timescale, and some observed regime shifts can be viewed in that way, but the assumption of stationarity is reasonable for many purposes on shorter timescales (Yndestad 1999, Rial *et al.* 2004). However the onset of anthropogenic climate change will mean that the assumption of stationarity is unlikely to be reasonable in the next decades. Other anthropogenic impacts that can fundamentally change ecosystem dynamics, such as introduced species or major changes in nutrient loads, habitats or food-webs, are also occurring in some fished ecosystems and can be expected to contribute to non-stationarity.

So reference points that are constant through time, and do not reflect the non-stationary nature of ecosystem and population dynamics, can be substantially inadequate. But reference points that can vary through time arbitrarily, without good

ecological justification or on a general claim of regime shift, could undermine the goals of ecologically sustainable harvesting.

There is not a good understanding of how best to incorporate the non-stationarity expected from climate change or other anthropogenic influences into fishery assessments and predictions. This remains a challenge to the definition and use of reference points in fishery management, although the challenge is not unique to fishery management (e.g. Matalas 1997 for water management).

Two approaches that have been developed and used in practical fishery management take quite different approaches—one uses fixed reference points to protect the specific cohorts or age groups in the population while the other changes the reference point for the population as a whole through time. A third approach, based on unfished reference areas, has not yet been used in practice but is described in the Empirical Reference Points section. The two applied approaches are:

A. Reference points applied to a restricted set of cohorts.
This is the approach taken by CCAMLR to management of the mackerel icefish. This species has few cohorts in the population at any one time and quasi-cyclic variability in recruitment strength results in large fluctuations in the size of cohorts and the total population on a time scale of 3-6 years (Kock and Everson 2003). For example, even without fishing the population fell below 20% of the long-term median unexploited level (a common limit reference point) in about 50% of years (de la Mare *et al.* 1998). The approach adopted was to modify the usual CCAMLR precautionary reference points to require that:
-   the median escapement from the fishery of the spawning biomass shall not be less than 75% over a 2 year projection; and
-   the probability of the fished spawning stock falling below 20% of the median unfished level shall not be more than 0.05 higher than for an unfished stock over a 20y projection.

The first of these requirements ensures that the cohorts present in the population have the required high escapement (low depletion) for each 2 year window of time. The 2 year period is intended to protect the individual cohorts present, and to prevent averaging across several cohorts that could allow excessive depletion of weak cohorts. Also it allows recruitment fluctuations that occur on a longer timeframe to be tracked. The second requirement ensures that fishing does not result in a 'significant' increase in the frequency of very low levels of abundance, which is considered to be important both to dependent predator populations and to the recruitment of icefish (de la Mare *et al.* 1998).

B. Population reference points that vary through time.
This is an approach used by the Northern Pacific Fisheries Management Committee to manage Gulf of Alaska pollock (Dorn *et al.* 2003, p124). The northern Pacific Ocean exhibits regime shifts on a multi-decadal timeframe, with pollock abundance apparently varying by orders of magnitude in response even in the absence of fishing (e.g. Klyashtorin 2001, Benson and Trites 2002). The approach used (Dorn *et al.* 2003) was to hindcast the size of the population each year in the absence of fishing ($B_{unfished}$) and then use the depletion limit of $0.3B_{unfished}$ as the limit reference point in that year. The unfished population sizes

were calculated by first fitting a range of stock recruitment relationships to the data from the fished population, then simulating the population in the absence of fishing but using those stock recruitment relationships. This gives a time varying limit reference point, and the same approach could be applied to a target reference point. However this time-varying approach to the limit reference point could still allow catches from very small populations and this may have undesirable effects on later stock recovery or on any dependent species. Consequently, in addition to the time varying limit reference point, a fixed limit on biomass is also applied to the Alaskan pollock—in this case 20% of the long term population size in the absence of fishing—and fishing is to be stopped if either limit is breached.

These methods assume that recruitment to the population, or other measures of productivity, in the absence of fishing can be estimated using the data from the fished population. And that they can be used to estimate what the present population would have been in the absence of fishing. This estimation is obviously sensitive to the assumptions that allow estimation of just what would have happened in the absence of fishing. While the adequacy of this approach cannot be known decisively, the use of several different stock-recruitment or stock-productivity relationships allows robustness and bounds to be examined. And this seems successful in at least some circumstances (e.g. Dorn *et al.* 2003, p124).

A summary of the biomass reference points used in fisheries identified as showing very good practice in the use of reference points is provided in Table 2. Key observations from these fisheries are that:
-   $0.25B_0$-$0.5B_0$ and approximately B_{MSY} are commonly used biomass limit reference points.
-   In several fisheries the minimum biomass limit reference point is set at a level to ensure that average recruitment does not decline. This is more conservative than the limit implied by previous recruitment overfishing definitions, which were at a level where serious or significant reduction in average recruitment occurs. In some cases the biomass limit reference point is identified on the basis of historical observations, and in others the lowest observed biomass is treated as this limit even though a decline in recruitment has not been observed at that biomass.
-   The US National Guidelines (NMFS 1998a) and the Australian Harvest Strategy Policy (DAFF 2007) both identify $0.5B_{MSY}$ as a limit reference point. However this can result in large reductions in biomass, with the consequent risk to economic benefits, ecological and genetic functionality and reversibility that is associated with low population biomass. For example $B_{35\%}$ is a common proxy for B_{MSY} and for many productive species B_{MSY} is in the vicinity of $0.35B_0$, so setting $B_{lim}=0.5B_{MSY}$ can result in limit reference points that are very low (i.e. $0.175B_0$ or $B_{17.5\%}$) compared to the approaches used in the other fisheries examined.
-   In situations where natural fluctuations in productivity or recruitment result in large fluctuations in stock size, the limit reference point is modified to track these changes through time, while also placing a limit on the absolute level of depletion that is acceptable. This results in the limit being the greater of two quantities, a time-varying fraction of the predicted unfished biomass and a static fraction of B_{MSY} or B_0. This approach can be expected to provide some

protection against non-stationarity in fish production due to events such as climate change.

3. Empirical reference points

'Classically' reference points have been defined for the quantities of direct interest, such as the stock biomass or fishing mortality, and these same quantities have been used as the parameters of decision rules. However these quantities are usually not directly measured, and instead they are estimated through population or stock assessment models. They are, in reality, 'model-based estimates' because the estimate depends on the model that is used.

However a wide range of other more direct approaches are possible. Both Hilborn (2002) and Butterworth (in press) point out that excessive focus on direct reference points and model-based estimation, and the kinds of management strategies and decision rules that they imply, can prevent the exploration of alternatives that could be cheaper to use, easier to understand and have similar or better effectiveness. These alternative approaches include empirical or 'data-based' indicators and reference points. These are based on quantities that can be easily measured, and like model-based reference points, they can be used to identify limits and targets or to trigger management actions.

The advantage of empirical indicators and reference points is their simplicity in operation. The disadvantage is that they may not provide reliable guidance in the management of the quantities that are of fundamental interest and concern, which are usually closely related to stock biomass and fishing mortality, because the easily measured indicators for a fishery are often difficult to interpret uniquely. This disadvantage may be overcome by extensive simulation testing to demonstrate the performance of a management strategy based on empirical reference points and decision rules (e.g. Sainsbury *et al.* 2000).

So complex models may be needed to test the reliability of the management strategies based empirical reference points and decision rules. However, once acceptable empirical reference points and decision rules are identified, they are usually easily implemented and do not require ongoing complex stock assessments.

Kelly and Codling (2006) argue that model-based reference points and population assessment is too expensive and that this approach has largely failed north Atlantic fisheries. They advocate management for this and other fisheries should be based on empirical indicators—particularly in the case of data-poor fisheries.

Schnute and Haigh (2006) discuss and examine model-based and empirical reference points, and conclude that they both should contribute to a modern 'strategic theory' for fisheries management. However practical experience with the reliable application of empirical reference points is, so far, relatively limited.

There has been a great deal of examination of the use of catch rate as an indicator in fishery management. But usually this is of catch rate as a measure of stock abundance within a stock assessment, rather than catch rate directly providing empirical reference points with triggered management decision rules. Nonetheless there are a

few examples of catch rates being used more directly as empirical indicators with reference points.

Starr *et al.* (1997) used commercial catch rate directly in a catch decision rule for a New Zealand rock lobster fishery. Similarly commercial catch rate is used to trigger automatic within-season changes to the allowable catch in an Australian toothfish fishery (Tuck *et al.* 2001). In both of these cases, extensive simulation testing of the performance and robustness of the management strategy based on these catch rate reference points was conducted before the approach was adopted for use in management.

The abalone fishery in New South Wales, Australia used the research survey catch rate at a set of fixed locations in 1994 (Worthington *et al.* 2002) to provide both target and trigger reference points for the fishery management plan. This reference year was chosen to represent a desired and presumed sustainable period. The survey locations are scattered throughout the fished area and the survey design is both fixed and not known to fishery operators.

Punt *et al.* (2001) developed and tested empirical reference points for an Australian swordfish fishery, including trigger reference points based on catch rates, percentiles of the distribution of fish length in the catch, and percentiles of the distribution of fish weights in the catch. They showed that approaches based on these reference points can have relatively high rates of failure (e.g. triggering action when it was not needed, or not triggering action when it was needed). However, this is essentially a risk-based trade-off in selecting the trigger reference point. For example, an empirical reference point could be selected to give a high level of stock protection, but this would also give low exploitation rate and yield.

Davies *et al.* (2007) extended the use of empirical reference points in this Australian swordfish fishery to include a different decision rule. The indicators used were catch rate and the size distribution of the catch. And the decision rule was a hierarchical decision tree designed to isolate and provide an appropriate management response for each of the different classes of reasons for change in the indicators – such as growth overfishing, recruitment overfishing or changing fishing effectiveness. These empirical indicators, reference points and decision rules were simulation tested and shown to robustly meet the stock and yield requirements of the Australian Commonwealth Harvest Strategy Policy (i.e. maintain biomass at an average of B_{MSY} or higher, and avoid the limits of ½ B_{MSY} and F_{MSY} with high probability).

Hobday *et al.* (2004) provide a series of indicators and reference points for use in ecological risk assessment. Some of these are empirical reference points that relate directly observable indicators to categories of increasing risk and decreasing acceptability (see Appendix 6). For target species, and regarding their 'major' and 'severe' risk categories as being appropriate for a limit reference point, their empirical limit reference points are:
  - reduction of geographical range by 25% of the unfished range; and
  - loss of 25% of the spawning units or spawning locations compared to the unfished situation.

Scandol (2003) used simulation to test the reliability and trade-offs in using 'control chart' decision methods and empirical trigger points for the management of several coastal Australian fisheries. The empirical trigger points were based on total catch, catch rate, the distribution of fish length in the catch, as well as various measures of the distribution of age in the catch. He showed that management strategies based on empirical indicators and reference points could have a high error rate, but that sustainable fisheries could be achieved for suitably conservative choices of the reference points. The use of these empirical indicators and reference points is being considered in the current re-development of fishery management plans.

Hilborn *et al.* (2002) used a general framework for the use of empirical reference points, including trigger reference points. They examined a strategy for rock-fish management that was based on maintaining a steady value in an empirical reference point. The performance of this strategy was compared with that of strategies based on classical use of reference points. The findings illustrated the not unexpected sensitivity of classical strategies to uncertainty in biomass estimates, and showed that a strategy based on empirical reference points can perform well. However these empirical indicators and reference points are not presently in use.

Fulton *et al.* (2004b, 2005) used simulation methods to test a wide range of common indicators of ecosystem health and function, including both empirical indicators (i.e. simple statistics from the observed data) and model derived indicators (i.e. estimates of unobserved quantities through fitting models to the observed data). They found that easily measured empirical estimates consistently out-performed model-based indicators of the effects of fishing on ecosystem structure and food webs. Model based indicators were highly dependent on the adequacy of the model used, and also in many cases were either too slow to detect trends or overly smoothed trends. A necessary condition (not necessarily a sufficient condition) for good performance from many model-based indicators is accurate and frequent observations. Simple empirical indicators can be derived from the same observations.

A further application of empirical reference points that has been frequently suggested is the use of fished and unfished reference sites to provide a direct measure or indicator of the impact of fishing. Fully protected Marine Protected Areas (reserves) are often identified as providing benefits to fishery management, including by providing unfished reference sites to allow measurement of the effects of fishing (see for example references in Sainsbury and Sumalia 2003). These approaches have not been applied in fishery management as yet. However one developing application is to the Californian coastal fisheries (Kaufman *et al.* 2004 and Appendix 4). They propose using the surveys from protected areas to measure the stock biomass that would exist at that point in time in the absence of fishing (i.e. $B_{unfished}$), which can then be used as an empirical reference point for comparison with the biomass in fished areas. It is likely that the survey observations will need to be adjusted for such things as the effects of exchange of animals across the boundary of the protected area, site specific details and regional changes that effect both inside and outside the protected area. So in this approach $B_{unfished}$ would probably be determined through a combination of reference site observations and models. This approach has some obvious challenges and problems. Unaccounted for degradation of the reference site would result in under-estimation of the effects of fishing. And model-based interpretations are sensitive to the assumptions that allow estimation of what would have happened in the

absence of fishing, as for the other model based approaches to estimating $B_{unfished}$ (e.g. Dorn *et al.* 2003, p124). But these approaches warrant further development and testing.

The use of empirical reference points deserves considerably more attention. They are the main option available for many small-scale, low-valued or information-poor fisheries, but have also proved useful in some large scale and data rich fisheries. In principle, empirical reference points should be able to deliver outcomes with the desired level of reliability. The details of their robustness and risk may often be very context-specific, however, and at present there are no empirical reference points that can be identified as best practice on the basis of current experience with their use.

Best practice reference points for target or commercially retained species

Best practice requires both biomass and fishing mortality targets and limits. It also requires the use of trigger reference points and decision rules that achieve the target reference points on average and have a low chance of breaching the limit reference points over an extended period.

Table 3 provides the reference points for fishing mortality and biomass identified here as best practice. These reference points are provided in terms of fishing mortality and biomass, but their application to particular fisheries may be via other indicators and measures—including empirical indicators such as those based on fish length or other direct observations. If empirical indicators and reference points are used, it is necessary to demonstrate that in application they give outcomes equivalent to or better than the biomass and fishing mortality reference points in Table 3.

The best practice limit reference point for biomass is the greatest of the 3 quantities:
- B_{lim}, the biomass below which average recruitment declines or stock dynamics are highly uncertain.
- $0.3 B_{unfished}$, where $B_{unfished}$ is the biomass expected to be present at a specific time in the absence of fishing. The biomass initially present when the fishery started, B_0, is commonly used as an unchanging proxy for $B_{unfished}$. But this is becoming increasingly unsatisfactory because the underlying assumption of stationarity is less tenable under the emerging understanding of natural ecosystem dynamics and the system-level effects of climate change and other anthropogenic effects. Instead a dynamic, time-varying estimate of $B_{unfished}$ should be used. This can be provided by model calculations based on the expected stock dynamics in the absence of a fishery, by reference to unfished sites, or a combination of both. For stocks that naturally exhibit large fluctuations in productivity, the quantity $0.3B_{unfished}$ can give very low levels of absolute biomass during periods of low productivity. In these cases, an additional limit reference point is required, which should be no lower than 0.2 of the median long-term unfished biomass.
- The biomass from which rebuilding to the target reference point could be achieved in a period no greater than one generation time for the species plus 10y.

$0.2B_{unfished}$ is commonly used as a limit reference point and there is good empirical support that this avoids recruitment overfishing for productive stocks (i.e. is an appropriate B_{lim} for such stocks). But it is not regarded as the best practice limit

reference point because this level of depletion (i) does not avoid recruitment overfishing in low productivity stocks, (ii) may not provide adequate protection for other fishing impacts that are likely to be slowly reversible or irreversible (e.g. genetic modification, reduced age structure with consequences to the quality of spawning, changed ecological role such as in food-web dynamics, ease of population recovery from the limit), (iii) is less robust to uncertainty in estimation and model specification, including to changes in the climate or ecosystem (Ludwig et al 1993), and (iv) is not consistent with the precautionary reference point approach of ICES, where B_{pa} was found to be about $1.4B_{lim}$ in fishery assessments based on good data sets.

The best practice limit reference point for fishing mortality is F_{MSY}, the fishing mortality giving maximum sustainable yield. Where this cannot be estimated directly, $F_{50\%}$, the fishing mortality that gives a 50% reduction in the spawning biomass per recruit is a default proxy for most species. For the stock-recruitment steepness seen in most fish (i.e. greater than about 0.3) $F_{50\%}$ provides more than 80% of the MSY and depletes the biomass to no more than about 30% of the unfished level. Use of a lower percentage in the 'per recruit' proxy value would require explicit justification as to why $F_{50\%}$ is unreasonable. Higher fishing mortality reference points (e.g. $F_{40\%}$) could be justified if there is information to suggest that the stock has high steepness in its stock-recruitment relationship. $F_{60\%}$ should be used as the default limit reference point for species suspected of having a particularly low ability to compensate for fishery removals (e.g. those with a very low natural mortality or very low 'steepness' in the stock-recruitment relationship).

These best practice reference points include several options or alternatives that are appropriate for different circumstances. Key principles in selecting reference points for target species are:
- Target reference points are set safely below limit reference points. There is a very low chance that a fishery assessed as being near the target is actually near or beyond the limit.
- There is a low chance that reasonable expectations of natural variability, in combination with the fishery, will result in the limit being approached or exceeded.
- A stock that is below the biomass target should be harvested at a lower rate than one above the target.
- If a biomass limit reference point is breached, or is predicted to be breached under expected natural and fishery conditions, then a recovery or avoidance plan is triggered.
- Rules used to set target catch levels should explicitly be risk averse, so that greater uncertainty regarding the status or reproductive capacity of a stock corresponds to greater caution ('safety margins') in setting target catch levels.

Biomass and fishing mortality reference points should, to the extent possible, be consistent (i.e. fishing at the target fishing mortality has a high chance of maintaining the population at the target biomass on average and avoiding the limit biomass). However because they will be estimated in different ways and from different data there is no guarantee that they will always be consistent. But in any event, limits should have a low chance of being breached, and trigger reference points and their resulting management responses should maintain biomass and fishing mortality at the target reference points on average.

Biomass target and limit reference points should take into consideration maintenance of genetic diversity of the retained species. This includes sub-stocks that may be genetically isolated to varying degrees. While limited understanding means that for most species genetic effects cannot be accounted for explicitly, two precautionary actions can be taken in setting reference points:

- If there is doubt about the stock structure, assume small stock units with separate reference points, rather than large units with aggregated reference points. In particular treat potentially discrete breeding populations as separate stocks and avoid loss of local spawning locations or aggregations.
- Maintain a high spawning biomass. Favor harvesting options that give high biomass, and avoid options that have biomass less than B_{MSY}.

Target and limit reference points should be set at levels that do not impair average recruitment. This is in contrast to acceptance, or high risk, of significantly impaired average recruitment which is implied by some approaches to defining recruitment overfishing and associated reference points. The biomass limits are set at levels such that the stock can be rebuilt from the limit to the target in a time relevant to human intergenerational equity. A limit of 30% of the unfished biomass level is appropriate even to stocks that can apparently maintain average recruitment at lower biomass. In populations with very high 'steepness', the MSY may occur near or below $30\%B_0$ but in such populations, similar catches can still be taken at higher biomass. Limiting the reduction in biomass to 30% is to maintain ecological and population processes (including as yet poorly understood genetic, physiological, population and ecosystem effects of low population size), to provide a safety margin for unforeseen dynamics (including changing environmental trends or variability), and to avoid levels of depletion from which it is potentially difficult to recover.

If the limit reference point is breached, or is predicted to be breached under current or expected circumstances, then a stock rebuilding plan should be triggered to return the stock to a safer condition (at least the MSY level of abundance). The timeframe for recovery should be consistent with human intergenerational equity and the capacity of the stock to rebuild. Current best practice for the stock recovery timeframe is illustrated by the US National Standard Guideline (i.e. a timeframe of no less than the time to recover without fishing and no more than the time to recover without fishing plus 10y—see NMFS 1998a) and the CCAMLR Convention (i.e. recovery within 20-30y—see CCAMLR 1984).

Biomass target reference points are set so that biomass will be mostly (e.g. at least 50% and more reasonably 75% of the time) above the biomass giving MSY. A population assessed as being close to the target reference point should have a very low chance of actually being near the limit reference point. Given the accuracy of most stock monitoring, the biomass target would usually be expected to be above $40\%B_0$ to avoid a limit of $30\%B_0$.

Fishing mortality target reference points are altered according to the status of fishing mortality and biomass. Appendix 3, and Figures 1 and 2, provide examples of this.

It should be demonstrated, for example by simulation testing or some other reviewable method, that the decision rules and trigger reference points have a good chance of being able to achieve target and avoid limit reference points under the range

of circumstances that the fishery might be reasonably expected to face (e.g. uncertainty or variability in stock productivity, in monitoring and in assessment, and in regulating fishing operations).

Best practice context for use of reference points for target or commercially retained species

Common features relevant to use of reference points in the fisheries regarded as showing best practice are:

- *Quality assurance of data.* Extensive use of appropriate mechanisms— such as the use of observation technologies (e.g. satellite vessel monitoring), observers and cross validation of information sources— to ensure that the data from the fishery on catches, discards and fishing operations are accurate and adequate.
- *Fishery-independent estimates of some key indicators of the fishery resource.* Most commonly this is the absolute or relative abundance of the stock (e.g. through surveys) and/or fishing mortality (e.g. through tag-recapture programs).
- *Prescribed management responses.* Emphasis is given to providing clear statements of how fishing will be altered in relation to the indicators and reference points so as to maintain the fishery near the target and avoid the limit. This especially includes specification of catch or effort decision rules that use trigger reference points to relate the catch, fishing mortality or effort to indicators of current stock condition.
- *Accounting for uncertainty in knowledge, variability and error.* The reference points and planned management responses are not selected based on average considerations or assuming perfect knowledge. Rather the selections are based on a realistic range of possibilities that might occur in the fishery, and they are shown to have a high chance of working despite this uncertainty, variability and error.
- *Priority on resource health.* Most of the fisheries considered here as illustrating best practice operate with clear and explicit statements that place highest priority on maintaining resource health and productivity in situations where this is inconsistent with other fishery objectives, and have chosen limit reference points that strongly avoid growth and recruitment overfishing. This point reaffirms the conclusion reached by Hilborn *et al.* (2003) for a sustainable fishery.
- *All sources of fishing mortality must be included in monitoring, fishery assessment and reconciliation of actual with intended catches.* This includes catch that is not retained in the fishery and catch by other resource users.
- *Management decisions are explicitly precautionary.* Operationally this does not mean that more data are required to prove that an activity is acceptable (i.e. a reversal of the 'burden of proof'). Rather it focuses on ensuring that the activities have a high chance of achieving the management objective given the data and understanding that are currently available. Specific elements of management decision-making with precaution can include simple catch setting rules that are more conservative if information is limited (e.g. the Alaskan tier system and the Californian management arrangements); procedures that explicitly set lower catch levels if uncertainty is high (either by changing the reference points as in the ICES process or specifically incorporating estimates

of uncertainty as in the CCAMLR procedure); and application of limits during development of fisheries.

- *New and exploratory fisheries have explicit and precautionary catch limits.* New and exploratory fisheries include those targeting previously untargeted species or areas, including new species or areas within an existing fishery that were previously untargeted. There is usually little direct information available about target species in developing and exploratory fisheries, but such developmental fisheries should operate within explicit and precautionary catch limits that are highly likely to be safe for the types of species and for the kinds and scale of fishing planned. The limits can be increased as the ability to estimate safe yields and impacts increases. The intention is to permit reasonable access and scope for fishery development while ensuring that this occurs within safe limits of impact. The UN Fish Stocks Agreement (UNFSA 1995) states that new and exploratory fisheries shall "adopt as soon as possible cautious conservation and management measures including, inter alia, catch limits and effort limits. Such measures shall remain in force until there are sufficient data to allow assessment of the impact of the fisheries on the long-term sustainability of the stocks, whereupon conservation and management measures based on that assessment shall be implemented. The latter measures shall, if appropriate, allow for the gradual development of the fisheries." For example, CCAMLR requires prior identification of precautionary catch limits for all developmental fisheries, and the requirement that formal stock assessments of sustainable catch be available to revise the precautionary catch limit before a fishery can progress from the developmental stage.

2. By-catch species

Image courtesy of AFMA

Background

The term 'by-catch' has a variety of definitions which usually invoke some combination of the concepts of untargeted catch, unintended catch and discarded catch (e.g. Alverson *et al.* 1994). By-catch in its widest sense includes all organisms killed or damaged by the fishery other than the target organisms that are caught and kept. Here by-catch is used in the sense of the Australian Commonwealth Policy on Fisheries By-Catch (Anon.2000). That is, by-catch species are species that are landed on the fishing vessel and discarded, or species that are adversely affected by fishing gear even though they are not landed. By-catch species do not include target or other commercially retained species that are managed explicitly through management plans, even if some of the catch of these species may be discarded.

Management of by-catch has become a major aspect of fisheries management. There are three main motivations for this. The first, from a fishery utilitarian viewpoint, is to ensure that management of target species is not undermined by by-catch, and to preserve future options for the utilisation of present by-catch species. Indeed, Hilborn *et al.* (2003) point out that eliminating the discard of species that are targets in some fisheries but discards in others and utilising currently discarded by-catch species would give a much greater increase of global fish catch than eliminating all known situations of overfishing.

The second motivation, emphasised in the Code of Conduct for Responsible Fisheries (FAO 1995a), is the responsibility of fishery managers to maintain healthy productive ecosystems. This includes maintaining healthy populations of species that are dependent on or associated with target species, and also extends to community structure and function more generally.

And thirdly, there is a common social concern about the waste that by-catch represents. Such concerns include killing animals needlessly, wasting a resource, and causing ecological impact without social benefit.

Policy on fishery by-catch is usually aimed at (1) minimising by-catch generally, and (2) recognising and protecting species that might be vulnerable to excessive impact or significant harm. These are the core objectives of the Australian Commonwealth Policy on fisheries by-catch (Anon 2000). The US National Standards Guidelines for fisheries management (NMFS (1998a) similarly require by-catch be minimised to the

extent practicable, and if by-catch is unavoidable, that mortality of by-catch species be minimised.

Simply ensuring that by-catch is low or close to zero is sometimes seen as sufficient to address the overall concerns about by-catch. However there are some potential weaknesses with this approach which must be recognised and managed. They are:
- By-catch can be reduced by causing local or global depletion of by-catch species. This would be an example of achieving an operational target but not the intent of by-catch management.
- By-catch can be reduced by converting by-catch species into retained species through developing new markets or products, including fish-meal production. This could be a desirable development in the fishery, so long as it is accompanied by sustainable management of the species at all the stages in the transition from a by-catch to a retained species.
- Even low by-catch levels may excessively impact highly vulnerable species.

A minimum requirement for by-catch management is to ensure that the level of by-catch does not excessively impact or deplete the species involved. There are often many species in the by-catch and little information or understanding about their ecology and dynamics. Consequently there is usually a great deal of uncertainty in the assessment of impacts and safe levels of catch. For a high probability of success in by-catch management, there is a need for the use of risk-based methods to identify the high risk species or activities, and a need for a high level of precaution in the management of by-catch.

There is a huge scientific literature describing the by-catch of fisheries and the effects of that fishing (e.g. see Alverson *et al.*1994, Jennings and Kaiser 1998, Hall 1999 and Kaiser and de Groot 2000). There is also a very large literature on possible indicators for measuring the impacts of fishing and other human impacts on non-target species, communities and ecosystems (e.g. Rapport *et al.* 1998, 1999). Among the many options proposed are the use of indicator species, simple measures of total mortality for individual species, and several measures of community structure and biodiversity. Fulton *et al.* (2004a) provide a recent and thorough review of the options for such indicators in a fishery context. The experience and approaches may be grouped under two headings: species or species group oriented, and community-structure oriented.

Species or species group oriented

No argument has been put forward for accepting greater levels of depletion in by-catch than in target or retained species. Indeed if the by-catch species are small and low in the food-chain, as they often are, then the acceptable depletion limit should be less than for target species high in the food chain (see Walters *et al.* 2005 and reference points for food webs below). So the reference points for by-catch species should be the same as those for target species at the same level in the food chain.

Usually the main difference in addressing by-catch species, rather than target species, is that there is much less information available on by-catch species and so there is a greater reliance on proxy measures for reference points (see Appendix 3). For this reason empirical reference points could be particularly valuable for by-catch management, but this approach has not been extensively developed.

By-catch management is particularly well developed for the Alaskan fisheries in the Northern Pacific Ocean and for the Southern Ocean fisheries managed under CCAMLR. By-catch species in the Alaskan Bering Sea ground-fish fisheries are managed using a series of complementary measures (NMFS 2006a, b). These include gear restrictions and area closures to limit by-catch of particular species or species groups; a very low catch limit for 'forage fish' low in the food chain which is designed to permit only very occasional and accidental capture; a by-catch limit on the aggregate quantity of nominated 'other species', and by-catch limits for some species or species groups that are considered particularly important ecologically or at higher risk. The aggregate 'other species' catch limit is set at 5% of the sum of the target species catch limits, which is arbitrary but is argued to be low enough to prevent targeting and to protect most of the species in the grouping from excessive depletion. It is recognised that this approach does carry risks, including that some of the most vulnerable species within the category could be depleted, and both monitoring of catches and scientific surveys are used to assess outcomes. The by-catch limits for species or species groups considered to be ecologically important or at higher risk are determined using the same tiered methods as applied to the target species (see Appendix 3). Because of data limitations, the lowest tier is used in which the catch limit is 0.75 of natural mortality multiplied by the estimated biomass. The biomass estimates are provided by assessment of the fishery catch and effort data, verified by observers, and scientific surveys. The fishing mortality of 0.75M is expected to be appropriate for species with high steepness in the stock-recruitment relationship, but would deplete species with low steepness (Beddington and Cooke 1983, Walters and Martell 2002).

The identification of by-catch species or species groups at higher risk in the Alaskan fisheries is based on analogy with similar species, analysis of the fishery catch and effort information, and scientific surveys. More formally structured Environmental Risk Assessments have also been used to identify high risk species and to target management effort in some Australian fisheries (Stobutzki *et al.* 2001, 2002, Fletcher *et al.* 2002 Hobday *et al.* 2004 and Appendix 6). Hobday *et al.* (2004) also provide benchmarks for risk that can be treated as reference points. Their 'major risks' are appropriate limit reference points and are defined as: reduction of the population below 50% of its unfished level; reduction of geographical range by more than 25% of the unfished range; loss of more than 25% of spawning units (e.g. locations or stocks); and impacts that take more than 5 generation time for recovery (Appendix 6).

CCAMLR sets by-catch limits for each species or species group in each management zone. These limits are aimed at preventing unmanaged targeting and depletion of by-catch species (Constable *et al.* 2000). CCAMLR has a structured process for changing the categorisation of a species from by-catch to 'developmental fishery', and then to be accepted as a target species, with each step requiring an increasingly rigorous assessment of the safe catch level (Constable *et al.* 2000). In the absence of sufficient information to conduct a stock assessment, a precautionary by-catch limit is set at a level that is agreed to be extremely safe for the species. For example in many situations this by-catch limit is 50t per year for each statistical reporting area. Higher by-catch limits can be set when a stock assessment is possible, and these use the same performance criteria as those applied to target species. That is a catch level such that it is predicted that:

(i)   The median fished biomass is no less than 50% of the median unfished biomass. In the case of designated key prey species the reduction is to no less than 75% of the unfished median.

(ii)   There is no more than a 0.10 probability that the fished biomass is below 20% of the median unfished biomass.

In addition to these overall by-catch limits, several CCAMLR Conservation Measures are aimed at avoiding localised depletion of by-catch species. For exploratory fisheries, Small Scale Research Units are identified and catch limits for both target and by-catch species are set for these units separately (Conservation Measure 41-01, see CCAMLR 2005). In addition, in the event that any one fishing operation (e.g. a trawl or long-line haul) catches more than 2t of by-catch for species with a catch limit set by stock assessment, or 1t of by-catch for species with a precautionary catch limit, then Conservation Measures 33-02 and 33-03 require that fishing be relocated by at least 5 miles, and it cannot return to the original location for at least 5 days (see CCAMLR 2005).

A common difficulty in managing by-catch species is the limited information available to estimate the sustainable catch level. Some approaches for dealing with this situation are:

-   When there is limited information about the population parameters and population size, a standard population model is used to determine a precautionary and constant catch level (Constable and de la Mare 1996). This constant catch level is then used until it is possible to conduct a more detailed assessment of the population—usually as the species becomes a fishery target species rather than a by-catch species.

-   Pope *et al.* (2000) present two methods for calculating fishing mortality using either length frequency information (based on the length-based cohort analysis of Jones, 1981) or information on catch rate and the area fished (a weighted swept-area method).

-   Sainsbury (1984) provides several methods for estimating population size and mean recruitment from limited catch and catch rate information.

-   Tiers 5 and 6 of the US Alaskan assessment methodology (see Appendix 3) provide methods for calculating an acceptable catch limit from limited data. Tier 5 requires estimates of population size and natural mortality rate. Tier 6 requires only a history of catches.

-   Hobday *et al.* (2004) provide a suite of methods for identifying the risk categories of by-catch species based on general biological properties (which may be obtained from the scientific literature) and the kind of fishing. These methods explicitly incorporate precaution in using 'worst case' interpretations in the absence of information to the contrary.

-   In the absence of information, the precautionary CCAMLR approach adopts a small by-catch limit (e.g. 50t per year and large reporting area). This is based on expert judgement about the catch level that is confidently considered to be safe.

Approaches such as these allow progress to be made in setting by-catch limits in varying circumstances of limited information.

Community-structure oriented

Fishing mortality on by-catch species affects not only those species, but can also alter the characteristics and functioning of the communities in which they live (e.g. Jennings and Kaiser 1998, Hall 1999 and Kaiser and de Groot 2000). The fishery can displace or replace predator species (Pauly 1988, Fogarty and Murawski 1998, Jennings and Kaiser 1998, Gislason and Rice 1998, Cury *et al.* 2000, Furness 2002, Dayton *et al.* 2002); can cause 'competitive release' in unfished species by removing their competitors (Fulton *et al.* 2004a); or can increase scavenger species that feed on discarded by-catch and offal. Scavengers attracted into the vicinity of fishing operations can then become vulnerable to damage or death themselves (e.g. seabird by-catch on long-lines). But while these broader impacts of fishing are recognised, there is not a well developed understanding or methodology to measure them, or to set limit or target reference points for them.

A large number of indicators that reflect changes in the structure and function of whole communities have been suggested and reviewed (e.g. Fulton *et al.* 2004 alb). Some work has been done to investigate their ability to detect fishery induced changes, and their sensitivity to natural fluctuations or other human impacts (e.g. Rice 2000, SCOR-IOC 2005). However, this is at an early stage of development and there is not a strong body of scientifically agreed approaches.

Another example of a community level indicator is the size composition of the ecological community being fished. Pope and Knight (1982) suggested that a linear relationship exists between log numbers per size class and fish size in a community, and that changes in this relationship measure the effects of fishing on community composition. Both theory and observation now support this. The slope of this relationship decreases, and the intercept increases, as a result of fishing pressure (Pope *et al.*, 1987; Rice and Gislason, 1996; Gislason and Rice 1998; Bianchi *et al.* 2000). Comparison across ecosystems (Bianchi *et al.* 2000) has shown that different ecosystems respond differently to fishing. Consequently the slope and intercept of the size spectrum cannot be simply interpreted across ecosystems and the comparison does not suggest a single intercept or slope that is desirable for all systems. However the relationship has been shown to be a reliable indicator of fishing effects when applied to time series from a given ecosystem, and so could be used as the basis for specification of reference points for fishery management. While these conclusions have been supported by the recent theoretical work of Pope *et al.* (2006), to date there has been no development of desired (target) or undesired (limit) reference points for indicators based on the ecosystem size spectrum. That is there has been no specification for any given ecosystem of how much change in the community size composition is desired to enhance fishery production or how much change is unacceptable because of the consequences to fishery production and/or the broader ecosystem. However Pope *et al.* (2006) suggest that this is possible and is a useful area for further development.

Another commonly considered community indicator is mean trophic level; and this is discussed further in the section on reference points for food webs. Fishing usually selectively removes larger, higher trophic level and long generation-time organisms leaving smaller, lower trophic level and short generation-time organisms (Pauly *et al.*

1998, Fogarty and Murawski 1998, Jennings and Kaiser 1998, Gislason and Rice 1998, Cury *et al.* 2000, Furness 2002, Dayton *et al.* 2002). By-catch species are usually of the latter type. At present, there is no firm basis or experience for setting targets or limits for by-catch species based on mean trophic level as an indicator.

Other commonly used indicators of community composition are species richness (the number of species), species evenness (the relative similarity in numbers of individuals of each species) and species diversity (some combination of richness and evenness). These indicators have been reviewed in the context of fishery impacts by Fulton *et al.* (2004a, b). There is a good theoretical and empirical basis for these indicators, and they are widely used in 'state of environment' reporting. However they are often very insensitive to the variables of interest e.g. only showing small changes in the indicator with loss of species, invasion by introduced species, and major changes in community composition. In addition, cause and effect can be very difficult to interpret. Experience with use of these indicators is too limited to set either target or limit reference points based on them.

Fulton *et al.* (2004b) show that community and ecosystem impacts of fishing could be effectively measured, and potentially managed, through monitoring and assessment of a suite of indicator species. No single species or group can capture all of the characteristics of an ecosystem; thus use of a suite of diverse indicators is preferable to using a single one or a range of similar ones, because all have different strengths and weaknesses (Fulton *et al.* 2004b). For example, indicators based on groups with rapid population growth (such as bacteria and plankton) are useful for giving early warnings of ecosystem damage but are susceptible to false positives. In considering these issues, Fulton *et al.* (2004b) list four categories that must be spanned by the representative suite of indicator species or groups. These are (1) rapid turnover groups (e.g. plankton, bacteria); (2) benthic habitat generating groups (e.g. sponges); (3) fish at both high and low positions in the food-chain; and (4) sensitive high trophic level groups (e.g. seabirds and marine mammals). In simulation studies Fulton *et al.* (2004b) found that the use of reference points for these indicators based on those used for target species (i.e. the limits on mortality and depletion, including more conservative limits for groups low in the food chain) gave encouraging results. However there is no practical experience or demonstration of this approach.

Ecosystem level indicators suffer from multidimensionality. Collie *et al.* (2003) argue that the reason that the multispecies, multifleet, age-structured models for the North Sea, developed over a decade ago by ICES, have not been used in management is because of the sheer multi-dimensional complexity of the results. To overcome this, there have been several attempts to reduce the ecosystem properties of interest to just a few integrated measures (Link *et al.* 2002, Fulton 2004 alb). Principal Components Analysis (PCA) has been successfully applied (Collie *et al.* 2003; Link *et al.* 2002) to track the changes in ecosystems that have been closely monitored for long periods of time. Collie *et al.* (2003) present their results in the form of radar plots in which the reference point becomes a reference circle. Reference areas/surfaces can be identified on the PCA plot and management actions could be triggered to avoid undesirable regions on the PCA plot and remain in desirable regions (Link *et al.* 2002).

Similarly RAPFISH (Pitcher and Preikishot 2001) condenses information from different fisheries across five issues relevant to sustainable fishing: ecological,

economic, ethical, social and technological. Each of these issues is represented by a large number of indices that are scored through a combination of expert judgment and direct measures. By-catch is scored as highly sustainable if discards are 0-10% of the retained catch, medium if they are 10-40% of the catch, and poorly sustainable/unsustainable if they are greater than 50% of the catch. Sustainability is scored high if only 1-10 species are caught; medium if 10-100 species are caught; and low if more that 100 species are caught. Multi-dimensional Scaling (MDS) is used to produce two-dimensional ordinations from these scores, interpreted so that sustainability is one dimension and the other represents differences across fisheries that are independent of sustainability. The resulting sustainability indicator has been used to compare fisheries and to track the changes in a single fishery through time. Both limit and target regions could be identified for composite indicators such as those developed using PCA and MDS methods. But so far there is no practical experience or demonstration of the use of reference points based on such composite indicators.

Greenstreet and Rogers (2006) provide a comprehensive examination of simple empirical indicators for the demersal fish community of the northeastern North Sea. This was based on trawl surveys from 1927 to 1997 and comparison of areas with contrasting fishing intensity, with low intensity areas being used to set empirical benchmarks or reference points for a relatively pristine fish community. The approach they take could also be applied to the use of unfished reference sites to establish empirical indicators and reference points for the effects of fishing. Greenstreet and Rogers demonstrated major reductions from the reference points in measures such as the percent of fish larger than 30cm, the average fish weight, the average asymptotic size of the fish making up the community, the age at maturity, and the length at maturity. They found no change in the trophic level as measured by the nitrogen isotope ratio, which indicated that large predators had been replaced by smaller predators operating at the same trophic level. Both species richness and species evenness measures of biodiversity decreased in the more heavily fished areas. However, there was evidence that intensively fished areas initially had higher biodiversity than lightly fished areas, and that the fisheries focused in the more diverse areas. Consequently the lightly fished areas did not provide an appropriate reference level for a relatively pristine fish community.

Best practice reference points for by-catch species

The distinction between retained and by-catch species is a result of human values and utilisation, rather than one of biology or ecology. Thus it follows that there is no biological basis for by-catch and retained species being given different limit reference points. Consequently the best practice limit reference points for by-catch species are the same as those for target and retained species, or for 'prey species' if the by-catch species are also considered to be key supporting elements low in the food web.

These limit reference points may not be directly measurable for all by-catch species because there is often very limited information available about historical fishery catches, population abundances or the key biological and ecological properties of by-catch species. And it may not be either warranted or feasible to provide the information necessary for direct measurement of the limit reference points. In these cases, proxies for the limit reference points can be developed in a risk assessment that is explicit in terms of the justification for the proxies, the evidence for assessment of

risk, and the use of precaution to achieve the intention of the limit reference points despite uncertainties.

The Environmental Risk Assessment methodology currently being applied by the Australian Fisheries Management Authority is best practice in this context. The methods used to estimate precautionary catch limits for threatened, endangered and protected species and the CCAMLR method to estimate precautionary catch limits are best practice for the setting of by-catch limits from limited data.

The best practice target reference point for by-catch is zero, with the recognition that this is a target to be approached as far as is feasible or acceptable. Best practice also establishes interim limits to catch or fishing mortality for by-catch that reflect what is currently regarded as being feasible or acceptable, and these are expected to change through time to reflect continuous improvement toward zero by-catch.

The interim limit on catch or fishing mortality must be lower than that implied by the limit reference point, and in the absence of more specific information, the best practice interim fishing mortality is 0.75 of the natural mortality rate. The interim level is a limit not a target, and it is not desirable to achieve it.

Best practice establishes trigger reference points to initiate management measures to reduce the chance of further by-catch if undesirable levels of by-catch occur, with at least one such trigger reference point being at the identified current feasible level.

These reference points relate to the species making up the by-catch; however a common management concern is the effect of by-catch on the structure and function of the ecosystem as a whole. There is considerable scientific effort going into the evaluation of possible indicators and reference points for these whole-system effects, but currently there is no agreed or demonstrated best practice in their selection or use.

Best practice context for use of reference points for by-catch species

The best practice context for use of reference points for by-catch has four elements:
- an accurate (unbiased) and quality assured way to measure the quantity and composition of the by-catch;
- risk assessment to identify species or groups that are at risk of excessive depletion or other damage;
- catch limits that are set for by-catch so as to prevent unmanaged expansion and targeting by the fishery, to protect species or groups at risk of excessive impact, and to maintain key ecological functions (especially seabed habitats and food web dependencies);
- management measures to ensure catch limits are not exceeded. This can include 'move on' provisions and allocation of by-catch limits for sub-areas within the fishery.

While the reference points developed for target species (including those for species low in the food chain) are appropriate for by-catch species, the main difficulty in their application is usually the lack of information and assessment effort. Catch limits and risk assessments are often based on very limited information and analysis, and they are often applied to groups of species with quite different ecological properties and vulnerabilities. Consequently the management measures for by-catch will usually

need to be set with a high degree of precaution—that is, lack of certainty should not delay or prevent management action to address the risk of significant or potentially irreversible harm.

While sustainable by-catch limits may be uncertain and precautionary, best practice management does include limits on by-catch so as to avoid unmanaged targeting and excessive depletion of by-catch species.

3. Threatened, endangered or protected species and communities

Image courtesy of AFMA

Background

Threatened, endangered or protected (TEP) species and communities are those that have been identified as requiring special protection and management. Species are usually recognised, 'listed' and managed as protected, endangered or threatened through a legislative process, or by international agreement (e.g. the Convention on the International Trade in Endangered Species of Wild Fauna and Flora, CITES). These processes determine what benchmarks or requirements must be applied by a particular jurisdiction. However there are also mechanisms for identifying endangered or threatened species that are not legislatively based, for example the IUCN (World Conservation Union) 'Red List'. In many cases the species involved are long-lived, K-selected species with relatively low fecundity, or are dependent on habitats that have been severely modified or reduced by humans.

The threatened and endangered categories are applied to species that have been reduced to such low levels of abundance that there is a risk of extinction, or because they have been identified as being especially vulnerable to such an outcome from human activities. Other species or groups are protected because of a general societal view that they should be protected irrespective of their current population status (although many of these have also been depleted). Under the Australian EPCB Act (Anon. 2005), protected marine species include cetaceans, sea snakes, pinnipeds, dugong, turtles, syngnathids, penguins and seabirds.

The EBPC Act also allows for the recognition and protection of threatened ecological communities, where these communities have been reduced, modified (e.g. by introduced species) or fractionated to the point where their continued existence is at risk.

The EPCB Act does not prescribe explicit thresholds for the level of reduction or fractionation of populations or habitats that relate to the different categories of threat, but it does include such thresholds in terms of the risk of extinction. These are:

> Critically endangered: The probability of its extinction in the wild is at least 50% in the immediate future

> Endangered: The probability of its extinction in the wild is at least: 20% in the near future

> Vulnerable: The probability of its extinction in the wild is at least 10% in the medium-term future.

The IUCN (the World Conservation Union) provides criteria for listing and delisting of species to various categories of threat (IUCN 2000), and these are also used as a part of the guidance for assessment of conservation status under the EPBC Act (Anon 2005). These criteria include small or rapidly declining population size, small or rapidly declining geographic range, and range fragmentation. The IUCN criteria include:

> Critically endangered: Population reduced by 80% or more in the longer of 10y or 3 generation times; population less than 250 mature individuals and with decline of 25% or more in last 3y or one generation time; or probability of extinction 50% or more in 10y or 3 generations.

> Endangered: Population reduced by 50% or more in the longer of 10y or 3 generation times; population less than 2,500 mature individuals and with decline of 20% or more in last 5y or 2 generation times; or probability of extinction 20% or more in 20y or 5 generations.

> Vulnerable: Population reduced by 30% or more in the longer of 10y or 3 generation times; population less than 10,000 mature individuals and with decline of 10% or more in last 10y or 3 generation times; or probability of extinction 10% or more in 100y.

Although it is recognised that some fish and (especially) elasmobranchs have low fecundity and productivity, the IUCN criteria have been criticised as being more applicable to relatively low fecundity groups such as mammals and birds than to highly fecund organisms such as fish (Musick 1999). There have been very few cases of global extinction of marine fish, despite severe fishing pressure in some cases (Musick 1999, Mace *et al.* 2002, Dulvy *et al.* 2003). Furthermore, significant population reduction is an inevitable and intended consequence in obtaining a high and sustainable catch from many fisheries fishery. For example the IUCN criterion for listing as 'Endangered' is reduction of population size by 50% or more in a period of 10y or 3 fish generation times. But reducing a fish population by 50% or more might

be both sustainable and intentional in a managed fishery, and in a developing fishery this reduction is often intentionally faster than 10y or 3 generation times. And almost all fisheries reduce the target population below 70% of the unfished level and so would meet the IUCN criteria as 'Vulnerable', but experience with fisheries is that this level of depletion is sustainable and does not cause extinction.

However Dulvy *et al.* (2003) warn that the assumption that marine species—on account of their high fecundity and dispersal capability—are less vulnerable to extinction than terrestrial animals might not be correct. In particular they conclude that fish may be as vulnerable as other groups to depensatory processes at low population levels. This is supported to some extent by the analysis of Liermann and Hilborn (1997) who found evidence for depensation in some severely depleted fish populations. While from their study depensation in fish populations seems relatively rare, its existence in some cases further emphasises the need to avoid depleting populations to very low levels. Dulvey *et al.* (2003) also point out that while many target species may be protected from excessive depletion by high productivity, this does not necessarily protect all the by-catch species, some of which may have low productivity. In this context it is significant that the elasmobranchs are well represented among the marine species that have become locally extinct or extremely depleted—for example IUCN has recently declared the angel shark to be locally extinct in the North Sea; the 'common' skate was locally extinct for several years in the Irish Sea (Brander 1981); and the barn-door ray is severely depleted on the continental shelf of the NW Atlantic (Casey and Myers 1998).

After a series of international workshops to develop more appropriate depletion measures for fish species, Musick (1999) identified threshold depletion limits that would trigger an immediate listing as 'vulnerable' and a more thorough assessment of conservation status. These thresholds could be regarded as 'precautionary' in the sense that they apply in the absence of more thorough assessment and until that more thorough assessment is provided and shows that the listing is not justified. The depletion (% reduction) thresholds suggested by Musick (1999) differ according to the productivity of the species, where productivity is assessed from general life history parameters such as the maximum age, intrinsic rate of natural increase (r) and the von Bertalanffy growth parameter (k). Musick's criteria for recognition of a vulnerable species were:
- 99% reduction in abundance for highly productive species (e.g. maximum age 1-3y, r>0.5, k>0.3)
- 95% reduction in abundance for medium productivity species (e.g. maximum age 4-10y, r =0.16-0.5, k=0.16-0.3)
- 80% reduction in abundance for low productivity species (e.g. maximum age 11-30y, r=0.05-0.15, k=0.05-0.15)
- 70% reduction in abundance for very low productivity species (e.g. maximum age >30y, r<0.05), k<0.05)

Two further major reviews followed the report of Musick (1999), and focused on the application of CITES to fisheries. FAO (2001) recommended that the depletion criterion for CITES listing be 5-20% of the unfished population level. Mace *et al.* (2002) concluded that the threshold for concern about the long-term viability of a species is reduction to 5-30% of the unfished or potential population. The 5% end of the range is appropriate for high productivity species, with productivity bring a proxy

for resilience or steepness in the stock-recruitment relationship. FAO (2001) and Mace *et al.* (2002) explicitly did not support thresholds as low as reduction to 1%, even for highly productive species. The appropriate threshold for low productivity species is at the 20% and 30% end of the range according to FAO (2001) and Mace *et al.* (2002) respectively.

Recognising all these arguments, depletion to 5-30% of the unfished or potential population size are better thresholds than the IUCN ones for significant risk of very slowly reversible or irreversible depletion in fish—including the risk of local or global extinction in marine fish species. And at a depletion of 30% the risk of extinction is expected to be very low even for very low productivity species (Clark 2002, Myers *et al.* 2002).

Limit reference points for fishery management should not pose a serious risk of extinction, or other changes that are irreversible or very slow to reverse. And so limit reference points should be set above the depletion thresholds associated with these outcomes. The recommended best practice reference points for biomass and fishing mortality identified in earlier sections of this report, and in particular the use of $0.3B_{unfished}$ as a limit reference point for biomass, are expected to achieve that.

In principle, the target level of catch or injury for TEP species should be zero, in that there is no higher catch that is intended or desired. In Australia, there are legislative penalties for intentionally or 'recklessly' killing or harming TEP species or communities (Anon. 2005). However recovery or management plans can identify non-zero catch limits (for example to reflect unintended and incidental capture) that are feasible and acceptable, provided that they do not significantly compromise the recovery or ongoing protection of the listed species or community. These acceptable levels of capture are interim and periodically reviewed, and would be expected to decrease over time as improvements are made to the fishing operations that interact with TEP species or communities. They are usually subject to close monitoring, verification and reporting.

One of the most fully developed operational and scientifically based approaches to setting such catch limits for TEP species is in the application of the US *Endangered Species Act* and especially the *Marine Mammal Protection Act* (MMPA). These Acts require that listed species be recovered, and remain recovered having been recovered, and that mortality or serious injury to marine mammals be reduced to insignificant levels, approaching zero. These 'insignificant levels' must not, directly or indirectly, 'reduce appreciably' the likelihood of both the survival and recovery of a listed species.

In implementation of these Acts, all fisheries are reviewed annually and categorised as to whether they have (I) frequent (II) occasional or (III) a remote likelihood of causing mortality or serious injury to marine mammals. Fishery operators in Category I and II require authorisation under the MMPA to take marine mammals and can be required to carry observers. Strategic Stocks under the MMPA are those that have been classified as threatened or endangered, or are declining toward levels at which they might be threatened or endangered. If combined mortality/injury for any Strategic Stock is higher than the a level judged to have no significant effect on the recovery and subsequent maintenance of that stock, then a take reduction plan is

required for all fisheries in Category I or II for that stock. The aim of the take reduction plan is to reduce serious injury and deaths to below this level of significant impact level within 6 months (usually the immediate focus in Category I fisheries), and to insignificant levels, approaching zero, on a longer time-frame.

In this approach the level of significant impact is used to identify protected species and fisheries requiring very urgent action. In effect it is used to prioritise management focus on the species and fisheries where the risk is high, as distinct from situations that require improvement but where recovery is at lower risk. The ultimate target of insignificant mortality, approaching zero, is maintained throughout this.

Implementation requires that there be a means of determining the level of mortality or catch that could be regarded as having no significant effect on the recovery and subsequent maintenance of the protected species. The limit must be precautionary and practical in the frequent situation of limited information about the ecology of the protected species, its population size and the level of fishery interaction.

The methodology to determine this limit is based on the concept of Potential Biological Removal (PBR) (Wade, 1998). This is the maximum number of animals that may be removed from the population while still achieving recovery of the depleted population (i.e. from 30% depletion to the natural carrying capacity in no more than 100y) or subsequent maintenance of the population at its carrying capacity (for at least 20y). From extensive consultations and simulation trials, Wade found that a very robust estimate of this limit is:

$$PBR = N_{min} \cdot \frac{1}{2} R_{max} \cdot F_r$$

where N_{min} is the minimum population estimate of the stock, $\frac{1}{2} R_{max}$ is the maximum net productivity rate, and F_r is a recovery factor between 0.1 and 1. A value of 1 allows no extra margin for error. A low value of F_r is precautionary (Wade 1998).

Wade (1998) used simulation testing to show that this formula with an F_r of 0.5 would allow marine mammal populations to reach or maintain their carrying capacity with high probability. A mortality that is consistently greater than the PBR has a 5% chance of depleting a marine mammal population. PBR is therefore a relatively conservative measure. It has not been simulation tested for fish populations but might well prove to be an effective precautionary catch limit for groups other than marine mammals, especially in 'data poor' situations.

A PBR-like method is also used in New Zealand to establish a maximum incidental catch of Hookers sea lion by the arrow squid fishery (Maunder *et al.* 2000, Fletcher 2004). But the maximum catch limit in New Zealand is linked to more immediate management action than is the case for US applications, and the fishery is closed if the pre-set catch limit is reached (Maunder *et al.* 2000, Sainsbury *et al.* 2000).

A major focus of management action to protect TEP species and communities is through reduction in the likelihood of capture. This is addressed through a variety of measures, alone or in combination. For example:
-   Modification of fishing practices, such as the CCAMLR requirements for use of tori poles, night setting and weighted lines to avoid seabird by-catch, and that no offal be discharged so as to reduce 'provisioning' and the

attractiveness of fishing vessels to sea-birds and marine mammals.
- Modification of fishing gear such as the adoption of excluder devices such as the turtle excluders used in the northeastern USA sea scallop fishery and many shrimp fisheries worldwide.
- Time-area closures of 'hot spots' for by-catch that may be pre-identified on the basis of historical experience or triggered in more 'real time' by reaction to by-catch rates as they occur during a fishing season or voyage. For example, the Fishery Management Plan for Atlantic tunas, swordfish and sharks establishes a closure to protect sharks in the mid-Atlantic off North Carolina. Similarly the International Pacific Halibut Commission has closed some areas to fishing in order to protect depleted rockfish stocks, and extensive areas in the Bering Sea and Gulf of Alaska fisheries are closed to fishing to protect haul-out and feeding areas of the Stella sealion. In 2002, the Grand Banks was closed to pelagic long-line gear because analyses showed that the level of by-catch occurring on endangered leatherback and threatened loggerhead turtles jeopardised their populations, and that exclusion of this gear type would reduce turtle by-catch by 60-75%. Within season 'move-on' rules are sometimes used to manage within-season or within-voyage by-catch, and require fishing to be relocated by some minimum distance if by-catch thresholds per fishing operation or local area are exceeded during fishing operations. For example CCAMLR Conservation Measures (CCAMLR 2005) require that fishing must be relocated by at least 5 miles, and cannot return to the initial location for at least 5 days, if any fishing operation (e.g. trawl or long-line haul) produces more than a nominated amount of by-catch.

The legislation governing TEP species in most countries (e.g. Australia, Anon. 2005) also includes provision for protecting the habitats or ecological communities that the protected species are dependent upon. For example Stella sealion protection measures in the northern Pacific include lower harvest of key sealion prey species and extensive areas closed to fishing to protect sealion foraging, nursery and haul-out habitats. However the approaches, targets and limits that have been used so far have been very case specific, and mostly based on qualitative argument. As yet there are no clear and accepted best practice reference points for addressing habitats or ecological communities that the protected species are dependent upon.

Best practice reference points for threatened, endangered or protected species
The best practice limit reference point for threatened, endangered or protected species must allow the species to recover, if it is depleted, and to remain undepleted. The target reference point is minimal or no fishing mortality.

The best practice limit reference point is a mortality or number of deaths calculated by using the Potential Biological Removals (PBR) method with 'recovery factor' (F_r) of 0.5, or variations of that method with similar intent. This is a highly precautionary method that can be applied with limited information (e.g. life history and an estimate of population size) to calculate the number of deaths that would significantly impair recovery of depleted or severely depleted populations and to maintain already healthy populations. Where stocks are suspected of existing in the population, this is applied at the stock level rather than the population level.

The best practice target reference point for catch of threatened, endangered or protected species is zero or approaching zero fishing mortality, with the recognition that this target is to be closely approached within a defined time period. Best practice also establishes levels of catch or fishing mortality to reflect what is currently regarded as being feasible and acceptable, and this is expected to change through time to reflect continuous improvement. The 'currently feasible' level would usually be a relatively small fraction of the limit reference point level. As for the 'currently feasible' levels specified for by-catch of any sort, these are not targets to be achieved. Rather they are benchmarks for continuous improvement and can be triggers to initiate additional management intervention in the event that the intended improvement is not achieved.

Best practice context for use of reference points for threatened, endangered or protected species and communities

The best practice context for use of reference points for TEP species or communities is similar to that for any by-catch, but with a much greater emphasis on precaution, accurate monitoring and effective management intervention. Best practice considerations include:
- an accurate (unbiased) and verified measure of the quantity and composition of the by-catch;
- catch, impact or mortality limits that would prevent recovery being known, and set with a high level of precaution in the absence of adequate information, and urgent effective action being taken to reduce the impacts where that limit is exceeded;
- ultimate reduction of all catch, impact and mortality limits to zero or approaching zero;
- management measures to ensure catch limits in total or sub areas/times are not likely to be exceeded, and effective elimination of the chance of further catch or interactions with TEP species or communities if a limit is exceeded.

Best practice in addressing TEP species and communities includes facilitating change in fishing operations to reduce the likelihood of capture or other interactions during fishing operations. This potentially includes capture reduction devices (e.g. excluder devices), changed fishing gear (e.g. circle hooks or net mesh-size), time-area closures or catch limits, and strategies for keeping TEP species away from fishing operations (e.g. tori poles and streamers for longlines, acoustic signals, eliminating offal and by-catch discharge). Best practice in this respect normally involves very active engagement and support for industry-developed solutions to interactions with TEP species or communities.

Where possible, mechanisms should be in place to reduce the scope for a single operator to close a whole fishery or sub area/time by exceeding the overall catch or impact limits, either through bad practice, bad luck or intent. For example this could be addressed by mechanisms such as allocated and tradable catch or impact limits.

4. Habitats

Image courtesy of CSIRO Wealth from Oceans Flagship, Voyage of Discovery

Background

A habitat is the biological and physical environment in which an organism lives, and is closely related to the concept of the ecological niche (e.g. Kolasa and Waltho 1998). Habitats are one of the basic determinants of the structure and productivity of marine ecosystems, and consequently of the kind and amount of fishery production available. Habitats can be defined at different scales and levels of refinement and a hierarchy of habitats can be recognised - for example a continental shelf habitat, sponge reef or sand bank habitats within that, and particular kinds of sponge or sand areas within these (e.g. Allen and Starr 1982, Noss 1990, Connor 2003).

Because habitats can be hierarchically defined, there is scope for confusion if the scale and context of the interpretation being used is not clear. In a fishery management context the ecosystem service of interest is usually the productivity and persistence of populations, so the habitat usage by these populations defines the relevant level in the hierarchy of habitats. Additionally it is usually not feasible to manage fishing activities on very small space scales, because of the movement patterns of the target species, the large area affected by the fishing gear or the costs of fishing constraints and compliance. While there are some fisheries that can effectively use small scale spatial management, notably hand-collection fisheries, the relevant scale of habitat definition for fishery management is usually in the middle range in the hierarchical classification of habitats – that is (Levels 3, 4 or 5 of Last *et al.* in press).

Different aged fish usually occupy different habitats, giving rise to the 'chain of habitats' or 'critical habitats' required by a species to complete its life cycle (e.g. Naiman and Latterell 2006). Marine fisheries currently do not affect the oceanographic aspects of fish habitats. But they can affect the biological aspects of habits e.g. through removal of predators, competitors and prey, or removal of seabed habitat-forming organisms (e.g. sponges and corals) and some types of geological structure.

The effects on predator, competitor and prey are addressed here in the sections relating to target species, by-catch species and food webs; however reference points to protect and maintain ecological roles and functions are not well developed for any of these categories.

For marine fisheries, most attention with respect to habitats has focused on seabed habitats, mainly because of the direct impact of dredging and trawling on seabed habitat-forming organisms and mobilisation of seabed sediments and nutrients (Jennings and Kaiser 1998, Hall 1999, Kaiser and de Groot 2000).

Pitcher *et al.* (2000) and Collie *et al.* (2000) summarise the relevant scientific work. They concluded:
- Trawling and dredging can alter benthic communities and habitats, and sometimes cause significant degradation;
- Inter-tidal dredging and scallop dredging have the greatest impacts of all gear types, and the effect of bottom trawling depends on the type of habitat involved;
- Seabed fauna are more adversely impacted and recover more slowly in situations with stable environments, habitats and sediments;
- Recovery rates—and the depletion caused by a given amount of fishing—vary greatly. They are greatest in environments that are naturally unstable and disturbed, and can be very slow in naturally stable environments.
- Continued fishing, especially intense fishing, can maintain seabed habitats in a permanently altered state.

Management actions available for conservation of marine habitats include closing areas to fishing, reducing fishing effort, and changing to less destructive fishing gears. These measures are commonly used in fishery management.

There have been examples of recovery in closed areas (e.g. Kaiser 2000 for scallop habitats and Sainsbury 1997 for tropical continental shelf habitats). However, the benefits of closed areas are not always clear (e.g. Sainsbury and Sumalia 2003). Usually this is because there is no formal performance evaluation accompanying the closed area (see Ward *et al.* 2001 for a review in the Australian context). But it is also because indicators of habitat disturbance are very weakly developed, so there is little agreement on what to measure and what would constitute disturbance.

In ecology, it has been recognised for many years that spatial and temporal patterns of disturbance and recovery play a key role in ecosystem dynamics and the maintenance of diversity, but to date no generality has emerged about the best indicators and limits (White and Jentsch 2001). In a fisheries context, Link (2004) suggests that the area covered or occupied by long lived seabed biota (such as corals) could provide a good indicator of habitat functionality and disturbance, and suggest that are reduction to 70% of the natural level is an appropriate 'warning threshold' and that a reduction to 50% of the natural level should be a limit reference point. Similarly, Done and Reichelt (1998) suggest using the ratio of long-lived to short-lived habitat generating organisms (e.g. sponges, corals) to measure disturbance, but they did not provide reference points for what might be regarded as acceptable or unacceptable ratios.

Sainsbury (1991) and Pitcher *et al.* (2000) provide explicit models of the effects of fishery induced habitat modification on the abundance and productivity of dependent species. In these models, the carrying capacity of dependent species is proportional to habitat availability, and fishing can decrease the amount of habitat with high relief (e.g. sponges and corals) while increasing the amount of open sand habitat. The consequence of habitat loss for dependent species is a reduction in their population

carrying capacity, and consequent reduction in both the yield available and the population size for a given level of fishing mortality.

This can be illustrated for the simple logistic model of habitat dependent and exploited species used by Sainsbury (1991):

$$1/B \, dB/dt = r(1 - B/K) - qE = r(1 - B/\Delta\Sigma a_i H_i) - qE$$

where B is the biomass of the exploited and habitat-dependent species, r is the intrinsic rate of natural increase, K is the carrying capacity (equal to the unfished biomass B_0), q is the catchability and E is fishing effort. K can also be expressed as a combination of the area of each habitat type, H_i, the relative density of the unfished population in each habitat type, a_i, and a constant of proportionality, Δ. The a_i represents the preference or utilization of habitat type i and K is $\Delta\Sigma a_i H_i$. For this model the Maximum Sustainable Yield is

$$MSY = rK/4 = rB_0/4 = (r\Delta\Sigma a_i H_i)/4.$$

which occurs at fishing mortality

$$F_{MSY} = r/2.$$

Fishing at F_{MSY} without affecting the habitat reduces the population to biomass

$$B_{MSY} = K/2 = B_0/2 = (\Delta\Sigma a_i H_i)/2.$$

Change in habitat abundances (i.e. changed H_i) does not change F_{MSY}, but it does change the carrying capacity of the target species and consequently both the yield available and the biomass that results from fishing at F_{MSY}.

When fished at F_{MSY}, the target species abundance would be greater than $B_0/2$ if habitats favored by the target species increased, but would be less than $B_0/2$ if favored habitats decreased. So habitat change can result in a target population being overfished even if overfishing of that population has never occurred. That is, the population biomass population could decline to below a reference point such as 0.5 B_{MSY} or 0.3 B_0 as a result of habitat loss and fishing mortality, even if the fishing mortality was always at or below F_{MSY}.

Habitats can change for reasons other than fishing. If they change for reasons other than fishing, this could be considered an externally imposed 'regime change'. In this situation the changed carrying capacity might simply be accepted, and the yield and biomass expectations and reference points would be changed accordingly. But a situation of particular interest is where fishing activities themselves change habitat abundances and reduce the favored habitats of target species or other dependent species. In this situation, managing the impacts of fishing on habitats becomes a part of managing the target species and the effects of fishing on the ecosystem more broadly. So it is not appropriate to simply adjust the reference points to match the depleted habitat condition—the changing baseline syndrome (Pauly 1995, Mace 2004). Rather it is necessary to recognise and manage the fishery impacts on habitats.

Some implications of habitat dependency and the effects of habitat modification on a fisheries target species can be considered from the simplest case where there are only two habitat types; the first that totally unfavorable to the target species (i.e. $a_1 = 0$) and the second that is favorable (i.e. $K = B_0 = \Delta a_2 H_2$). For this model, fishing the habitat dependent species at the exploitation rate giving MSY, but simultaneously reducing

the favorable habitat to half its unfished level, results in depletion of the dependent population to $0.25 B_0$ instead of the intended $0.5 B_0$. And $0.25 B_0$ is in the vicinity of common limit reference points for excessive depletion. To maintain a habitat dependent target species above $0.30 B_0$, while fishing at F_{MSY}, the favorable habitat cannot be reduced below 60% of its unfished area. The habitat area would need to be more than 60% of its unfished area to deliver biomass levels for the habitat dependent target species at a reasonable target, rather than a limit, reference point.

Similar arguments can be made about the depletion of by-catch species that are dependent on the habitats that are reduced by fishing, although specific level of depletion experienced will depend on the level of incidental fishing mortality. There is no reason to expect that the incidental fishing mortality will be equal to F_{MSY} for non-target species. However the incidental fishing mortality must be greater than or equal to zero, and the level of depletion experienced by non-target species will be greater than or equal to the level of depletion of the habitat they are dependent on. So to avoid depletion of non-target species below $0.3B_0$ their habitat cannot be reduced below 0.3 of the unfished area of habitat, and this reduction in habitat area must be less if there is any incidental fishing mortality on the non-target species. Furthermore the habitat-generating organisms (e.g. sponges, corals etc) are by-catch species that are likely to have relatively low productivity, so it would be undesirable to reduce their abundance below 0.3 of their initial abundance.

In summary this model implies that:
- To maintain the populations of habitat-forming species, and species that are dependent on these habitats, above 0.3 of their unfished population sizes then these habitats cannot be reduced below 0.3 of their unfished areal extent.
- If habitat dependent species are subject to an incidental fishing mortality in addition to the loss of habitat then to maintain these species above 0.3 of their unfished population size requires that the habitat be maintained at greater than 0.3 of its unfished areal extent.
- If habitat dependent species are subject to a fishing mortality equal to F_{MSY} for that species, which is a common limit reference point for target species, then to maintain these species above 0.3 of their unfished population size requires that the habitat be maintained at greater than 0.6 of its unfished areal extent.
- If habitat dependent species are subject to a fishing mortality greater than F_{MSY} for that species, which may occur for some by-catch species or less desired species in a multi-species fishery, then to maintain these species above 0.3 of their unfished population size requires that the habitat be maintained at greater than 0.6 of its unfished areal extent.
- For habitat dependent target species any reduction in the required habitat gives a reduction in the maximum sustainable yield and in the yield that is obtained for any given level of fishing effort or fishing mortality.

While this is a simple model, and in particular it does not account for the high level of uncertainty in estimation and system dynamics relating to habitats and habitat dependencies, it nonetheless provides general guidance on limit reference points for habitats in fished ecosystems. Maintaining 0.3 of the areal extent of unfished habitats is a minimum requirement to deliver populations of habitat forming organisms and dependent species that are not depleted to below 0.3 of their unfished abundances. And for dependent species that will be sufficient only in the relatively unusual

situation where there is no incidental fishing mortality. For habitat dependent target species and significantly caught by-catch species, where fishing mortality in the vicinity of F_{MSY} is a reasonable expectation, the habitats should not be reduced below 0.6 of their unfished areas extent. This is a limit reference point in that it derives from avoiding depletion of the target species below 0.3 of its unfished abundance. This does not imply a requirement for 60% of areas to be unfished, but rather that fishing without habitat-modifying impact is required across 60% of the area of relevant habitats.

Overall there is justification in using an approach similar to that applied to account for trophic dependencies when harvesting key prey species, but where the exact nature of the habitat dependencies are not fully understood or explicitly modeled. That is, in the absence of explicit models of the relevant system to provide specific guidance, habitats should not be reduced to less than 75% of their unfished areas.

There are relatively few examples of fishery management frameworks and systems that explicitly address habitat management.

The US framework for federally managed fisheries is the most comprehensive approach yet taken for managing fish habitats. The US *Magnuson-Stevens Act* revision in 1996 requires that Essential Fish Habitats (EFH) and Habitat Areas of Particular Concern (HAPC) be identified and protected (NMFS 2005b). The *Magnuson-Stevenson Act* requires that all agencies work together to protect Essential Fish Habitat, not only those responsible for fisheries or the environment, and so activities such as coastal development, aquaculture and electricity generation are included as necessary in protection measures. EFH is defined as "those waters and substrate necessary to fish for spawning, breeding and feeding, or growth to maturity". EFH must be identified for each federally managed species. HAPC are localized areas within EFHs that are of particular importance to the life cycles of fish species (Dobrzynski and Johnson 2001). They are areas that are "particularly important in ecological function, sensitive to human-induced environmental degradation, stressed by development activities, or rare". HAPC do not automatically receive additional protection but their identification can help prioritize conservation measures. It is required that fishing "minimise adverse effects" or not cause "greater than minimal adverse effect" on EFH, where adverse effect is any impact that "reduces quality and/or quantity of EFH".

The *Magnuson-Stevenson Act* does not provide explicit limits of acceptable change to EFH, but the requirements allow for only minimal and temporary change. In practice this is a stringent requirement that can only be met through use of fishing gears that have little or no effect on habitats during each fishing operation or if the habitats are extremely resilient.

The typical steps taken for identification and implementation of the EHF requirements (NMFS 2005b) are (1) identify the habitats used by the different life history stages for each of the managed species in the fishery (2) produce integrated maps of the extent and types of EFH in the fishery, (3) identify the fishing operations that occur in these habitats (e.g. gear types and fishing intensity) and the likely risk posed to EFH by these operations, and (4) develop management measures to protect EFH risk and incorporate them into the statutory management plan. EFH is usually identified from

scientific surveys of fish and habitats (e.g. DeLong and Collie 2004) but other information, including information from fishers and other stakeholders can also be used. Implementation of the EFH requirements has progressed variously in different US fisheries, with the north Pacific and western US having recently completed amendments to their management plans (see NMFS 2005b). Seabed habitats and fish distributions have been used to identify EFH, and key foraging and breeding areas for numerous species and species groups (see NMFS 1998b). Both qualitative and quantitative assessments have been used to identify effects on EFH that might be "more than minimal and not temporary". This has resulted in a large number of areas being identified and protected from fishing gears that pose a risk to EFH; including in the North Pacific the particularly productive and biodiversity region of the Sitka Pinnacles being closed to all fishing (NPFMC 2002, NMFS 2003, 2007). Large areas have been closed to bottom trawling on the basis of its effects and risk on EFH. In the Aleutian Islands Habitat Conservation Area 284,000 sq. n. miles are closed to bottom trawling while 12,000 sq. n. miles remains open to bottom trawling open. In the Pacific NW fishery more than 130,000 sq. n. miles is closed to trawling, which is about 42% of the EEZ area under the management plan. Areas closed to bottom trawling include many areas where other fishing gears with limited or no habitat impacts are permitted, and include some areas where bottom trawling would be permitted if certain requirements can be met through the design of new trawling technologies or practices.

CCAMLR fisheries operating in some sensitive high latitude areas are not permitted to use bottom trawls because of the impact this method of fishing might have on benthic habitats. For example, the Patagonian toothfish fishery in CCAMLR area 58.4.2 is restricted to long-lining only. The area is divided into five sub-regions and benthic communities are given additional protection by closure of half each sub-region to fishing. In addition to these measures, fishing is prohibited in depths greater than 550m in order to protect deepwater communities that are expected to be slow to recover from fishing impacts.

Comprehensive habitat management measures have also been implemented in the New Zealand Extended Economic Zone (i.e. 12-200 nautical miles from the coastal baseline). In November 2007 32% of this area was closed to bottom trawling and dredging to protect seabed habitats through a system of Benthic Protection Areas (Mfish 2007). The closed areas were selected to be representative of broad habitat types, defined largely on a combination of depth, oceanographic conditions, and geological features. Fishing methods other than bottom trawling and dredging are permitted in Benthic Protection Areas under specific conditions. For example near-bottom trawling that does not impact sea-bed habitats can be conducted subject to prior notification, the presence of observers, and use of vessel and net monitoring equipment.

Hobday *et al.* (2004) provided qualitative criteria for assessing the ecological risk to habitats (Appendix 6, Table A6.4). Their major and severe risk categories are appropriate for limit reference points. These set the spatial scale of impact as up to 25% of the area of fragile habitats (e.g. most biogenic habitats and delicate geological habitats) impacted and the recovery time as being of the order of years to decades.

The FAO Ecosystem Approach to Fisheries (FAO 2003) recognises the importance of habitats to ecosystem functions and fishery production. It emphasises that habitat damage should be prevented, existing damage should be reversed, and where required, habitat should be increased. It also accepts that damage to habitat cannot be prevented completely and that many kinds of fishing will inevitably result in some habitat damage. FAO (2003) recommends that the goal for fisheries management is to set acceptable limits on this damage and to ensure that the fishing does not result in this limit being exceeded. However it does not provide explicit guidance or reference points for acceptable limits of fishing impacts on habitats.

Best practice reference points for habitats

It is recognised that habitats are a critical element of the ecosystems supporting fishery production. But direct management of fishery impacts of habitats is at an early stage of development and implementation, and there is no widely agreed approach to the selection or use of reference points for habitat management. Nevertheless examples of best practice are emerging and simple theoretical guidance is available about the likely limits of habitat modification for sustainable fisheries.

The best practice target reference point for habitat impacts is for no impact on relevant seabed habitats, modified as appropriate to include acceptance of minimal and temporary impacts. This is consistent with the theoretical predictions that yield from a habitat-dependent target species is reduced if the relevant habitat is reduced, and that reduction of the habitat to less than 0.6 of the unfished areal extent could result in the target species becoming excessively depleted. To obtain and maintain high yield from a habitat-dependent target species there should be minimal loss of its favourable habitat. The best practice context for management of habitats is to identify 'critical habitats' for species of interest, and to ensure such habitats are exposed to no more than minimal and temporary impacts. If a wide enough range of species is considered this becomes a 'no net loss' requirement as all habitats are likely to be critical to one species or another. At this time the management of wild capture fisheries does not include the intentional modification of habitats to enhance the production of particular species and/or to reduce the production of others, and 'no net loss' of habitats is consistent with this practice.

Similarly limit reference points for habitat impacts can be inferred from the examples of best practice management and from general principles. The best practice limit reference point for habitat impacts is for relevant habitats to be reduced to no more that 0.3 of the unfished areal extent. This is consistent with avoiding excessive depletion of the habitat-forming organisms themselves and of habitat-dependent species that are not subject to fishing mortality.

While this is the existing best practice limit reference point for habitat impacts there are theoretical grounds for regarding it to be inadequate for protection of habitat-dependent species that are also subject to fishing mortality in the vicinity of F_{MSY} or greater. And some habitat-dependent by-catch species may have low productivity and consequently a low F_{MSY}, so that significant fishing mortality may result from relatively small catches. In cases where the species is exposed to significant fishing mortality in addition to habitat loss a more appropriate limit reference point would be no less than 0.6 of the unfished areal extent of the relevant habitats.

The full spatial range of the habitat type should be included in calculating these proportions of the unfished areal extent of habitats. This could include equivalent habitat beyond the spatial range of the fishery, in protected areas, and un-impacted habitat within the fishing grounds.

Best practice context for use of reference points for habitats

The best practice context for the use of reference points for habitats firstly includes explicit consideration of habitats in management of the fishery, including both articulation of intentions and limits to acceptable change and a preparedness to take management action to achieve the desired outcomes.

A key and basic element of this is knowledge of what habitats are present, what species of relevance to the fishery they support, and the likely impact or change that is caused by fishing. Strategies for habitat protection cannot be developed or evaluated unless the habitats and impacts on them are identified.

Furthermore, the extent and role of habitats are not likely to be known with a high degree of scientific certainty. Management of fishery impacts on habitats very commonly shows two of the key situations that are used to invoke application of the precautionary approach—that is a lack of scientific certainty and a risk of degradation that is effectively irreversible or very slowly reversible. So the best practice context for use of reference points for habitats includes clear application of the precautionary approach (FAO 1995b).

The management measures for habitat protection should include:
- Encouraging the use of fishing gears and practices that cause minimal impact on habitats; and
- Restricting or eliminating the use of fishing gears and practices that cause excessive levels of impact. Use in this context could include the quantity and frequency of use, or the area of use. Management measures could include areas where certain gears and practices are not permitted if that was necessary to prevent unacceptable damage and change to habitats.

While these habitat protection measures could be achieved through fishery management interventions, such protection can also be provided through systems of representative Marine Protected Areas that are designed to achieve broader conservation and biodiversity goals for regional ecosystems. Best practice makes complementary use of both measures.

There is an increasing focus on indicators of ecosystem status and of the effects of fishing (e.g. Fulton *et al.* 2004a, b and 2005). Many of these indicators require comparison with unfished baseline levels. If insufficient data exist from the period prior to fishing to establish the baseline levels then they must determined by comparison with unfished sites, by model predictions of the unfished condition, or some combination of these two approaches. If unfished areas are to be relied on to provide baseline data it is important that they be representative of the system with which they are compared and be large enough to mitigate edge effects. Such reference areas are unlikely to be effective for highly mobile species or in relation to human impacts that are geographically widespread and degrade the ecosystem both inside and outside the reference site (Sainsbury and Sumalia 2003).

5. Food webs
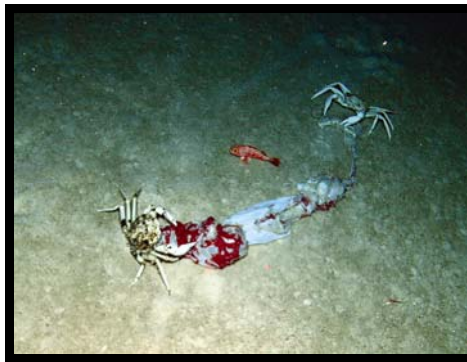
Image courtesy of CSIRO Wealth from Oceans Flagship, Voyage of Discovery

Background

The progression from traditional single species management to integrated management of entire ecosystems is through the intermediate step of managing a target species in such a way as to take ecosystem impacts and processes into account. This is the Ecosystem Approach to Fisheries (FAO 2003). A key element of this approach is taking into account trophic interdependencies such as dependent predators of the target species or required prey of the target species. For many years the need for this was argued on general ecological grounds, and often not accepted as realistic or important. More recently, there has been growing empirical evidence that fishery induced changes and simplifications to trophic interactions have played an important part in the collapse and non-recovery of large fisheries—with the evidence being seen first in enclosed lake systems, then in whole continental shelf systems (Marten 1979, Frank *et al.* 2005).

In addition to the continued focus on the target species, there is growing recognition and acceptance of the need to maintain trophic structures and flows for a wide range of other reasons (e.g. FAO 1995a, 2003), including resilience to natural variability, maintaining genetic, species and community biodiversity, and providing 'ecosystem services' to human needs other than fishing (Hughes 1994, Holling and Meffe 1996, Costanza *et al.* 1997, 1998).

While there is increased recognition that ecosystems have real thresholds and limits that, when exceeded, can result in major and persistent change, and that food web interactions are often an important part of this behavior, there is not an accompanying ability to predict these limits and thresholds. Food webs are 'flexible' and resilient to a certain amount and kind of stress, so that their structure and behavior may at first not change greatly. Only after the threshold is passed does the system begin to deteriorate quickly (Anon. 1991).

Food webs, and ecosystem processes more generally, are inherently non-linear systems. Consequently they are expected to demonstrate the dual properties of being poorly predictable and showing relatively abrupt change when a threshold is passed (Holling and Meffe 1996). But while there is considerable ecological theory and numerous models relating to food webs and their dynamics (e.g. Kitchell 1999, Cury *et. al* 2003, Trites 2003) there are few widely accepted general conclusions or

predictions. Moreover the evidence for change in the trophic level and structure of fished marine ecosystems due to fishing is mixed—it has been clearly demonstrated in some ecosystems (Pinnegar *et al.* 2002, Gascuel *et al.* 2005) but not in others, despite intense fishing and thorough study (Cury *et al.* 2005, Greenstreet and Rogers 2006). Consequently, while there is active development and examination of food web models to guide judgments, the specific management approaches that are used are usually simple and intuitive—and accompanied by the hope that they are sufficiently precautionary.

Food web interactions are often of particular and direct concern for fisheries that target species low in the food web, i.e. at a relatively low trophic level. These species typically have rapid population growth, and if viewed individually as single species in isolation, they can appear to be capable of sustaining a high fishery catch to low population levels. However there are two related problems with this single species view.

- The first is that these species are food for other species at higher trophic levels, and excessive harvest of the prey can have severe consequences for the populations of the higher level predators (Dayton *et al.* 2002, Walters *et al.* 2005). The consequences can include loss of fishery yield from the higher trophic levels and/or depleted populations of charismatic species which may lead to them being listed as protected or endangered.

- Secondly, the fishery is in a dynamic competition with the top predators, and if this is ignored, it can in some circumstances obscure the true dynamics of the prey species and its interaction with the fishery. This can lead to unintentional overfishing and abrupt collapse of the prey species (Murphy 1972, Pauly and Tsukayama 1987) under the combined effects of fishing and predation.

Most single species models used to calculate biological reference points (e.g. yield per recruit models) assume that natural mortality is unchanging. This is not the case for forage species because predation mortality can change quite significantly due to changes in the rest of the ecosystem, particularly if predator populations are also subject to harvesting. Consequently the appropriate biological reference points for prey species become 'moving targets', changing as predation mortality changes through time and in response to ecosystem changes (e.g. Collie and Gislason 2001, Jurado-Molina and Livingston 2002).

There are several examples where changes in trophic interactions as a result of fishing have been well demonstrated.
- In the North Sea, the herring and mackerel stocks collapsed as a result of a combination of fishing and oceanographic changes. These groups are major predators of gadoid eggs and larvae. Reduction in this predation, in combination with the changed environmental conditions, led to a huge increase in the recruitment of gadoid species in what became known as the 'gadoid outburst' (Cushing 1984, Hislop 1996).
- In the North Sea there is good evidence of the effective replacement of sandeel predators by the industrial fishery. For several decades the main predators of sandeels were mackerel, whiting and haddock. As the predator stocks declined

through fishing, the predation mortality decreased and the industrial fishery catch increased. The total mortality (predation plus fishery) has remained virtually the same throughout (ICES 1997). A similar replacement of natural predators by a developing fishery is documented by Murphy (1972) for the Peruvian upwelling fisheries.

- The breeding success of kittiwakes and other seabirds in the Shetland Islands region is highly dependent on the availability of sandeels. In this area, as for the North Sea more generally, the fish predators of sandeels were depleted and subsequently an industrial fishery for sandeels developed. But the size of the industrial fishery in the Shetland Islands region was relatively small, and there was an increase in local sandeel abundance (i.e. increased catch did not fully compensate for the reduced fish predation so the sandeel population increased). Breeding success and population size of the seabirds also increased (Furness 1999, 2002).

- The Barents Sea capelin is a key prey species for cod, marine mammals and seabirds. An industrial fishery for capelin developed in the late 1970s and early 1980s, and as a result of overfishing the stock collapsed to about 1% of its former abundance. Following reduction in the fishery catch to a level considered sustainable, the capelin rapidly recovered. However soon after, the gadoid populations rapidly increased (the 'gadoid outburst') and this resulted in greatly increased predation on the capelin, which caused the stock to collapse again under the combined impacts of fishing and predation. Seabirds were also seriously affected by the capelin collapses—although seabirds take relatively small amounts of capelin in the Barents Sea compared to other predators, they are strongly dependent on access to capelin. During the capelin collapses the seabird populations decreased to about 20% of their former abundance (Gjosaeter 1997, Bogstad and Mehl 1997, Anker-Nilssen *et al.* 1997).

Fishing changes community structure and function (Pauly 2000, Link 2005). The mean trophic level of all organisms in the catch has been shown to decrease with increasing fishing pressure (Pauly *et al.* 1998). Two different changes, which are not mutually exclusive, can cause this decrease—expansion of the fishery to retain or target species that are at successively lower levels in the food web; and reduction of the mean length, age and trophic level of individuals in the harvested populations as fishing mortality is applied. Both of these would be expected to occur in a well managed fishery, but excessive change could also indicate sequential depletion of species at successively lower trophic levels ('fishing down the food chain') or over-exploitation of the target species at any or all trophic levels.

Because healthy and natural ecosystems differ in their trophic structure, there is little merit in comparing mean trophic level in isolation across ecosystems, but it has been shown that reductions over time in the mean trophic level of a particular system may be a good indicator of fishing pressure (Fulton *et al.* 2005). These reductions can have varied effects on the fisheries involved. The low trophic level and highly productive organisms that come to dominate a heavily fished ecosystem might be high value species such as shrimps or molluscs that are desirable from the viewpoint of commercial fishing. For example, the value of fisheries landings in the region of the collapsed Newfoundland cod stocks has increased since that collapse as a result of increased catch of high valued lobster, shrimp and crab which were formerly heavily

predated upon by cod (Hilborn *et al.* 2003). But there are also many cases where the 'replacement' low trophic level species are of lower value than their predators (Parsons 1992, Shiganova and Bulgakova 2000, Pitcher 2001). And in both cases there remain unanswered questions about the effects of such change on the stability and resilience of the associated marine ecosystems, and on the full range of services and functions these ecosystems provide to humans and to the global ecosystem.

Fishing pressure on species high in the food web can cause a cascade of effects on unharvested species lower in the food web (e.g. Frank *et al.* 2005). One mechanism for this is the fishery reducing the abundance of a top predator (a 'keystone predator') that controls the abundance of lower trophic levels (Scott Mills *et al.* 1993), which in turn allows increase in prey populations and release of previously limited competition between the prey species. This can result in major change propagating through the trophic relationships and structure of the ecological community, including competitive elimination of species which previously coexisted in the presence of the keystone predator. These 'trophic cascades' are weakly predictable or unpredictable, and usually the existence of the 'keystone predator' responsible for maintaining the previous community structure is recognised only in retrospect. Also they can be very slow to reverse, even if left undisturbed. Such a cascade of effects is thought to have occurred in the Aleutian Islands, western Alaska (Estes *et al.* 1998, 2003, Estes 1990, Doroff *et al.* 2003) through a series of interactions involving several marine mammal species, sea urchins and kelp. Harvesting of sea otters at the beginning of the 20th century reduced their populations to very low levels. When harvesting stopped their numbers increased, but has declined again in more recent years (Dorroff *et al.* 2003). There are a number of possible reasons for the more recent decline in sea otters, but one is increased predation by orcas due to recorded declines in their more traditional prey species—pinniped and cetacean populations (Estes 1998). Estes also links the declines in sea otter abundance to subsequent increases in their prey species, sea urchins. In turn increased urchin abundance is linked with declines in kelp forests and to major changes in the abundance of habitats and species in the whole coastal ecosystem.

While there are cases of 'keystone' top predators in marine food webs, they appear to be relatively uncommon, and mostly in inter-tidal, relatively shallow sub-tidal ecosystems or enclosed seas. For example sharks do not appear to be keystone predators in pelagic ecosystems (Kitchell *et al.* 2002). However effects and consequences that are very similar to 'trophic cascade' effects, including slow reversibility and multiple stable states, can occur by two other mechanisms that do appear to be common in marine food webs.

- The first is in food webs with a 'wasp-waist' structure. These are food webs in which the abundance of species at mid-level trophic levels (i) are controlled by the food available to them from lower trophic levels and (ii) also control the abundance of higher trophic level species that prey upon them (e.g. Curry *et al.* 2002). Usually there are just a small number of key prey species that play this role in wasp-waist food webs, and usually these species are significant dietary components for many top predators when the prey species are abundant. These prey species act as 'keystone prey' in the ecosystem in that their presence or absence causes major change to the rest of the ecosystem. Natural variability or fluctuations in the productivity and abundance of the key prey species, as for example from changes in oceanographic conditions, can

drive major natural changes in the ecosystem. Fishery induced reductions in the abundance of these key prey species, in concert with natural fluctuations, can similarly drive major changes in wasp-waist food webs. Wasp-waist food webs appear to be common in marine food webs, especially in pelagic and highly productive upwelling ecosystems, and some of the changes observed in cod-herring fisheries can be explained this way (Curry *et al.* 2002, Bakun 2006).

- The second is the diffuse effect of predation low in the food chain when the upper trophic levels are depleted. One of the unusual features of aquatic food webs, compared to most terrestrial food webs, is that many species show trophic role reversals as the individuals of that species get larger. For example large cod eat sprats but sprats eat small cod (i.e. eggs, larvae and very young cod), and the young individuals of a top predator species can be in food competition with prey species eaten by older individuals of that top predator. This introduces the potential for food web interactions to that have unintuitive results – such as a top predator population being out competed by its prey and collapsing (Mangel and Levin, 2005, and contained references). These interactions are incorporated in the models and interpretations of the size spectrum of ecosystems that have been applied in many ecosystems (Pope *et al.*, 1987; Rice and Gislason, 1996; Gislason and Rice 1998; Bianchi *et al.* 2000 and discussion above about reference points for by-catch species). Recently Pope *et al.* (2006) have shown that the diffuse effects of small species can have important effects on the recovery dynamics at the large end of the ecosystem size spectrum. They show that the change in the community size spectrum caused by fishing, that is an elevated intercept due to increased small organisms and an increased slope due to reduction in abundance of large organisms, can be very slow to recover even if fishing is stopped. The mechanism is that a wide range of small species increase in abundance when the larger species are depleted, these small species increase the predation mortality on the young of the depleted large species. The resulting low survival of the young of the large species prevents recovery of the large species and subsequent reduction in the populations of small species. This situation can persist as a locally stable state, presumably until an external event helps re-establish the large species population. This could be a cause of the failure of many depleted fish species to recover quickly, or at rates consistent with the productivity exhibited when their populations were larger, even when fishing on them is stopped (Caddy and Agnew 2003, 2004).

The dynamics of marine food webs are not well known or understood, but there are both empirical observations and theoretical grounds to recognise that trophic interactions are a significant aspect of the ecological response to fishing. They can and have resulted in unintended and highly undesirable outcomes for fisheries and high profile dependent species such as birds and mammals. Once fishing has altered the food web there are several mechanisms that appear common in marine ecosystems that can result in the community not returning to its original state, or being very slow to return, even if fishing is stopped. The two different mechanisms likely to cause these effects in marine ecosystems suggest the need for upper and lower biomass reference points for species low in the food chain. In different circumstances too many or too few animals in the mid and lower trophic levels could cause undesirable consequences - a lower biomass limit would be needed for key prey species in wasp-

waist food webs and an upper limit for aggregate biomass of prey species would be appropriate when significant steepening of the ecosystem size spectrum is expected. There has been some development of reference points for the first of these circumstances, but not the second.

Although the lack of understanding about marine food webs usually prevents fishery manager from taking a holistic approach to the issues, fisheries management can directly address food web concerns in three ways: by maintaining populations of large target species in the ecosystem at levels where they can fulfill their trophic function, by setting catch limits for prey species below those that would be set in a single species context; and by responding to changes seen in indicators of food web structure or function. Examining each of these:

Maintaining populations of large target species
Maintaining the populations of large target species, including the large individuals in those populations, at ecologically functional levels is an aspect of the argument for fisheries to operate on the high biomass side of the production curve. That is to maintain biomass above B_{MSY} – that is for MEY and related reference points to be used as targets (so long as the MEY occurs at a higher biomass than MSY), for F_{MSY} to be used as a limit, and for the use of absolute biomass limits to be used especially in situations where there is wide natural variation in the size of target species populations. These issues are addressed in the setting of target species reference points.

Lower catches for prey species
As outlined above, exploitation of key prey species can have significant consequences for other species (including other target species) throughout the food web.

Walters *et al.* (2005) investigated the effect of fishing a wide range of species at the F_{MSY} that would be correct for each stock in isolation. According to single species management approaches, this should be an acceptable policy. However Walters *et al.* showed that in many ecosystems it results in over-depletion of top predators as they suffer from both fishing and a loss of prey. The reduced prey abundance in effect reduces the productivity or carrying capacity of the top predators, and renders unsustainable the F_{MSY} that would be sustainable if that predator species was harvested but its prey was not.

Jurado-Molina and Livingston (2002) showed a similar result with a multispecies Virtual Population Analysis model (MSVPA) which included predator and prey species. However this pattern was not apparent in all of the ecosystems examined by Walters *et al.* (2005). The details of the food web did matter, highlighting the complicated and weakly predictable nature of ecosystems. The authors recommend that all fisheries management strategies be treated in an adaptive management context to help overcome this lack of predictability, starting with conservative settings supported by monitoring that triggers changed management responses as appropriate.

While it is generally agreed that in most cases the exploitation rate for species low in the food web should be lower than those high in the food web, there is little quantitative or general guidance on just how much lower is sufficient or necessary in a given circumstance.

CCAMLR manage target species differently depending on whether they are designated as "prey" or "predator" species. Prey species—such as Antarctic krill, lanternfish and icefish—are those fed on by a large number of Antarctic predators. The target biomass for predator species is a median of 50% of unfished biomass (a proxy for B_{MSY}) whereas that for prey species is 75% (de la Mare 1996, de la Mare *et al.* 1998, Constable *et al.* 2000; and see discussion above for Target or Commercially Retained Species). The 75% level was chosen as halfway between the supposed MSY level of 50% and the unfished level of 100%.

MSY will not always occur at 50% of the unfished biomass, and so a generalisation of this rule could be the spawning biomass that is halfway between the unfished and the MSY levels. This approach was taken by Sainsbury and Sumalia (2000).

The groundfish fishery in the Bering Sea is managed using Optimum Yields that are generally calculated on a single species basis. However there are several modifications of this approach that are based on food web considerations (NMFS 2003, NPFMC 2002, NMFS 2006b, Anon. 1999):
- Individual species stock assessment reports for Alaskan groundfish species include qualitative evaluation of the trends of predators and prey of that species. Although there are no explicit rules for interpretation and management response, these trends can result in modification of the single species yields if, for example, a dependent predator is showing signs of prey shortage.
- A category of "forage fish" is designated. Commercial trade in these species is not permitted and the maximum aggregate catch from this category cannot exceed 2% of the total fishery catch. This is despite these species being abundant, and is aimed at ensuring the food needs of predators are met.
- Species that are fishery targets but also key prey species of the protected Stella sea lion are managed using more conservative catch decision rules than those used for other target species.

The approach to establishing catch limits in the Icelandic cod, capelin and shrimp fisheries is one of the best and most complete examples of the incorporation of food web interactions in fishery management (Baldursson *et al.*1996, Daníelsson *et al.* 1997, Stefansson *et al.* 1998, and Jakobsson and Stefansson 1998). There are directed fisheries for all three species, and an intention to rebuild the cod stock. Capelin is a major prey item for cod, and shrimp is a major prey item for capelin. So the fisheries for shrimp and capelin must leave sufficient food for rebuilding of the cod, and the extent/rate of rebuilding of the cod will affect the options available to the capelin and shrimp fisheries. This situation was examined through simulation testing of a range of catch decision rules and reference points, using both environmental and economic performance measures, across a range of models of the stocks and trophic interactions. The result was a more conservative approach to the capelin and shrimp catch decision rules than would have been the case if based on single species considerations alone, but relatively little change in the catch decision rule for cod (in that it is designed to maintain the cod biomass at a little above the single species B_{MSY}).

Use of indicators of food web structure or function

Intense fishing pressure reduces the average size and age of animals in the fished populations and often differentially removes top predators. Both of these effects can be expected to reduce the trophic level of the ecosystem, so that as a fishery develops there will be a decrease in mean trophic level (and also in the ratio of high to low trophic level species). An excessive decrease in trophic level can also be indicative of overfishing of individual species and sequential depletion of high trophic level species.

To help distinguish these interpretations, Pauly *et al.* (2000) developed an index that related the change in trophic level to the change in fishing yield. The Fishery Is Balanced (FIB) index is based on the expectation that as the fishery removes or avoids animals high in the food web and increasingly takes animals (the same or different species) lower in the food web, there should be increased fishery yield —and the increase should be about 10% with every trophic link that is removed or avoided between primary production and the fishery catch:

$$FIB = \log(Y_i / TE^{TL_i}) - \log(Y_0 / TE^{TL_0})$$

where Y is catch, TL is the mean trophic level in the catch, TE is transfer efficiency, *i* is the year of interest and 0 is the baseline year.

If in a time series the FIB value falls, it indicates that catches are not increasing as much as would be expected given the shift in fishery catch towards lower trophic level species. Changes in targeting and catch constraints from fishery management could give rise to spurious changes in the FIB index and there is no agreed limit to acceptable change, but the FIB index is potentially a useful indicator of food web functionality. Fulton *et al.* (2005) found that change in the slope of the FIB index plotted against time is a more reliable index than the absolute FIB value itself.

Link (2005) recommends two categories of indicator to measure changes in food webs: changes over time in the biomass/abundance of particular groups; and complexity of the food web. But he recommends that these should be used together to build an overall picture of the system, rather than in isolation, because both have their strengths and weaknesses. His examples and recommendations in relation to these two categories of indicator are:
-   The biomass/abundance indices are the percentage change in the biomass or abundance of groups that usually have characteristic and different roles and levels in the food web; for example, flatfish, pelagic species, all species at trophic level 4 or above (i.e. top predators), piscivores, scavengers, and gelatinous zooplankton. Link (2005) recommends limit reference points that relate to percentage changes in these groups, and are mostly set at a doubling or halving of the indicator from its unfished value. An upper and lower limit is given for most indicators. Neither large increases nor large decreases are desirable, but for different reasons. The reference points suggested by Link (2005) are derived from the Georges Bank-Gulf of Maine ecosystem that has undergone periods of intense fishing pressure, and they are well founded in empirical observation for that system.
-   The food web complexity indicators suggested by Link (2002b, 2005) are the mean number of interactions per species, species richness, and the number of

cycles in the food web. Again the limits are calibrated for desirable and undesirable states that have been observed using these indicators in the Georges Bank-Gulf of Maine ecosystem.

Link (2005) points out that it is unclear how generally applicable the Georges Bank-Gulf of Maine ecosystem experience might be. However it is useful to have well established experience and guidance from at least one ecosystem. Link also points out that the form of the appropriate management response to change in the indicators—for example as a reference point limit is approached—is more complex in this multi-species and ecosystem context than in the single species context where rules can relate desired fishing mortality to current stock size.

The appropriate management responses may be to target change to fishing in particular parts of the ecosystem rather than to 'simply' reduce overall fishing pressure. For example, if the biomass of pelagic fish decreases enough to trigger a reference point, fishing mortality on this group specifically should be reduced. If the biomass of fish at trophic level 4 or above drops then it could require some combination of reducing direct fishing mortality on these trophic levels and reducing the fishing mortality on their prey (i.e. species at trophic level 3). In most cases it is not clear just what management response would be appropriate or effective, or whether there is effective management control through fisheries management alone. There is difficulty in uniquely attributing cause for an observed change in many of these 'system level' indicators, and so there is difficulty in identifying an appropriate and targeted management response.

There is not a well established or widely agreed approach to indicators and reference points for food webs. However the indicators that have been suggested should help increase awareness of the changes that occur in the food webs of fished ecosystems, support improved consideration of actions to manage the effects of fisheries on food webs, and ultimately improve the selection of appropriate reference points and triggered management responses.

Best practice reference points for food webs

Food webs provide the direct basis of fishery production and determine many other attributes of marine ecosystems. Issues of concern in relation to the effect of fisheries on food webs include impairing the size, productivity or resilience of predators (e.g. fish, birds, marine mammals) through removal of their prey, and destabilising or switching food webs and related ecosystem structure to different 'stable states'.

The fishery productivity and sustainable yield of predators can be reduced by simultaneously fishing their prey. There is a considerable body of science that describes mechanisms and examples where food web interactions have led to significant, undesired and unintended outcomes in fisheries. There is also evidence demonstrating that marine food webs can be very flexible and resilient. Marine food webs are complex systems and there is no simple and general summary of their dynamics. However there is no doubt that fisheries can and do have effects on both caught and non-caught species through food web interactions, and consequently on ecosystem structure and function as a whole. The growing understanding of the importance of small pelagic fish in controlling both the abundance of their larger

predators and of their smaller prey in productive pelagic ecosystems means that fisheries for small pelagic species are of particular concern in this regard.

Best practice in the management of food web interactions is not well developed. However a minimalist requirement in the management system is explicit recognition of the potential for food web interactions and an ability to modify fishing controls in order to manage significant food web interactions that are considered likely. There are two broad approaches to relevant reference points and management responses—one concerned with the food web as a whole, and the other that focuses on identified key elements or connections in the food web.

Food web as a whole
Approaches that address the food web as a whole have generated a large number of potential indicators to measure change, mostly derived from food web models. While this is a very active field and advancement is likely, it has not yet provided demonstrated best practice through the use of these indicators and associated reference points in management decision-making. Current thinking is that a suite of indicators and reference points may be needed and that the comparison with unfished reference sites may be a necessary part of that to provide timely and reliable interpretations. One potentially useful indicator is the FIB index that compares actual catches to theoretical catches as a fishery changes the trophic level it is harvesting—a departure between these two could indicate food web disruption but there is currently no reference point for this departure.

Key elements of the food web
Typically this approach focuses on key prey species for predators of particular concern (e.g. dependent fishery target species, birds and marine mammals). Best practice involves explicit nomination of significant prey species or forage species in fisheries management plans, and having specific management conditions and reference points for them.

In some cases the management objectives totally preclude the development of commercial fisheries on species that are designated as significant prey species, either as a permanent limitation or as a precautionary measure while better understanding is sought. In such cases best practice sets the permitted by-catch levels and trip limits for designated prey species that are very low compared to likely species productivity, and that are consistent with only non-commercial and incidental take. The intention is to discourage targeted commercial fishing.

When targeted commercial fishing of designated significant prey species is permitted, best practice reference points are selected so as to maintain the productivity and ecological viability of predators. In a few very well studied situations in relatively simple ecosystems it has been possible to explicitly model these interactions and estimate the appropriate reference points and management controls. While this represents best practice it will not be feasible in many situations, and it remains unclear whether the reference points derived from these cases could or should be generalised.

Best practice in the absence of appropriate trophic models is a biomass target reference point for nominated key prey species that is no less than the mid-point

between the unfished biomass and B_{MSY} for that species. The justification is that in an unfished state, the whole unfished biomass is available to direct predators and the food web more generally, and that both experience and theory suggest that reducing the biomass of prey species to the level that gives MSY can cause undesirable impacts on direct predator species and elsewhere in the food web. The mid-point of the unfished level and the B_{MSY} level biomasses is an arbitrary balance between meeting fishery and food web needs. It can be modified in light of more specific understanding, but it is the default approach that is currently best practice. In the absence of sufficient information to estimate B_{MSY}, current best practice is to assume a logistic production model for which B_{MSY} is at 50% of the unfished level, so that the best practice limit reference point for designated key prey species is reduction to 75% of the unfished biomass. The limit reference point for key prey species should be no less than that which would be applied to a target species higher in the food chain, and best practice for this is B_{MSY}. But in any event the limit should at least match the practice of CCAMLR where the requirement is that there is no more than 0.1 probability of being below 20% of the median unfished biomass.

Best practice context for use of reference points for food webs

The best practice context for the use of reference points for food webs shares many features with that for habitats, as the approach to both is at a relatively early stage of development and is characterised by a high level of scientific uncertainty. A basic element is that there is explicit consideration of food webs and trophic dependencies in management of the fishery, including both articulation of intentions and limits to acceptable change, and a preparedness to take management action to achieve the desired outcomes.

The best practice context includes explicit recognition that species high in the food web need to be treated differently from those low in the food web, with the reference points and management of the lower trophic level species intended to deliver higher biomass and lower exploitation rates than would be the case if the species were considered in isolation.

The catch should be categorised into at least coarse trophic levels (e.g. key forage species, top predators) with monitoring, indicators and appropriate intended outcomes and reference points defined for each category. Targeted fisheries on key forage species should be developed cautiously, incrementally and adaptively, if they are permitted at all. Exploitation rates should be low until it can be established that it is safe to increase them.

A key and basic element of this is knowledge of what food web linkages are present, what species of relevance to the fishery they support, and the likely impact or change that is caused by fishing. Food web protection cannot occur unless the food webs and impacts on them are recognised, even if that recognition contains uncertainty.

Because food webs occur in a regional ecosystem context it is possible (even likely) that several different fisheries—conducted, researched and managed more or less separately—all impinge on the same food web. Best practice would ensure that these interactions are recognised and that any necessary management responses include all relevant fisheries.

There is a high degree of scientific uncertainty about the dynamics and function of food webs. And so the best practice context for use of reference points for food webs includes clear application of the precautionary approach (FAO 1995b). As with management of habitats, management of fishery impacts on food webs combines a lack of scientific certainty with a risk of degradation that may be effectively irreversible on all but very long time scales.

Conclusions

Reference points are a key operational element in fisheries management. They provide the explicit and measurable targets and limits for management. Consequently they provide a strong basis for ongoing adaptive management, the detection and correction of undesirable trends so as to achieve intended management outcomes, and for assessing management performance. Conversely if reference points and associated indicators cannot be specified then management and performance assessment is likely to be compromised. In many different management environments it is held that 'if you are not measuring it you are not managing it'. And in that context the development of reference points to support management of the ecosystem effects of fishing is a key part of the transition from single species fishery management to the ecosystem approach to fisheries (FAO 2003) or ecosystem based fishery management (e.g. Pikitch *et al.* 2004).

'Best practice' reference points are considered here for five elements of environmental management that are central to modern fishery management – the target species; by-catch species; protected, endangered or threatened species; habitats; and food webs.

The 'best practice' concept is based on the best practice that has been demonstrated through use, and recognises that views of what is 'best' will continuously improve with experience. Fisheries research and management has a long history and focus on target species. The broader elements of environmental management have come to prominence in fishery management only in the last few decades. This means that best practice reference points are much better developed and tested for target species than for the other elements of the ecosystem. Greater change and evolution of what is regarded as best practice is expected in future for these other elements than is expected for the target species. In particular, best practice with respect to habitats, food webs and overall ecosystem composition (biodiversity at the genetic, species and community levels) is at an early stage of development in current fishery management practice.

Nevertheless, there is sufficient experience with the management of all of the examined ecosystem elements to identify current best practice and best practice reference points in practical use. In the case of habitat management current best practice is likely to be insufficient to provide high and sustainable yields from habitat dependent target species, and a suggested alternative reference points is provided based on simple models of habitat dependent species. And current best practice reference points for food-webs focus on protecting key prey species rather than the wood-web system as a whole.

While best practice is expected to evolve with experience the currently available best practice reference points allow a start to be made in the practical implementation of ecosystem based approaches to fishery management.

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Tables

Table 1. Limit and target reference points for retained species fishing mortality in fisheries identified as demonstrating good practice. The Marine Stewardship Council (MSC) 80 Guidepost is included because it is intended to reflect best practice for the fishery type being assessed.

Fishery	Limit RP	Target RP	References and comments
US Gulf of Alaska groundfish	Maximum of F_{MSY} (MAY interpretation). F_{MSY} interpreted as in the appropriate Tier of Appendix 3. For most stocks (Tier 3) F_{MSY} is $F_{35\%}$ and this is the limit reference point for biomass above $B_{40\%}$. The limit reference point decreases linearly for biomass below $B_{40\%}$ (i.e. slightly above the implied B_{MSY}) and reaches zero at $0.05B_{40\%}$.	F_{oy} as interpreted in the appropriate Tier of Appendix 3. For most stocks (Tier 3) the maximum is $F_{40\%}$, decreasing linearly for biomass below $B_{40\%}$ and reaching zero at $0.05B_{40\%}$. Additional constraints on target F are that (i) for some key prey species (pollock, Atka mackerel and Pacific cod) F=0 for biomass below $B_{20\%}$, and (ii) stock rebuilding triggered below $0.5B_{MSY}$ (i.e. $B_{17..5\%}$).	Annual catch limits set for each target species. Target stocks treated in appropriate Tier of Appendix 3. One species at Tier 1, 10 in Tier 3, 8 in Tier 5, and 2 in tier 6. NMFS (2003)
US west coast groundfish	Maximum of F_{MSY} (MAY interpretation) for each species. F_{MSY} interpreted to be $F_{40\%}$ (e.g. whiting), $F_{45\%}$ (e.g. sablefish) or $F_{50\%}$ (e.g. rockfish).	Maximum is $F_{40\%}$ (e.g. whiting), $F_{45\%}$ (e.g. sablefish) or $F_{50\%}$ (e.g. rockfish), decreasing linearly for biomass below $0.4 B_0$ and reaching zero at $0.1B_0$.	Target stocks treated similar to Tier 3 of Appendix 3 but with $F_{x\%}$ modified to account for differences in stock productivity. Biomass expressed as fraction of B_0 rather than $B_{x\%}$.
US northeast	F_{MSY} which is	$0.8 F_{max}$	The target F for the stock

scallops	interpreted to be F_{max}		is currently achieved by averaging across areas that are open to fishing (local F about twice target) and closed to fishing (local F=0), with plans to achieve target in fished areas (see Framework 15, Amendment 10 of www.nefmc.org).
Icelandic cod	No specified limit but the catch decision rule has a low (<5%) chance of giving an F>F_{MSY}	An annual catch of 0.25 of current exploitable biomass. This corresponds to F=1.6M, F=1.3$F_{0.1}$ and F<F_{MSY}.	Developed from extensive simulation testing of the catch decision rule to ensure a low chance of stock collapse and achieving biomass greater than B_{MSY} with stock recruitment relationships as observed. See Baldursson (1996) and Danielsson *et al.*(1997)
Australian Federal Harvest strategy policy	Less than or equal to F_{MSY} (i.e F≤F_{MSY})	Less than or equal to the fishing mortality giving the maximum economic yield (i.e. F≤F_{MEY}).	DAFF (2007)
Pacific halibut – International Pacific Halibut Commission and MSC 80 Guidepost	No specific fishing mortality limit but the catch decision rule has a low chance of giving an F greater than F_{MSY}	A maximum annual catch of 0.20 of current exploitable biomass. This corresponds to about $F_{40\%}$, 0.5F_{MSY}, and F=1.6M. It provides about 75% of the MSY catch. A decision rule applies the maximum fishing mortality if the current biomass is more than 1.5 of the biomass limit reference point, and reduces	The catch decision rule was developed from extensive simulation testing to ensure near-zero probability of spawning stock biomass being reduced below the limit reference point for biomass, which is the lowest recorded level, and to provide close to MSY catches. This limit biomass level was not associated with reduced recruitment. See www.iphc.washington.edu/ Clark and Hare (2002), and www.MSC.org.

		linearly to zero at the limit reference point.	
ICES – general guidelines rather than specific fisheries	Precautionary F_{pa} such that with the assessment methodology there is a low chance that the realised F would reduce average recruitment if applied in the long term.	No explicit fishing mortality target.	Approach and methodology as in Appendix 5.
Western Australian rock lobster - MSC 80 Guidepost	Although not explicitly stated the effect of management measures is approximately F_{msy} (MAY interpretation).	No explicit target for fishing mortality; the harvesting strategy accepted as reliably avoiding the biomass limit reference point.	www.MSC.org
New Zealand Hoki - MSC 80 Guidepost	No explicit limit for fishing mortality; biomass limit of $0.2B_0 = \frac{1}{2} B_{MSY}$ implied by application.	No explicit target for fishing mortality; biomass target of $B_{MSY} = 0.4B_0$ implied by application.	www.MSC.org
Alaskan Pollock - MSC 80 Guidepost	F_{MSY}	Fishing mortality as specified in Tier 1 (Appendix 3), modified to have zero F at biomass below $B_{20\%}$ rather than below $0.05B_{MSY}$, accepted as adequate to avoid fishing mortality and biomass limits. Tiers 4, 5 & 6 not accepted as adequate because they do not reduce F if	www.MSC.org

		biomass or F approaches limits.	
South Georgia toothfish - MSC 80 Guidepost and CCAMLR	No explicit limit for fishing mortality, but a limit is implied by the MCY that meets biomass targets and limits. In general the fishing mortality implied by MCY is less than the F_{MSY} given by an MAY interpretation. In addition the biomass limits will result in this being more conservative than an unconstrained MCY for usual stock-recruitment relationships.	No explicit target for fishing mortality.	A long-term maximum constant yield (MCY) is recalculated and applied as a catch limit each year, updated with estimates of recent recruitment and age structure. www.MSC.org
Heard and MacDonald Is icefish – CCAMLR and MSC 80 guidepost	The fishing mortality that results in the 5th percentile of the fished biomass of the yearclass being no less than 75% of the 5th percentile in the absence of fishing.	No explicit target but a level is implied by the biomass criteria combined with uncertainty and variability.	Icefish are a designated key prey species and have highly variable recruitment and relatively few coexisting year-classes. This approach establishes a fishing mortality and catch limit for each year-class. www.MSC.org
CCAMLR Southern Ocean krill - CCAMLR	The fishing mortality implied by the MCY that meets the biomass limits for this fishery (Table 2). In general the fishing mortality implied by MCY is always less than the F_{MSY} given by an MAY	No explicit target but a level is implied by the biomass criteria combined with uncertainty and variability.	This gives a maximum constant yield (MCY) level for a designated key prey species (de la Mare 1996, Constable and de la Mare 1996, Constable *et al.* 2000).

	interpretation.		
Pacific cod – MSC 80 guidepost	As per tier 3b in Appendix 3. The limit reference point is $F_{35\%}$ for stocks above $B_{40\%}$, decreasing linearly for stock biomass below $B_{40\%}$ and becoming zero at $0.05\ B_{40\%}$	As per tier 3b in Appendix 3. The target is $F_{40\%}$ for stocks that are above $B_{40\%}$, decreasing linearly for stock biomass below $B_{40\%}$ and becoming zero at $0.05\ B_{40\%}$	

Table 2. Limit and target reference points for retained species

biomass, usually spawning biomass, in fisheries identified as demonstrating very good practice. The Marine Stewardship Council (MSC) 80 Guidepost is included because it is intended to reflect best practice for the fishery type being assessed.

Fishery	Limit RP	Target RP	References and comments
US Gulf of Alaska groundfish	The limit is 0.5 B_{MSY}. One stock is treated as Tier 1 where B_{MSY} is directly estimated. Most stocks are treated as Tier 3 in which $F_{35\%}$ is the proxy for F_{MSY} and so $B_{35\%}$ is the proxy for B_{MSY}. Assuming constant average recruitment this is equivalent to a limit reference point of $0.175B_0$. For Tier 5 the B_{MSY} proxy is 0.75M. B_{MSY} cannot be calculated for Tier 6.	None specified. For most stocks (Tier 3) the maximum fishing mortality decreases linearly from a biomass slightly above B_{MSY} and reaches zero at $0.05B_{MSY}$. This is intended to maintain stocks above the limit reference point and near B_{MSY}. The fishing mortality is zero if biomass is below the limit reference point.	Annual catch limits set for each target species. Target stocks treated in appropriate Tier of Appendix 3. One species at Tier 1, 10 in Tier 3, 8 in Tier 5, and 2 in tier 6. NMFS (2003).
US west coast groundfish	$0.25B_0$ which is slightly higher than $0.5B_{MSY}$ (B_{MSY} interpreted to be $0.4B_0$)	None specified. The maximum fishing mortality decreases linearly from a biomass of $0.4B_0$ and reaches zero at $0.1B_0$. This is intended to maintain stocks above the limit reference point and near B_{MSY}. The fishing mortality is zero if biomass is below the limit reference point.	Target stocks treated similar to Tier 3 of Appendix 3 but with $F_{x\%}$ modified to account for different for stock productivity.
US northeast scallops	$0.5B_{MSY}$ where B_{max} is the proxy for B_{MSY}.	None specified	
Icelandic cod	Very low probability that the	The target spawning stock	

	spawning stock biomass would be below both B_{lim} (approx $0.3B_{MSY}$) and B_{rep} (approx. $0.55B_{MSY}$).	biomass is $0.8B_{MSY}$. It is about equal to the spawning biomass that gives maximum recruitment for a Ricker stock-recruitment curve, and is $1.5B_{rep}$ and about $3B_{lim}$.	
Australian Federal Harvest strategy policy	Greater than or equal to ½ B_{MSY}	Greater than or equal to the biomass at the maximum economic yield (i.e. $B \geq B_{MEY}$).	DAFF (2007)
Pacific halibut - IPHC and MSC 80 Guidepost	Limit biomass for female spawning stock is the lowest historically recorded level in each reporting area. The lowest past spawning biomass occurred during the 1970s and was not associated with reduced recruitment - this is effectively B_{lim}.	The 20% harvest rate results in a spawning biomass that is about $0.45B_0$ and $1.5B_{MSY}$, so these could be regarded as targets.	Recruitment is strongly affected by decadal and longer fluctuations in the environment. The catch harvest rule was developed from extensive simulation testing to ensure near-zero probability of spawning stock biomass being reduced below the lowest level recorded and to provide close to MSY catches on average. Clark and Hare (2002). www.MSC.org
ICES – general guidelines rather than specific fisheries	B_{pa} If the estimated B is less than B_{pa} there is a low chance that the actual B is be less than B_{lim} (the biomass below which recruitment is reduced on average).	None defined.	Approach and methodology as in Appendix 5. Genetic issues have been actively considered (e.g. ICES (2002) and while no genetically based reference points have been identified B_{lim} was

			seen as providing some protection against generic loss. Appendix 5.
Western Australian rock lobster – MSC 80 Guidepost	Biomass above levels at which major decline in recruitment seen or expected. $0.25B_0$ (measured as egg production) accepted as both meeting this and exceeding B_{MSY}.	None specified but $0.3B_0$ (measured as egg production) accepted as adequate in context of decision rules.	www.MSC.org
New Zealand Hoki - MSC 80 Guidepost	Biomass above levels for which major decline in recruitment seen or expected. 90% probability of being above $0.2 B_0$ used in performance assessment as a proxy limit.	$0.4B_0$ used in performance assessment as a level that the target is at or above. $0.4B_0$ is also treated as a proxy for B_{MSY}, so that B_{MSY} is a target reference point.	Catch levels determined that meet these requirements on the basis of 5y forward projections that include several sources of uncertainty.
Alaskan Pollock - MSC 80 Guidepost	B_{MSY}, with a probability of 70% of being above this limit taking account natural variability. B_{MSY} is approximately $B_{35\%}$.	None specified but the harvesting strategy is required to avoid the biomass limit reference point. Tier 1 approach, modified to have F reduce to zero at $B_{20\%}$ rather than below $0.05B_{MSY}$, accepted as adequate.	www.MSC.org
Alaskan Salmon - MSC 80 Guidepost	Escapement 20% of that demonstrated to give high and sustainable long-term catches.	Escapement demonstrated to give high and sustainable long-term catches.	www.MSC.org
South Georgia South Georgia toothfish - MSC 80 Guidepost and CCAMLR	The 10th percentile of the estimated exploitable biomass is no less than 20% of the median level that would result in the absence of	The median of the estimated exploitable biomass distribution is greater than or equal to 50% of the median unfished	Both the target and limits must be met in selection of the long-term maximum constant yield (MCY). This is recalculated each

	fishing. For constant recruitment this biomass level is equivalent to $B_{20\%}$.	distribution. For constant recruitment this biomass level is equivalent to $B_{50\%}$.	year, updated with estimates of recent recruitment and age structure. Constable and de la Mare (1996), Constable *et al.* (2000). www.MSC.org
Heard and MacDoald Is icefish – CCAMLR and MSC 80 guidepost	The 5th percentile of the fished biomass of each yearclass not less than 75% of the 5th percentile in the absence of fishing.	No specific target requirement but the MCY that meets the limit requirements will result in a fished biomass distribution with a median greater than or equal to 75% of the median unfished level.	Icefish are a CCAMLR designated key prey species with highly variable recruitment and few year-classes. This approach establishes a fishing mortality and catch limit for each year-class. www.MSC.org
CCAMLR Southern Ocean krill - CCAMLR	The 10th percentile of the fished biomass is no less than 20% of the median level that would result in the absence of fishing. For constant recruitment (i.e. infinitely high steepness) this biomass level is equivalent to $B_{20\%}$.	The median of the fished biomass distribution is greater than or equal to 50% of the median unfished distribution. For constant recruitment (i.e. infinitely high steepness) this biomass level is equivalent to $B_{50\%}$.	This gives a maximum constant yield (MCY) level for a CCAMLR designated key prey species. The high escapement or biomass limit is designed to maintain the integrity of predator-prey dependencies and processes in the ecosystem. de la Mare (1996), Constable and de la Mare (1996), Constable *et al.*(2000).
Pacific cod – MSC 80 guidepost	As per tier 3 in Appendix 3. $0.5B_{MSY}$ is the limit reference point, with $B_{35\%}$ being the proxy for B_{MSY}. Assuming constant average recruitment	As per tier 3 in Appendix 3. The target biomass is at or above $B_{40\%}$.	As per tier 3 in Appendix 3. Catch decision rule that linearly decreases catch if biomass is below $B_{40\%}$, and catch is zero if biomass is below

	this is equivalent to a limit reference point of $0.175B_0$.		$0.05\ B_{40\%}$. www.MSC.org

Table 3. Best practice target and limit reference points for spawning biomass and fishing mortality of target and retained species high in the food-chain.

	Limit RP	Target RP
Fishing mortality	F_{MSY} This can be estimated directly or a proxy can be justified and used. $F_{50\%}$ is a reasonable default proxy for situations in which 'steepness' in the stock recruitment curve is unknown, and use of higher values of fishing mortality (i.e. lower % SPR) requires specific justification.	Fishing mortality giving optimum yield with (i) a high probability (i.e. greater than 90%) of avoiding the fishing mortality and biomass limit reference points over an extended period (at least 2 generation times) (ii) achieve the target biomass.
Biomass	The greater of B_{lim}, $0.3B_{unfished}$ and the level from which rebuilding could be achieved without fishing in a period not greater than a fish generation time plus 10y. B_0 can be used as a constant proxy for $B_{unfished}$ for stocks that do not show large natural fluctuations or 'regime shifts'. For stocks that naturally show large fluctuations two limit reference points, related to recent and long-term productivity, must both be met. These limits are 0.3 $B_{unfished}$ and 20% of the median long-term B.	Set consistent with optimum yield and a median biomass of at least B_{MSY}. For important prey species a higher level of median biomass is required, such as midway between B_{MSY} and the unfished biomass.

Table 4. Target and limit reference points for by-catch. There is not a desired level to by-catch, where by-catch is the unintended and incidental catch that is discarded, and so target reference points do no apply to by-catch. The by-catch policies for most fisheries have objectives to minimize by-catch and by-catch mortality as much as possible, practicable or feasible. This could be regarded as implying a low acceptable level of by-catch at any point in time. However explicit targets or limits are not identified in most management arrangements. Instead there is a focus on measurement of impacts and subsequent decision-making processes to address any unacceptable impacts that emerge, rather than specifying reference points and management responses in advance. The Marine Stewardship Council (MSC) 80 Guidepost is included because it is intended to reflect best practice for the fishery type being assessed.

Fishery	Limit RP	References and comments
US Gulf of Alaska groundfish	- Catch limit, based on insignificant impact, for large number of species that are targets in other fisheries. - Catch limit on forage fish of 2% of retained catch. - Catch limit on 'other species', of ecological and potential commercial importance, of 5% of retained catch - Catch limit on short-tailed albatross under Threatened and protected species legislation	Zero target implied by National Standards Guideline to minimize by-catch (NMFS (1998a). By-catch of all species is monitored by observers. By-catch of target species accounted for in catch limits, and compulsory retention of Pollock, cod and some rockfish. Catch limits for species targeted by other fisheries. Catch limits to prevent targeting of designated forage fish. Precautionary catch limit on pooled 'other species' set using the lower tiers of assessment in Appendix 5. Extensive use of time-area closures to manage by-catch. Fishing areas are closed when by-catch limits are reached.
US west coast groundfish	- Catch limits on overfished target species determined to meet US National Standard Guidelines for recovery target and timeframe. - By-catch of target species in other fisheries prohibited or limited to a level having insignificant impact.	By-catch is limited for species that are recovering target species or targets in other fisheries, and in these cases fishing areas are closed when by-catch limits are reached. However other species are not explicitly limited, although for some a review of circumstances is triggered if the by-catch abruptly changes.
Pacific halibut - IPHC and MSC 80 Guidepost	- Catch limits based on achieving 'insignificant impact', for species that are targets in other fisheries.	Criteria (www.MSC.org) - Recording and verification of by-catch. - Trends in abundance of main by-catch species known, and within

	- Trends in most by-catch species assessed and reported regularly, although without explicit reference points.	acceptable limits.
Western Australian rock lobster – MSC 80 Guidepost	No explicit limit reference points.	Criteria (www.MSC.org) - Amount and type of by-catch is measured and verified. - Ecological risk assessment conducted, and at least in part based on comparison of fished and unfished areas, and shows no unacceptable impact. - Attempts being made to identify explicit limits of change. - The fishery management plan includes objectives relating to by-catch where issues have been identified to pose a risk, and mechanisms to adjust fishery operations if adverse impacts are detected.
New Zealand Hoki - MSC 80 Guidepost	No explicit limit reference points. Information on trends or stock assessment for most by-catch species, but these are interpreted on a case-by-case basis without consistent or explicit reference points.	Criteria (www.MSC.org) - Amount and type of by-catch is measured. - Ecological risk assessment conducted, where possible based on comparison of fished and unfished areas. - No unacceptable impacts on by-catch species. - The fishery management plan includes objectives relating to species diversity where issues have been identified to pose a risk, and mechanisms to adjust fishery operations if adverse impacts are detected.
Alaskan Pollock - MSC 80 Guidepost	Discards of approx. 1% of total catch considered exemplary. Trends monitored, including by fishery independent surveys. Most by-catch species assessed reported regularly, although without explicit reference points. In practice acceptable	Criteria (www.MSC.org) - By-catch measured. - Impacts of by-catch on communities, including species abundance and composition, have been assessed. - Effects of discards and waste discharge have been assessed. - Species identified as being affected by fishing are monitored for population size and density. - Models or hypotheses of the effects

	limits are usually when the catch is a low (usually less than 5%) percentage of estimated population abundance or below the overfishing catch limit under tiers 5 or 6 of Appendix 3.	of fishing on populations and communities developed that are consistent with historical information and used to guide interpretations and management decisions to reduce fishing effects.
Alaskan Salmon - MSC 80 Guidepost		Criteria (www.MSC.org) - By-catch species are known - Quantitative measures of by-catch of main species - A risk assessment is conducted to evaluate the effects of fishing on the species diversity and relative abundance.
South Georgia toothfish - MSC 80 Guidepost and CCAMLR	Catch limits set for on by-catch species or species groups. Limits set by precautionary assessment, using conservative assumptions and the same reference points used for designated target species, or a default value of 50t.	Criteria (www.MSC.org) - Quantitative information on the amount of by-catch. - Information to allow verification and interpretation of information from fishery. - Information available on distribution and ecology of main by-catch species, sufficient to understand main fishery impacts and their reversibility. - Assessment of the significance and risk of impacts on ecosystem structure, and no unacceptable impacts. - The impacts on biological diversity and productivity have been considered and no unacceptable impacts have been found. - Acceptable levels of impact are set for key aspects of the environment, and are estimated and regularly reviewed. - Management objectives are set to detect and reduce impacts.
Southern Ocean icefish – CCAMLR and MSC 80 guidepost	- Catch limits on all by-catch species or species groups. Limits set by precautionary assessment, using conservative assumptions and the same reference points as used for designated	Criteria (www.MSC.org) -By-catch levels reported and verified. - Catch limits set for all by-catch species or species groups. The intention is to preclude development of targeting until there is sufficient information to allow development of a fishery development plan,

	target species, or a default value of 50t which is regarded as both ecologically safe and sufficient to preclude targeted fishing. - Specific risk assessment conducted on species of concern because of low productivity (sleeper sharks), which determined very low risk from current catches.	supported by a preliminary and precautionary catch limit, and ultimately a full stock assessment. - In addition to overall catch limits there are lower 'fine space-scale' by-catch limits which if reached require the vessel to 'move on' and not return to the initial location of relatively high by-catch for several days. Constable *et al.*(2000).
Pacific cod – MSC 80 guidepost	No explicit reference points. Trends monitored, including by fishery independent surveys. Most by-catch species assessed reported regularly, although without explicit reference points. In practice acceptable limits are usually when the catch is a low (usually less than 5%) percentage of estimated population abundance or below the overfishing catch limit under tiers 5 or 6 of Appendix 3.	Criteria (www.MSC.org) - by-catch monitored - impacts regularly assessed and within acceptable limits

Table 5. Target and limit reference points for threatened, endangered or protected species. The objective of policy and law relating to threatened or protected species is usually to minimize deaths and allow populations to rebuild or be maintained at high levels. In this context the target catch is a zero catch. And the limit reference point relates to the maximum deaths that will allow an acceptable rate of population recovery or level of protection. The Marine Stewardship Council (MSC) 80 Guidepost is included because it is intended to reflect best practice for the fishery type being assessed.

Fishery	Limit RP	References and comments
US Gulf of Alaska groundfish	PBR on direct removals and fishery caused mortality	
Western Australian rock lobster – MSC 80 Guidepost	No explicit limit, but the acceptable level of by-catch for sea-lions is zero.	Criteria (www.MSC.org) - Presence of threatened or protected species is known, there are verified measures of the amount caught, and there are assessments of the fishery impacts. - The fishery is managed to meet the requirements of recovery plans for relevant threatened or protected species. - The fishery management plan includes objectives relating to threatened or endangered species where issues have been identified to pose a risk, mechanisms to adjust fishery operations if adverse impacts are in place and have been acted upon if triggered.
New Zealand Hoki - MSC 80 Guidepost	No explicit limit. Desired reductions or maximum catch levels developed on a negotiated and ad hoc basis, without explicit reference points or methodology.	Criteria (www.MSC.org) - By-catch and incidental mortality is measured and verified. - There has been an assessment of risk to threatened and protected species based, where possible, on information from fished and unfished areas. - No unacceptable impacts on threatened or protected species demonstrated. - If unacceptable impacts are identified adequate corrective actions are being taken. - The fishery management plan includes objectives relating to threatened ore endangered species where issues have been identified

		to pose a risk, and mechanisms to adjust fishery operations if adverse impacts are detected.
Alaskan Pollock - MSC 80 Guidepost	Direct mortality less than PBR.	Criteria (www.MSC.org) - Presence and distribution of threatened or endangered species in the area of the fishery known. - Fishery independent monitoring of interactions, mortality and by-catch of threatened or endangered species. - Knowledge of the functional relationships between population dynamics of the threatened or endangered species and additional mortality, foraging success and prey abundance/spatial distribution. - Risk assessment of fishery impacts of the genetic, species and populations of all threatened or protected species. - Demonstrated responsiveness to risk assessment through attempts to minimise impacts. - Strategy to manage impacts on and reduce risk to threatened or protected species. - Where uncertainty is high management to restrain impacts is precautionary. - Direct impacts are below levels that harm population size. - Indirect impacts (foodchain etc) have been assessed and are less than levels that harm population size.
Alaskan Salmon - MSC 80 Guidepost	PBR used to establish take limit for threatened and protected species.	Criteria (www.MSC.org) - catch of threatened and protected species within permitted take level.
Pacific halibut - IPHC and MSC 80 Guidepost	PBR used to establish take limit for threatened and protected species (short-tailed albatross).	Bird by-catch mitigation devices required to be used on all vessels. Monitoring and assessment of population and impacts.
South Georgia toothfish - MSC 80 Guidepost and CCAMLR	No explicit reference point, but an objective of no significant reduction in population size or recovery rate. Maximum	Criteria (www.MSC.org) - Threatened and protected species in fishing area recognized. - Quantitative estimates of direct fishery interactions (e.g. deaths,

	allowable catch limits determined on an intermittent and case-by-case basis by the relevant groups and process in CCAMLR and the individual listing countries, without a formal or uniform protocol.	injuries, disturbance, provisioning etc) and the consequences of these impacts. - Direct impacts are within acceptable levels. - Agreed and enacted mitigation responses.
and Heard and MacDonald Is icefish – CCAMLR and MSC 80 guidepost		
Pacific cod – MSC 80 guidepost	PBR on direct removals and fishery caused mortality	Criteria (www.MSC.org) - Mortality levels measured - mortality levels within acceptable limits

Table 6. Target and limit reference points for habitats. While the existence of habitat is accepted to be an essential requirement for the persistence of fisheries, and fish populations and ecosystems more generally, there are very few examples of relevant reference points being developed and applied. The emphasis in management arrangements, where it exists at all, is on information collection rather than setting targets, limits or triggers for specified management response. The Marine Stewardship Council (MSC) 80 Guidepost is included because it is intended to reflect best practice for the fishery type being assessed.

Fishery	Limit RP	References and comments
Western Australian rock lobster – MSC 80 Guidepost	No explicit reference point	Criteria (www.MSC.org) - Knowledge of types of major habitats and their spatial distribution. - Ecological risk assessment conducted, and at least in part based on comparison of fished and unfished areas, and shows no unacceptable impact. - Attempts being made to identify explicit limits of change. - The fishery management plan includes objectives relating to habitat where issues have been identified to pose a risk, and mechanisms to adjust fishery operations if adverse impacts are detected.
New Zealand Hoki - MSC 80 Guidepost and associated documents	No explicit reference points. Major, course-scale, habitat types mapped and extent of impact calculated. The percentage of major habitat types that have been unfished range from about 2% to 30%, and are about 25% in aggregate. Some protected areas and mechanisms to limit expanded impacts on habitats.	Criteria (www.MSC.org) - Knowledge of location of major habitats and their spatial distribution. - No unacceptable impacts on habitats. - Ecological risk assessment conducted. - The fishery management plan includes objectives relating to habitat where issues have been identified to pose a risk, and mechanisms to adjust fishery operations if adverse impacts are detected.
Alaskan Pollock - MSC 80 Guidepost	No explicit reference point. However extensive areas have been closed to fishing as a precautionary measure,	Criteria (www.MSC.org) - The distribution of habitats has been mapped over the range of the fishery, especially habitats considered vulnerable to fishing. Changes are measured over time.

	variously chosen to protect seabed habitats, forage species and interactions with species listed as threatened, endangered or protected.	- Development and monitoring of metrics for habitats that are most likely to be impacted. - Effects of fishing on habitats assessed and documented, with particular attention to vulnerable habitats and essential fish habitats. - Loss of fishing gear and its impacts on habitats is measured. - Models or hypotheses of the effects of fishing on habitats developed that are consistent with historical information and used to guide interpretations and management decisions to reduce fishing effects.
Pacific halibut - IPHC and MSC 80 Guidepost	No explicit reference point.	Criteria (www.MSC.org) - habitats known and mapped - impacts identified and shown to be within acceptable limits (mainly limited to coral damage by bottom set longlines)
South Georgia toothfish - MSC 80 Guidepost and CCAMLR	No explicit reference point.	Criteria (www.MSC.org) - Nature and distribution of benthic habitats known. - Impacts on habitat identified, including extent and location, and estimates of recovery times available. - Assessment of the significance and risk of impacts on habitat, and no unacceptable impacts.
Heard and MacDonald Is icefish – CCAMLR and MSC 80 guidepost	No explicit reference point but use of protected areas argued as maintaining habitats. About 17% of the relevant EEZ in protected areas selected to be representative of seabed habitats and fish community types, and to protect inshore foraging for threatened, endangered or protected species. These protected areas include about 38% and	Criteria (www.MSC.org) - Nature and distribution of benthic habitats known. - Impacts on habitat identified, including models of resilience and recovery time available. - areas closed to fishing used for addressing fishery impacts

|	27% of the total area of the two main habitat types that are fished by the icefish fishery.	
Pacific cod – MSC 80 guidepost	No explicit reference point. However extensive areas have been closed to fishing as a precautionary measure, in relation to this and other fisheries in the region, variously chosen to protect seabed habitats, forage species and interactions with species listed as threatened, endangered or protected.	Criteria (www.MSC.org) - habitat types and locations, and impacts of fishing, known - impacts within acceptable limits (mainly limited to coral damage by longlines)

Table 7. Target and limit reference points for food webs. There are very few examples of reference points being developed and operationally applied for food webs, although there are examples where reference points for perceived key prey or predator species are modified to reflect their role in the food web. The emphasis in management arrangements, where it exists at all, is on information collection rather than setting targets, limits or triggers for specified management response. The Marine Stewardship Council (MSC) 80 Guidepost is included because it is intended to reflect best practice for the fishery type being assessed.

Fishery	Limit or target RP	References and comments
Western Australian rock lobster – MSC 80 Guidepost	No explicit reference point	Criteria (www.MSC.org) - Significant predators and prey of target species are known. - Ecological risk assessment conducted, and at least in part based on comparison of fished and unfished areas, and shows no unacceptable impact.
New Zealand Hoki - MSC 80 Guidepost	No explicit reference point	Criteria (www.MSC.org) - Research on predators and prey has or is being done. - There has been an assessment of risk to threatened and protected species based, where possible, on information from fished and unfished areas. - No unacceptable impacts.
Alaskan Pollock - MSC 80 Guidepost	No explicit reference point Stability in the trophic level of the fish and invertebrate catches through time accepted as demonstrating no 'fishing down the foodweb'.	Criteria (www.MSC.org) - Monitoring of food web and predator-prey dynamics most likely to be impacted by fishing. - Impacts on forage-fish abundance and distribution are measured, in particular for prey consumed by threatened or protected species. - Assessment of the food web effects of fishery removals. - Models or hypotheses of the effects of fishing on food webs developed that are consistent with historical information and used to guide interpretations and management decisions to reduce fishing effects.
Alaskan Salmon - MSC 80 Guidepost	No explicit reference point	Criteria (www.MSC.org) - There is knowledge of the main predators and prey of the target species. -There is knowledge and

		monitoring of the functional relationship between predators and prey of the target species such that important changes can be detected.
Pacific halibut - IPHC and MSC 80 Guidepost	No explicit reference point.	Criteria (www.MSC.org) - Predators, prey and role in the food-web known.
South Georgia Toothfish - MSC 80 Guidepost and CCAMLR	Toothfish not regarded as a key prey species, so target reference point not modified from the criteria applied to species high in the food chain. i.e. limit RP is 0.1 probability of spawning biomass being below 20% of median unfished level, and target RP of median biomass not less that 50% of median unfished level.	Criteria (www.MSC.org) - Knowledge of the main predators and prey of the target species. - Assessment of the significance and risk of impacts on functional relationships, and no unacceptable impacts.
Heard and MacDonald Is icefish – CCAMLR and MSC 80 guidepost	Icefish regarded as a key prey species, so target reference point modified to be that for species low in the food chain. i.e. limit RP is 0.1 probability of spawning biomass being below 20% of median unfished level, and target RP of median biomass not less that 75% of median unfished level.	Criteria (www.MSC.org) - Functional role of target species in food web know. - Diets of major predators known and monitored.
Pacific cod – MSC 80 guidepost	No explicit reference point. However food-web models indicate that cod make up a small fraction of top predator diets, and that this and other parts of the food web are insensitive to fishery catches at recent levels.	Criteria (www.MSC.org) - predators and prey, and functional role in the ecosystem known - dependencies of predators (esp. marine mammals) known and met by stock abundance

Figures
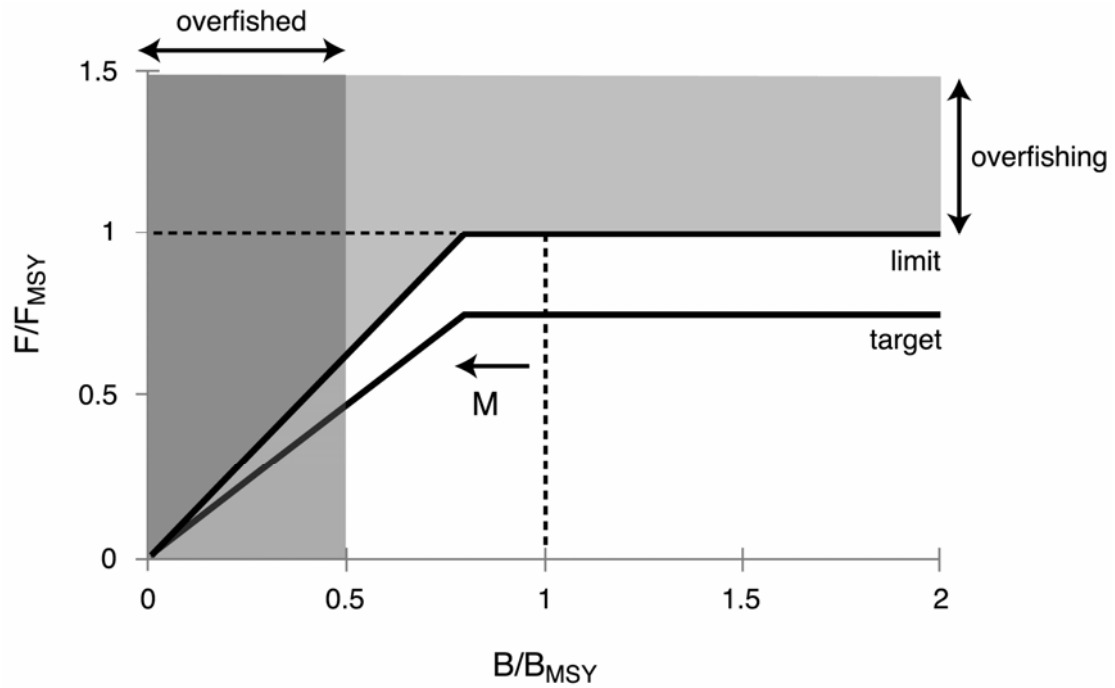
Figure 1a. An illustrative example of target and limit reference points and of a decision rule that relates the target fishing mortality to current stock size. The fishery is overfished if the biomass is below a biomass limit reference point, here $0.5B_{msy}$, and is suffering overfishing if the fishing mortality is above the fishing mortality limit reference point for the current biomass. The fishing mortality limit reference point is no greater than F_{MSY}, and decreases below a threshold as stock size approaches the biomass limit reference point. The target fishing mortality for any current stock size is designed to maintain the stock in the vicinity of B_{MSY} and to avoid the biomass limit reference point. (Based on Mace 1994, 2001)
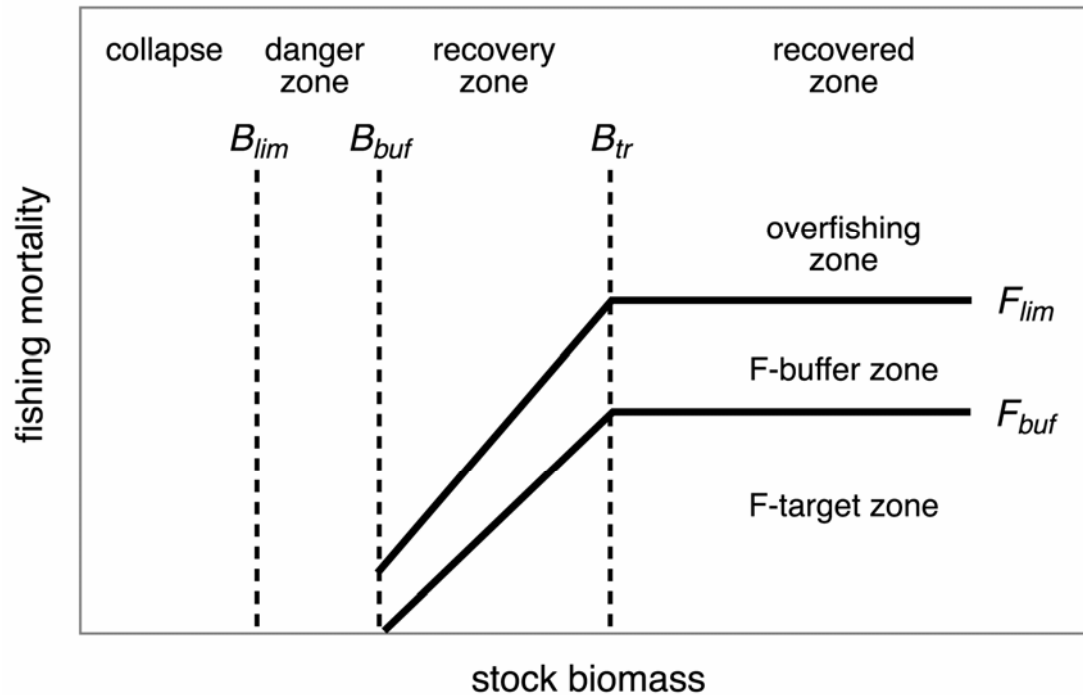
Figure 1b. An elaboration of the reference points and decision rule to include a 'buffer zone' that reflects uncertainty. The decision rule is modified from that in Figure 1a so that fishing mortality is zero if the population reaches a level, B_{buf}, and so that there is an upper limit, F_{buf}, for the target fishing mortality at any current stock biomass. The 'buffer' values are less than the limit values and reflect the uncertainty in knowing or controlling the fishing mortality and biomass. The buffer fishing mortality and biomass estimates are selected so that if an estimate of fishing mortality and biomass is less than the buffer value then there is a very low chance that the limits are exceeded in the real world. (Based on Mace 1994, 2001)
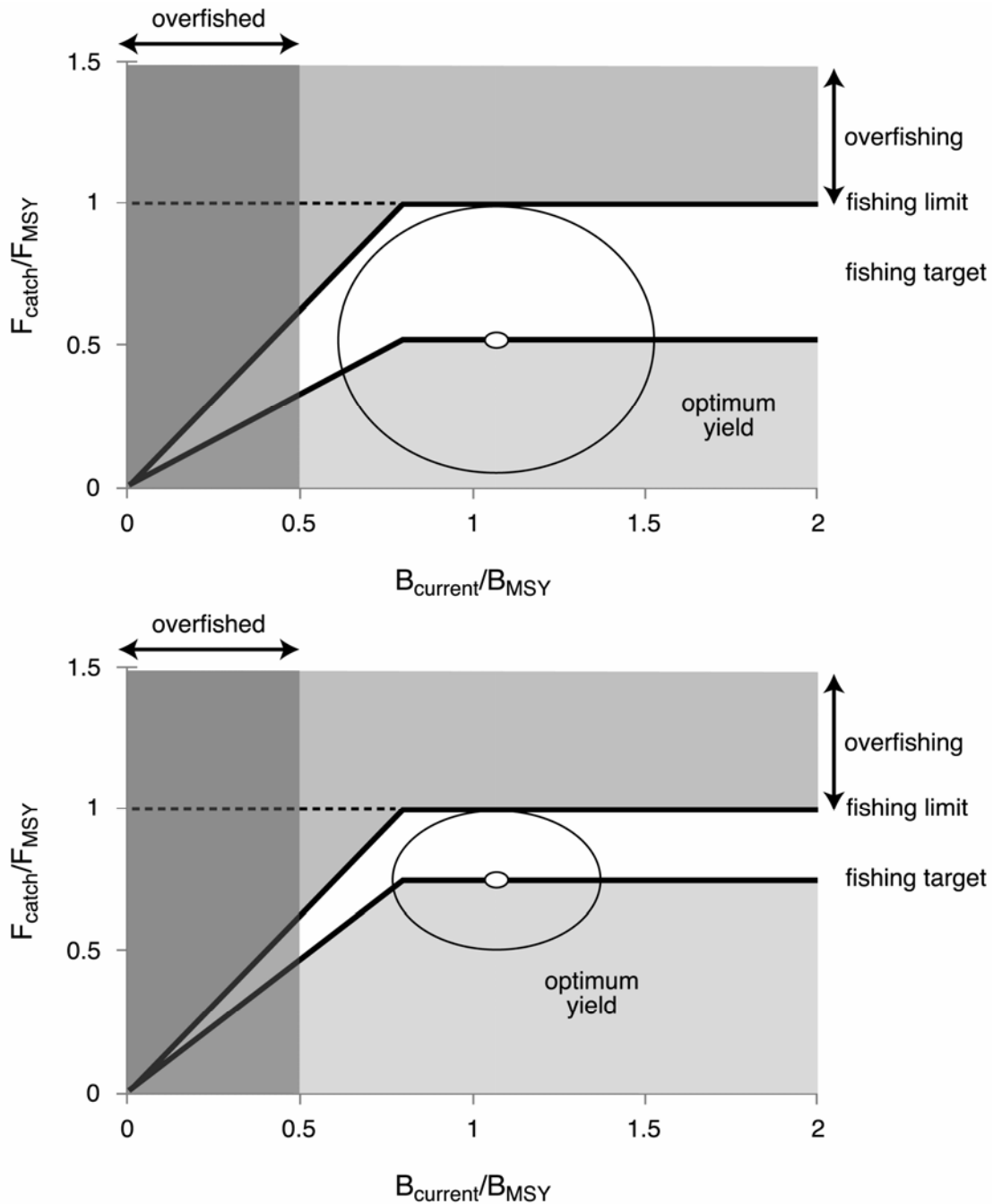
Figure 2. An illustrative example of how the decision rule and target reference point shown in Figure 1a might be changed to reflect different levels of uncertainty. The circle indicates a contour of uncertainty about the present biomass and fishing mortality relative to MSY levels. If uncertainty is large (upper) the target fishing mortality reference point must be set further from the limit reference point than could be the case if this uncertainty was smaller (lower). These changes affect the decision rule and the target fishing mortality, but not the limit fishing mortality or limit biomass. In these examples the biomass thresholds for zero and maximum fishing mortality are held constant in the decision rule as the uncertainty changes, but this need not be the best solution for a particular situation. (Based on Gerrodette *et al.* 2002)

121
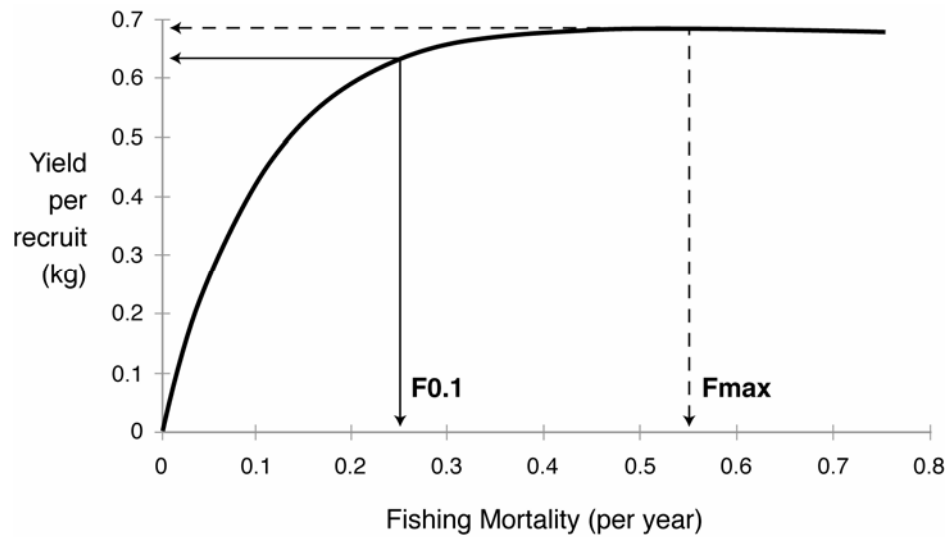
Figure 3. The relationship between yield per recruit (YPR) and fishing mortality, with some associated reference points. YPR is the expected yield from a single recruit, with fixed growth, natural mortality and fishing selectivity. F_{max} is the value of F that gives maximum YPR. $F_{0.1}$ is the value of F for which the slope of the YPR vs. F curve is 0.1 of the slope at the origin. $F_{0.1}$ gives close to the maximum yield per recruit but at a much lower fishing mortality, and consequently lower capture cost and ecological impact, than F_{max}.

A. Relationship between SPR and F

B. Spawning-recruitment Relationship
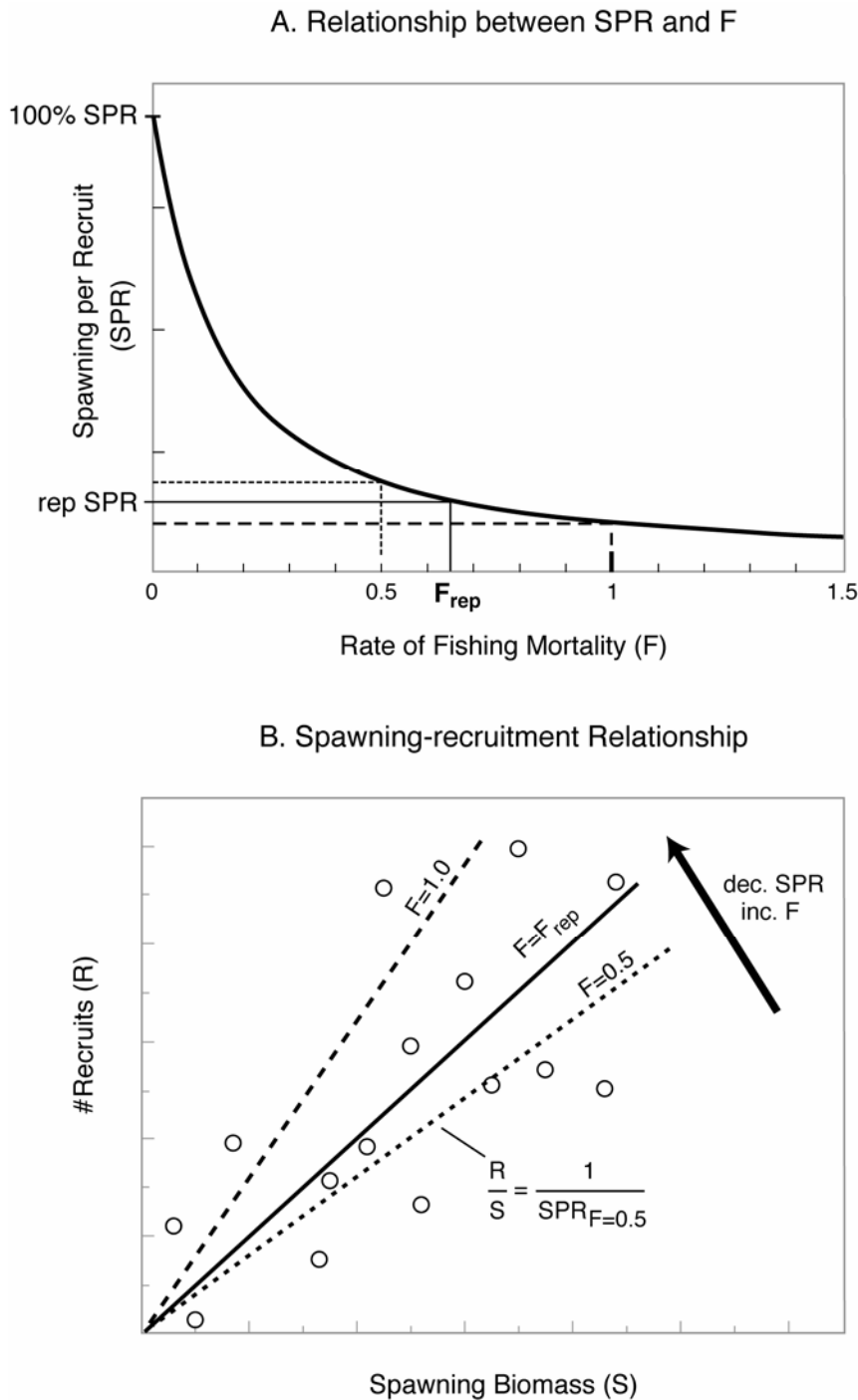
Figure 4. Spawners per recruit, stock-recruitment relationships, and 'replacement lines' (based on Mace and Sissenwine 1993).

(a) The relationship between spawning biomass per recruit (SPR) and fishing mortality F. $F_{x\%}$ is the fishing mortality that reduces the SPR to x% of its unfished value (i.e. at F=0).

(b) An example plot of observed recruits per spawning biomass and superimposed lines of the recruits-spawners ratio for different fishing mortalities from (a). For a given fishing mortality (a) gives the spawning biomass per recruit, and the reciprocal gives the gradient of the straight lines through the origin shown in (b).

Points along the straight lines in (b) give the recruits per spawning biomass necessary to replace the spawners for that given fishing mortality. F_{rep} is intended to allow replacement of the spawners on average. Here it is equal to $F_{med,}$ the fishing mortality that gives the straight line passing through the median of the observed recruitment-stock points. A related reference point is F_{low}, the fishing mortality for a straight line with 90% of the observed recruitment-stock ratios above the line and that is expected to give population increase in 90% of years. Another is F_{high} which has 10% of observed recruitment-stock ratios above the line and is expected to give population decrease in 90% of years.
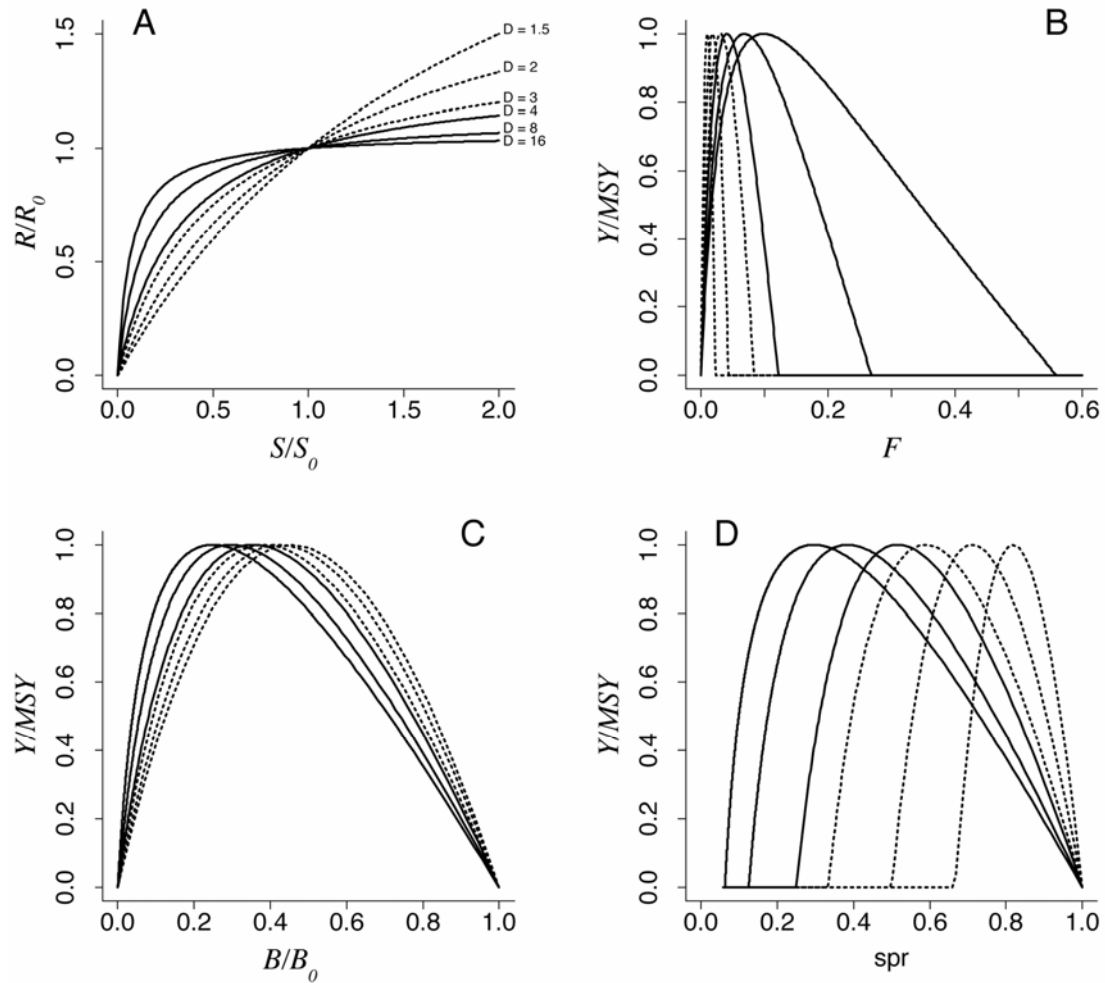
Figure 5. Some consequences of stock recruitment relationships of different steepness (based on Clark 2002). (A) The deterministic relationship between recruitment and stock size for different steepness values. An approximate conversion of the 'D' values shown and the more commonly used steepness h is (D= 1.5, h= 0.18), (D= 2, h= 0.23), (D= 3, h= 0.33), (D= 4, h= 0.43), (D= 8, h= 0.65), (D= 16, h= 0.8). (B) The relationship between yield and fishing mortality for these stock recruitment relationships. (C) The level of biomass reduction from the unfished level for different yield levels. (D) The relationship between yield and the proportionate reduction in the spawners per recruit (SPR). Steeper stock recruitment relationships imply a higher fishing mortality to achieve MSY, a greater biomass reduction at MSY and a greater reduction in SPR at MSY.
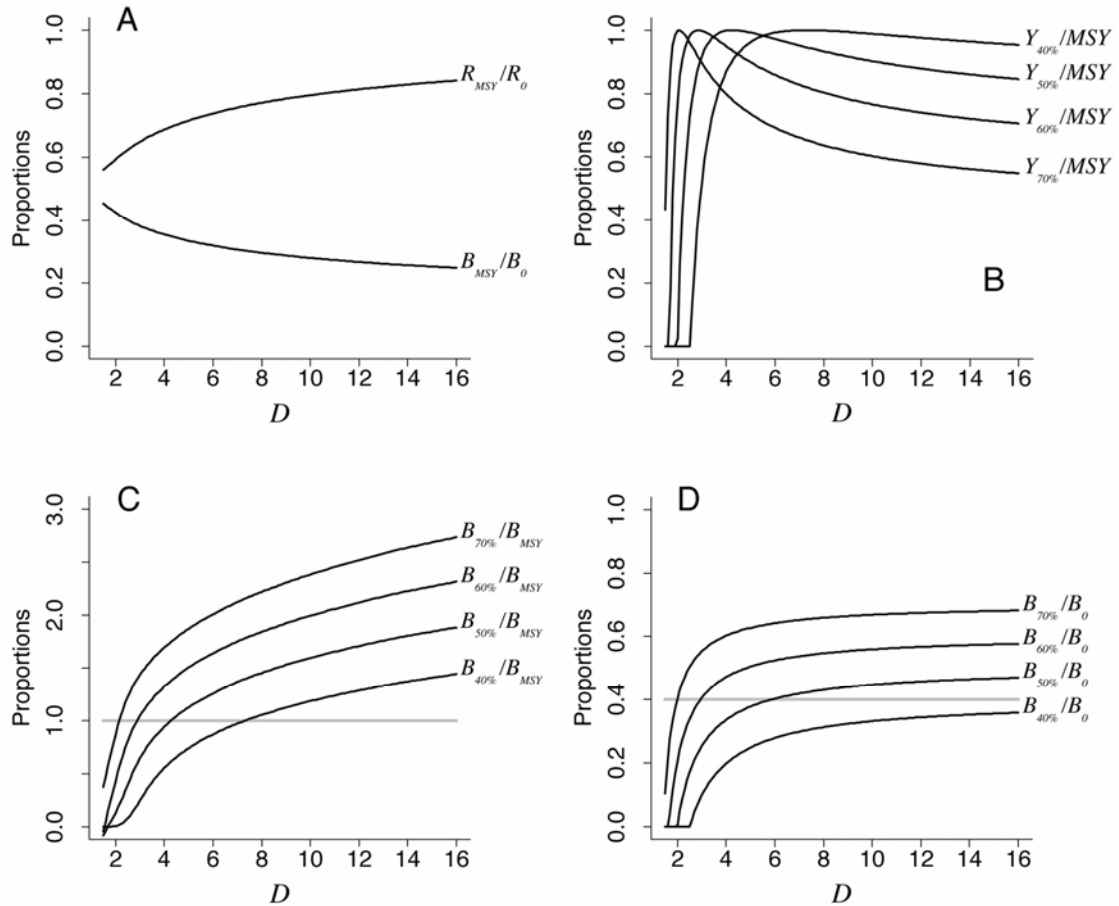
Figure 6. The effect of steepness of the stock-recruitment relationship on key sustainability measures (based on Clark 2002). (A) The reduction in recruitment and biomass at MSY. (B) The yield ($Y_{x\%}$) given at various $F_{x\%}$ levels compared to MSY, (C) the biomass ($B_{x\%}$) when fished at various $F_{x\%}$ levels compared to B_{MSY}. (D) The biomass when fished at various $F_{x\%}$ levels compared to B_0. For most species steepness is greater than 0.3 (i.e. D greater than about 3) and so $F_{50\%}$ delivers a high fraction of MSY and a biomass that is above B_{MSY}.
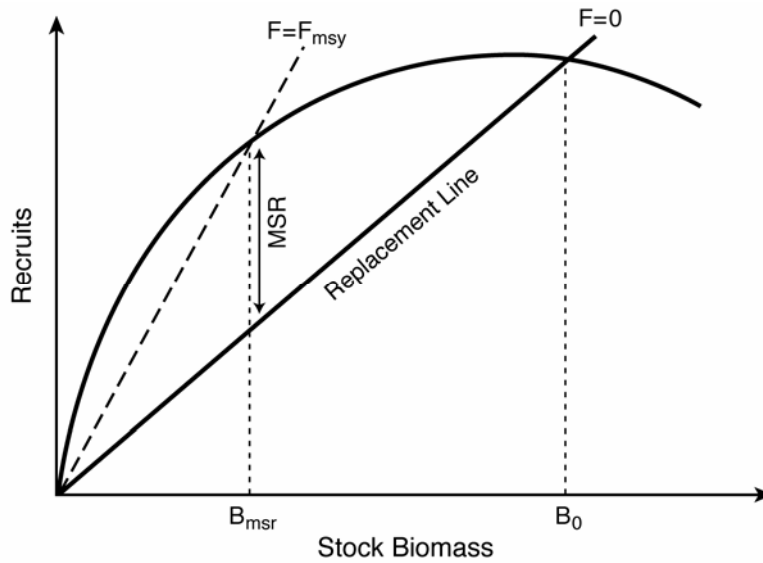
Figure 7. A stock-recruitment relationship and related reference points. The
Maximum Surplus Recruitment (MSR) occurs at B_{MSY}. The F=0 replacement line has
a gradient that is the reciprocal of the spawners per recruit with F=0, and it intersects
the stock-recruitment line at the unfished spawning biomass B_0. The F=F_{MSY}
replacement line has a gradient that is the reciprocal of the spawners per recruit with
F=F_{MSY}, and it intersects the stock-recruitment line at the point of MSR and the
spawning biomass at MSY, B_{MSY}.
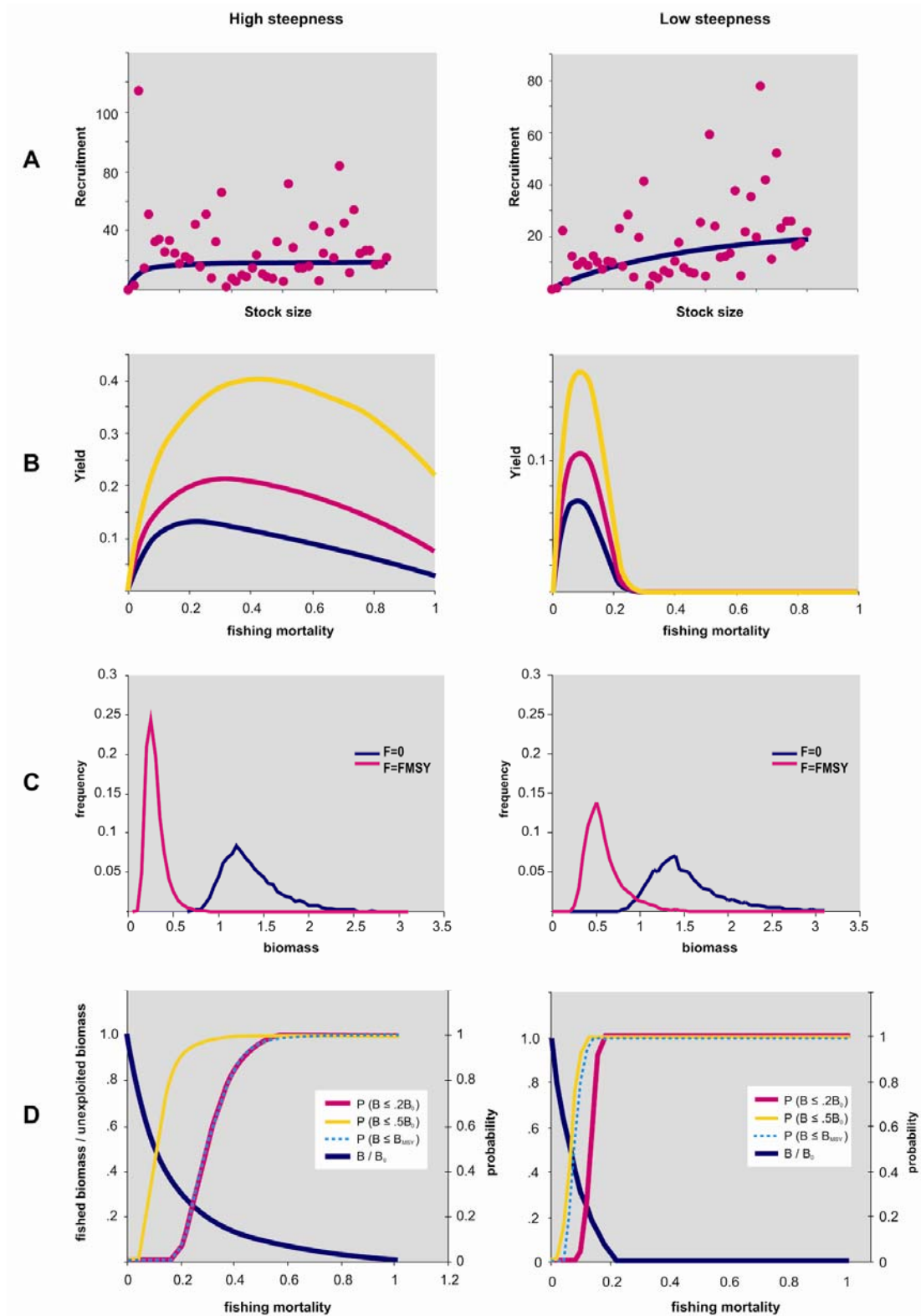
Figure 8. Some fishery related consequences of variability and the 'steepness' parameter of the stock-recruitment relationship. Annual variability in recruitment is represented by a lognormal distribution with a coefficient of variation of 75% and shown here are the states of various properties after 100y of simulation.

  A. The form of the stock-recruitment relationship and typical stochastic realizations from it.

128

B. The long term average yield, and its 95% confidence interval, for fixed levels of fishing mortality. The F giving maximum long-term yield is F_{MSY} for this constant F strategy and natural variability results in the yield in any one year ranging from half to double the long term MSY level. For high steepness F_{MSY} is higher and relatively high yields are maintained at fishing mortalities substantially above F_{MSY}.

C. The frequency distributions of biomass for the unfished population and for fishing at F_{MSY}. The unfished biomass (B_0) and the biomass at MSY (B_{MSY}) are the means of these distributions. In the absence of fishing B_0 is expected to range form half to double the mean value. When fished at a constant F_{MSY} the natural variability in recruitment similarly results in a wide range of biomass levels, including some extremely low levels for the high steepness case because mean B_{MSY} is low.

D. The mean level of depletion (mean B/mean B_0), and the probabilities $P(B<0.5$ mean B_0), $P(B< B_{MSY})$ and $P(B<0.2 B_0)$ for fixed levels of fishing mortality.

| Constant quota | Proportional escapement | Constant escapement | Proportional harvest rate |

Catch

Biomass

Fishing mortality

B'

B'

B'

B'

Fig

Biomass

130
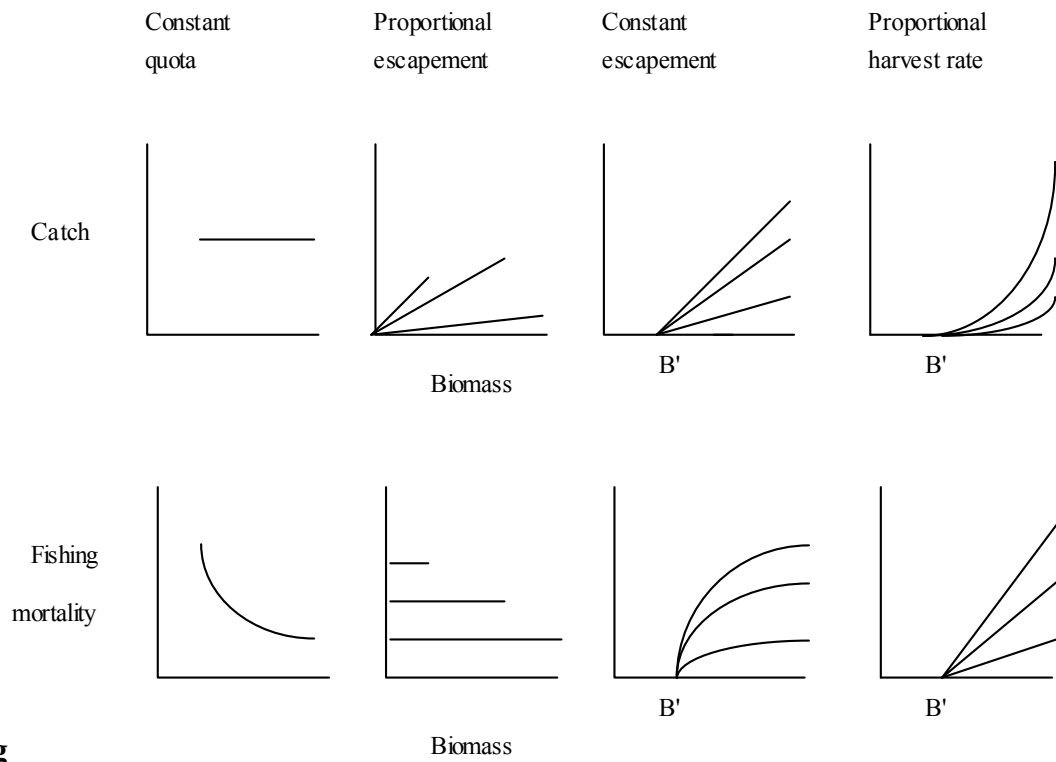
Definition of terms

Biomass: The live weight of organisms in the population or a defined part of the population (in particular the live weight of the sexually mature part of the population the spawning stock biomass). The spawning stock biomass can be measured in units other than live weight (e.g. egg production).

B_{MSY}: The population biomass at which MSY is available. In an MAY interpretation of MSY, the average biomass that results from fishing at F_{MSY}.

$B_{x\%}$: The average spawning biomass if the population was fished with constant mortality $F_{x\%}$. The value of $B_{x\%}$ is calculated by multiplying the average recruitment by the spawning biomass per recruit under $F_{x\%}$ (i.e. SPR at $F_{x\%}$). $B_{x\%}$ cannot be calculated without measurement or assumption of the average recruitment. If average recruitment is assumed to be unchanged over the relevant range of biomass then $B_{x\%}/B_{100\%}$ is the reduction in biomass from the unfished level under fishing mortality $F_{x\%}$. For populations with high steepness $B_{x\%}$ will be similar to $x\%B_0$ for a wide range of population sizes, but for populations with low steepness $B_{x\%}$ will be larger than $x\%B_0$ (see Mace 1994).

B_{loss}: The lowest spawning biomass in the observed time series.

B_{med}: The long-term average spawning biomass if the population is fished with constant mortality F_{med}. ? need this?

$B_{50\%R}$: The average spawning biomass at which recruitment is 50% of its maximum level, a limit reference point for a recruitment overfished stock.

B_{lim}: The average spawning stock biomass below which average recruitment begins to decline, especially as estimated by segmented regression methods. Below B_{lim} there is a substantial increase in the probability of reduced recruitment, while at B_{lim} the probability of reduced recruitment is still small. Alternatively B_{lim} can be the biomass below which the stock dynamics are unknown (ICES 2003a,b). B_{lim} is a limit reference point for a recruitment overfished stock.

B_{pa}: A precautionary limit reference point set to ensure that there is a low chance of the stock being at or below B_{lim} with the methods of monitoring and estimation that are used. When a stock is estimated to be at B_{pa} there should be a high probability that it is above B_{lim}.

B_{max}: The long-term average biomass resulting from fishing at constant F_{max}. Assuming constant recruitment B_{max} can be calculated from the spawning biomass per recruit multiplied by the median historical recruitment.

$B_{unfished}$: The average biomass likely to exist at any point in time in the absence of fishing. This could be derived from interpretation of observation of unfished reference sites, theoretical calculations or a combination of both of these.

Depensation: The situation where the per capita productivity decreases with decreasing population size or density. This is sometimes seen in populations that are reduced to such small size that reproduction becomes less efficient, natural predator defenses become less effective, or loss of genetic diversity limits the ability to accommodate natural variability in the physical environment. Even if fishing mortality is stopped depensatory processes can result in very slow recovery from depletion, no recovery from depletion, or further population decline to extinction..

F_{MSY}: The fishing mortality that on average generates MSY and B_{MSY}, especially in the MAY interpretation of MSY.

F: Fishing mortality. The part of the total mortality rate that is due to fishing. Fishing mortality is usually expressed as an instantaneous rate (F), so that the proportion of fish surviving a period t is given by exp[-(M+F)t] where M is the natural mortality rate. F is zero in the absence of fishing and can be greater than one, while exp[-(M+F)t] is always between zero and one.

F_{max}: The fishing mortality that results in the maximum Yield per Recruit (YPR), where that maximum exists.

$F_{0.1}$: The fishing mortality at which the slope of the YPR vs F curve is 0.1 of the slope at the origin for a given schedule of age (or size) specific selectivity. 'Optimal $F_{0.1}$' (sensu Deriso 1987) is the $F_{0.1}$ value that also has the selectivity chosen to globally maximize the YPR.

$F_{x\%}$: The fishing mortality that reduces the spawning biomass per recruit to x% of the spawning biomass per recruit at the unfished level.

F_{med}: The fishing mortality that produces a spawning biomass per recruit that is equal to the inverse of the median of the recruits per spawning biomass observed in the fishery. Year classes fished at this level will just replace themselves on average for the recruits per spawning biomass observed in 50% of years.

F_{low}: The fishing mortality that produces a spawning biomass per recruit that is equal to the inverse of the 90th percentile of the recruits per spawning biomass observed in the fishery. Year classes fished at this level will just replace themselves on average for the recruits per spawning biomass observed in 90% of years.

F_{high}: The fishing mortality that produces a spawning biomass per recruit that is equal to the inverse of the 10th percentile of the recruits per spawning biomass observed in the fishery. Year classes fished at this level will just replace themselves on average for the recruits per spawning biomass observed in 10% of years.

F_{crash} or $F_{extinction}$ or F_{τ}: The value of F for which the replacement line on a stock-recruitment is equal to the slope of the stock-recruitment curve at the origin, so that the fishing mortality cannot on average be supported by recruitment and the stock will decline to extinction. An adequate limit reference point for fishing mortality must be lower than F_{crash}.

F_{lim}: The fishing mortality that has or could result in reduction of the average spawning biomass to B_{lim}. This is a limit reference point for recruitment overfishing.

Fpa: A precautionary limit reference point set to ensure that there is a low chance of the fishing mortality being at or above F_{lim} with the methods of monitoring and estimation that are used. When a fishing mortality is estimated to be at F_{pa} there should be a high probability that it is below F_{lim}.

Generation time: The average time in an unfished population between birth of an individual and that individual replacing itself through reproduction. In practical fisheries applications this has been interpreted as being the average age of the contributors to reproduction in an unfished stock, and calculated as [the sum for all ages of (age x survival x contribution to reproduction)] / [the sum for all ages of (survival x contribution to reproduction)], where the contribution to reproduction is commonly taken to be the age specific egg production.

M: Natural mortality. The part of the total mortality rate that is due to natural causes, including disease, predation and starvation. Natural mortality is usually expressed as an instantaneous rate (M), so that the proportion of fish surviving a period t is given by exp[-(M+F)t] where F is the fishing mortality rate.

MAY: Maximum Annual Yield which is the long-term average yield obtained when the yield each year results from a constant fishing mortality (F_{MAY} or often simply F_{MSY}) being applied to the available population biomass. The catch in each year under this approach is the Current Annual Yield (CAY).

MBAL: Minimum Biologically Acceptable Level used by ICES and defined as the level of spawning stock below which the probability of poor recruitment increases as spawning stock decreases. (Serchuk and Grainer 1992 and Appendix 5)

MCY: Maximum Constant Yield which is a single unchanging maximum yield that can be taken, with an acceptable level of risk, from all probably future levels of biomass and recruitment.

MFMT: Maximum Fishing Mortality Threshold. This is specified by the US National Standard Guidelines as not exceeding F_{MSY}, or a proxy of it, under the MAY interpretation of MSY. The MFMT can be specified by a decision rule so that it can vary with current biomass or other factors.

MSST: Minimum Stock Size Threshold. This is specified by the US National Standard Guidelines as the larger of either 0.5 B_{MSY} (or a proxy of it) or the minimum biomass that would rebuild to B_{MSY} in 10y while fished at the MFMT.

MSY: Maximum Sustainable Yield. Conceptually MSY is the maximum average long-term yield that can be taken from a population. See MAY and MCY for clarification of the dynamic and static interpretations of MSY, and Ricker (1975).

Resilience: The ability of a population or ecosystem to absorb, maintain itself or recover from perturbations, changes or 'shocks'. The perturbations may be from

causes that are internal or external to the population or ecosystem, and both can occur at the same time. Fishing is an external perturbation.

Overfished: The condition that results from persistent overfishing. The population is below the limit reference point or some other expression of unacceptable impact, usually related to some combination of reduced long-term yield, reduced resilience or ability to recover, and unacceptable impacts on associated or dependent species.

For individual populations two different overfished situations are commonly recognized (see FAO glossary):
> - Growth overfished, in which the yield per recruit of a population or age class could be increased by reducing fishing mortality and/or increasing the age of fish selected by the fishing gear. For fixed selectivity a population will become growth overfished if the fishing mortality exceeds F_{max}.
>
> - Recruitment overfished, in which the average annual recruitment to the stock is significantly reduced, usually as a result of excessive reduction in the number or quality of spawners. Less extreme definitions of recruitment overfished, based on an average reduction in recruitment rather than a 'significant' reduction in recruitment, are provided by Cooke (1984) and Article II of the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR 1984). Cooke's definition is a situation in which 'a reduction in the proportion of fish caught would be more than compensated for by the increased number of recruits to the fishery as a result of increased escapement of mature fish'. And the CCAMLR (1984) interpretation is of the population being below a level that ensures 'greatest annual net increment', which is the population size producing Maximum Surplus Recruitment and MSY. ICES (see Serchuk and Grainer 1992) initially defined its Minimum Biologically Acceptable Level (MBAL) as the 'level of spawning stock below which the probability of poor recruitment increases as spawning stock decreases'. The ICES limit reference point has been clarified more recently (Anon. 2003 and Appendix 5) as the spawning biomass below which recruitment becomes systematically reduced, which is consistent with the Cooke and CCAMLR interpretations.

Taking an ecosystem perspective Murawski (2000) considered an ecosystem to be overfished of it showed the following features: (1) the biomass of any species falling below its limit reference point; (2) significant declines in diversity; (3) increases in inter-annual variability in biomass or catch; (4) significant decrease in resistance or resilience to environmental perturbation; (5) lower social or economic benefit than would be achieved with lower harvesting rates; (6) low long-term viability of ecologically important non-target species.

Overfishing: A rate or pattern of fishing that if continued would result the population becoming overfished. The population may or may not be overfished while overfishing is taking place.

SPR: Spawners per recruit. The expected quantity of spawning potential (measured by numbers, biomass, egg production or similar quantities) that a new recruit to the

population will produce over its lifetime under a particular schedule of fishing mortality and selectivity. It is calculated by:

$$SPR = \Sigma \exp(-(M+p_aF)a)\ W_a\ f_a$$

Where the symbols are as described for YPR. In an analogous manner other 'per recruit' quantities can be calculated, for example the expected fishable biomass per recruit (BPR) is

$$BPR = \Sigma \exp(-(M+p_aF)a)\ W_a\ p_a$$

Steepness (h): A parameter of the Beverton and Holt stock-recruitment relationship. The proportion of recruitment, relative to the recruitment to an unfished population, which results on average when the spawning biomass is reduced to 20% of its unfished level i.e. $R_{0.2B0} / R_{B0}$.

The Beverton and Holt function between spawning biomass (B) and recruitment (R) is

$$R = B/ (\alpha+\beta B).$$

In a re-parameterization of this function (Mace and Doonan 1988, Francis 1992b) the α and β parameters are related to steepness (*h*) and through:

$\alpha = SPR_{F=0} [\ 1- (h-0.2)/0.8h]$ and
$\beta = (h-0.2)/0.8\ h\ R_0.$

where $SPR_{F=0}$ is the spawners per recruit in the absence of fishing and R_0 is the number of recruits in the absence of fishing.

YPR: Yield per recruit. The expected yield (measured by numbers, biomass or similar quantities) that a new recruit to the population will produce over its lifetime under a particular schedule of fishing mortality and selectivity. It is calculated by:

$$YPR = \Sigma\ \exp(-(M+p_aF)a)\ W_a\ [1- \exp(-(M+p_aF)a]\ [p_aF/ (M+p_aF)]$$

where M is the natural mortality rate, F is the fishing mortality rate for fully vulnerable aged fish, W_a is fish weight at fish age a, p_a is the selectivity of the fishing gear at age a, f_a is the proportion of fish sexually mature at age a, and the sum is across all ages in the population.

Appendices

Appendix 1. *Project Participants*

Dr Andrew Constable, Australian Antarctic Division.

Dr Bill Clarke, International Halibut Commission.

Dr Patricia Livingston, National Marine Fisheries Service, USA.

Dr Pamela Mace, National Marine Fisheries Service, USA.

Dr Andre Punt, University of Washington, USA.

Dr Jake Rice, Fisheries and Oceans, Canada, and the International Council for the Exploration of the Sea (ICES).

Dr Gunnar Stefansson, Icelandic Marine research Institute.

*Appendix 2. The proforma used to report on the use of reference points and
identify potential best practice.*

From the fisheries that you are directly involved with please chose a fishery or
fisheries that illustrate what you regard as best practice for management of the various
issues being addressed in this project (i.e. retained species, by-catch species,
threatened or protected species etc). It may be that different fisheries illustrate best
practice for different issues, and so several fisheries are reported on here. It would
help greatly if all of the issues in the proforma were commented on for each fishery
that is reported on. It is likely that many of the issues are not relevant to some
fisheries, and in this case please just make that comment in the relevant part of the
proforma (i.e. say not relevant rather than leaving the entry ambiguously blank).

Fishery
- Name and location
- Relevant fishery management plan, policy, legislation (please provide copies
  of these or a source, such as a www site or contact point, so that we can obtain
  copies)
- Main target species
- Main retained species
- General form of management (open access, input control (what vessel/gear
  restrictions, limited entry), output control (competitive quota, individual
  quota)

Target and retained species
- General approach to retained species management
- For all or a representative selection of species:
    - Ecological properties of the species (e.g. where on r-K spectrum; top pred,
      intermediate pred/prey, prey species)
    - Level of natural variability (e.g. 'usual' level of interannual recruitment
      variability →highly variable recruitment interannually → episodic
      recruitment and regime shifts)
    - Planned management responses (decision rules and recovery rules and
      targets)
    - Level of information/uncertainty (see appendix 2 and elaborate as
      necessary)
    - Reference points (target, limit and trigger if used).
    - Status of species in relation to reference point (eg. under , acceptably near
      target, over)
    - Comment on strengths/weakness of reference points and score out of 10 (1
      for poor and 10 for best practice in your fisheries). Please comment on
      whether any specific consideration has been given to genetic biodiversity
      in the retained species, and if so what approach to its management was
      taken.

By-catch species

- General approach to by-catch management
- For general approach or a representative selection of species/groups:
    - Ecological properties of the species or groups (e.g. where on r-K spectrum)
    - Level of natural variability
    - Planned management responses (decision rules and recovery rules and targets)
    - Level of information/uncertainty (see appendix 2 and elaborate as necessary)
    - Reference points (target, limit and trigger if used)
    - Status of species/groups in relation to reference point (eg. under , acceptably near target, over)
    - Comment on strengths/weakness of reference points and score out of 10 (1 for poor and 10 for best practice in your fisheries). Please comment on whether any specific consideration has been given to species biodiversity among the by-catch species, and if so what approach to its management was taken.

Threatened, endangered or protected species and communities
- General approach to management of threatened or protected species/communities
- For general approach or a representative selection of species/communities:
    - Ecological properties of the species or groups (e.g. where on r-K spectrum)
    - Level of natural variability
    - Planned management responses (decision rules and recovery rules and targets)
    - Level of information/uncertainty (see appendix 2 and elaborate as necessary)
    - Reference points (target, limit and trigger if used)
    - Status of species/ communities in relation to reference point (eg. under , acceptably near target, over)
    - Comment on strengths/weakness of reference points and score out of 10 (1 for poor and 10 for best practice in your fisheries). Please comment on whether any specific consideration has been given to community biodiversity, and if so what approach to its management was taken.

Habitats
- General approach to management of habitats
- For general approach or a representative selection of habitats:
    - Ecological properties of the habitats
    - Level of natural variability
    - Planned management responses (decision rules and recovery rules and targets)
    - Level of information/uncertainty (see appendix 2 and elaborate as necessary)
    - Reference points (target, limit and trigger if used)
    - Status of habitats in relation to reference point (eg. under , acceptably near target, over)
    - Comment on strengths/weakness of reference points and score out of 10 (1 for poor and 10 for best practice in your fisheries). Please comment on

whether any specific consideration has been given to the effects of habitats on biodiversity, and if so what approach to its management was taken.

Food webs
- General approach to management of food webs in general and of direct feeding interactions (predator-prey relationships involving the target species) specifically.
- For direct feeding interactions (e.g. predator-prey relationships) that directly involve the target or other highly valued species:
  - Ecological properties involved
  - Level of natural variability
  - Planned management responses (decision rules and recovery rules and targets)
  - Level of information/uncertainty (see appendix 2 and elaborate as necessary)
  - Reference points (target, limit and trigger if used)
  - Status of species/ communities in relation to reference point (eg. under , acceptably near target, over)
  - Comment on strengths/weakness of reference points and score out of 10 (1 for poor and 10 for best practice in your fisheries)
- For food webs in general:
  - Ecological properties involved
  - Level of natural variability
  - Planned management responses (decision rules and recovery rules and targets)
  - Level of information/uncertainty (see appendix 2 and elaborate as necessary)
  - Reference points (target, limit and trigger if used)
  - Status of species/ communities in relation to reference point (eg. under , acceptably near target, over)
  - Comment on strengths/weakness of reference points and score our of 10 (1 for poor and 10 for best practice in your fisheries)

Appendix 3. The tiered system used in the US Alaskan fisheries.

This approach is used to set overfishing limit (OFL) reference points and determine fishing mortality for the Acceptable Biological Catch (ABC) for different levels of information availability (Witherall, 1999, Witherall et al, 2000, and NMFS (2003). For these fisheries the ABC provides a specific interpretation of the optimum yield (OY) target reference points defined by the National Standard Guidelines (NMFS (998A). The Guidelines allow catches to be set lower than the OY for ecological, economic and social reasons.

For Tier 1, a "pdf" refers to a probability density function. For Tiers 1 and 2, if a reliable pdf of biomass (e.g., the biomass level that would describe a stock of fish at its maximum sustainable level) B_{MSY} is available, the preferred point estimate of B_{MSY} is the geometric mean of its pdf. For Tiers 1–5, if a reliable pdf of B is available (e.g., current biomass level), the preferred point estimate is the geometric mean of its pdf. The harmonic mean is always less than the arithmetic mean, and if for example the pdf is gamma with a coefficient of variation of 50% then the harmonic mean is at most 0.75 of the arithmetic mean. For Tiers 1–3, the coefficient a is set at a default value of 0.05, with the understanding that a different value for a specific stock or stock complex may be justified by the best available scientific information. For Tiers 2–4, a designation of the form "$F_{X\%}$" refers to the F associated with an equilibrium level of spawning per recruit (SPR) equal to X percent of the equilibrium level of spawning per recruit in the absence of any fishing. If reliable information sufficient to characterize the entire maturity schedule of a species is not available, then SPR calculations based on a knife-edge maturity assumption may be considered reliable. For Tier 3, the term $B_{40\%}$ refers to the long-term average biomass that would be expected under average recruitment and $F = F_{40\%}$. (Note that $B_{40\%}$ is not the same as $40\%B_0$).

1. Information available: Reliable point estimates of B and B_{MSY} and reliable pdf of F_{MSY} .

    1a. Stock status: B/B_{MSY} > 1

        $F_{OFL} = m_A$, the arithmetic mean of the pdf

        $F_{ABC} \leq m_H$, the harmonic mean of the pdf

    1b. Stock status: a < B/B_{MSY} ≤ 1

        $F_{OFL} = m_A \times (B/B_{MSY} - a)/(1 - a)$

        $F_{ABC} \leq m_H \times (B/B_{MSY} - a)/(1 - a)$

    1c. Stock status: B/B_{MSY} ≤ a

        $F_{OFL} = 0$

        $F_{ABC} = 0$

2. Information available: Reliable point estimates of B, B_{MSY} , F_{MSY} , $F_{35\%}$, and $F_{40\%}$.

    2a. Stock status: B/B_{MSY} > 1

        $F_{OFL} = F_{MSY}$

        $F_{ABC} \leq F_{MSY} \times (F_{40\%}/F_{35\%})$

    2b. Stock status: a < B/B_{MSY} ≤ 1

        $F_{OFL} = F_{MSY} \times (B/B_{MSY} - a)/(1 - a)$

        $F_{ABC} \leq F_{MSY} \times (F_{40\%}/F_{35\%}) \times (B/B_{MSY} - a)/(1 - a)$

    2c. Stock status: B/B_{MSY} ≤ a

        $F_{OFL} = 0$

$$F_{ABC} = 0$$

3. Information available: Reliable point estimates of B, $B_{40\%}$, $F_{35\%}$, and $F_{40\%}$.

    3a. Stock status: $B/B_{40\%} > 1$

$$F_{OFL} = F_{35\%}$$
$$F_{ABC} \leq F_{40\%}$$

    3b. Stock status: $a < B/B_{40\%} \leq 1$

$$F_{OFL} = F_{35\%} \times (B/B_{40\%} - a)/(1 - a)$$
$$F_{ABC} \leq F_{40\%} \times (B/B_{40\%} - a)/(1 - a)$$

    3c. Stock status: $B/B_{40\%} \leq a$

$$F_{OFL} = 0$$
$$F_{ABC} = 0$$

4. Information available: Reliable point estimates of B, $F_{35\%}$, and $F_{40\%}$.

$$F_{OFL} = F_{35\%}$$
$$F_{ABC} \leq F_{40\%}$$

5. Information available: Reliable point estimates of B and natural mortality rate M.

$$F_{OFL} = M$$
$$F_{ABC} \leq 0.75 \times M$$

6. Information available: Reliable catch history.

    OFL = the average catch over the reliable catch history, unless an alternative value is justified on the basis of the best available scientific information

    $ABC \leq 0.75 \times OFL$

In general, the above definitions represent an attempt to institute a precautionary approach consistent with the requirements of the Magnuson-Stevens Act and the practical constraints of existing data. Tiers 1–6 satisfy the first characteristic of a precautionary approach by placing a substantial buffer between OFLs and the annual ABC. Tiers 1–3 satisfy the second characteristic of a precautionary approach by decreasing fishing mortality rates for stocks that fall below the MSY level (or, in the case of Tier 3, for stocks that fall below a reference level somewhat higher than the MSY level). Tier 1 satisfies the third characteristic of a precautionary approach by reducing the target fishing mortality rate in direct relation to the level of uncertainty regarding the stock's productive capacity (i.e. greater uncertainty leads to a lower target fishing mortality rate).

The F_{ABC} given by these decision rules can be reduced by biomass based limits to fishing. For example if biomass is below the overfishing limit then a recovery plan is implemented which replaces the F_{ABC} decision rules.

***Appendix 4. The approach to Ecosystem Based Fishery Management
under the California Marine Life Management Act.*** Based on Kaufman *et
al.* (2004).

The California Marine Life Management Act (1999) requires maintenance of
ecosystem health and biodiversity and requires human use to be sustainable (i.e. to
provide the fullest range of present and long term consumptive and non-consumptive
benefits). Fisheries are required to be managed under a Fishery Management Plan and
a plan for staged implementation of Ecosystem Based Management was adopted in
2002.

The management plan for California's nearshore finfish fishery recognizes three
stages of information availability, and establishes annual Total Allowable Catch
(TAC) decision rules that are designed to assure sustainable fisheries with the
information available at each stage. As the information and understanding about the
fishery changes the fishery can change stages, and higher stages can have higher
TACs. This approach is based on the principle that a high level of precaution is
necessary at the onset of a fishery for a new resource or if there is little understanding
about the effects of fishing. That precaution can be reduced as improved information
allows risk to be more explicitly recognized and managed, although some
uncertainties are irreducible and so the need for precaution cannot be entirely
eliminated.

These stages and the approach to TAC setting and information gathering are described
by Kaufmann *et al.* (2004) as:

Stage I. Data poor, with information typically limited to catch history from a period
that can be argued to be not overfishing the stock and general experience with the
kind of fishery and resource. The TAC is set based almost entirely on 'blind'
precaution and performance measurement is solely on the basis of whether the TAC is
adhered to, because there is insufficient information for more meaningful risk
management and performance measurement. As a default the TAC is set at 50% of
the catch level that might be considered sustainable from past catch history or other
argument. Where pooled TACs are set, management focuses on the least abundant
species in the catch.

Stage II. Data moderate, with information typically able to support informed risk
assessment and management of single or multiple target species. However
precautionary measures are still significant, even in setting TACs for target species,
because there is still very limited information about the effects of fishing and changed
environmental conditions on the ecosystem. The preferred reference point for target
species is $B_{unfished}$, the biomass likely to exist under prevailing environmental
conditions and if there had been no fishing in recent history, rather than B_0 (virgin
biomass) or similar historical measures. $B_{unfished}$ can be derived from theoretical
calculations but they are preferably based on sampling and interpretations from
unfished reference sites. $B_{unfished}$ may need to be recalculated periodically in order to
take account of natural fluctuations in parameters such recruitment and growth rate.

A default target reference point for fishing mortality in stage II is $F_{50\%}$, which is
applied if the resource species is believed to be above 0.6 of $B_{unfished}$. The catch

decision rule for the target species reduces the applied F linearly from 0.6 of $B_{unfished}$ to 0.2 of $B_{unfished}$.

Stage II allows greater TACs when stocks are healthy and during periods of greater environment-induced productivity. This provides incentive for the fishing industry to participate in data collection. In addition to collecting more data, Stage II requires a good database and the development of tools to make use of existing data.

Stage III. Sufficiently data-rich to support ecosystem management. Stage III analyses encompass *non-target* species and physical oceanography. Implementation of this stage is still in the future and it is expected that the move from Stage II to III will be gradual, as information becomes available. Key to moving to Stage III is the identification of reference areas, both unfished reserves and areas subject to varying levels of fishing pressure. Correlation studies and investigation of the causes of changes in these areas will allow an understanding of the effects of fishing. Alterations that are not due to human disturbance but rather to changes in, for example, climate, can be identified using unfished reference areas.

The default TAC for Stage III is the same as for Stage II. Integrating all the information and models available at Stage III will pose future challenges and may involve optimization across a number of models.

The California Marine Life Management Act emphasizes the need for Marine Protected Areas as a management tool. MPAs serve as a reference for comparison with fished areas and consequently are an import part of the Stage III strategy. They might also act as a buffer against management mistakes and protect some part of the population, possibly increasing reproductive potential.

Appendix 5. Target and limit reference points required by the International Council for the Exploration of the Sea (ICES).

Based on ICES (2003 a, b).

<u>General</u>

The ICES approach is that for stocks and fisheries to be within safe biological limits, there should be a high probability that spawning stock biomass (SSB) is above a limit B_{lim} below which recruitment becomes impaired or the dynamics of the stock are unknown, and that fishing mortality is below a value F_{lim} that will drive the spawning stock to that biomass limit. The word 'impaired' is synonymous with the concept that on average recruitment becomes systematically reduced as biomass declines below a certain point due to the effect of fishing (i.e. the medium-term average recruitment is lower than has been observed at higher levels of biomass). Because of uncertainty in the annual estimation of F and SSB, ICES defines the more conservative operational reference points, B_{pa} (higher than B_{lim}), and F_{pa} (lower than F_{lim}), where the subscript pa stands for precautionary approach. When a stock is estimated to be at B_{pa} there should be a high probability that it will be above B_{lim} and similarly if F is estimated to be at F_{pa} there should be a low probability that F is higher than F_{lim}. The reference values B_{lim} and F_{lim} are therefore estimated in order to arrive at B_{pa} and F_{pa}, the operational values that should have a high probability of ensuring that exploitation is sustainable based on the history of the fishery.

Stocks that are both above B_{pa} and below F_{pa} are considered to be inside safe biological limits. Stocks that are both below B_{pa} and above F_{pa} are considered to be outside safe biological limits, and stocks that are above B_{pa} but also above F_{pa} are considered to be harvested outside safe biological limits. When a fishery is at or above F_{pa}, ICES will advise that F should be reduced, and when a stock is estimated to be at or below B_{pa} ICES will advise that F should be reduced. When a stock is estimated to be above B_{pa}, but is subject to an F that is at or higher than Fp_a, ICES will again advise that F should be reduced. The reference points F_{pa} and B_{pa} are boundaries to the safe limits domain, and not targets.

ICES previously defined and used the Minimum Biologically Acceptable Level (MBAL) of biomass for a number of stocks. MBAL was originally chosen as the SSB below which the probability of poor recruitment increased. It is therefore comparable to the current usage of B_{lim}.

Target reference points represent long term management objectives. Target reference points are constrained by the precautionary reference points, so that a target fishing mortality should be below F_{pa} and a target SSB should be above B_{pa}. Target reference points have not yet been defined by clients of ICES advice nor used by ICES in the provision of advice.

<u>Operational compatibility between fishing mortality and biomass reference points</u>

The operational reference point B_{pa} is derived from B_{lim} in order to ensure that when a spawning stock is observed to be at B_{pa} there is a low probability that it is really at B_{lim}. If SSB is at or below B_{pa}, ICES should advise that F be reduced in order to

increase SSB above B_{pa}, (since ICES does not intend that B_{pa} is to be used as a target). Similarly, F_{pa} is derived from F_{lim} in order to ensure that when a stock is observed to be at F_{pa} there is a low probability that it is really above F_{lim}. If F is at or above F_{pa}, ICES should therefore advise that F is reduced below F_{pa} (since ICES does not intend that F_{pa} is to be used as a target). Assessment uncertainty taken into account by the independent calculations of B_{pa} and F_{pa} is unlikely to be the same, so that when a stock is observed to be at F_{pa} this does not necessarily imply that SSB will be at B_{pa} all of the time. Also B_{pa} is the average biomass under fishing mortality F_{pa} and so it is expected that the SSB at any particular time will vary from B_{pa}. Therefore, when F is at F_{pa}, but SSB is below B_{pa}, ICES will also give advice to further reduce F. Although we do not expect that F_{pa} implies that equilibrium SSB is B_{pa}, it will still be helpful to evaluate the performance of these reference points by monitoring the actual operational relationship between F_{pa}, SSB, and B_{pa}.

Calculating the limit reference points

ICES (2003a, b, c) describe the details of how the limit reference points can be calculated with different levels of information available, and particularly for different patterns in the stock-recruitment plot (including no apparent stock-recruitment signal and limited observational range in the observations).

But the intent of the approach is illustrated in the treatment of stocks where a change point in the stock-recruitment relationship is evident (i.e. recruitment can be reasonably described by a constant average above the change point biomass and recruitment decreases below the change point biomass). The change point may be estimated on the basis of a segmented regression of the recruitment (R)-spawning stock biomass (SSB) data. If the fit is statistically robust, the estimate of the change point is used as B_{lim}. For other stocks, B_{loss} may be used as a proxy for B_{lim}. Calculated this way B_{lim} is the biomass below which the average recruitment begins to decline, which represents an important shift away from the MBAL and other previous approaches in which the limit reference point is the biomass at which recruitment was already seriously or significantly reduced on average.

F_{lim} is then derived from B_{lim} as follows:
• Calculate R/SSB at B_{lim}, the slope of the replacement line at B_{lim}.
• Invert to give SSB/R.
• Use this SSB/R to derive F_{lim} from the curve of SSB/R against F.

 F_{pa} may be estimated from F_{lim} as follows:
• Carry out a set of retrospective assessments to tabulate and plot the distributions of realised F in past assessment years corresponding to each intended F in that year.
• Compare the distributions between intended F values and identify the highest intended F that still carries a lower risk that the realised F is above F_{lim}, and that F value is used as $F_{pa.}$

To Estimate B_{pa} from B_{lim}:
• Use the set of retrospective assessments to obtain the observed SSB in each past year and compare with the 'true' SSB estimated by the recruitment-spawning stock biomass data set.
• Plot the pairs of SSB_{obs}/SSB_{true} against SSB_{true}

• Draw through the origin the line that leaves α% (where α is the acceptable risk) of the points above the line; the slope of this line is β in $B_{pa} = β * B_{lim}$.

While experience with calculating these reference points, and especially the pa reference points is limited, ICES (2003c) examined many Arctic cod stocks for which overall F_{pa} was about 0.6 F_{lim} and B_{pa} was about 1.4 B_{lim}.

Appendix 6. **Ecological Risk Assessment for the Effects of Fishing.**
Based on Hobday, Smith and Stobutzki (2004)

The aim of ecological risk assessment is usually to identify the likely severity of risk from human activities to components of the ecosystem. Risk assessment allows the identification of issues of concern, particularly vulnerable components, risky practices, and key knowledge gaps.

There are many forms of risk assessment, ranging from qualitative and 'opinion based' (e.g. Fletcher *et al.* 2002) methods through to highly quantitative and 'objective evidence based' methods (e.g. Hayes 1997, 2002, Hobday *et al.*, 2004). Qualitative indicators can be categories such as "poor", "good" and "best". Often suite of indicators is used, rather than relying on one, with weights assigned to each indicator.

Recently there has been a rapid development and expansion in the use of Ecological Risk Assessment, with international focus coming particularly through Marine Stewardship Council assessments and Australian domestic focus coming particularly from assessment of fisheries for Ecologically Sustainable Development and against sustainability requirements of the EPBC Act.

 The Ecological Risk Assessment developed to address EPBC assessment needs for Australian Commonwealth fisheries takes a hierarchical three stage approach (Hobday *et al.*, 2004). A qualitative first stage is used to quickly identify components of greatest concern. These can then be managed directly and/or carried through to the second stage of more rigorous assessment if it is thought that additional analysis would be useful to refine understanding of the risk or to demonstrate that the qualitative assessment had given a 'false negative' outcome., The semi-quantitative analysis is based on the methods of Stobutzki (2001, 2002). Issues of concern that are identified in the second stage can be managed directly or carried to the third stage, a fully-quantitative assessment. Hobday *et al.* (2004) provide reference points (some are quantitative and some qualitative) to classify level of risk (see Tables A6.1 to A6.5).

This approach is efficient because many potential risks are screened out at Level 1, so that the more intensive and quantitative analyses at Level 2 (and ultimately at Level 3) are limited to a subset of the higher risk activities associated with fishing. It also leads to rapid identification of high-risk activities, which in turn can lead to immediate remedial action through the risk management response.

The approach makes use of a general conceptual model of how fishing impacts on ecological systems, which is used as the basis for the risk assessment evaluations at each level of analysis. Five general ecological components are evaluated: (1) target species; (2) by-product and by-catch species; (3) threatened, endangered and protected species (TEP species); (4) habitats; (5) ecological communities.

The methodology is described in detail by Hobday *et al.* (in 2004), but a brief description is:

Level 1: Scale, Intensity and Consequence Analysis (SICA)

Level 1 is a rapid assessment tool to identify low risk components and screen them out of further, and more complex, data compilation or collection, risk analysis or risk management activities. Each component of the fishery is assessed using a scale, intensity and consequence analysis (SICA). In this each component is examined and any logical units of analysis are identified (e.g. for the target species component the units of analysis might be the individual species that comprise the target group). A series of scenarios are developed about how fishing activities might affect the units of analysis, and the "worst case" combination of scenario and unit of analysis is identified. Only this combination of impact scenario and unit of analysis are then formally analysed for consequence. So the risk assessment for the whole component is based on the most vulnerable unit of analysis. The rationale for this is that if the activity is safe for the most vulnerable unit of analysis it is expected to be safe for the other units in that component, and so the whole component is assessed to be at low risk. If not, then further and more complex responses are required. The analysis uses tables to help standardise assessment of the consequence for common units of analysis (see Tables A6.1 to A 6.5)

Level 2: Productivity Susceptibility Analysis (PSA)

The components that were identified to be at moderate or greater risk in Level 1 are examined semi-quantitatively at Level 2.

The PSA approach is based on the assumption that the risk to an ecological component will depend on two characteristics: (1) the extent of the impact due to the fishing activity, which is determined by the susceptibility to fishing and (2) the productivity of the unit, which will determine the recovery rate after damage. The susceptibility is primarily determined by the combination of the fishing methods and the fish ecology.

A standardised score for susceptibility is derived as the product of scores for availability (e.g. spatial overlap), encounterability (e.g. overlap in the water column of the fish and the fishing gear), selectivity (e.g. what sizes are caught on encounter) and post-capture mortality. Extensive tables are used to derive standardised scores for susceptibility from these elements.

A standardised score for productivity is derived from the biological and ecological properties of the entity being assessed. The productivity assessment can make use of information in the literature or data-bases if there are no relevant data from the fishery. Uncertainty or unreliability is included in the score of productivity, and so for example missing data results in a higher risk being assigned.

Each unit of analysis is scored for risk in relation to susceptibility and productivity using the standardised guides as to severity of risk and the output is graphed to produce a PSA plot (Figure A6.1). There are pre-identified thresholds for high, medium and low risk. Units of analysis that fall in the high risk areas of this graph are the priority for further analysis at level 3 and for directed management response. If the reason for scoring high risk is mainly because of a lack of information the risk management response may focus on collecting additional information but if it is

because the data show high risk then the management response would also address the cause of the risk.
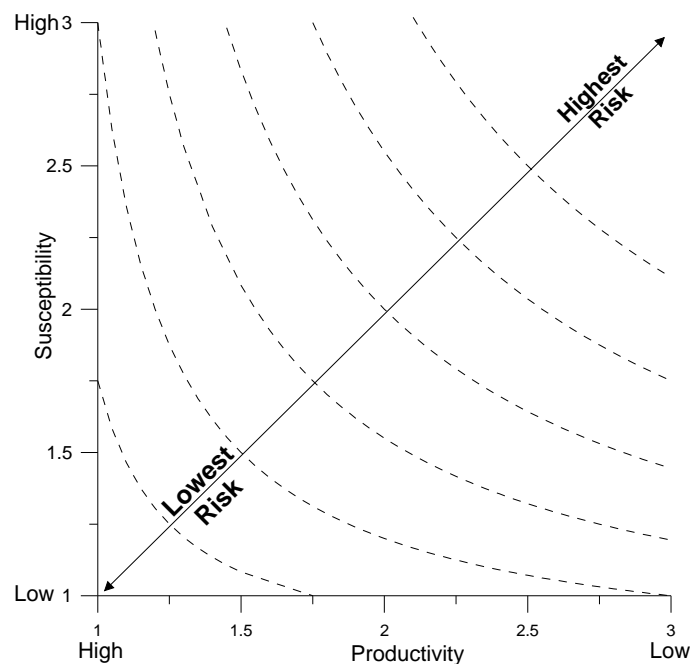
Figure A6.1. The axes on which risk to the ecological units is plotted. The *x*-axis includes attributes that influence the productivity of a unit, or its ability to recover after impact from fishing i.e. resilience. The *y*-axis includes attributes that influence the susceptibility of the unit to impacts from fishing. The combination of susceptibility and productivity determines the relative risk to a unit, i.e. units with high susceptibility and low productivity are at highest risk, while units with low susceptibility and high productivity are at lowest risk. The contour lines represent a multiplicative relationship between the axes and group units of similar risk levels.

Level 3: Quantitative risk assessment

This uses the methods of quantitative modelling and statistics, as for example applies in target species assessments.

1. **Table A6.1** . A guide for scoring the level of consequence for target species. (From Hobday *et al.* 2004)

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
Population size	**1. Population size** Insignificant change to population size/growth rate (r). Unlikely to be detectable against background variability for this population.	**1. Population size** Possible detectable change in size/growth rate (r) but minimal impact on population size and none on dynamics.	**1. Population size** Full exploitation rate but long-term recruitment dynamics not adversely damaged.	**1. Population size** Affecting recruitment state of stocks and/or their capacity to increase	**1. Population size** Likely to cause local extinctions if continued in longer term	**1. Population size** Local extinctions are imminent/immediate
Geographic range	**2. Geographic range** No detectable change in geographic range. Unlikely to be detectable against background variability for this population.	**2. Geographic range** Possible detectable change in geographic range but minimal impact on population range and none on dynamics, change in geographic range up to 5 % of original.	**2. Geographic range** Change in geographic range up to 10 % of original.	**2. Geographic range** Change in geographic range up to 25 % of original.	**2. Geographic range** Change in geographic range up to 50 % of original.	**2. Geographic range** Change in geographic range > 50 % of original.
Genetic structure	**3. Genetic structure** No detectable change in genetic structure. Unlikely to be detectable against background variability for this population.	**3. Genetic structure** Possible detectable change in genetic structure. Any change in frequency of genotypes, effective population size or number of spawning units up to 5%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units up to 10%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units up to 25%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units, change up to 50%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units > 50%.
Age/size/sex structure	**4. Age/size/sex structure** No detectable change in age/size/sex structure. Unlikely to be detectable against background variability for this population.	**4. Age/size/sex structure** Possible detectable change in age/size/sex structure but minimal impact on population dynamics.	**4. Age/size/sex structure** Impact on population dynamics at maximum sustainable level, long-term recruitment dynamics not adversely affected.	**4. Age/size/sex structure** Long-term recruitment dynamics adversely affected. Time to recover to original structure up to 5 generations free from impact.	**4. Age/size/sex structure** Long-term recruitment dynamics adversely affected. Time to recover to original structure up to 10 generations free from impact.	**4. Age/size/sex structure** Long-term recruitment dynamics adversely affected. Time to recover to original structure > 100 generations free from impact.
Reproductive capacity	**5. Reproductive capacity** No detectable change in reproductive capacity. Unlikely to be detectable against background variability for this population.	**5. Reproductive capacity** Possible detectable change in reproductive capacity but minimal impact on population dynamics.	**5. Reproductive capacity** Impact on population dynamics at maximum sustainable level, long-term recruitment dynamics not adversely affected.	**5. Reproductive capacity** Change in reproductive capacity adversely affecting long-term recruitment dynamics. Time to recovery up to 5 generations free from impact.	**5. Reproductive capacity** Change in reproductive capacity adversely affecting long-term recruitment dynamics. Time to recovery up to 10 generations free from impact.	**5. Reproductive capacity** Change in reproductive capacity adversely affecting long-term recruitment dynamics. Time to recovery > 100 generations free from impact.

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
Behaviour/movement	**6. Behaviour/ movement** No detectable change in behaviour/ movement. Unlikely to be detectable against background variability for this population. Time taken to recover to pre-disturbed state on the scale of hours.	**6. Behaviour/ movement** Possible detectable change in behaviour/ movement but minimal impact on population dynamics. Time to return to original behaviour/ movement on the scale of days to weeks.	**6. Behaviour/ movement** Detectable change in behaviour/ movement with the potential for some impact on population dynamics. Time to return to original behaviour/ movement on the scale of weeks to months.	**6. Behaviour/ movement** Change in behaviour/ movement with impacts on population dynamics. Time to return to original behaviour/ movement on the scale of months to years.	**6. Behaviour/ movement** Change in behaviour/ movement with impacts on population dynamics. Time to return to original behaviour/ movement on the scale of years to decades.	**6. Behaviour/ movement** Change to behaviour/ movement. Population does not return to original behaviour/ movement.

2. **Table A6.2.** A guide for scoring the level of consequence for by-catch and byproduct species. (From Hobday *et al.* 2004)

3.

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
Population size	**1. Population size** Insignificant change to population size/growth rate (r). Unlikely to be detectable against background variability for this population.	**1. Population size** Possible detectable change in size/growth rate (r) but minimal impact on population size and none on dynamics.	**1. Population size** No information is available on the relative area or susceptibility to capture/ impact or on the vulnerability of life history traits of this type of species Susceptibility to capture is suspected to be less than 50% and species do not have vulnerable life history traits. For species with vulnerable life history traits to stay in this category susceptibility to capture must be less than 25%.	**1. Population size** Relative state of capture/susceptibility suspected/known to be greater than 50% and species should be examined explicitly.	**1. Population size** Likely to cause local extinctions if continued in longer term	**1. Population size** Local extinctions are imminent/immediate
Geographic range	**2. Geographic range** No detectable change in geographic range. Unlikely to be detectable against background variability for this population.	**2. Geographic range** Possible detectable change in geographic range but minimal impact on population range and none on dynamics, change in geographic range up to 5 % of original.	**2. Geographic range** Change in geographic range up to 10 % of original.	**2. Geographic range** Change in geographic range up to 25 % of original.	**2. Geographic range** Change in geographic range up to 50 % of original.	**2. Geographic range** Change in geographic range > 50 % of original.
Genetic structure	**3. Genetic structure** No detectable change in genetic structure. Unlikely to be detectable against background variability for this population.	**3. Genetic structure** Possible detectable change in genetic structure. Any change in frequency of genotypes, effective population size or number of spawning units up to 5%.	**3. Genetic structure** Detectable change in genetic structure. Change in frequency of genotypes, effective population size or number of spawning units up to 10%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units up to 25%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units up to 50%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units > 50%.
Age/size/sex	**4. Age/size/sex structure** No detectable change in	**4. Age/size/sex structure** Possible detectable change	**4. Age/size/sex structure** Detectable change in	**4. Age/size/sex structure** Long-term recruitment	**4. Age/size/sex structure** Long-term recruitment	**4. Age/size/sex structure** Long-term recruitment

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
structure	age/size/sex structure. Unlikely to be detectable against background variability for this population.	in age/size/sex structure but minimal impact on population dynamics.	age/size/sex structure. Impact on population dynamics at maximum sustainable level, long-term recruitment dynamics not adversely damaged.	dynamics adversely affected. Time to recover to original structure up to 5 generations free from impact.	dynamics adversely affected. Time to recover to original structure up to 10 generations free from impact.	dynamics adversely affected. Time to recover to original structure > 100 generations free from impact.
Reproductive capacity	**5. Reproductive capacity** No detectable change in reproductive capacity. Unlikely to be detectable against background variability for this population.	**5. Reproductive capacity** Possible detectable change in reproductive capacity but minimal impact on population dynamics.	**5. Reproductive capacity** Detectable change in reproductive capacity, impact on population dynamics at maximum sustainable level, long-term recruitment dynamics not adversely damaged.	**5. Reproductive capacity** Change in reproductive capacity adversely affecting long-term recruitment dynamics. Time to recovery up to 5 generations free from impact.	**5. Reproductive capacity** Change in reproductive capacity adversely affecting long-term recruitment dynamics. Time to recovery up to 10 generations free from impact.	**5. Reproductive capacity** Change in reproductive capacity adversely affecting long-term recruitment dynamics. Time to recovery > 100 generations free from impact.
Behaviour/movement	**6. Behaviour/ movement** No detectable change in behaviour/ movement. Unlikely to be detectable against background variability for this population. Time taken to recover to pre-disturbed state on the scale of hours.	**6. Behaviour/ movement** Possible detectable change in behaviour/ movement but minimal impact on population dynamics. Time to return to original behaviour/ movement on the scale of days to weeks.	**6. Behaviour/ movement** Detectable change in behaviour/ movement with the potential for some impact on population dynamics. Time to return to original behaviour/ movement on the scale of weeks to months.	**6. Behaviour/ movement** Change in behaviour/ movement with impacts on population dynamics. Time to return to original behaviour/ movement on the scale of months to years	**6. Behaviour/ movement** Change in behaviour/ movement with impacts on population dynamics. Time to return to original behaviour/ movement on the scale of years to decades.	**6. Behaviour/ movement** Change to behaviour/ movement. Population does not return to original behaviour/ movement.

4. **Table A6.3.** A guide for scoring the level of consequence for Threatened, Endangered or Protected species. (From Hobday *et al.* 2004)

5.

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
Population size	**1. Population size** Almost none are killed.	**1. Population size** Insignificant change to population size/growth rate (r). Unlikely to be detectable against background variability for this population.	**1. Population size**. State of reduction on the rate of increase are at the maximum acceptable level. Possible detectable change in size/ growth rate (r) but minimal impact on population size and none on dynamics of TEP species.	**1. Population size** Affecting recruitment state of stocks or their capacity to increase.	**1. Population size** Local extinctions are imminent/immediate	**1. Population size** Global extinctions are imminent/immediate
Geographic range	**2. Geographic range** No interactions leading to impact on geographic range.	**2. Geographic range** No detectable change in geographic range. Unlikely to be detectable against background variability for this population.	**2. Geographic range** Possible detectable change in geographic range but minimal impact on population range and none on dynamics. Change in geographic range up to 5 % of original.	**2. Geographic range** Change in geographic range up to 10% of original.	**2. Geographic range** Change in geographic range up to 25% of original.	**2. Geographic range** Change in geographic range up to 25% of original.
Genetic structure	**3. Genetic structure** No interactions leading to impact on genetic structure.	**3. Genetic structure** No detectable change in genetic structure. Unlikely to be detectable against background variability for this population.	**3. Genetic structure** Possible detectable change in genetic structure but minimal impact at population level. Any change in frequency of genotypes, effective population size or number of spawning units up to 5%.	**3. Genetic structure** Moderate change in genetic structure. Change in frequency of genotypes, effective population size or number of spawning units up to 10%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units up to 25%.	**3. Genetic structure** Change in frequency of genotypes, effective population size or number of spawning units up to 25%.
Age/size/sex structure	**4. Age/size/sex structure** No interactions leading to change in age/size/sex structure.	**4. Age/size/sex structure** No detectable change in age/size/sex structure. Unlikely to be detectable against background variability for this population.	**4. Age/size/sex structure** Possible detectable change in age/size/sex structure but minimal impact on population dynamics.	**4. Age/size/sex structure** Detectable change in age/size/sex structure. Impact on population dynamics at maximum sustainable level, long-term recruitment dynamics not adversely damaged.	**4. Age/size/sex structure** Severe change in age/size/sex structure. Impact adversely affecting population dynamics. Time to recover to original structure up to 5 generations free from	**4. Age/size/sex structure** Impact adversely affecting population dynamics. Time to recover to original structure > 10 generations free from impact

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
					impact	
Reproductive capacity	**5. Reproductive capacity** No interactions resulting in change to reproductive capacity.	**5. Reproductive capacity** No detectable change in reproductive capacity. Unlikely to be detectable against background variability for this population.	**5. Reproductive capacity** Possible detectable change in reproductive capacity but minimal impact on population dynamics.	**5. Reproductive capacity** Detectable change in reproductive capacity, impact on population dynamics at maximum sustainable level, long-term recruitment dynamics not adversely damaged.	**5. Reproductive capacity** Change in reproductive capacity, impact adversely affecting recruitment dynamics. Time to recover to original structure up to 5 generations free from impact	**5. Reproductive capacity** Change in reproductive capacity, impact adversely affecting recruitment dynamics. Time to recover to original structure > 10 generations free from impact
Behaviour/movement	**6. Behaviour/ movement** No interactions resulting in change to behaviour/ movement.	**6. Behaviour/ movement** No detectable change in behaviour/ movement. Time to return to original behaviour/ movement on the scale of hours.	**6. Behaviour/ movement** Possible detectable change in behaviour/ movement but minimal impact on population dynamics. Time to return to original behaviour/ movement on the scale of days to weeks	**6. Behaviour/ movement** Detectable change in behaviour/ movement with the potential for some impact on population dynamics. Time to return to original behaviour/ movement on the scale of weeks to months	**6. Behaviour/ movement** Change in behaviour/ movement, impact adversely affecting population dynamics. Time to return to original behaviour/ movement on the scale of months to years.	**6. Behaviour/ movement** Change in behaviour/ movement. Impact adversely affecting population dynamics. Time to return to original behaviour/ movement on the scale of years to decades.
Interaction with fishery	**7. Interactions with fishery** No interactions with fishery.	**7. Interactions with fishery** Few interactions and involving up to 5% of population.	**7. Interactions with fishery** Moderate level of interactions with fishery involving up to10 % of population.	**7. Interactions with fishery** Major interactions with fishery, interactions and involving up to 25% of population.	**7. Interactions with fishery** Frequent interactions involving ~ 50% of population.	**7. Interactions with fishery** Frequent interactions involving the entire known population negatively affecting the viability of the population.

6. **Table A6.4.** A guide for scoring the level of consequence for habitats. (From Hobday *et al.* 2004)

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
Substrate quality	**1. Substrate quality** Reduction in the productivity (similar to the intrinsic rate of increase for species) on the substrate from the activity is unlikely to be detectable. Time taken to recover to pre-disturbed state on the scale of hours.	**1. Substrate quality** Detectable impact on substrate quality. At small spatial scale time taken to recover to pre-disturbed state on the scale of days to weeks, at larger spatial scales recovery time of hours to days.	**1. Substrate quality** More widespread effects on the dynamics of substrate quality but the state are still considered acceptable given the percent area affected, the types of impact occurring and the recovery capacity of the substrate. For impacts on non-fragile substrates this may be for up to 50% of habitat affected, but for more fragile habitats, e.g. reef substrate, to stay in this category the % area affected needs to be smaller up to 25%.	**1. Substrate quality** The level of reduction of internal dynamics of habitats may be larger than is sensible to ensure that the habitat will not be able to recover adequately, or it will cause strong downstream effects from loss of function. Time to recover from local impact on the scale of months to years, at larger spatial scales recovery time of weeks to months.	**1. Substrate quality** Severe impact on substrate quality with 50 - 90% of the habitat affected or removed by the activity which may seriously endanger its long-term survival and result in changes to ecosystem function. Recovery period measured in years to decades.	**1. Substrate quality** The dynamics of the entire habitat is in danger of being changed in a major way, or > 90% of habitat destroyed.
Water quality	**2. Water quality** No direct impact on water quality. Impact unlikely to be detectable. Time taken to recover to pre-disturbed state on the scale of hours.	**2. Water quality** Detectable impact on water quality. Time to recover from local impact on the scale of days to weeks, at larger spatial scales recovery time of hours to days.	**2. Water quality** Moderate impact on water quality. Time to recover from local impact on the scale of weeks to months, at larger spatial scales recovery time of days to weeks.	**2. Water quality** Time to recover from local impact on the scale of months to years, at larger spatial scales recovery time of weeks to months.	**2. Water quality** Impact on water quality with 50 - 90% of the habitat affected or removed by the activity which may seriously endanger its long-term survival and result in changes to ecosystem function. Recovery period measured in years to decades.	**2. Water quality** The dynamics of the entire habitat is in danger of being changed in a major way, or > 90% of habitat destroyed.
Air quality	**3. Air quality** No direct impact on air quality. Impact unlikely to be detectable. Time taken to recover to pre-disturbed state on the scale of hours.	**3. Air quality** Detectable impact on air quality. Time to recover from local impact on the scale of days to weeks, at larger spatial scales recovery time of hours to days.	**3. Air quality** Detectable impact on air quality. Time to recover from local impact on the scale of weeks to months, at larger spatial scales recovery time of days to weeks.	**3. Air quality** Time to recover from local impact on the scale of months to years, at larger spatial scales recovery time of weeks to months.	**3. Air quality** Impact on air quality with 50 - 90% of the habitat affected or removed by the activity.which may seriously endanger its long-term survival and result in changes to ecosystem function. Recovery period	**3. Air quality** The dynamics of the entire habitat is in danger of being changed in a major way, or > 90% of habitat destroyed.

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
					measured in years to decades.	
Habitat types	**4. Habitat types** No direct impact on habitat types. Impact unlikely to be detectable. Time taken to recover to pre-disturbed state on the scale of hours.	**4. Habitat types** Detectable impact on distribution of habitat types. Time to recover from local impact on the scale of days to weeks, at larger spatial scales recovery time of hours to days.	**4. Habitat types** Detectable impact on distribution of habitat types. Time to recover from local impact on the scale of weeks to months, at larger spatial scales recovery time of days to weeks.	**4. Habitat types** Impact on distribution of habitat types. Time to recover from impact on the scale of months to years, at larger spatial scales recovery time of weeks to months.	**4. Habitat types** Impact on relative habitat abundance resulting in severe changes to ecosystem function. Recovery period measured in years to decades.	**4. Habitat types** The dynamics of the entire habitat is in danger of being changed in a major way, the distribution of habitat types has been catastrophically shifted away from original spatial pattern. If reversible, will require a long-term recovery period, on the scale of decades to centuries.
Habitat structure	**5. Habitat structure** No detectable change to the internal dynamics of habitat or populations of species making up the habitat. Time taken to recover to pre-disturbed state on the scale of hours.	**5. Habitat structure** Detectable impact on habitat structure. Time to recover from local impact on the scale of days to weeks, at larger spatial scales recovery time of hours to days.	**5. Habitat structure** Detectable impact on habitat structure. For impacts on non-fragile habitat structure this may be for up to 50% of habitat affected, but for more fragile habitats, to stay in this category the % area affected needs to be smaller up to 20%. Time to recover from local impact on the scale of weeks to months, at larger spatial scales recovery time of days to weeks.	**5. Habitat structure** The level of reduction of internal dynamics of habitats may be larger than is sensible to ensure that the habitat will not be able to recover adequately, or it will cause strong downstream effects from loss of function. For impacts on non-fragile habitats this may be for up to 50% of habitat affected, but for more fragile habitats, to stay in this category the % area affected up to 25%. Time to recover from impact on the scale of months to years, at larger spatial scales recovery time of weeks to months.	**5. Habitat structure** Impact on habitat structure resulting in severe changes to internal dynamics of habitats. Time to recover from impact on the scale of years to decades, at larger spatial scales recovery time of months to years.	**5. Habitat structure** The dynamics of the entire habitat is in danger of being changed in a major way. If reversible, will require a long-term recovery period, on the scale of decades to centuries.

7. **Table A6.5.** A guide for scoring the level of consequence for communities. (From Hobday *et al.* 2004)

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
Species composition	**1. Species composition** Interactions may be occurring which affect the internal dynamics of communities leading to change in species composition not detectable against natural variation.	**1. Species composition** Impacted species do not play a keystone role – only minor changes in relative abundance of other constituents. Changes of species composition up to 5%.	**1. Species composition** Detectable changes to the community species composition without a major change in function (no loss of function). Changes to species composition up to 10%.	**1. Species composition** Major changes to the community species composition (~25%) (involving keystone species) with major change in function. Ecosystem function altered measurably and some function or components are locally missing/declining/increasing outside of historical range and/or allowed/facilitated new species to appear. Recovery period measured in years.	**1. Species composition** Change to ecosystem structure and function. Ecosystem dynamics currently shifting as different species appear in fishery. Recovery period measured in years to decades.	**1. Species composition** Total collapse of ecosystem processes. Long-term recovery period required, on the scale of decades to centuries
Functional group composition	**2. Functional group composition** Interactions which affect the internal dynamics of communities leading to change in functional group composition not detectable against natural variation.	**2. Functional group composition** Minor changes in relative abundance of community constituents up to 5%.	**2. Functional group composition** Changes in relative abundance of community constituents, up to 10% chance of flipping to an alternate state/ trophic cascade.	**2. Functional group composition** Ecosystem function altered measurably and some functional groups are locally missing/declining/increasing outside of historical range and/or allowed/facilitated new species to appear. Recovery period measured in months to years.	**2. Functional group composition** Ecosystem dynamics currently shifting, some functional groups are missing and new species/groups are now appearing in the fishery. Recovery period measured in years to decades.	**2. Functional group composition** Ecosystem function catastrophically altered with total collapse of ecosystem processes. Recovery period measured in decades to centuries.
Distribution of the community	**3. Distribution of the community** Interactions which affect the distribution of communities unlikely to be detectable against natural variation.	**3. Distribution of the community** Possible detectable change in geographic range of communities but minimal impact on community dynamics change in geographic range up to 5 % of original.	**3. Distribution of the community** Detectable change in geographic range of communities with some impact on community dynamics Change in geographic range up to 10 % of original.	**3. Distribution of the community** Geographic range of communities, ecosystem function altered measurably and some functional groups are locally missing/declining/increasing outside of historical range. Change in geographic range for up to 25 % of the	**3. Distribution of the community** Change in geographic range of communities, ecosystem function altered and some functional groups are currently missing and new groups are present. Change in geographic range for up to 50 % of species including keystone species.	**3. Distribution of the community** Change in geographic range of communities, ecosystem function collapsed. Change in geographic range for >90% of species including keystone species. Recovery period measured in decades to centuries.

Sub-component	Score/level					
	1 Negligible	2 Minor	3 Moderate	4 Major	5 Severe	6 Intolerable
				species. Recovery period measured in months to years.	Recovery period measured in years to decades.	
Trophic/size structure	**4. Trophic/size structure** Interactions which affect the internal dynamics unlikely to be detectable against natural variation.	**4. Trophic/size structure** Change in mean trophic level, biomass/ number in each size class up to 5%.	**4. Trophic/size structure** Changes in mean trophic level, biomass/ number in each size class up to 10%.	**4. Trophic/size structure** Changes in mean trophic level. Ecosystem function altered measurably and some function or components are locally missing/declining/increasing outside of historical range and/or allowed/facilitated new species to appear. Recovery period measured in years to decades.	**4. Trophic/size structure** Changes in mean trophic level. Ecosystem function severely altered and some function or components are missing and new groups present. Recovery period measured in years to decades.	**4. Trophic/size structure** Ecosystem function catastrophically altered as a result of changes in mean trophic level, total collapse of ecosystem processes. Recovery period measured in decades to centuries.
Bio-geochemical cycles	**5. Bio- and geochemical cycles** Interactions which affect bio- & geochemical cycling unlikely to be detectable against natural variation.	**5. Bio- and geochemical cycles** Only minor changes in relative abundance of other constituents leading to minimal changes to bio- & geochemical cycling up to 5%.	**5. Bio- and geochemical cycles** Changes in relative abundance of other constituents leading to minimal changes to bio- & geochemical cycling, up to 10%.	**5. Bio- and geochemical cycles** Changes in relative abundance of constituents leading to major changes to bio- & geochemical cycling, up to 25%.	**5. Bio- and geochemical cycles** Changes in relative abundance of constituents leading to Severe changes to bio- & geochemical cycling. Recovery period measured in years to decades.	**5. Bio- and geochemical cycles** Ecosystem function catastrophically altered as a result of community changes affecting bio- and geo- chemical cycles, total collapse of ecosystem processes. Recovery period measured in decades to centuries.