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# Technical Reviews for the Commonwealth Harvest Strategy Policy


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# Contents

# 1 Executive summary

The following sections attempt to identify key points raised in each of the sections of this set of reviews. It should be noted that this material is diverse and relatively complex so, unfortunately, brief summaries of each section are not possible. The following are not conclusions but rather constitute important points that require noting.

## 1.1 Reference Points Appropriate to Life-History Characteristics

The range of suggestions for what would constitute an appropriate target biomass and fishing mortality value is very great but the difficulty in estimating the real risks of running relatively high fishing mortality rates at low stock sizes indicates that the suggestion of $B_{40\%}$ rather than something lower is a reasonable compromise. The current default biomass target reference point of $B_{48\%}$ would appear to be highly conservative (biologically) for many species, although it may be quite appropriate for slower growing sharks and rays and may not be sufficiently conservative for some key low trophic level species. For example, the Commonwealth small pelagic fishery, in line with a number of regulations world-wide, has adopted a biomass level of at least 80% $B_0$ as the $B_{LIM}$ for each species in this fishery (with higher values in the more data poor situations), and for such ecologically important species such apparent high levels seem appropriate. However, such a level would ignore the fact that such species are naturally highly variable and could quite naturally vary in abundance, sometimes down to very low abundance levels. An alternative could be not to accept a limit with reference to a fixed $B_0$ but rather to only take a standard proportion of available biomass. Such constant escapement strategies are not currently included in the HSP but would be useful for naturally highly variable species such as scallops, small pelagic species, and squid, for which the concept of a stable unfished biomass, $B_0$, may not be meaningful. Full implementation of this would thus mean that management of such stocks would not be in relation to specific biomass limit and target reference points but rather in relation to estimates of current stock size. In addition, such a strategy might need to include some minimum level of predicted harvest before fishing could occur so as to avoid encouraging unprofitable fishing.

For productive species where $0.5B_{\mathrm{MSY}}$ is less than $B_{20\%}$ the current HSP suggests that levels of biomass < $B_{20\%}$ would be acceptable. Given the uncertainty inherent in estimation of stock productivity, the precautionary approach would firstly require good evidence that $0.5B_{\mathrm{MSY}}$ is indeed below $B_{20\%}$. In the face of these various doubts and uncertainties it would be difficult to argue that there would be no increase in the risk of depletion affecting consequent recruitment levels if the limit biomass reference point was permitted to vary below the current $B_{20\%}$. For small pelagic fisheries, because of ecosystem based fishery management considerations the limit reference point would tend to be either the same as or very close to the target (which has similarities to having a constant escapement strategy.

## 1.2 Buffered Targets or Meta-Rules

The present arrangements where those harvest strategy control rules in which a break point is clearly defined at the proxy target reference point certainly stabilizes catches and another meta-rule that prevents TACs varying by more than 50% between any two years has also been helpful in preventing serious dislocation and disturbance in the fish-

ery for some relatively unstable species. These particular meta-rules have already been simulation tested using MSE.

If it was decided to pursue the issue of buffers and meta-rules around the targets in an attempt to stabilize catches through time then it would be beneficial to use simulation testing (MSE) to consider the effect of such changes to the expected dynamics of different fisheries.

## 1.3 Data Poor Fisheries and Tiered Harvest Strategies

We define fisheries or species as data poor if information is insufficient to produce a defensible quantitative stock assessment.

For data poor fisheries, difficulties can arise in almost every component of the harvest strategy – for example, little or no regular monitoring means time series are rare, the assessment method is undertaken with an unknown degree of uncertainty, reference points are poorly defined and the associated control rules do not necessarily address risk clearly. Yet, a recognized component of the present Harvest Policy is the application of a consistent degree of risk across all fisheries, irrespective of fishery type.

Often the efficacy of a data poor harvest strategy can be very fishery specific. The use of a tiered system of assessment methods and associated control rules allows for the development of detailed, integrated stock assessments (Tier 0 and 1) down to the lowest Tiers where data is limited to catch rates, catches, or even just catches (Tiers 6 and 7). Below these tiers is the Ecological Risk Assessment, which aims to determine whether there are particular species that are exceptionally vulnerable to the effects of fishing.

## 1.4 TAC Setting and Multi-Year TACs

Generally, when TACs are set for individual species, catches of other species are not considered. In multi-species fisheries, there are often technological interactions where fishing effort directed towards one quota species will normally result in a mixed catch of fish that may include other quota species. Fishers can usually ‘target’ to some degree through fishing different areas and depths, seasons, times of day and by modifying gear. But it is the degree to which fishers can target that is the issue. The species mix in catches may not necessarily match the mix in combined TACs or in quota holdings. This difficulty in balancing quotas for multiple species with actual catches may then lead to increased discarding, TAC over-runs, effort restrictions or fishery closures when quota is constrained on some species. It is possible to characterize recent multispecies catch data into primary and companion components. The approach of identifying companion species within a given fishery provides an empirical means to examine the impact of individual species TAC decisions across all of the quota species in a fishery.

In general, multi-year TACs will require a “discount” (reduction) of some level of catch to balance the greater risk associated with less frequent review and adjustment. There are obvious risks of stock depletion if the multi-year TACs are set too high. While there is debate about how best to set multi-year TACs no decisions have yet been made. Currently there has been little testing of the robustness of fisheries to the application of multi-year TACs.

## 1.5 Rebuilding Strategies and Bycatch-only TACs

A primary objective of the Commonwealth Harvest Strategy Policy (HSP) is to maintain key commercial fish stocks at ecologically sustainable levels and within that context, maximize the economic returns to the Australian community. If a fishery falls below the default limit reference point of $B_{20\%}$ the HSP states that: "Typically recovery times are defined as the minimum of 1) the mean generation time plus ten years, or 2) three times the mean generation time." However, attempting to meet these guidelines has been problematic, for example, in at least three conservation dependent species in the SESSF.

The HSP already states that not all species in a multi-species fishery need be maintained at the target reference point (default of $B_{48\%}$ as a proxy for $B_{MEY}$) as long as all assessed species stay above the limit reference point. So the rebuilding target for each species is not always clear.

The HSP makes the assumption that rebuilding of a depleted species will always occur. However, in a changing marine environment this may not always be true. Potential regime shifts have already been identified in particular species (Jackass Morwong) on Australia's east coast (a world hot spot for sea water temperature rise) and this provides an example of a species whose long term productivity has declined. There is thus a need to recognize that there are circumstances under which rebuilding to previously experienced levels would not be expected to occur.

It is also possible that some species, particularly when they were fished under a basket species category (e.g. gulper sharks) may have been reduced to such a low level that the probability of them recovering would become influenced by random events. In addition, if the projected timeline for recovery is extremely long it becomes possible that long term changes in the marine environment will become influential on the probability of eventual recovery.

Finally, there are some species which are naturally extremely variable (e.g. squid and scallops). Simulation testing can be used, and has been used, to demonstrate that the harvest strategies in place are potentially capable of achieving the intent of the HSP, even though it is very hard to identify adequate proxies for a particular limit or target biomass reference point. However, some unpredictable events, such as the recent almost complete die-off of scallop beds in south-east Australia, unrelated to any fishing, are not amenable to anything other than reactive management.

## 1.6 Spatial Management

Spatial management may be applied in various contexts within a harvest strategy. It can form the main harvest strategy framework (such as in a system of rotational closures), it can be used to augment a harvest strategy framework, or spatial management measures can be invoked as a control rule (a variation of rotational closures). For some species a management scheme that controls fishing mortality with large spatial and temporal fishery closures offers a management strategy more robust to uncertainty than direct control of catch, since only a small component of the stock gets exposed to the fishery. However, this relies on good compliance with fixed closure boundaries (the Commonwealth Vessel Monitoring System ensures this) and is mainly applicable to species that do not move large distances.

# 2 Introduction

## 2.1 Document Structure

The Commonwealth Harvest Strategy Policy (HSP), and the guidelines for its application, provides a management framework that uses evidence based methods when assessing individual fish stocks and then applying a risk-based, precautionary approach to the setting of harvest levels controlled by effort or catch for each stock. One of the reasons for its implementation was to provide "… the fishing industry and other stakeholders with a more certain operating environment where management decisions for key species are more consistent, predictable and transparent." (DAFF, 2007, Minister's Foreword, p iii).

The HSP and Guidelines is a complex document with many facets and these technical reviews reflect this in their scope and in their details. There are a number of reviews with separate headings and mostly separate subject matter but there is an unavoidable element of overlap between some subjects because of the inter-relationships between the sections. This technical review document is composed of eight sections each providing the details for each of the subject matters covered. However, the main conclusions are extracted from the text and placed under sections in the executive summary. A ninth section relating to Alternative Economic Targets and Reference Points will be presented as a separate report.

The eight sections relate to: 1) this introduction, including an introduction to fisheries and harvest strategies, 2) reference points appropriate to life-history characteristics, 3) buffered targets, 4) data-poor fisheries and tiered harvest strategies, 5) TAC setting and multi-year TACs, 6) rebuilding strategies and bycatch-only TACs, 7) assessing byproduct species, and 8) spatial management and metarules.

Two other sections in this document deal with "Other Issues" and with research projects potentially valuable to the HSP and its further development.

## 2.2 Objectives for Fisheries

For a range of reasons the management of natural fisheries resources is a difficult problem everywhere fishing occurs. The fundamental problem of fisheries management is that instead of being able to measure the status of different harvested stocks directly it is only possible to infer their status from samples, which usually only provide an uncertain view of a stock. While the development of time-series of fishery observations (such as catches, catch rates, age-structure data, and many others) can improve our understanding of events (if the quality and representativeness of such data is good enough) there always remains a degree of uncertainty in any assessment. In addition, there are also many data-poor or data-limited fisheries and species globally (Vasconcellus and Cochrane 2005; Pikitch 2012). Nevertheless, fishery managers are required to make decisions in the face of that uncertainty. Unfortunately, this uncertainty and its implications have not always been recognized though now, around the world, the countries with the most effective fisheries management attempt to account for uncertainty in an explicit fashion.

The history of fisheries management documents the movement away from not realizing that management of these natural resources was required through to the current environment of a wide array of management approaches in use in different fisheries around the World (Smith, 1988; Hilborn, 2012). A major change through time relates to the objectives which systems of fisheries management attempt to achieve. When declines in large fisheries were first identified at the end of the 19th century the concerns that arose involved a combination of wanting to maintain catch rates (to fish economically) and to maximize the yields from different fisheries (Garstang, 1900). At that time the primary objective was to maximize yield but it took some years before it was recognized that for many species applying more fishing effort did not necessarily lead to increased catches (the yield-per-recruit problem; Russell, 1931, Beverton & Holt, 1957). It may be difficult now to grasp the simplistic view of how to manage fisheries that existed in the 1910s right up to the 1970s but serious attention, acted on at national levels, was only paid to fisheries dynamics and management from the late 1950s onwards. Prior to the late 1950s most thought was given to increasing catches and the efficiency of fishing gear and it still seemed contrary to intuition to recommend limiting catches. At the second FAO conference in 1946 the FAO, for example, was strongly urging the development of fisheries as a source of protein and food: "The fishing grounds of the world are teeming with fish of all kinds. Fisheries are an international resource. In underdeveloped areas especially, the harvest awaits the reaper." (FAO, 1985).

Early stock assessment approaches effectively ignored uncertainty and tended to produce deterministic management advice based on the assumption that natural populations are in equilibrium with each other and with any fishing effort imposed on them (Schaefer, 1954, 1957; Gulland, 1965; Megrey, 1989). These assumptions of stability were clearly invalid in many cases but nevertheless this approach led to concepts such as the Maximum Sustainable Yield (MSY), which related to catch levels, and $F_{MAX}$, the fishing mortality which related to the effort expected to lead to the maximum yield; which would often be larger than the MSY. Both these concepts were early fisheries targets or objectives with fisheries legislation in many countries including the achievement of MSY as the aim of management; though generally, that same legislation neglected to define the concept of MSY. How to achieve such objectives was rarely made explicit. In the 1970s it became apparent, following the collapse of a number of fish stocks, that MSY, as it was then interpreted, was not necessarily the safest objective to adopt (Larkin, 1977) and more serious efforts were made to find alternatives although the concept of MSY is still used but has evolved into use as an upper limit to fishing mortality or has been redefined to account for risks of alternative catch levels (Smith and Punt, 2001). In the 1970s and early 1980s, input controls relating to effort, gear, vessel numbers, and closed seasons were the management tools in most fisheries and some of the more successful management objectives focussed on defining an optimum fishing mortality rate. This work led to the concept of $F_{0.1}$, which was an effectively *ad hoc* advance over $F_{MAX}$ in terms of sustainability as well as profitability as it usually led to a large reduction in fishing effort (reduction in fishing mortality) but only led to a minor loss in yield (Hilborn & Walters, 1992). Even though this was an improvement over $F_{MAX}$ or $F_{MSY}$ it was still based on the notion that fish stocks were able to achieve equilibrium with the fishing mortality imposed on them. While this was well known to be an approximation there was still a great deal of development needed to produce the methodologies required for taking uncertainty into account.

The importance of acting to provide management advice in the face of uncertainty was a growing theme in fisheries resource management through the late 1980s and early 1990s; the need to act before scientific consensus could be achieved rather than calling for more research was identified as a key problem for management (Ludwig *et al*., 1993); this notion of not using a lack of scientific certainty about the risk of serious environmental damage as an excuse for not acting to prevent that damage is the basis of the precautionary approach in fisheries (FAO 1995, 1996).

## 2.3 Explicit Recognition of Harvest Strategies

As stock assessments were becoming more sophisticated so were the management options that were developed. In the late 1980s and early 1990s the effects of variability, uncertainty, and associated risks began to be addressed in stock assessments (Francis, 1992) and the notion of presenting a table of management options with their associated risks was also developed. Hilborn & Walters (1992, p453) defined a harvest strategy as "…a plan stating how the catch taken from a stock will be adjusted from year-to-year depending upon the size of the stock, the economic or social conditions of the fishery, conditions of other stocks, and perhaps the state of uncertainty regarding biological knowledge of the stock." The harvest strategies discussed at that time revolved mainly around the classical three: constant catch (e.g. TACs; output controls), constant fishing mortality (e.g. $F_{0.1}$; input controls), and constant escapement (e.g. always leaving at least 75% of estimated Mackerel Icefish biomass in the Heard and McDonald Island fishery; mixed input and output controls). There are at least three modifications or alternatives to the classical three harvest strategies. The first would involve periodic or pulse fishing, which, as the name implies, entails only fishing a stock or region at intervals (e.g. rotational harvesting is effectively pulse fishing, such as used recently in scallops; Harrington *et al.*, 2007; Haddon, 2011). The second modification to a classical harvest strategy would entail taking into account the economics of the fishery and perhaps trying to optimize profitability rather than yield. Finally, the third alternative harvest strategy would entail adding details that account for aspects of the species' biology to other harvest strategies (this is only considered an alternative because such actions can often dominate the control of fishing). Examples include sex selective fishing (e.g. only male mud crabs can be taken in Queensland) and size limits that exclude a significant proportion of mature females (e.g. size limits in scallops in Bass Strait and minimum size of Bugs in the northern prawn fishery).

Harvest strategies in the early 1990s focused mainly on setting out fishery objectives (defining biological reference points; Smith *et al.*, 1993) and what constraints should be used. In more recent parlance, this was about determining how to assess each stock's status and what limit reference points to put in place. This may have been driven, at least in part, by new legislation in the USA that required definitions of overfishing that would explicitly guard against recruitment overfishing (Mace & Sissenwine, 1993)

A number of very influential documents were published by the FAO in the mid-1990s, including: the *Code of Conduct for Responsible Fisheries* (FAO, 1995), the *Precautionary Approach to Capture Fisheries* (FAO, 1996), and *Fisheries Management* (FAO, 1997); these latter two documents being parts of the *Technical Guidelines for Responsible Fisheries* series. The authors stated: "Long term management objectives should be translated into management actions, formulated as a fishery management plan or other management framework" (FAO, 1995, p 11). Giving more details, the *Guidelines* appear to be one of the first documents to describe the components of what are now

termed Harvest Strategies. Thus it identified the needs for *targets*, described as the desired outcomes for a fishery, *operational constraints or limits*, described as the undesirable outcomes that are to be avoided, and *control rules* which specify in advance what action should be taken when specified deviations from the operational targets and constraints are observed (FAO, 1996). Early work on simulation testing of management arrangements (now known as management strategy or procedure evaluation) appears to have contributed to this approach to describing harvest or management strategies. Thus, in the *FAO Guidelines* it defines a *management procedure* as a description of the data to collect, how to analyze it, and how the analysis translates into actions. This is a standard way to describe a modern harvest strategy: define the data needed, the analysis of status, and the control rules used to generate management advice; however, in the *guidelines* the emphasis that was given to *management procedures* was placed on the investigation of how uncertainties influenced the management process (which stemmed from how these management procedures were implemented in South Africa; Butterworth & Bergh, 1993).

The main difference brought about by the adoption of formal harvest strategies was the inclusion of explicit decision (control) rules. Prior to the introduction of harvest strategies the data required for stock assessments was certainly collected and the primary thrust of research was the development and articulation of improved stock assessment methodology. With the addition of formal control rules, management responses become predetermined based on the outcome of the assessment. The control rules in the Australian HSP represented a major change to the management of Commonwealth fisheries and constitute the primary basis for improving the consistency, predictability, and transparency of management that the Minister spoke of in 2007 (DAFF, 2007).

## 2.4 Ecosystem Based Fisheries Management

In addition to pointing the way to what was required for the responsible management of fisheries the importance of taking into account the ecosystem effects of fishing, such as bycatch and habitat modification was identified in both the *FAO Code* and the *FAO Guidelines*. This was generally expressed in terms of using the precautionary approach to avoid unrecoverable damage to stocks and related ecosystems (Garcia, 1994; FAO, 1995, 1996, 1997).

Formal fisheries management policies have been proposed, and in some cases adopted, by a range of countries such as Australia, the USA, New Zealand, South Africa, and Europe. Each has included the major aspects of ecosystem based fisheries management as an important component within the proposed systems (DAFF, 2007; Ministry of Fisheries, 2008; US Department of Commerce, 2007). Most of these pieces of legislation were preceded by earlier fishery acts that included EBFM as directly relevant. Thus, the US Magnuson-Stevens Act was preceded by the Sustainable Fisheries Act in 1996, at the time when many of these changes recognizing the broader context in which fisheries operate were being formally adopted. Nevertheless, it was only more recently that more emphasis has been placed on EBFM.

Ecosystem Based Fishery Management is, however, very difficult to put into detailed practice. In practice, in many instances, EBFM is being implemented as an evolutionary extension of conventional fisheries management and entails single species stock assessments combined with sometimes detailed considerations of any bycatch, which may

include a full ecological risk assessment (especially of threatened and endangered species), and the potential interactions of the fishing gear used with physical habitats (Pikitch et al, 2004; Haddon, 2007). This, however, remains a great improvement over simply ignoring the issue and neglecting these potentially important contributors to the retention of the ecosystems supporting the fisheries. A more recent manifestation of EBFM within Australian fisheries involves setting more conservative reference points for species of ecological importance, such as low trophic level species (Smith *et al*., 2011; Pikitch *et al*., 2012).

## 2.5 Australia's Limit and Target Reference Points

Each country with a formal fisheries management system of harvest strategies has implemented them in ways that suit their own particular collection of circumstances. Australia, for example, is characterized by numerous different fisheries but none are particularly large by world standards. This is a reflection of Australia's geographical location and great age. Australia has fisheries ranging from the tropics, such as indigenous hunting for dugongs in the Torres Straits, about 10° south, to industrial fishing for sub-Antarctic Patagonian toothfish around Macquarie Island at about 54° 30" south. The generally low productivity of Australian fisheries reflects the low run-off of nutrients from the generally dry and previously eroded continent, the fact that most major coastal current systems flow south from nutrient-poor tropical waters, and finally the small number of permanent areas of upwelling from deeper nutrient rich waters (Haddon, 2007). This diverse range of fisheries constitutes a serious challenge to the specification of a Harvest Strategy Policy that can apply to all.

The selection of the particular limit and target reference points for the Australian Commonwealth's fisheries, that form the foundation of the HSP, differs in some respects from practice elsewhere. The selection of $B_{20\%}$ as the limit reference point reflects earlier literature. The earliest reference to this Limit Reference Point depletion level of 20%$B_0$ appears to be Beddington & Cooke (1983). Their analyses, looking at potential yields from different stocks, were given a constraint such that:

> *"… an escapement level of 20% of the expected unexploited spawning stock biomass is used. This is not a conservative figure, but it represents a lower limit where recruitment declines might be expected to be observable. … We have chosen a twenty year period in which to investigate the probability that the escapement will fall below the 20% level. … In presenting the results of this analysis, we have calculated the appropriate level of catch, that will ensure that the probability that the SSB falls below 20% of its unexploited level is less than 0.1" (Beddington & Cooke, 1983, p9-10; this approximates the statements on* $B_{LIM}$ *in the HSP, p4)*

Myers *et al* (1994) examined the stock recruitment relationships of 72 different fish stocks in an effort to determine a workable depletion level limit or threshold that would prevent recruitment overfishing in most cases. They concluded that in relation to methods that used estimates of $B_{20\%}$: "… based on both empirical and theoretical considerations we do not recommend them for general use." (Myers *et al*., 1994, p 204). Instead of using $B_{20\%}$ as a threshold beyond which the risk of recruitment overfishing was unacceptably high they suggested using 50% $R_{MAX}$ (the maximum average recruitment), however, they were using very poor methods to estimate the unfished biomass, which in turn gave poor estimates of $B_{20\%}$.

The most influential document giving rise to the notion that $B_{20\%}$ is a reasonable depletion level to use as an indicator of potential recruitment overfishing was a document prepared for the NMFS in the USA (Restrepo *et al*., 1998). In fact, they recommend ½ $B_{MSY}$ but consider $B_{20\%}$ to be an acceptable replacement for that figure. However, it is important to note that this is only a 'rule of thumb' and there is no empirical basis that links the proxy $B_{LIM}$ of $B_{20\%}$ and $0.5B_{MSY}$. Indeed, selecting $0.5B_{MSY}$ for some species could result in $B_{LIM}$ much lower than 20%. Nevertheless, this relationship and proxy has been adopted in Australia.

It is in the selection of the maximum economic yield as the explicit target reference point where the Australian commonwealth is unusual; this is discussed in policy documents from other places, usually pointing out that MEY requires a lower fishing mortality rate, less yield, but higher profitability. Despite this recognition and often setting targets that are more conservative than using MSY, explicitly setting MEY as the target is uncommon. Elsewhere there has been discussion and attention paid to setting the targets by considering the risk of falling below the limit reference point. In Australia the strategy is to "...ensure that the stock stays above the limit biomass level at least 90% of the time." (DAFF, 2007, p 4) This suggests a probabilistic approach to setting targets. Caddy and Mahon, 1995 and Caddy and McGarvey (1996) described methods, improved on by Prager *et al*. (2003), for estimating a suitable target reference point that should prevent the particular stock involved from breaching the selected limit reference point with a probability equal to that chosen.

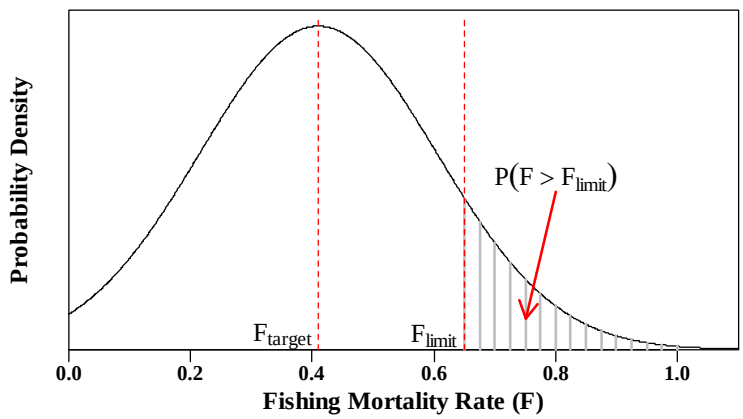


**Figure 1.** The probability density function describing the expected variation in annual fishing mortality for a given fishery and the relationship between the target and limit reference points. After selecting a given set of limit reference points (possibly $F_{0.1}$ or $F_{MSY}$), a search is made for the target fishing mortality that produces the pre-specified probability of falling below the limit reference point (after Caddy and Mahon, 1995). Prager *et al.* (2003) improved this by including uncertainty in the estimation of the limit reference point but the basic idea of having a probability density function around the target reference point which is defined by selecting the long-run probability of staying above the selected limit reference point is common to these approaches.

### 2.5.1 GENERAL APPLICABILITY OF REFERENCE POINTS

The Commonwealth HSP simply states that the limit reference point will not be breached with a probability > 0.1. Despite this requirement it also selects $B_{40\%}$ as a default proxy for MSY and a target of $B_{48\%}$ as a proxy for the target reference point of MEY, with no reference to whether or not this will achieve the stated risk level of falling below $B_{LIM}$ or even be far more conservative. Currently there is no operational way

to estimate the rates expected for the management imposed to breach the limit except to develop a mathematical projection model of the stock dynamics. This is done, for example, in the sub-Antarctic Patagonian toothfish fisheries (*Dissostichus eleginoides*) where the CCMALR control rule is specified in terms of the expected outcomes following 35 years of projecting the proposed management regime forward (the spawning biomass must be at or above $B_{50\%}$ (50% escapement) after 35 years with a < 10% chance of falling below $B_{20\%}$). While this has been translated in the Australian context into a manner consistent with the Australian Commonwealth Harvest Strategy it is intrinsically difficult to translate a control rule based on a projected future status into one based on the most recent status relative to reference limits and targets. This means that currently the requirement of not falling below the limit reference point more than 10% of the time is useful only when testing harvest strategies using Management Strategy Evaluation, or if projections are added to the assessments (which are not part of the current harvest strategies endorsed by the harvest strategy policy that only consider the next year's catch or fishing mortality.

The CCAMLR rule applied to Patagonian toothfish aims to achieve a target of 50% of spawning biomass rather than 48%; in other ways too (the scale of MPAs in both the Macquarie Island and Heard and McDonald Islands fisheries, and the level of observer coverage) it exceeds the expectations of the HSP. Despite these advantages before such a harvest strategy can be accepted in the current HSP framework it is necessary to conduct a management strategy evaluation to demonstrate that this alternative management strategy is at least as capable of achieving the intent of the HSP for the fisheries concerned. Even after this further analysis has been done, in practice such strategies need to be translated, sometimes artificially, into terms consistent with the explicit structure of the HSP. The explicit requirements of the HSP are not sufficiently broad to allow direct acceptance of alternative criteria for successful management. The HSP currently requires Commonwealth fisheries to be managed based on reference points that relate the present estimates of fishing mortality or spawning biomass (or their proxies) to the unfished state (in particular it involves the concept of $B_0$ the equilibrium unfished spawning biomass). Concepts such as a constant escapement, either now or at some projected future date, if correctly applied, are perfectly capable of managing a fishery to achieve the intent of the current HSP but are not currently part of the HSP.

The lack of this recognition is a problem for Australian fisheries on internationally distributed species (e.g. Patagonian toothfish and various tuna species) as well as a few difficult to manage highly variable Australian species. For example, with extremely variable species such as Bass Strait scallops and squid the concept of unfished biomass ($B_0$) does not appear to have any meaningful interpretation. Haddon (2011), in an evaluation of scallop management strategies, interpreted the regulation of having at least 40% of viable areas closed to fishing at all times (with at least 500t of biomass) as being a spatially explicit proxy for the $B_{LIM}$ limit reference point. This aims to achieve the intent of the sustainability objective. While this spatial proxy does not relate to any notion of $B_0$ or of $0.5B_{MSY}$, it is a pragmatic way forward within the HSP. There are control rules for when to allow fishing in a scallop bed (there must be < 20% under the legal size), but defining a suitable target for scallops remains difficult:

> *The target for the fishery might be characterized as aiming to have a fishery each year and to achieve a catch level that matches the processor and market capacity. The first rule* [minimum size requirement] *acts to maintain profitability by avoiding waste and*

> *focusing on the larger scallops that generate a higher yield of scallop meat for a given number of scallops processed; in this manner the objective of achieving the most profitable fishery is approached, but this is difficult to interpret as a specific target. (Haddon, 2011, p 20)*

The fundamental idea of Limit and Target reference points assumes that there is such a thing as a long term average or that fishing mortality can be considered as continuous through time. Fishing a scallop bed usually means completely depleting it to low levels, while other beds are left alone. Fishing mortality is thus relatively episodic in such species as the stock size tends to step down in jumps rather than smoothly declining (it also increases in jumps as new beds establish). While in principle such reference points might be thought reasonable if a long enough time period was considered it also seems reasonable that the time period over which stock dynamics should be averaged should be related to how rapidly management needs to react to stock changes. Even with such idiosyncratic management arrangements as those used in the Bass Strait scallops, which still attempt to meet the intent of the HSP, it is not possible to predict events such as virtually the whole Bass Strait stock (probably > 20,000 tonnes) dying off in only a few months, as happened in 2011. Such difficulties might be alleviated if some means was developed, other than time consuming and expensive approaches such as MSE, which could lead to the certification of alternatives to a strict interpretation of the HSP. Alternatively, a wider range of acceptable harvest strategy objectives and control rules, such as the inclusion of a constant escapement strategy into the HSP, might achieve the same aim.

# 3 Reference Points and Life-History Characteristics

## 3.1 MEY and MSY Proxies

**Key Questions from the Discussion Document**

- *…whether the Guidelines should be revised to strengthen and clarify advice relating to:*
  - *the selection and use of alternative proxy target reference points (other than the default proxies already defined in the Policy), taking into account the differing productivities and biological characteristics of various species and species groups*
  - *whether fine scale adjustments (e.g. B48 v B51) are justifiable.*

and

- *…whether there is a need to review and/or develop further advice within the Guidelines on the selection and use of limit reference points, to ensure consistency with the Policy objectives. Stakeholders may give consideration to the following questions:*
  - *Is the proxy setting in the Policy and Guidelines of $0.5B_{MSY}$ appropriate, given that for some species this implies an actual $B_{LIM}$ of less than $B_{20}$? Should a more conservative approach be taken in which $B_{LIM}$ is generally constrained to a value equal or greater than $B_{20}$, except where a scientifically defensible case can be made for a lower value.*
  - *Similarly, should alternatives be considered for groups/species on the basis of productivity (e.g. chondrichthyans) or ecological role (e.g. small pelagic fish) and how might these be determined?*

The concept of maximum sustainable yield (MSY) has a long history, beginning in the 1930's with Russell (1931), who discussed the notion of whether it was possible to maintain a maximum catch from a fishery, Hjort et al. (1933), whose publication was entitled "The Optimum Catch", and Graham (1935) who graphically described a yield curve as the rate of change (the production) of a fished population (see Smith, 1994, for a detailed history). The use of equilibrium surplus production models for stock assessment in the 1950's enabled and led to major fisheries management organisations adopting MSY as a fisheries management target (Schaefer, 1954, 1957; Mace, 2001; Smith and Punt, 2001). The scientific community began to question the use of MSY as a management target in the 1970's (e.g. Larkin 1977, Sissenwine 1978). At that time it was realised that a static MSY based on a theory that assumed the fishery was in equilibrium with fishing effort was generally not an appropriate management target because fish populations naturally fluctuate, and cannot produce equilibrium fixed catches in the long-term.

Density dependent recruitment compensation (i.e. stock recruitment steepness), where survivorship of juveniles increases as stock size declines, operates to offset the losses of

individuals from a population as the population is reduced naturally or due to fishing and this therefore acts to stabilize the population. This phenomenon must exist to allow naturally stable populations to exist under harvesting, and is the basis for concepts such as surplus production and sustainable harvest (Rose et al. 2001).

Current fisheries management uses MSY more generally in terms of a dynamic fishing mortality rate, $F_{\mathrm{MSY}}$, which should achieve MSY; $F_{MSY}$ is now more generally used as a threshold beyond which fishing mortality should be reduced (Mace, 2001). Many proxies for $F_{\mathrm{MSY}}$ have been developed, for example $F_{0.1}$, $F_{\max}$, $F_{30\%}$ and $F_{40\%}$ (different target fishing mortality rates some of which derive from yield per recruit calculations, and others that have a more empirical origin). Of particular interest for the Commonwealth Harvest Strategy Policy (HSP) is $F_{40\%}$, the harvest rate that would result in the spawning stock biomass (SSB) being reduced to 40% of the virgin level ($B_0$). Clark (1993) showed, using simulations, that for a range of groundfish species, a reliably high annual yield could be achieved by fishing at $F_{40\%}$, which allows for some variability in recruitment; even when that recruitment was serially correlated (periods of low or high recruitment). Fishing at $F_{40\%}$ instead of $F_{35\%}$ didn't change the predicted yield by much but reduced the number of times the stock approached a limit of $B_{20\%}$, set by Clark as a threshold to indicate overfishing and which became a far more widely accepted rule-of-thumb.

Our familiar harvest control rule diagrams with spawning stock biomass (SSB) on the X axis and fishing mortality ($F$) on the Y axis derive from earlier work such as Serchuk et al. (1997) and Restrepo et al. (1998). Overfishing is indicated by fishing at $F>F_{\mathrm{MSY}}$ (or $F_{\mathrm{MSYproxy}}$), and the stock is considered overfished at $0.5B_{\mathrm{MSY}}$, or $0.5B_{\mathrm{MSYproxy}}$. Our current HSP default proxy for $F_{\mathrm{MSY}}$ is $F_{40\%}$, as recommended by Clark (1993) and others, and a corresponding $B_{\mathrm{MSY}}$ proxy of $B_{40\%}$. The SSB biomass limit is assumed by the HSP to be $B_{20\%}$, which is 50% of $B_{40\%}$, the proxy for $B_{MSY}$.

The first major review question is whether a spawning stock biomass target of $B_{40\%}$ would be appropriate across the range of species to which it is applied. As mentioned above, this proxy for MSY was initially derived through simulation analysis of a range of groundfish species (Clark 1993). Groundfish species are a subset of the kinds of organisms that the HSP has been applied to, that include taxa such as molluscs, crustaceans, elasmobranchs (sharks and rays), in addition to finfish. Productivity and therefore SSB at MSY would be expected to vary with life history. Adams (1980) was among the first to investigate such differences, finding that *K*-selected types (long-lived, late maturing, low *M*, large body size) would be highly sensitive to overfishing and, once depleted, recovery would require a long time. Winemiller (2005) provides a useful classification of fish stocks based on three major life history strategies: Periodic (long-lived, high fecundity, high recruitment variation), Opportunistic (small, short-lived, high reproductive effort, high demographic resilience) and Equilibrium (low fecundity, large egg size, parental care). Species with different life histories have different responses to fishing pressure, and potentially could be managed according to different reference point targets. There has been a commonly held belief that long-lived *K*-selected species would tend to have low steepness, implying relatively low productivity, but studies such as Shertzer and Conn (2012) have been unable to find such a relationship.

Guidelines used by NZ fisheries management (Ministry of Fisheries NZ 2008; developed but not yet adopted) use productivity categories as defined by FAO (2001) and Musick (1999) to separately define biomass targets ranging from $B_{25\%}$ for high productivity species to $>B_{45\%}$ for very low productivity species. They also note however, that it is becoming increasingly difficult to justify MSY-compatible biomass targets less than 30-40% $B_0$. Hilborn and Stokes (2010) however, suggest using historical production levels as a guide to sustainable catches and point out that the dynamics of many species more productive species would entail that 25% $B_0$ would be consistent with an MSY target.

Within finfish only, several meta-analyses (e.g. Myers 2001, Goodwin et al. 2006) have examined productivity of fish stocks from the RAM Legacy Stock Assessment Database (Myers et al. 1999). Using surplus production models, Thorston (2012) found average $B_{\mathrm{MSY}}/B_0$ values for Pleuronectiformes (flatfish) of 39.5%, Gadiformes (grenadiers, cods, hakes) 43.9%, Perciformes (perch-like fish – many of our commercial species including morwong, whiting, tunas, swordfish) 35.3%, Clupeiformes (herring and anchovy) 26.1%, Scorpaeniformes (gurnards, flathead, rockfish, ocean perch) 46.3% and Other 40.5%. The high value for Scorpaeniformes is unsurprising given work by Dorn (2002) showing little recruitment compensation (low steepness) for US west coast rockfish. The standard deviation of these results was in the order of 0.1 for each group, so an approximate 95% confidence interval of ±0.2 times each estimate applies. There is an assumption that species within these taxonomic groups have similar characteristics, but it is clear that a wide range of life history characteristic types such as those defined by Winemiller (2005) occur within large taxonomic groups such as Perciformes (e.g. whiting and swordfish). Orange roughy and redfish are not within the groups examined – these are in the Order Beryciformes.

For elasmobranchs, Brooks et al. (2010) estimated an analogous form of $B_{\mathrm{MSY}}/B_0$ using numbers of fish rather than biomass termed $S_{\mathrm{MER}}/S_0$. As this was an analysis applicable to data poor species, the required information to determine the target depletion level was based on life history characteristics only – the maximum lifetime reproductive rate. Values for the 11 species examined ranged from 21% for Blue shark to 47% for Short-finned mako. They refer to Au *et al*. (2008) who summarized a likely range of spawning depletion required for optimum safe yields as being between $B_{20\%}$ - $B_{50\%}$, "with the range for sharks probably lying at the upper end of that interval" (Brooks *et al*., 2010, p172). More recent meta-analyses by Zhou et al. (2012) have shown that sustainable exploitation rates for elasmobranchs are less than half natural mortality, while for teleosts they are closer to parity.

A further and related review question is about the appropriateness of the HSP $B_{20\%}$ or ½$B_{\mathrm{MSY}}$ limit below which the stock is assessed as being overfished. Beddington and Cook (1983) may have been the first to use the 20% $B_0$ threshold and probability of falling below it as an indicator of where recruitment declines might be expected to be observable.

Questions about limit reference points for fishing mortality lead to discussion about the level of $F$ that would lead to the population to continue to decline possibly to extinction – termed $F_{\mathrm{crash}}$. Population features that are important in determining $F_{\mathrm{crash}}$ and the relationship of $F_{\mathrm{crash}}$ to $F_{\mathrm{MSY}}$ are the fishery selectivity pattern in relation to maturity and whether stock-recruitment depensation is a possibility (Punt 2000). The ratio of

$F_{crash}/F_{MSY}$ decreases with the productivity of the population. An explicit study of the relationship of $F_{crash}$ to $F_{20\%}$ does not appear to have been made.

For productive species where $0.5B_{MSY}$ is less than $B_{20\%}$ the current HSP suggests it is theoretically possible to set the limit reference point at $0.5B_{MSY}$. Given the uncertainty inherent in estimation of stock productivity, the precautionary approach (FAO 1995) would firstly require good evidence that $0.5B_{MSY}$ is indeed below $B_{20\%}$. Some of the most productive fish species, often with highly variable recruitment, are small pelagics, also known as "forage fish" whose abundance levels can naturally vary widely. Such species may have $B_{MSY}$ values much lower than $B_{40\%}$, in which case they may be candidates for limit reference points lower than $B_{20\%}$. However Walters et al. (2005) found that general application of single species MSY to a multispecies ecosystem leads to system degradation, and that forage species may require further protection to maintain the populations of larger piscivores. In the CCAMLR fisheries, for example, the minimum escapement for such forage fish species, such as the mackerel icefish (*Champsocephalus gunnari*) is 75% (that is the TACs are set such that at least 75% of available stocks are left in the water for ecosystem services), which is a very different rule to $B_{75\%}$.

Much has been written about the need to move to ecosystem-based fisheries management (EBFM) to take account of direct and indirect effects of commercial fisheries on the ecosystem that supports the exploited fish populations (e.g. Crowder et al., 2008). While there is general agreement on the principles, implementation of operational procedures based on them is still in progress. Several recent studies (Smith et al. 2011, Pikitch *et al.*, 2012) have used ecosystem models and in some cases empirical data to examine the effects of fishing low trophic level or forage species on predators and other parts of the marine ecosystem. While impacts vary for different species and across different ecosystems, there is an emerging consensus that exploitation rates should be set more conservatively than conventional single species MSY levels for such species. The Marine Stewardship Council identifies criteria for identifying "key" low trophic level species and then requires that default target biomass reference points be set at 75% of $B_0$, corresponding to exploitation rates at about half $F_{MSY}$. Pikitch *et al.* (2012), a major report from the Lenfest Forage Fish Task Force, recommend a tiered approach relating to the data availability. Thus, for high data situations they recommend no more than 75% $F_{MSY}$ and no less than 30% $B_0$ to be left in the ocean. For intermediate data situations these numbers were no more than 50%$F_{MSY}$ and $B_{LIM}$ at least 40%$B_0$. Finally, for relatively data poor situations they recommended no new forage fisheries and existing fisheries to be restricted to a $B_{LIM}$ no less than 80%$B_0$. The CCAMLR rule, which relates to taking no more than a defined proportion of current estimates of biomass, takes account of the natural variation of forage fish species. Their dynamics tend to be so variable that they can naturally increase or decrease their stock size by large amounts over relatively short periods. To require that stocks be maintained at 80%$B_0$ is not something that can necessarily be managed; even the concept of $B_0$ when applied to such variable species is problematic (see section 2.5.1 above).

Accounting for climate change on marine ecosystems and impacts on commercial fisheries is also a topic of much recent research (e.g. Brown et al. 2010; Plaganyi *et al.*, 2012). While an active area of research, climate-linked ecosystem models are not currently in operational use as a fisheries management tool for setting commercial catches. There are a number of example single species commercial fisheries where there has been acceptance of an environmentally induced productivity shift in the population (for

a specific Southern and Eastern Scalefish and Shark Fishery, SESSF, example see Wayte, 2013). The recognition of environmentally induced or population density induced variation in such productivity factors as growth and fecundity is growing in importance when conducting single species stock assessments (Whitten *et al*., 2013). These areas are active and relatively new areas of research and no general conclusions have yet been drawn. The expectation is that the marine climate will continue changing, especially in the hot spot areas of the east and west coasts of Australia. Fisheries will undoubtedly change but quite what changes are possible is still to be determined (André *et al*., 2010) so the implications for the HSP are limited to a need to retain some flexibility so that if circumstances in a fishery change significantly the HSP can respond appropriately.

The range of estimates of suitable target biomass and fishing mortality values is very great but the difficulty in estimating the real risks of running relatively high fishing mortality rates at low stock sizes (Beddington *et al*., 2007) suggests that Clark's (1993) suggestion of $B_{40\%}$ rather than something lower is a reasonable compromise. The current default biomass target reference point of $B_{48\%}$ would appear to be highly conservative (biologically) for many species, although it may be quite appropriate for slower growing sharks and rays and may not be sufficiently conservative for some key low trophic level species. For example, the Commonwealth small pelagic fishery (AFMA, 2009) has adopted a biomass level of at least 80% $B_0$ as the $B_{LIM}$ for each species in this fishery (with higher values in the more data poor situations).

This all means that if MEY or MSY can be reliably estimated from the life history characteristics and fishery data, then there would be no reason not to lower the target reference point. In fact, the objective of optimizing the economic performance of the fishery would require it. But it should also be kept in mind that estimating MEY or MSY can be very difficult and would undoubtedly require dedicated resources for each fishery. In addition, numerous studies show that, especially in mixed fisheries, it is difficult to balance the fishing mortality on an array of species (Walters *et al*., 2005). In New Zealand they use a soft target at $B_{20\%}$ and a hard target of $B_{10\%}$ below which the fishery is closed (Ministry of Fisheries, 2008). New Zealand has many more specifically targeted fisheries and so closing particular fisheries is a reasonable option. Nevertheless, because of these various doubts and uncertainties it would be difficult to argue that there would be no increase in the risk of depletion affecting consequent recruitment levels if the limit biomass reference point was permitted to vary below the current $B_{20\%}$.

### 3.1.1 DATA POOR STOCKS

A large number of commercially unimportant species, often bycatch or byproduct species, "cannot reasonably be assessed" (Beddington et al. 2007, p1716). Indeed, for such species many stock assessments are not sufficiently informative to support control rules with limit, threshold and target reference points for stock size and fishing mortality (Cadrin and Pastoors 2008). This subject is dealt with in more detail in Chapter 5.

Assessment approaches currently used for extremely data poor stocks depend on the limited data available (Dowling et al 2008; Smith et al 2009). Where a reliable series of catch estimates exist methods such as depletion-corrected average catch (DCAC), depletion-based stock reduction analysis (DBSRA) (Dick and MacCall 2011), and maximum constant yield (MCY) (Ministry of Fisheries NZ 2008) are in operational use in US and NZ fisheries. For species where occurrence distributions and spatial overlap with fisheries are known and there is some information about biological characteristics,

risk assessments such as productivity susceptibility analysis (PSA) (Berkson et al. 2011, Cope et al. 2011) and ecological sustainability assessment for fishing effects (SAFE) (Zhou and Griffiths 2008; Zhou et al. 2011) approaches have been used.

# 4 Buffered Targets or Meta-Rules

## 4.1 Target reference points in multi-species fisheries

**Key Questions in the Discussion Paper:**

- *The Review may consider whether further guidance is required on:*
  - *developing and setting target reference points for individual species within multispecies fisheries*
  - *acceptable levels of risk for stocks whose biomass is allowed to vary below* $B_{MSY}$.

A joint project between the Commonwealth Scientific and Industrial Research Organisation (CSIRO) and the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) is expected to start soon, which may provide further information on these matters.

There is a very large literature on the management of multi-species fisheries and the related ecosystem based fisheries management (Caddy and Mahon, 1995; Link, 2002; Hilborn *et al*., 2004; Haddon, 2007; Smith *et al*., 2007). All concede that multi-species fisheries management is difficult but no universal solution has yet been proposed. A major problem is that the species mix in fishery catches may not necessarily match the mix in combined TACs or in quota holdings. Klaer and Smith (2011) propose characterizing multispecies catch data into primary and companion components. This method provides an empirical means to examine the impact of individual species TAC decisions across all of the quota species in a fishery. The establishment of spatial closures has also been proposed. However:

> *For fisheries that are multi-species … marine reserves have some potential advantages. Their successful use requires a case-by-case understanding of the spatial structure of impacted fisheries, ecosystems and human communities. Marine reserves, together with other fishery management tools, can help achieve broad fishery and biodiversity objectives, but their use will require careful planning and evaluation. Mistakes will be made, and without planning, monitoring and evaluation, we will not learn what worked, what did not, and why. (Hilborn* et al*., 2004, p198)*

Any potential advantages of spatial closures for fisheries (insurance, protection of habitat, spill over of adults and larvae) can be effectively cancelled out if the fishery involves highly mobile species, or those with limited larval dispersal, or if fishing is not the only potential threat to the system (see chapter 8 for a more detailed discussion). However, for a mixed fishery in which a large number of species are caught but only some are formally managed, usually through quotas but possibly by limiting effort (e.g. the South East Fishery and, currently, the Northern prawn fishery) there is the potential for marine closures to offer some refuge from fishing mortality for many unassessed species. Despite these advantages, there are also possible downsides to imposing closures. If large closures are imposed and catches of the key commercial species are not reduced accordingly then fishing mortality in the areas remaining open will increase, possibly causing harm to the stock still exposed to fishing (Haddon *et al*., 2003). At the same time if there are sufficient closures in a fishery they may affect fishing behaviour and influence the fisheries data used to assess the stocks. Evidence that this is occurring

is only now being generated in the SESSF. A current FRDC funded research project is exploring the impact that marine closures (all types) can have on the stock assessment process to determine whether the closures are compromising our understanding of the stock dynamics.

In Australia, the present system for managing multi-species fisheries and other ecosystem-based fisheries management is to assess the key commercial species within the context of a standard harvest strategy with associated assessments and control rules or a system of tiered harvest strategies that treat different species according to how much information is available to assess the stock status. There is also an array of data-poor harvest strategies available but so far, these have not been mixed with more formal harvest strategies (whole fisheries are considered data-poor and treated as such but particular data-poor species within a mixed fishery do not tend to be managed using data-poor harvest strategies). However it should be noted that the tiered harvest strategy framework used in the SESSF already allows for different treatment of species according to the amount of information available (Smith et al. 2008). For any remaining species there is the ecological risk assessment process (Hobday *et al*., 2011; Williams *et al.*, 2011). This entails a hierarchical system of levels entailing different degrees of detail. At the second level in the hierarchy, the approach assesses the relative productivity and susceptibility of each species to the fishing pressure being imposed and classifies each species as low, medium or high risk (see Table 2 in Chapter 5).

Because this remains an area of fisheries management still searching for solutions it is certainly a candidate for greater clarification of options within the HSP. For example, in mixed species fisheries, where the key commercial species are managed using more formal harvest strategies if it was decided that it would be acceptable to manage relatively data-poor species using the data-poor harvest strategies available (Dowling *et al*, 2008; Smith et al 2009) then this option, or others, needs to be made clear in the HSP.

## 4.2 Buffer Zones

**Key Questions in the Discussion Paper:**

- *The Review may consider whether 'buffer zones' might be applied to the interpretation of reference points, such that when an indicator moves within a specified range of the reference point level, the reference point level is considered to have been achieved? The use of this approach in other countries (e.g. New Zealand) might provide a useful case study if the Review considers this issue further.*

### 4.2.1 INTRODUCTION

Target and limit reference points and the related control rules that use these to guide management are usually depicted using a phase diagram that compares fishing mortality against spawning biomass (Figure 2). This type of diagram was originally described by Serchuk *et al*., (1997, 1999) and Restrepo *et al*., (1998). In the case illustrated (Figure 2) the biomass and fishing mortality target and limit reference points are precisely defined. Fishing mortality being constant above the target biomass means that catches will increase with stock size. In the illustration the break-point occurs exactly at the target and there is a linear decline in F with decreasing spawning biomass, down to the limit biomass, after which no targeted fishing should occur. In mixed fisheries there is usually a

bycatch TAC set to allow for unavoidable bycatch (to reduce or prevent the need for discarding).

Such control rules (Figure 2) imply that the assessment of the stock biomass levels is relatively precise, which is not the case for any stock assessment, whether it is a sophisticated integrated assessment or a simple analysis of catch rates. The HSP recognizes that the assumption of equilibrium at the target is unrealistic. It states that:

> *…control rules should ensure that the fishery is maintained at (on average), or returned to, a target biomass point $B_{TARG}$ equal to the stock size required to produce maximum economic yield…. … For highly variable species that may naturally (i.e. in the absence of fishing) breach $B_{LIM}$, the harvest strategy for these species must be consistent with the intent of the Policy (DAFF, 2007, p 23).*

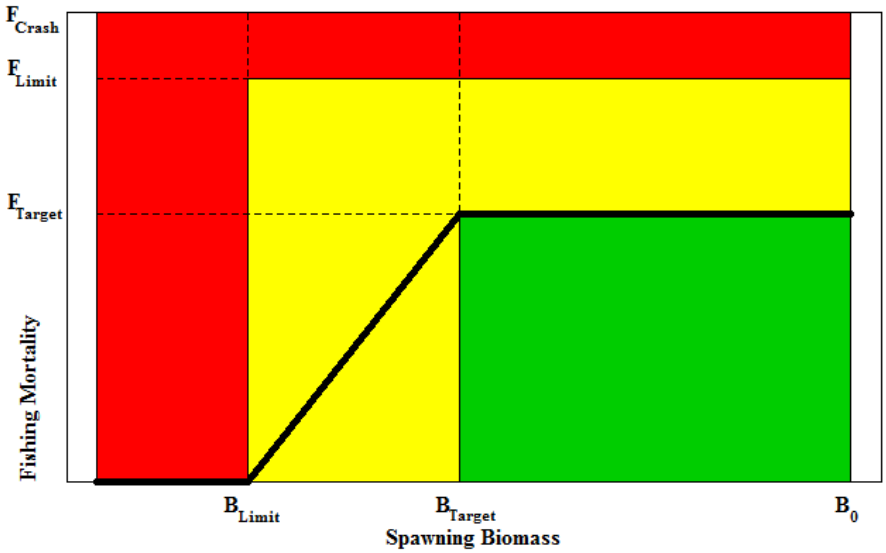


**Figure 2.** Common relationships between fishing mortality and spawning biomass related reference points; this is not the control rule used in the SESSF. The red area reflects situations where a stock would be experiencing overfishing and be overfished. The green area would be considered as under-fished and under-fishing, while the yellow areas reflect areas where the harvest control rule (thick black line) would act to reduce catches and fishing mortality to move the stock back towards the targets. After Beddington *et al*. (2007). There is a constant target fishing mortality until the biomass breakpoint (in this case the $B_{Target}$) is reached followed by a linear decline to the $B_{Limit}$, after which there is no targeted fishing.

Natural variation is expected due to environmental forcing and recruitment variability from year to year so the expectation is that even with a perfectly managed fishery the stock would fluctuate around the target. The HSP states: "For stocks above $B_{LIM}$ but below the level that will produce maximum sustainable yield ($B_{MSY}$) it is necessary to first rebuild to $B_{MSY}$. Once stocks are above $B_{MSY}$, rebuilding shall continue toward $B_{TARG}$…." (DAFF, 2007, p24) If a precise harvest control rule were to be interpreted without a buffer or meta-rules the stock would be expected to be below the target, and therefore presumably in need of rebuilding 50% of the time so the recommended TACs would also fluctuate up and down randomly. Given that natural variation is acknowledged in the HSP then ideas of having targets with buffers or meta-rules relate to proposed solutions for dealing with this problem of variation leading to highly variable management.

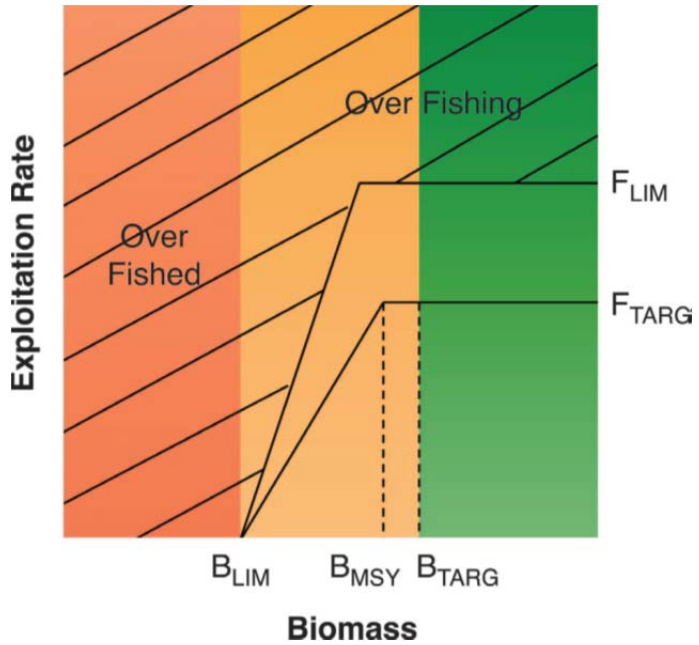


**Figure 3.** Example of a harvest control rule consistent with Australia's harvest strategy policy ($B_{LIM}$ - limit biomass reference point; $B_{MSY}$ - biomass that corresponds to maximum sustainable yield; $B_{TARG}$ - target biomass reference point; $F_{LIM}$ - limit fishing mortality rate; $F_{TARG}$ - target fishing mortality rate). The HSP specifies $B_{TARG}$ as $B_{MEY}$, the biomass that corresponds to the maximum economic yield. The control rule specifies that as the biomass reduces below $B_{MSY}$, $F_{TARG}$ is progressively reduced to zero at $B_{LIM}$ (after Smith, *et al*., 2008)

A common option when precise targets and inflection points are used in the control rules within a harvest strategy is to apply a meta-rule that says no change to TACs will be made unless the proposed change is at least 10% (or some such value; in the GAB there is a CPUE update rule that is used to account for the very latest catch rates when setting TACs, that requires a minimum change of a 20% increase or decrease in the CPUE for a 10% increase or decrease in a TAC). In the SESSF, there are a few species with relatively large catches so 10% might be a very large number (e.g. 10% of the flathead TAC would be 275 t) so the meta-rule there is before a change to the TAC is made it must be at least a 10% change to the TAC or 50 t, whichever is smaller.

Such meta-rules have the advantage of increasing stability of catches through time. However, there is potential for confusion with this approach because of the way the harvest control rules are actually used to generate Recommended Biological Catches (RBCs) from which a separate process is used to set a Total Allowable Catch. The harvest control rules are well documented in each case but extra clarity and transparency could be achieved if the final step of generating the TACs from the RBCs were as thoroughly documented. The harvest control rules in each formal harvest strategy clearly define the RBC one the assessment has been completed. Without clear documentation of the step from RBC to TAC this permits uncertainty to enter the process.

Meta-rules certainly have a place in the simpler harvest control rules used in fisheries for which there is no formal mathematical model of the stock dynamics. In such fisheries the control rules are precisely specified and without such meta-rules the issues of variable catch levels and change every year would arise. The advent and increase in the number of multi-year TACs will interact with tis, however, and should reduce its importance. The same effects of greater stability could be brought by using buffers around each target reference point for each species. Both such buffers and meta-rules have the same problem of trying to use a single value for all species, even though some species are much more variable than others. For example, depending on the prevalence of scallop beds it is quite possible for the current scallop harvest strategy to have the stock appear to move from above the biomass target to below the biomass target in a single fishing season, even when there are an array of undersize scallop beds waiting to grow into the fishery. The meta-rules currently used in the different fisheries appear to work acceptably well except for some of the more extremely variable species such as squid and scallops.

For those species with more sophisticated stock assessments their control rules can be more sophisticated also and appear more akin to the original proposed by Caddy and Mahon (1995). The implementation of these control rules, however, is less rigid. They may still be specified precisely but their specific detail can effectively add in a buffer to the stock status at which fishing mortality (catches) are reduced to rebuild the stock towards the target. This is exemplified within the SESSF (Day, 2009). The Tier 1 harvest control rule in the SESSF specified limit and target biomass depletion reference points, as well as a target fishing mortality rate. This is represented as a series of values depicted as the series: ($B_{LIM}$:$B_{TARG}$:$F_{TARG}$). Since December 2005, when the Harvest Strategy Policy was first implied in the Ministerial Directive, various values had been suggested and used for the target and breakpoint in the Tier 1 rule (Figure 4).

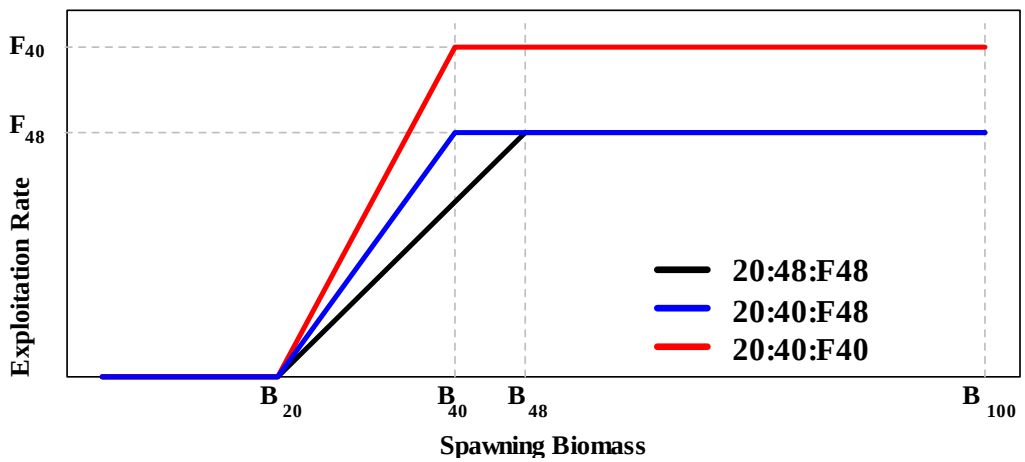


**Figure 4.** Early alternative Tier 1 control rules used in the SESSF under the HSP in 2006 (after Day, 2009), illustrating different target biomass and fishing mortality levels.

Initially, $B_{40\%}$ was used as a proxy for $B_{MSY}$, leading to the 20:20:40 rule. A little later $B_{48\%}$ was suggested as a proxy for $B_{MEY}$, the selected target in the HSP leading to the 20:40:48 and the 20:48:48 rules. The breakpoints at which the impact of the Harvest Control Rule on fishing mortality begins were thus altered. For the 2009 TAC setting session, AFMA directed that the initial trajectory of the 20:40:40 rule (the redline in

Figure 4) up until fishing mortality reached $F_{48\%}$, which meant that the breakpoint in the control rule needed to be estimated as it lay to the left of $B_{40\%}$ (Figure 5).

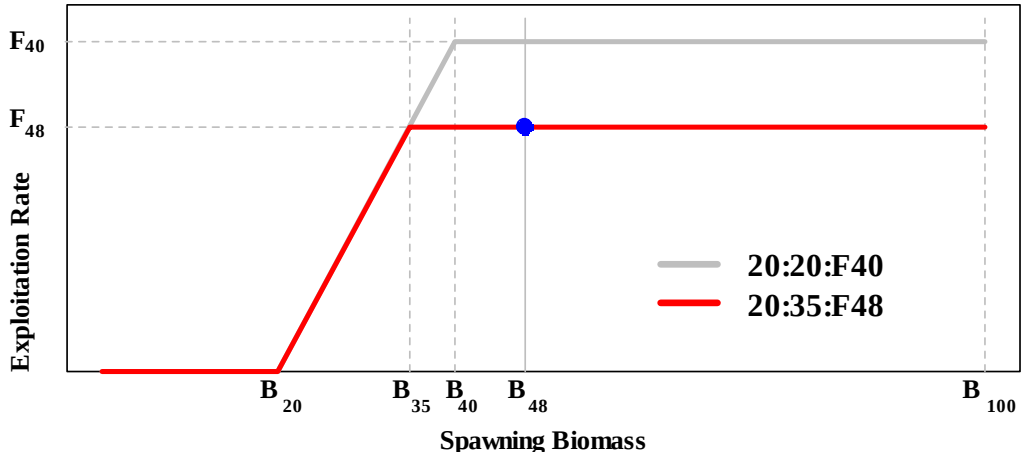


**Figure 5.** Re-estimated control rule for Tier 1 assessments in the SESSF in 2009, with a break-point at $B_{35\%}$ as a modification of the older 20:40:40 rule to become 20:35:48. The blue dot represents the biomass and fishing mortality targets (after Day, 2009).

This 20:35:48 ($B_{LIM}$:$B_{TARG}$:$F_{TARG}$) control rule introduced a large buffer between the target and the breakpoint. This does not mean that the catches (TAC) do not come down if the stock falls below the target biomass, but it does mean that the steepness of reductions in catch only increase once the biomass falls below $B_{35\%}$.

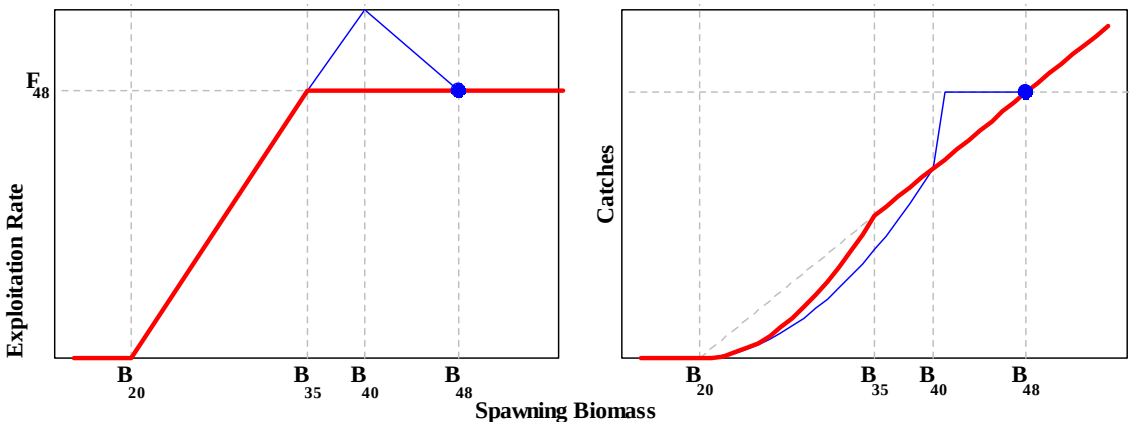


**Figure 6.** The 20:35:48 Tier 1 control rule in terms of its effect on fishing mortality and on relative catches. A constant fishing mortality implies a constant proportion of the available biomass will be allocated as a TAC, the steep linear decline in the control rule implies an exponential decline in catches the further below the breakpoint at $B_{35\%}$ the biomass becomes. When a buffer is installed in which the TAC is not decreased until, for example, $B_{40\%}$, this would be equivalent to allowing the fishing mortality to increase (blue lines in the plots) and then, past the chosen buffer, to decrease very rapidly when it returned to a linear decline with biomass.

It is thus apparent that a form of buffer around the target reference point can already be introduced into the harvest control rules. If an alternative form of buffer were introduced that kept TACs constant until the buffer range (for example one might use $B_{40\%}$ as a suitable buffer from $B_{48\%}$) then the implication would be that the fishing mortality would be allowed to increase up until the buffer limit and then decrease more rapidly than without the buffer (Figure 6). Such dynamics within the fishery would need to be

examined in more detail than the brief consideration given here and, to be in line the HSP it would need to be simulation tested to determine whether such a strategy increased or decreased the risk overall or breaching the limit reference point.

The present arrangements where those harvest strategy control rules in which the break point is clearly defined at the proxy target reference point certainly stabilizes catches and another meta-rule that prevents TACs varying by more than 50% between any two years has also been helpful in preventing serious dislocation and disturbance in the fishery for some relatively unstable species.

The terms of reference for these technical reviews included the identification of relevant research that could be conducted to improve the implementation and operation of the harvest strategy policy. If it was decided to pursue the issue of buffers and meta-rules around the targets then it would be beneficial to use simulation testing (MSE) to consider the effect of such changes to the expected dynamics of different fisheries.

# 5 Data-Poor Fisheries and Tiered Harvest Strategies

## 5.1 Introduction

**Key questions:**

- *Is there a need for further guidance on the development and testing of additional assessment Tiers to allow the use of more appropriate indicators for some particularly data poor stocks?*
- *How can increased precaution be demonstrated with decreased data without a MSE or HS evaluation process? Is the application of discount factors sufficient?*
- *Should there be a greater emphasis on the use of empirical performance measures and related control rules for fisheries with limited data and resourcing?*
- *Is the use of grouped species catch per unit effort (CPUE) data as indicators in some fisheries consistent with the objectives of the Policy and demonstrably precautionary and are there practical alternatives?*
- *How can a transparent and cost-effective, risk-based approach to data collection, research, assessment and decision-making can be integrated into the Policy.*
- *Should and if so how would specific requirements for data specification and provision relevant to harvest strategies be specified in the Policy or Guidelines. This might include specification of minimum documentation standards (e.g. consideration of the point at which additional data collection—monitoring and assessment—is required when catches of non-quota species start significantly increasing due to targeting or other reasons).*

Harvest strategies usually consist of monitoring, assessment and control rules. The assessment component contains a) a performance measure(s) from the system - or a model(s) using data from the system to generate a performance measure(s) - together with b) target and c) limit reference points, against which the performance measure is compared. Here an assessment is used in its broadest definition, being any system that provides information about the status (or a proxy) of the stock. In some cases, the assessment might be a simple linear regression of catch rates, whereas in others it uses a full dynamic stock assessment model.

For data poor fisheries, difficulties can arise in almost every component of the harvest strategy – for example, little or no regular monitoring means time series are rare, the assessment method is undertaken with an unknown degree of uncertainty, reference points are poorly defined and the associated control rules do not necessarily address risk clearly. Yet, an essential component of the present Harvest Policy is the application of a consistent degree of risk across all fisheries, irrespective of fishery type.

This data poor section describes the risk-cost-catch trade-off involved when attempting to manage fisheries with only limited information. The HSP states: “A tiered approach to control rules is encouraged in order to cater for different levels of certainty (or

knowledge) about a stock…… Such an approach provides for an increased level of precaution in association with increasing levels of uncertainty about stock status, such that the level of risk is approximately constant across the tiers." (DAFF, 2007, p36) However, the use of a tiered system of harvest strategies highlights an issue of minimum information requirements despite the fact that some fisheries already have few resources to obtain more information, if required.

The second component describes several data poor assessment methods, but placed in the framework of the risk-cost-catch trade-off. Much recent work, both within Australia and internationally has been undertaken, but many of these have yet to be formally published.. Indeed, a FRDC funded project (2012/202) entitled 'Operationalising the risk cost catch trade-off' began in July 2012 in CSIRO and is due to finish in June 2014.

Simulation testing already undertaken shows that the risk level by Tier method is not always predictable and can also be very case specific (Deroba and Bence, 2008; Fay *et al*., 2012). However, undertaking these tests for each fishery or species is impractical and expensive. Generic data poor MSE software is being developed (although mostly still unavailable) which points to one approach. The other is to apply a degree of caution although it may be unclear how to evaluate the risk without simulation tests.

It is important to note that economic reference points are not discussed in this section. Data-poor methods tend to focus mostly on the limit reference points, data-poor target reference points are to be described in the 'Alternative Economic Targets and Reference Points' document.

## 5.2 Defining Data-Poor

"Data poor" or "data limited" are relative terms (the two terms are used here interchangeably) and are applied to different circumstances in different fisheries (Welch *et al.* 2005). One definition of data-limited fisheries is that they lack sufficient biological information to infer the exploitation status of the targeted stocks (Vasconcellos and Cochrane, 2005; Dowling et al, 2011). Similarly, Punt el al (2011) define data-poor as stocks with catch estimates but little or no information on relative abundance and few or no samples of age and length from the fishery. Richards and Maguire (1998) (cited in Pilling *et al.* (2008)) define fisheries as data poor when the best scientific information available is inadequate to determine meaningful reference points and/or current stock status with respect to such reference points. Thus, a "data poor" fishery is one for which a defensible and quantitative stock assessment cannot be provided because of limitations in the kind and/or quality of the data available (Haddon *et al.* 2005; Kelly & Codling 2006). Therefore, here we define fisheries or species as data poor if information is insufficient to produce a defensible quantitative stock assessment.

While these definitions are largely similar, there are implications that need to be considered when specifying harvest strategies. Clearly, model-derived reference points are not available in most data-poor situations. However, suitable proxies can often be specified for a harvest strategy to be developed for a data-poor fishery. Restrepo *et al.* (1998) describe fishery and stock assessment attributes to delineate data richness (Table 1).

**Table 1:** Fishery and stock assessment attributes used by Restrepo *et al.* (1998) to delineate data richness (cited Welch *et al.* 2005).

| | **Data rich** | **Data moderate** | **Data poor** |
|---|---|---|---|
| Life history characteristics | Yes | Yes | Unreliable or limited |
| Fishery dependent data (e.g. logbook catch and effort) | Yes | Yes | Unreliable or limited |
| Data time series | >20 years | Generally <20 years | Generally <20 years |
| Fishery independent data (e.g. monitoring) | Yes | Available or limited | No |
| Stock assessment | Sophisticated | Simple sophisticated | Minimal or lacking |
| Reliable MSY related quantities | Yes | Limited | No |
| Stock size estimate | Yes | Yes | No |
| Fishery parameters (e.g. selectivity, fishing mortality) | Yes | Yes | Unreliable or limited |
| Control rules | *Fmsy*, *Bmsy* etc | $F_{35\%}$, $B_{35\%}$ | M, avg catch etc |
| Data quality | High | High moderate | Moderate poor |
| Uncertainty | Accounted for | Reasonable characterisation | Qualitative or lacking |

Data limited fisheries often arise because of inherent characteristics such as being new or developing, low value, or the cost of data collection being prohibitive because of geographic spread, a lack of monitoring and enforcement resources, the remoteness of fishing grounds and/or vast coastlines with multiple access points. Specifically data limited fisheries can include, but are not necessarily limited to:

a. new fisheries with no time series of information;
b. large scale but recently developed fisheries where fisheries research and management have lagged exploitation;
c. low-value fisheries for which little data are collected;
d. small-scale developing fisheries with usually several target species of otherwise mixed fisheries;
e. large scale fisheries where the quality of data is poor or variable and difficult to assure (e.g. misreporting and/or discarding);
f. spatially structured fisheries where data collected may not be representative of the whole stock; and
g. near to or totally bycatch species, in a mixed fishery, to which little or no attention is paid.

(Haddon *et al.* 2005; Pilling *et al.* 2008).

Vasconcellos and Cochrane (2005) estimated that 20-30% of the world's capture fisheries were data-limited. Data limitations were more pronounced in invertebrate fisheries, and more among demersal than pelagic finfish fisheries. It was also more prominent in areas with high species diversity and small stocks where fisheries play an important role for food security, such as in many tropical and low-income countries of Africa, Asia, Oceania and the Caribbean. For example, Salas *et al.* (2007) characterise small-scale fisheries in Latin America and the Caribbean as being multi-gear and multispecies, and having low capital and labour intensive, remote landing sites, large numbers of migrant and seasonal workers, and weak market and bargaining power among fishers.

Many data-limited fisheries are also of low economic value, implying limited human resources for undertaking stock assessment (Scandol 2005). While such fisheries may not necessarily be data-poor, stock assessment and complex statistical analysis of trends may not be locally deliverable in an ongoing sense. In such cases an empirical harvest strategy could be developed using quality fishery dependent and/or independent data. However, in many cases fisheries are both data-poor and lacking in local analytical capacity.

## 5.3 Review of Relevant Research

### 5.3.1 TIERS AND INDICATORS

Data-poor stocks comprise an important component of the species targeted by the Commonwealth fisheries. Often these are caught within complex, multi-species fisheries and can therefore be both a target and a byproduct within a region. A separate review of bycatch species in Commonwealth fisheries has also been undertaken.

The SESSF have implemented a Tier system to classify their assessment methods from data rich to data poor (Smith *et al*., 2008; Little *et al*, 2011). A Tier 1 assessment is a robust stock assessment, whereas a Tier 3 and 4 uses catch curve analysis to estimate F and a time series of catch rate data respectively. An overview of all the assessment methods used in AFMA's managed fisheries, to which the HSP applies, has shown that eight Tiers – from Tier 0 to 7 can be identified in Commonwealth fisheries (Dowling *et al.* in press).

For some fisheries, where data is very limited a series of catch triggers for levels at which management intervention may be required can be used as the harvest strategy (Dowling *et al*, 2008; Dowling, 2011) and these effectively impose a series of Tiers on such fisheries aimed at increasing information requirements and assessment if a fishery grows.

Many methods used in data poor tiers have been tested using Management Strategy Evaluation (Haddon, 2011, Little *et al*., 2011, Klaer et al, 2012) to compare their management effectiveness and compare their relative risk and uncertainty (Fay *et al*., 2012). Although Tiers are applied in the SESSF they are not the norm across the Commonwealth managed fisheries, where often only one tier is used in each fishery. This is potentially an issue, from both the point of view of consistency among fisheries and also a consistent application of the risk-cost-catch trade-off.

A study on AFMA's information needs (Dichmont *et al.* in press) has developed a Guideline to developing a fishery's information needs where the Tier system is further enhanced from that in use in the SESSF. This work is the output from a meta-analysis across all the Commonwealth's harvest strategies currently in use. It breaks the system into three component - Harvest strategy assessment Tiers (**Table 2**), economic target Tiers (**Table 3**) and ERA/M Tiers. The first two relevant tables are reproduced below. This extension shows that there are two components regarding the Tier assessment system – the stock assessment method to develop the index of abundance and the method to determine the target or MEY. This section only discusses stock assessment rather than economic methods.

The degree to which monitoring supports harvest strategies is clearly illustrated in the way that AFMA has implemented the harvest strategy policy (HSP) across its fisheries. Each tier in **Table 2** defines the types of data that are collected and the form of assessment undertaken to feed into the harvest control rule for that tier. The harvest control rules themselves can vary widely for a given tier, but in all cases should be designed to meet the requirements of HSP to achieve the target maximum economic yield (MEY) while avoiding biologically defined limits (limit reference points or LRPs) with a probability that is defined in the HSP. This second criterion is referred to below as the "risk" criterion. For many fisheries, the performance against the requirements of the HSP of the current harvest strategy (based on the current monitoring strategy) for each target species will have been tested using simulation testing such as management strategy evaluation (MSE) (Smith et al, 1999; Sainsbury et al, 2000).

**Table 2:** Tier structure for harvest strategies with associated data requirements. Costs included in the original table have been removed.

| Tier number | Tier description | Minimum data requirements |
|---|---|---|
| 0 | Robust assessment of F and B based on fishery dependent AND independent data | Time series of independent surveys and verified catch, effort and/or catch rate data. Data required to standardise catch rates (if used). |
| 1 | Robust assessment of F and B based on fishery dependent data ONLY | Time series of verified catch, effort and/or catch rate data. Data required to standardise catch rates (if used). |
| 2 | Assessment of F and B based on fishery dependent and/or fishery independent data | Time series of catch, effort and/or catch rate data. |
| 3 | Empirical estimates of F based on size and/or age data | Time series of catch only. Representative sample of size and, if relevant, age |
| 4 | Empirical estimates of<br>• relative biomass based on fishery dependent data<br>• within season changes to relative biomass based on fishery dependent data<br>• relative biomass based on fishery independent surveys | Time series of catch only or time series of fishery dependent data such as catch rates or independent survey data. |
| 5 | Empirical estimates of F based on spatial distribution of effort relative to species distribution | Patchy catch and effort data or distribution of catch/effort relative to the species distribution |
| 6 | No estimate of biomass and F; use of fishery-dependent species-specific triggers | Patchy catch and/or effort data by species |
| 7 | No estimate of biomass and F; use of fishery-dependent triggers for groups of species | Patchy catch and/or effort data by groups of species |

Reuter *et al.* (2010) describe a similar system of Tiers for management of Alaskan fisheries, where Tier 1 equates to having point estimates of biomass and biomass at maximum sustainable yield (MSY), together with a probability density function of fishing mortality at MSY. Alaska's Tier 6 is the data-poorest level, with catch history only.

Cadrin *et al.* (2004) and Cadrin and Pastoors (2008) also refer to tiered approaches, viewing the estimation of biological reference points as a hierarchy, ranging from data-poor proxies of relative indices of stock size and exploitation rates, to applying more informative demographic production models such as stochastic, age-based simulations of maximum sustainable yield. Interim limits can be derived from the most reliable tier of approaches, and research programs can be designed to advance the analysis to a more reliable tier for approximating or estimating MSY reference points.

Despite the general principle being agreed, it is not always clear is how to rank these Tiers. There is an implication that risk increases as one moves to more data poor methods, however results from simulation tests of these Tier methods are unpredictable.

In Klaer and Wayte (2011), several forms of uncertainty are described – stock assessment uncertainty, using an inappropriate assessment method, uncertainty in the data used in the assessment, and uncertainty in translating stock assessment result into stock status. These different forms of uncertainty and different assessment methods were tested within an MSE and showed that, for example, an average-length-based harvest strategy can achieve the policy within the correct risk profile (Klaer, Wayte & Fay, 2012), that surplus production methods work well as long as certain conditions are met (but this was not usually the case for the species tested; Klaer & Wayte, 2011) and that a cpue-based Harvest Control Rule worked well but was sensitive to, for example, the choice of parameter values and the reference period for the reference points (Little *et al.*, 2011). On the other hand, Dichmont *et al.* (2006) and Dichmont and Brown (2010) showed that, for the Northern Prawn Fishery and the Queensland spanner crab fishery, simple regressions of catch rate data could perform well at guiding management actions. In the NPF case, the catch rate HCR was compared to surplus production and delay difference methods. In that case, the latter was preferred as risk was more clearly defined but the catch rate method otherwise performed well. The harvest strategies used in some other data-poor fisheries within the Commonwealth were examined within the reducing uncertainty in stock status project using Management Strategy Evaluation, but these were constrained in the range of testing possible simply because of the lack of information (Dowling, 2011; Haddon, 2011; Plaganyi *et al*, 2012). Nevertheless, within those constraints the harvest strategies in use were found to be capable of achieving the intent of the policy even in the data-poor circumstances (Haddon, 2012b).

From a Tier perspective, there is therefore a lot of scope for using or developing different Tier assessment methods. However, the various MSE tests have shown very case specific results indicating that a precautionary system should be applied unless these methods are tested through MSEs.

Bentley and Stokes (2009) compare the assessment versus the procedural[1] paradigms – the latter applies to the Commonwealth HSP. Rather than focusing on the assessment

[1] In New Zealand and South Africa, management strategies are called management procedures. Harvest strategies are called operational management procedures (Rademeyer *et al.*, 2008).

method itself, they propose that harvest strategies are much more likely to apply to data poor fisheries. However, they argue that more attention needs to be given to the method of presenting evaluation results to decision makers, and more attention should be given to the design, evaluation and selection of harvests strategies to be tested. In the USA, development of a standard format for assessments in some cases has been proposed. However, in Australia, each RAG produces its own format, level of output detail etc. and some consistent approach (while still considering the differences between fisheries) might communicate better to the public and other scientists.

**Table 3:** Level number, description with associated minimum data requirements for the category of the economic component of stock assessments associated with estimating the target reference point for a species or group. Costs included in the original are removed here.

| Level number | Level descriptor | Minimum data requirements |
|---|---|---|
| 1 | Full dynamic bio-economic model using a Tier 0-2 assessment | Recent industry level costs and prices. Projected costs and prices over a reasonable projection period. This requires information about projections on exchange rates. |
| 2 | $B_{MEY}$ proxy using a Tier 0-4 assessment | Expert driven opinion on previously profitable catch rates that has good stakeholder or scientific backing |
| 3 | MEY proxy using a Tier 5-7 assessment | Little or no information on profitable levels |

### 5.3.2 COST-CATCH-RISK TRADE-OFF

The above aspect of Tiers leads directly to the next issue – how is the risk-cost-catch trade-off, as described in Sainsbury (2005), maintained between Tiers, or whether it even should be done. In the SESSF, there has been much debate about discount factors and other methods of developing RBCs per Tier that maintain constant risk between Tiers. There is a gap between the theory of the trade-off and its practical implementation. The tier in current use for a particular species in a given fishery will have been determined by a range of factors, including the monitoring and assessment methods in use prior to implementation of the HSP. However the tier applied to a particular species is a matter of choice and could be varied over time, taking into account the cost-catch-risk (CCR) trade-off. The tiers in **Table 2** span from high information need (Tier 0) to low information need (Tier 7), with costs of both monitoring and assessment varying across tiers. Within constraints, fisheries are able to choose the tier that best suits the needs and capacity of the fishery. While consideration of the costs of monitoring and assessment might tend to favour higher tiers (lower information requirements), this will depend on how precaution is applied in determining the harvest control rules that complete the harvest strategy definition for each tier. This is because higher tiers are associated with higher levels of uncertainty about stock status, requiring more precautionary harvest control rules and hence lower catch levels to meet the risk criterion defined in the HSP. This interplay across tiers between economic costs (of monitoring and assessment) and benefits (derived from catch levels) to achieve an acceptable level of risk is the essence of the CCR trade-off. To date, the quantitative nature of this trade-off has not been explored fully for any fishery.

In an attempt to reduce the risks to the stock associated with using some of the higher tier harvest strategies (i.e. Tiers 3 and 4 in the SESSF rather than the Tier 1) an array of discount factors have been proposed with a larger discount for the higher tiers. An AFMA draft document describing the TAC setting process states: "The application of the discount factor is to be determined on an individual species basis but will be applied unless RAGs advise that alternative equivalent precautionary measures are in place (such as spatial or temporal closures) or that there is evidence of historical stability of the stock at current catch levels." (AFMA, 2009a, p 5). There is no discount applied to Tier 1 assessments, a 5% discount to the TACs derived from Tier 3 methods, and a 15% discount to the TACs from Tier 4 (it should be noted that these discount levels were chosen arbitrarily). The relative risk of the various tiers used in the SESSF has been examined using management strategy evaluation (MSE) and this has found that the specific outcome is species and fishery dependent (Fay *et al*., 2012).

As part of a meta-analysis of all AFMA's harvest strategies, Dowling et al. (in press) used a statistical linear model to quantify the risk-cost-catch (RCC) frontier for each of three forms of risk – biological, economic and ecosystem. Although the most parsimonious models were statistically significant, the management and research costs tended to be reactive to risk. For risks to target species, it was not possible to develop a model for proactive use. This shows that the risk-cost-catch trade-off has generally not been applied to AFMA's fisheries and more work would be required before it could be. The findings showed that the information collection and assessment of a fishery, tended to reflect the history of a fishery rather than a program designed to address a RCC trade-off.

New Zealand uses a very simple harvest strategy for their most data-poor stocks, which only have catch information. They set the maximum constant yield for such species using the average catch from a period when the fishery was relatively stable with no major changes in fishing mortality which is multiplied by a constant (less than 1.0) which is chosen relative to available information based on any knowledge on the stock demographics and the history of the fishery (Ministry for Primary Industries, 2012). The application of the constant $c$ in the $MCY = cY_{AV}$ equation is a form of discount factor to allow for the uncertainty in such a harvest strategy. This approach is, however, purely empirical and is not an attempt to provide for equivalent risk between alternative assessment methods, although the stability of catches does suggest a low risk strategy.

The notion of applying a discount to the recommended catch levels that are produced by data-poor harvest strategies is becoming more common. In a proposed management framework for the Pacific Coast Groundfish Fishery Management Plan (FMP) in the USA it states that two management committees:

> *… further recommended that if the ABC [Allowable Biological Catch] control rule is structured to account for different levels of information available for each stock in the FMP, then the system of uncertainty buffers for each category or "tier" should provide increasing precaution with decreasing levels of information and increasing uncertainty.*
>
> *(PFMC and NMFS, 2010, p 7)*

The intent is to attempt to reduce the risk in accordance with increasing levels of uncertainty in different assessment methods and harvest strategies. This principle is simple to understand but demonstrating that different assessment methods have the perceived

relative degrees of risk requires detailed simulation testing. Fay *et al*. (2012) have demonstrated that the relative risks can be greatly affected by what appear to be small details in the different harvest control rules. Without the meta-rule that limits annual changes to the TAC for a stock to no more than 50% the Tier 3 harvest strategy does not always perform better than the Tier 4 harvest strategy. With the meta-rule then the ordering is as might be expected the Tier 3 generally out-performs the Tier 4; although the particular outcome is also species and stock dependent.

### 5.3.3 DATA-POOR ASSESSMENT METHODS

In the data- and capacity-poor context, most literature has focused on empirical indicators and assessments, and less on control rules and the incorporation of indicators and assessments in a harvest strategy framework. Data-poor assessments have been reviewed extensively elsewhere (see for example Kruse *et al*., 2005b, Pilling *et al*., 2008, Marine and Coastal Fisheries Special Section Volumes 1 and 2 in 2009, 2010). Publications of the US National Oceanic and Atmospheric Administration (e.g. Dick and MacCall, 2010; Berkson *et al.*, 2011) are useful as are those available on the FAO website.

A review of data poor indicators was undertaken in Dowling *et al.* (in press) and key points are provided here. When developing a harvest strategy performance measures, target and limit reference points, or suitable proxies that may be applied to a fishery, have to be identified. In data-limited situations the initial focus will be upon empirical measures of fishery performance. An empirical indicator is calculated directly from a specific set of raw data, and the calculation, may produce one or two parameters that can be easily defined (e.g. nominal CPUE, mean age, mean length). This differs from an estimated or model-derived indicator, which is derived from a range of data sets and is dependent on additional parameters or models that may or may not be available (e.g. biomass, fishing mortality) (Scandol, 2005).

For the simpler empirical reference points, where stock status cannot be directly inferred, target and limit reference points can be replaced by putting thresholds on changes to the empirical indicator (for example, total effort) that would indicate further investigation and analysis, before further changes are allowed. These thresholds are already applied to some of AFMA's harvest strategies (e.g. those in Tiers 5, 6 and 7). Such threshold or trigger levels should, if possible, relate to all possibilities for change to which managers should be alerted. Given a possible suite of indicators and reference points or triggers, and given the characterization of the fishery, consideration must be given to how these could be used as input to a control rule.

Scandol (2003) investigated indicators and reference points 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. It was shown that management strategies based on empirical indicators and reference points could have a high error rate, but that sustainable fisheries could be achieved when suitably conservative choices were selected for the reference points.

Scandol (2005) processed empirical stock status indicators including catch, CPUE, mean age, mean length, recruitment fraction, total mortality and fishery independent surveys using quality control methods that worked by constraining those indicators within stated bounds. Biomass surveys were found to perform best, followed by mean

age and length, and recruitment fractions. CPUE and catch had the worst performance but were still acceptable.

A review of data poor methods undertaken in the USA NOAA (Dorn *et al.*, 2011) looked at catch-only methods which included minimal life history information only and methods that include catch, life history and time series of survey indices or length composition data. Most of these packages are freely available on the NOAA website (http://nft.nefsc.noaa.gov/index.html), including PopSim that is a generic age-based MSE operating model. Many of these methods performed well but needed key assumptions to be true for the model to be validly applied. On the other hand, a novel approach is used to assess data rich and poor stocks in the SESSF alongside each other – the so called 'Robin Hood' approach (Punt *et al.*, 2011) – thereby drawing from data rich information and inferring to data poor species.

Punt *et al.* (2001) used Monte Carlo simulation to examine the performance of alternative empirical indicators and associated reference points in terms of their ability to correctly identify the biological conditions that they were designed to measure. Indicators based only on catch rates are shown to be potentially very misleading. In contrast, indicators based on the mean length or mean weight of the catch changed in a more predictable manner with abundance. However, reference points based on these quantities were frequently 'triggered' either too early or too late.

Trenkel and Rochet (2003) compared the performance of population indicators for a Celtic Sea groundfish community based on achieved precision, statistical power and availability and estimation method of reference points. Among the population indicators of intrinsic population growth rate, total mortality, exploitation rate, mean length of catch, and change in fishing mortality to reverse population growth, the mean length of catch was most precisely estimated and the corresponding hypothesis tests had consistently large powers.

Life history characteristics inferred from size-specific catch data (e.g. percentage of mature fish in catch), have been suggested as a way to monitor change in stock status for data-poor species (Reuter *et al.*, 2010; Froese, 2004; Kelly and Codling, 2006). Basson and Dowling (2008) used a simulation approach to consider CPUE and eight size-based indicators: mean, median and 90$^{th}$ percentile length and weight, and the proportion of "big" and "small" fish in the catch. Size based indicators changed less than CPUE in response to changes in fishable abundance and were thus much more sensitive to measurement error or random noise. Further, size-based indicators were shown to be informative only for populations where individual growth was slow. Of the size-based indicators, mean length and weight performed best. The performance of size-based indicators also depended on the stock-recruitment relationship. Using classification trees as control rules, it was demonstrated that there was little to be gained by using more than 4-5 indicators together. The choice of indicators depended on the population dynamics, specifically lifespan and growth. Moreover, even good indicators could perform poorly when used in a badly-designed control rule.

Froese (2004) suggested that assessments could be based on three size-based indices from catch composition data, *Px*: (i) percentage of mature fish in the catch, *Pmat*, with 100% as target; (ii) percent of specimens with optimum length in the catch, *Popt*, with 100% as target; and (iii) percentage of large fish in the catch, *Pmega*, with 0% as target,

and 30–40% as representative of reasonable stock structure if no upper size limit exists. Cope and Punt (2009) showed that Froese's (2004) values were not always sufficient to ensure protection from overfishing, since the metrics were intended to avoid growth and recruitment overfishing, but there was no quantitative linkage to stock status and calculation of future sustainable catches. Moreover, their values cannot be interpreted adequately without knowledge of the selectivity pattern. They introduced *Pobj* (the sum of *Pmat*, *Popt*, and *Pmega*) to distinguish selectivity patterns. This approach gives further guidance to interpreting catch length composition data under variable fishery conditions without collecting additional information. It also provides a link to developing harvest control rules that inform proactive fisheries management under data-limited conditions.

McGarvey *et al.* (2005) used a simulation incorporating delay-difference models to evaluate the performance of stock assessment models based on logbook data sets of i) catch in weight and fishing effort, ii) plus catch in numbers, and iii) catch in weight and catch in numbers (no effort). Assessment models utilising catch in numbers substantially improved precision and accuracy in annual population estimates.

Griffiths *et al.* (2007) used catch by length data with anecdotal information to build a size distribution of the true population, which was incorporated into a Bayesian modelling approach to estimate abundance and biomass from gillnet catches in data-limited situations.

All these studies have shown that both within Australia and internationally an extensive research drive on data poor methods have been undertaken. However, most of these are still within the Tier 3-5 range. Few have no catch data, for example, or only group (rather than species) specific data such as the Coral Sea aquarium fishery (Haddon, 2012b).

In particular, while some examples exist (e.g. Wayte and Klaer 2010), there remains a real need to provide general guidance on formulating control rules that link empirical indicators with suitable management responses. Most research has focused on comparing data-poor assessment methods rather than comparing the effectiveness of different data-poor harvest control rules.

### 5.3.4 DATA RELATED ISSUES

Data related issues are described in the Discussion paper in terms of data requirements, developing fisheries, fisheries data used in the assessment and real time data provision. A further aspect, are data sources and quality. For data poor fisheries, difficulties can arise in almost every component of the harvest strategy – for example, little or no regular monitoring means time series are rare, the assessment method is undertaken with an unknown degree of uncertainty, reference points are poorly defined and the associated control rules do not clearly address risk.

For fisheries, as for natural resource management generally, the purpose of monitoring is to support management strategies, which in turn are designed to achieve management objectives. Monitoring is one of the key steps in the adaptive management cycle, and together with assessment and decision making define a management strategy. Monitoring is key to supporting any adaptive management strategy as it provides the data used to assess the state of the system and to check whether management strategies are achiev-

ing their objectives. Monitoring is needed to support both harvest strategies and environmental risk management (ERM). Monitoring strategies cannot be assessed without simultaneously considering their use in supporting management strategies.

In the Guidelines by Dichmont *et al.* (in press), data requirements for each Tier are provided (**Table 2** and **Table 3**). However, the key issue is rather whether there are minimum data requirements in the form of minimum Tiers for specific fisheries. Dichmont *et al.* (in press) discuss this issue, and while they provide guidance on how to approach this issue no clear past precedent could be obtained to provide empirical solutions. However, the review does state that there are certain types of data that all fisheries should collect on a routine basis. A minimum standard is that there be logbooks which collect data on all fishing operations, including where and when they occurred (at the finest spatial and temporal resolution possible), the type of fishing gear used, and a record of the amount of all species (or higher taxa where identification is difficult) retained. Additional (reasonable) requirements for most fisheries are a record of species caught by the gear but not retained, or observed to interact with the gear. These minimum standards are required to determine the nature and level of interactions of the fishery with the ecosystem. These constitute a minimum standard, for all fisheries independent of their scale and impact, that would provide for a defence against claims that a fishery was causing irreversible damage.

Additional minimum standards should apply to some fisheries depending on their scale and likely level of ecological impact. These additional requirements are to assess the impacts of fishing on the fished stock and the ecosystem in which it is a part and include collecting information to help determine the biological status of impacted ecological components. There are several means that could be used to determine to which fisheries minimum requirements apply. This could be on the basis of 1) the value of the fishery, 2) the volume of landings in the fishery, and/or 3) the overall ecological foot print of the fishery (which will in part be determined by the types of gear used in fishing operations). Two options to address these considerations are: the first is to make *a priori* determinations of risk, for example similar to the “fishery risk assessments” adopted by DSEWPaC in marine bioregional planning; the second (and likely preferred) option is to make case-by-case determinations using the steps and methods described in Dichmont *et al.* (in press).

However, the harvest strategies applied to some fisheries already are confined by their inability to collect the information required. At this stage, it is unclear what the consequences of this experience have been to these fisheries.

Implementation of harvest strategies in Commonwealth fisheries has shown that - i) there are additional Tier levels beyond those used in the SESSF acknowledging the large number of target species and types of fisheries managed by AFMA, ii) there are pragmatic harvest strategies that meet the intent of the Policy but that still need clear statements as to how these conform to the policy, and iii) commitments written into the harvest strategies to collect and store data as required to allow the fishery to establish more defensibly its stock status may need additional resources than those already available (Dowling et al, 2008a).

The various MSE tests described above have shown very case specific results indicating that a precautionary system should be applied unless these methods are tested through

MSEs. Thus, a rule of specific to particular fisheries is best could be considered. However, not all fisheries or species within a multi-species fishery can afford an MSE. So something more generic is also needed along with criteria for when to apply which approach. Two approaches are possible:

a) generic MSEs have been developed (NOAA's PopSim; Haddon and Dowling, 2012, and others), but are either at very early stages or require further work. Further research in this area would be of value.

b) A risk-cost-catch trade-off framework where many data poor methods are tested in an MSE framework and then potentially generalised (if at all possible). A start to this process has recently been funded by FRDC (PI Dichmont), but this work will only report at the end of 2014.

Presently, there is little direction on what constitutes a defensible harvest strategy because any such discussion tends to describe more data rich approaches. As more MSE tests are undertaken, this issue will become more clearly defined and some solutions provided. However, there are fisheries or species within multi-species fisheries managed by the Commonwealth that are sufficiently complex that the costs of moving beyond very little data make the move almost impossible. For example, there are minor fisheries of such relatively low value that there are insufficient resources to even enter all data into databases or query those databases and do the analyses necessary to fulfil the existing data-poor HS requirements (Dowling et al, 2008a). Thus, the issue is whether even lower Tiers than those used within AFMA (Table 2) are required and whether these still conform to the intent of the policy. If not, then a funding model needs to be provided that allows all components of those fisheries that implement the harvest strategy to be appropriately resourced. Fulfilling the requirements of the Harvest Strategy Policy for all Commonwealth fisheries has obvious resource requirements.

The hierarchical methods developed in several harvest strategies or within the ERA (Hobday *et al.* 2011) entail small scale fisheries starting at a data poor Tier which consists mostly of empirical triggers. The ERA is explicit in that it provides two options when a risk is shown, using a method that defaults to being precautionary, which is to a) move to a more data rich method and test if this risk still remains or b) mitigate this risk through direct management action. This is the principle behind the assessment Tier system and the hierarchical trigger system used in some fisheries.

# 6 TAC Setting and Multi-Year TACs

## 6.1 TAC SETTING AND MULTI-YEAR TACS

The following issues were identified:

- *The Review may consider if criteria should be developed and described in the Guidelines which RAGs can refer to when determining whether and how to apply discount factors to determine TACs in a consistent manner (ABARES, 2011, pp118). Some work has been done in the SESSF and may help inform the Guidelines in this respect.*
- *What empirical indicators might be most appropriate for assessing fishery condition through time when applying MYTACs?*
- *How risks associated with MYTACs might be incorporated into RBCs? The use of multi-year TACs has not been accompanied by an appropriate consideration of risk, to this point in time, noting that longer periods between assessments may increase the risk that changes in stock status occur?*
- *How to determine an appropriate time period for MYTACs and whether the period is dependent on the status of the stock (e.g. very depleted versus near target)?*

### 6.1.1 TAC SETTING

A key management lever used in Commonwealth fisheries is the application of total allowable catches (TACs) through individual transferable quotas (ITQs). Many of the Australian Commonwealth fisheries are multi-species fisheries and these present further particular problems when setting TACs. The use of ITQ management in multi-species fisheries has been the subject of much debate and the complexities and difficulties of managing multi-species fisheries are well known (Branch 2009; Chu 2009). In these fisheries, a major issue is in setting total allowable catches (TACs) that are directed towards individual species to achieve management outcomes across a range of species. Generally, when TACs are set for individual species, catches of other species are not considered. In multi-species fisheries, there are often technological interactions where fishing effort directed towards one quota species will normally result in a mixed catch of fish that may include other quota species. Fishers can usually 'target' to some degree through fishing different areas and depths, seasons, times of day and by modifying gear. But it is the degree to which fishers can target that is the issue. The species mix in catches may not necessarily match the mix in combined TACs or in quota holdings. This difficulty in balancing quotas for multiple species with actual catches may then lead to increased discarding, TAC over-runs, effort restrictions or fishery closures when quota is constrained on some species (Branch et al 2006; Sanchirico et al 2006). This may lead, therefore, to problems with achieving $B_{MEY}$ for multiple species.

While a number of solutions have been proposed or implemented to improve transferability of quota and other incentives to reduce over-quota fishing and discarding, it is surprising that there has been little focus on TAC-setting itself and coordinating this across multiple species/stocks as a means of dealing with some of these issues. Klaer and Smith (2011) analysed data from the trawl sector of the Australian Southern and Eastern Scalefish and Shark Fishery to determine the relationship between primary species and companion species and the implications this has for TAC setting. The primary species is the species being considered when setting an individual species TAC. The

companion species are ones that should also be considered when setting the TAC of the primary species, because a considerable proportion of the primary species catch is taken as a companion species non-target catch. The target species in each fishing operation was determined and was used to characterize recent multispecies catch data into primary and companion components. The approach of identifying companion species within a given fishery provides an empirical means to examine the impact of individual species TAC decisions across all of the quota species in a fishery.

### 6.1.2 MULTI-YEAR TACS

Currently there is a growing use of Multi-Year TACs in those fisheries where they can be implemented. However, this strategy and the various means by which is has been and is being implemented have not been subject to formal management strategy evaluation. In general, multi-year TACs will require a "discount" of some level of catch to balance the greater risk associated with less frequent review and adjustment. There are obvious risks of stock depletion if the multi-year TACs are set too high. There is also a potential loss of yield if good recruitment occurs but is not reacted to for a few years (though potential losses through natural mortality may be offset by potential gains by growth of fish left in the water for longer, this balance will vary by species).

While there is a good deal of debate within various Australian Assessment Groups concerning the implementation of multi-year TACs no clear decisions or standard protocols have yet been adopted with respect to avoiding the potential risks of setting a multi-year TAC so high it leads to depletion. There are a number of examples where fish species have declined rapidly over relatively short numbers of years, for example deepwater flathead in the GAB (Klaer, 2011), and school whiting in the SESSF (Day, 2012). While draft breakout rules have been produced within the SESSF these have not been tested and only relate to catch rates and so are of limited use in those species where catch rates are highly variable. Informal criteria for placing species into multi-year harvest strategies have been developed but limited financial resources are currently restricting the number of Tier 1 assessments able to be conducted and this leads to pressure to maintain TACs in the absence of new information. Even if a species does breach its break out rules there are currently no guarantees that there are sufficient financial resources available to do a more adequate assessment.

There remains debate over the best way to set a multi-year TAC. The options raised include simply applying the current TAC forward for three year, another (only suitable for Tier 1 assessed species) is to set the TAC in each year in line the median projected secure catch from the stock assessment model, another is to apply some arbitrary discount with different figures being suggested in every case discussed. It is therefore very simple to conclude that more simulation testing work needs to be conducted to determine the utility of different criteria for selecting species as suitable for multi-year TACs. The exploration of the risk cost catch trade-off currently underway in a FRDC project should be able to provide insights with respect to this problem of whether multi-year TACs should always be reduced below single year TACs so as to reduce the risk of declines. Any research undertaken on this topic should evaluate the different options for setting multi-year TACs. With reductions in available resources for conducting stock assessments this research program takes on extra urgency.

There is very little literature regarding application of multi-year TACs in other jurisdictions. The Pacific Fishery Management Council allows for multi-years TACs in that assessments are done in year y, and acceptable biological catches (ABCs) are forecast for years y+1 and y+2. The Council then selects the TACs for years y+1 and y+2, usually based on an Allowable Catch Limit (ACL) control rule. However, no formal simulation testing of this strategy has been completed (Andre Punt *pers. comm.*).

In New Zealand, while there doesn't appear to be a formal mechanism for allocating multi-year TACs their management system of identifying the Maximum Constant Yield (MCY) leads, in practice, to stable TACs over many years. Thus a consideration of volume 1 of the stock assessment plenary document for 2012 (Ministry for Primary Industries, 2012) shows that, for example, Alfonsino, Arrow Squid, Barracouta, Blue Cod. Blue Moki, Blue Warehou, and Butterfish have all had the same Total Allowable Commercial Catch (TACC) for ten years of more. While there are some species where the TACC has varied (e.g. Blue Mackerel) there are many more New Zealand fisheries which have exhibited stable TACCs for many years. It is important to note that the MCY calculation accounts for the risk of setting the same catch level over a number of years by resulting in lower catches on average than setting any annual TAC based on updated assessments.

# 7 Rebuilding Strategies and Bycatch TACs

## 7.1 Introduction

The following issues were identified:

- *… whether and how the guidelines should be amended to provide further direction on the recovery objective and on whether rebuilding timeframes could be determined in a more species specific manner, giving consideration to the species productivity and other factors which might affect the stock's ability to recover (e.g. climate change, stochastic events, etc.). (DAFF, 2012, p26)*

and

- *… whether and how the advice in the guidelines on formulating rebuilding strategies (and particularly the estimation of incidental catch allowances) should be expanded upon or strengthened, and whether and how the Policy itself should be made more prescriptive in this matter. (DAFF, 2012, p26)*

A primary objective of the Commonwealth Harvest Strategy Policy (HSP) is to maintain key commercial fish stocks at ecologically sustainable levels and within that context, maximize the economic returns to the Australian community (DAFF, 2007, p4). A key commercial species is defined in the HSP as "… a species that is, or has been, specifically targeted and is, or has been, a significant component of a fishery" (DAFF, 2007, p54). To meet this objective, harvest strategies were developed for key commercial species that were "… designed to pursue maximizing the economic yield from the fishery, and ensure fish stocks remain above levels at which the risk to the stock is unacceptably high" (DAFF, 2007, p4). These minimum levels are defined by Limit Reference Points (LRP).

The HSP specifies minimum standards for the Limit Reference Point (LRP) as being: "$B_{LIM}$ (or proxy) equal to or greater than ½ $B_{MSY}$ (or proxy)" and/or "$F_{LIM}$ (or proxy) less than or equal to $F_{MSY}$ (or proxy)" (DAFF, 2012a, p22). In practice, this was operationalized by declaring the spawning biomass that corresponds to the level at which the risk to the stock is unacceptably high as the $B_{LIM}$, and unacceptably high was "… for example the point at which recruitment overfishing is thought to occur" (DAFF, 2007, p23). While this specific point has been estimated to occur across a wide range of depletion values for a range of species (Myers *et al*., 1994), in Australia it was decided to adopt ½ $B_{MSY}$ as the default depletion level to use as $B_{LIM}$ (Restrepo *et al*, 1998), which defaulted to being represented as $B_{20\%}$. It should be remembered that there is no empirical demonstration that $B_{20\%} = B_{LIM}$, is the same as ½$B_{MSY}$. In fact, given that *MSY* can easily vary greatly from $B_{40\%}$ if ½$B_{MSY}$ were completely adopted it would be possible to have a limit biomass reference point well below $B_{20\%}$. Even where it is deemed possible to estimate $B_{MSY}$ the limit reference point of $B_{20\%}$ has been retained to avoid the risk of depletion reaching levels that constitute risks to subsequent recruitment. Given the choice of $B_{20\%}$ as the limit the HSP aims to: "… ensure that the stock stays above the limit biomass level at least 90% of the time (i.e. a 1 in 10 year risk that stocks will fall below $B_{LIM}$). In those circumstances where the depletion level cannot be estimated, the HSP allows for

proxies to be used within designed harvest control rules (see section 2.5.1 for comments on the incompatibility of the requirement that current harvest strategies use the present stock status to determine any recommended biological catches and yet the determination of the probability of falling below the limit reference point would require projections forward of any recommended catch levels; the only way to get around this incompatibility is to conduct simulation tests to ensure the harvest strategy adopted fulfils the <10% requirement).

The Harvest Control Rules (HCRs) in each Harvest Strategy adopted in each Commonwealth fishery are designed to reduce fishing mortality if the stock is assessed as declining away from the $B_{TARG}$ towards the $B_{LIM}$ (the default Target Reference Point – TRP – is $B_{48\%}$ which is taken as a proxy for the maximum economic yield, or MEY; = $B_{MSY}$ × 1.2, where $B_{40\%}$ is used as a conservative proxy for $B_{MSY}$); in this way it aims to prevent overfishing by encouraging the stock to rebuild. If, however, a stock does drop below $B_{LIM}$ then it becomes defined as overfished and an AFMA managed rebuilding strategy must be put in place to rebuild the stock towards $B_{TARG}$. Below $B_{LIM}$ a stock may also be considered for listing as conservation dependent or a more significant listing level, and such a listing may require the development of a formal recovery plan under the EPBC Act.

In the Commonwealth fisheries there are currently four fish species which are conservation dependent: School shark (*Galeorhinus galeus*), Orange roughy (*Hoplostethus atlanticus;* not on the Cascade Plateau), eastern gemfish (*Rexea solandri*), and southern bluefin tuna *(Thunnus maccoyii*). All of these species were seriously depleted before the implementation of the present HSP and since the introduction of recovery plans targeted fishing is supposed to stop (except for SBT, which is managed under an international harvest strategy). Unfortunately, this means that information and data about these species becomes greatly reduced. This lack of information means the difficulty in managing these species and pushing the recovery plans forward becomes greater. This is an unintended consequence of the HSP. In a cost recovery setting, it becomes even more difficult to fund research on fisheries for which directed commercial activity has ceased.

The Guidelines for Implementing the HSP state that: "For a stock below $B_{LIM}$ a rebuilding strategy will be developed to rebuild the stock to $B_{TARG}$. Once such a stock is above $B_{LIM}$ it may be appropriate for targeted fishing to re-commence in-line with the stock rebuilding strategy and HS." (DAFF, 2007, p 24)

This present document is concerned with details of the management of those stocks that fall below $B_{LIM}$, including the different strategies and timeframes for rebuilding. In terms of timeframes for rebuilding the *Guidelines* state: "Typically recovery times are defined as the minimum of 1) the mean generation time plus ten years, or 2) three times the mean generation time." (DAFF, 2007, p. 44). In addition, in mixed fisheries, to minimize discarding, the rebuilding strategies need to determine what level of incidental bycatch is likely to occur under normal fishing operations where the depleted species is no longer subject to a targeted fishery.

Attempting to meet these guidelines has been problematic in three of the conservation dependent species (the orange roughy fishery has effectively been shut down) as well as other currently depleted species (such as blue warehou, *Seriolella brama*) so the discussion here will focus on research related to these subjects.

### 7.1.1 POTENTIAL ISSUES OF RELEVANCE TO THE REVIEW

Relating to these issues the discussion document (DAFF, 2012b) listed two areas in which commentary was invited:

***"Rebuilding timeframes and strategy:***

There has been some debate about the scientific basis for these timeframes, and whether this statement pertains to the timeframe for moving the stock from below $B_{LIM}$ to $B_{LIM}$ (or above), to $B_{MSY}$, to $B_{TARG}$, or from $B_{LIM}$ to $B_{TARG}$. In addition, while the Guidelines state that rebuilding strategies should aim to rebuild stocks to $B_{TARG}$, this is perhaps inconsistent for multispecies fisheries which are allowed to maintain stocks at below $B_{MEY}$ (i.e. the Policy's $B_{TARG}$) but always above $B_{LIM}$. In addition, it is uncertain whether the implicit assumption that all stocks can be rebuilt is in fact correct**.** An important issue is:

***Rebuilding strategies for incidental catch:*** The Policy states that where stocks are below $B_{LIM}$, *targeted* fishing for that stock shall cease. The Policy states that a '*rebuilding strategy may impose additional constraints on* (incidental catch) *allowance up to and including closure of the fishery*'. However, the Policy does not require that harvest strategies necessarily impose a zero catch limit on stocks below $B_{LIM}$. Specifically, the Guidelines note that '*Clearly, a zero RBC below $B_{LIM}$ provides the maximum possible recovery rate. However, achieving zero catches in a multi-species fishery may be difficult'* (HSP, p. 44). The Guidelines also state '*the optimal time path to rebuild a stock has an economic component. In determining the optimal time path to rebuild a stock, there is a trade-off between lost profits in the short term and the speed at which the stock is rebuilt*' (HSP, p. 43).

Accordingly, where a commercial stock falls below $B_{LIM}$ targeted fishing must cease but an incidental catch allowance (sometimes referred to, somewhat misleadingly, as a 'bycatch allowance' or 'bycatch TAC') may be put in place as part of a suite of management measures to rebuild the stock. Experience has shown that stocks managed under rebuilding strategies have not always shown the expected rebuilding for recovery within the planned timeframe. For example, while rebuilding strategies were implemented for three species (eastern gemfish, school shark and blue warehou) in 2008, recent assessments and projections suggest that the total fishing mortality of these species has not been reduced sufficiently to allow rebuilding within the specified timeframes (ABARES, 2011). In the case of eastern gemfish, targeting has been prohibited since 1996 but there is still no sign of recovery to previous levels. The possibility of a regime shift is being considered in this case, amounting to a reduction in overall productivity of the stock not necessarily related to fishing.

## 7.2 Review of Research

Some of the questions asked within the discussion document are more related to policy decisions than to technical questions amenable to research. Thus, for example, the question of whether targeted fishing should cease until a stock has rebuilt to $B_{LIM}$ or $B_{TARG}$ is a policy decision, but the implications of such decisions can be discussed in terms of their implications for the stock and for the internal consistency and other possible implications for the rest of the policy.

### 7.2.1 REBUILDING FROM $B_{LIM}$ TO $B_{TARG}$ OR TO BACK ABOVE $B_{LIM}$

The HSP is clear about the targets for rebuilding. It states that "For a stock below $B_{LIM}$, a stock rebuilding strategy will be developed to rebuild the stock to $B_{TARG}$. Once such a

stock is above $B_{LIM}$ it may be appropriate for targeted fishing to re-commence in-line with the stock rebuilding strategy and Harvest Strategy." (DAFF, 2007, p 4 & 24). This reflects the fact that the harvest control rules that operate within particular fisheries when the stock status falls between $B_{TARG}$ and $B_{LIM}$ constitutes the strategy that aims to rebuild the stock if it falls below $B_{TARG.}$ A separate rebuilding strategy is required when the stock is estimated to be below $B_{LIM}$, because all targeted fishing is required to stop. Both aspects constitute rebuilding strategies, and it therefore makes sense that the HSP states that rebuilding should aim to return the stock to the TRP of $B_{TARG}$ (=$B_{48\%}$). Confusion appears to have occurred because of the possibility of interpreting the quoted statement as meaning the intent was that there should be no targeted fishing until the species achieved the $B_{TARG}$. This confusion is really a failure to understand the intent of the HSP with respect to depleted species. A clarification of this intent should remove this potential confusion. The HSP makes two clear statements about depleted and conservation dependent species:

> *Where the biomass of a listed species/stock is rebuilding towards to [sic] $B_{TARG}$, consideration may be given to deleting the species from the EPBC Act list of threatened species, or amending the category it is in. Deleting a species from the list of threatened species under the EPBC Act is effected via a legislative instrument issued by the Minister for the Environment and Water Resources. Advising the Minister that a recovering species that has rebuilt above $B_{LIM}$ should be considered for delisting will be the responsibility of AFMA on the advice of the AFMA board, however, any person can initiate the process. (DAFF, 2007, p24)*

Similarly, there is the statement in the section on the relationship of the Policy to the EPBC Act:

> *Where the biomass of a listed stock is above* $B_{LIM}$ *and rebuilding towards* $B_{TARG}$*, consideration could be given to deleting the species from the EPBC Act list of threatened species, or amending the category it is in.*
>
> *The relevant sections of the EPBC Act, primarily Part 13, will apply for any listing, amending, or deletion of a species from the list of threatened species.*
>
> *The best available science will underpin all key decisions in the application of the Policy and relevant provision of the EPBC Act. Stakeholders will be well informed and agencies will ensure transparency. (HSP, p. 7)*

Because this is the basis of the HSP, the assumption is often made that if a species were above the $B_{LIM}$ then the harvest strategy for whatever fishery is involved would be used to manage the fishery as per normal. For this reason, while the quotations above appear relatively clear in their intent, the use of the phrases "...may be given…" and "…could be given…" in lines 2 of each quote are often pointed to by Industry members when this failure to understand the intent of the HSP is mentioned.

While this appears simple to resolve by making the intent of the HSP explicitly clear there are difficulties because the issue is at least partly due to the interaction between the Fisheries and the Environment Acts. While it is clear that targeted fishing can begin, albeit slowly, once a species rises above $B_{LIM}$, it is not clear whether targeted fishing can occur on conservation dependent species even if they are above $B_{LIM}$. Clarifying that would appear to be beyond the scope of the HSP review because it involves the EPBC Act. However, if it is the intent of the policy, then it could still be made clear that for

those species that do not become conservation dependent, even though they dip below $B_{LIM}$, targeted fishing will be permitted to re-commence following the stock being assessed as being above the LRP.

This is an important clarification because it would reduce uncertainty over the conditions under which the HSP and its accepted harvest control rules would apply for the provision of management advice.

### 7.2.2 IN A MIXED FISHERY SHOULD DEPLETED STOCKS ALWAYS BE REBUILT TO $B_{TARG}$?

The HSP is clear that stocks that fall below the target reference point should be rebuilt to $B_{TARG}$. However, whether this is always the default target biomass level, $B_{48\%}$, is not made explicit but appears to be assumed. The discussion document (DAFF, 2012b) is correct to point out that the HSP allows for circumstances where the TRP may differ from this default under an array of circumstances where the default $B_{MEY}$ is not the expected yield from a fishery. This issue is covered in sectionon-Stevens act

> *3.3 Multi-species fisheries*
>
> *In fisheries that target or catch a number of species … it will be extremely difficult to maintain all species at the TRP because not all species can be effectively targeted and some species will be caught as incidental catches of the main target species. Importantly, MEY applies to the fishery as a whole and is optimised across all species in the fishery. As a result, some secondary species (e.g. lower value species) may be fished at levels that will result in their biomass remaining below their target biomass reference point (i.e. $B_{MEY}$). In such circumstances, the estimated biomass of these secondary species **must** be maintained above their limit reference point, $B_{LIM.}$ (HSP, p25)*

The management of secondary species may be conducted using harvest strategies designed for relatively data poor stocks (Dowling *et al*., 2008a), which, for example, may use catch level triggers that lead to increases in the data gathering and possible assessment requirements before further increases are permitted. If this path is adopted this would meet the requirements as listed under the quoted Section 3.3. To date this does not appear to be common in the major Commonwealth fisheries. They are either completely data poor (for example the Western Deepwater Trawl) or, if they are a mixture, the principle economic targets are assessed in some form of tiered assessment arrangement and any remaining secondary species and bycatch species are either dealt with under the lowest tier assessment available in the particular fishery or are included in the Ecological Risk Assessment (Haddon, 2012).

For the major mixed fisheries it would be valuable to conduct research to devise or recommend further data poor stock assessment methods to improve the effectiveness and hence the defensibility of the harvest strategies selected for a fishery.

### 7.2.3 REBUILDING TIMEFRAMES RELATIVE TO SPECIES' PRODUCTIVITY

In Australia there are guidelines for determining the timeframe over which stocks depleted below $B_{LIM}$ are expected to be rebuilt.

*The analysis of rebuilding strategy options and timelines can be complex and is further complicated by the social, economic and policy dimensions of such decisions. …*

*Typically recovery times are defined as the minimum of 1) the mean generation time plus ten years, or 2) three times the mean generation time. (HSP, 2007, 44)*

The notion of developing rebuilding strategies for overfished or depleted stocks is common to other formal harvest strategy policies around the world, for example, rebuilding strategies are part of the requirements for the Magnuson-Stevens Fishery Conservation and Management Act in the USA (US Department Commerce, 2007) as well as in the Harvest Strategy Policy introduced in New Zealand (Ministry of Fisheries, 2008). The details of how rebuilding strategies are implemented differ by country but the intent of moving an overfished stock back towards the target for that fishery is invariably the same. The definition of overfished is usually related to the stock depletion level being below the limit reference point.

In the USA the LRP is known as the MSST – minimum stock size threshold and the technical guidance (Restrepo *et al*, 1998) for implementing their management standards describes how to approach rebuilding strategies for overfished stocks:

*… the National Standard Guidelines require that special plans be implemented to rebuild the stocks up to the $B_{MSY}$ level within a time period that is related to the stock's productivity. This document does not propose a default rebuilding plan, because the time to rebuilding may depend on each stock's current level of depletion. Instead, the document presents the four key elements that should be considered in rebuilding plans: An estimate of $B_{MSY}$, a rebuilding time period, a rebuilding trajectory, and a transition from rebuilding to more optimal management. (Restrepo et al, 1998, p3)*

However, in addition it stated:

*To the extent possible, the stock size threshold* [MSST] *should equal whichever of the following is greater: One-half the MSY stock size, or the minimum stock size at which rebuilding to the MSY level would be expected to occur within 10 years if the stock or stock complex were exploited at the maximum fishing mortality threshold ... (Restrepo et al, 1998, p17)*

In the Magnuson-Stevens Act (U.S. Dept of Commerce, 2007), for a fishery that is overfished a plan is required that:

*(A) Specify a time period for rebuilding that fishery that shall –*

*(i) be as short as possible, taking into account the status and biology of any overfished stocks of fish, the needs of the fishing communities, recommendations by international organizations in which the United States participates, and the interactions of the overfished stock of fish within the marine ecosystem; and*

*(ii) not exceed 10 years, except in cases where the biology of the stock of fish, other environmental conditions, or management measures under an international agreement in which the United States participates dictate otherwise; (US Dept Commerce, 2007, p92)*

The guidelines (Restrepo *et al*, 1998) described a rebuilding plan based on these two clauses thus:

> *In the absence of data and analyses that can be used to justify alternative approaches, we recommend that a default rebuilding plan for stocks below the MSST be based upon the precautionary target control rule of Section 3.3 with the following extensions:*
>
> *The maximum rebuilding period,* $T_{max}$*, should be 10 years, unless* $T_{min}$ *(the expected time to rebuilding under zero fishing mortality) is greater than 10 years, when* $T_{max}$ *should be equal to* $T_{min}$ *plus one mean generation time. (Restrepo et al, 1998, p37)*

This strategy includes reference to the notions of 10 years, or $T_{min}$, the time to rebuild in the complete absence of fishing, and of adding one mean generation time to $T_{min}$ if 10 years would be insufficient. This appears to be the origin of one of Australia's potential timeframes for rebuilding. Ten years plus the mean generation time suggests that the well-known variability of recruitment events and the obscured but important relationship between spawning biomass and consequent recruitment events (Myers and Barrowman, 1996) has not been accounted for.

New Zealand has elected to base its rebuilding time frames on a notion of $T_{min}$. The standards document state:

> *The Harvest Strategy Standard specifies that where the probability that a stock is at or below the soft limit [*$B_{20\%}$*] is greater than 50%, the stock should be rebuilt to the target [*$B_{40\%}$*] within a time period between* $T_{min}$ *and* $2 \times T_{min}$ *(where* $T_{min}$ *is the theoretical number of years required to rebuild a stock to the target with zero fishing mortality).*
>
> *Mathematical projection models will generally need to be developed to estimate* $T_{min}$ *and to compare and contrast alternative rebuilding strategies. These will usually be probabilistic models that incorporate uncertainty in the projections. (Ministry of Fisheries, 2008, p11 – 12)*

In explanation for the notion of $T_{min}$ the same document states:

> $T_{min}$ *reflects the extent to which a stock has fallen below the target, the biological characteristics of the stock that limit the rate of rebuild, and the prevailing environmental conditions that also limit the rate of rebuilding. Allowing a rebuilding period up to twice* $T_{min}$ *allows for some element of socio-economic considerations when complete closure of a fishery could create undue hardships for various fishing sectors and/or when the stock is an unavoidable bycatch of another fishery. (Ministry of Fisheries, 2008, p12)*

There are some depleted species in Australia (e.g. Eastern gemfish; SESSF RAG papers, 2011 and 2012) that, given the previous variation inferred from the Tier 1 assessment, would not recover in a maximum of 10 years plus the mean generation time. For this reason the New Zealand strategy appears more general than that espoused in either the USA or in Australia. The strategy in the USA and Australia appears to default to one where recruitment is expected to be deterministically dependent on spawning stock size or at least considers that recruitment will operate relative to the median expected recruitment. The explicit suggestion of using stochastic projection models is directly related to accounting for the known risks arising from recruitment variability using Monte Carlo simulation methods (Francis, 1992). This latter approach would be more con-

sistent with the emphasis placed in the Australian HSP of using precautionary and risk based strategies.

In those fisheries where specific targeting is a characteristic then the option of closing such a fishery should severe depletion occur is an option that would have little impact on other fisheries (although the closure of the orange roughy fisheries using the 700 m trawl closure has also greatly reduced the catch of other deepwater species such as the various species of oreo). In mixed species fisheries, however, it is only by using mathematical simulation methods that the potential influence of allowing different bycatch TACs can be determined.

The rebuilding strategies in the USA are aimed primarily at the sustainability objective of the fisheries Act while the strategy in New Zealand's Fisheries Act provides for greater flexibility to take economic, social, and cultural needs into account. In a study of the economics outcomes of stock rebuilding (Larkin *et al*., 2007) used simulation models and determined that extending the rebuilding timeframe over the 10 years plus mean generation time could substantially increase annual harvests and economic benefits, depending on the productivity of the stock concerned and the economic discount rate used. The longer timeframes adopted in New Zealand for rebuilding depleted stocks thus allowed for both sustainability and economic objectives to be more balanced. Again, however, this would entail conducting a simulation study and continued monitoring of the depleted stock. It is clear that the need to satisfy the requirements for rebuilding plans leads to a substantial increase in the demands for technical analysis (Restrepo *et al*., 1998) and even with that analysis there remains great uncertainty because of the reduced information available (Punt and Ralston, 2007).

There is also a need to recognize that there are circumstances under which rebuilding would not be expected to occur. The marine environment is not a constant and the east and west coasts of Australia in particular are potential hot spots for significant change (Harris *et al*., 1988; Hobday and Lough, 2011). Within the SESSF there is already an instance where a relatively depleted species that was near the LRP (Jackass Morwong, *Nemadactylus macropterus*) exhibited a 20 year series of below average recruitment. This was eventually characterized as a change in the species productivity due to a regime change or regime shift, or at least an alteration in prevailing conditions that has lasted for decades (Wayte, 2012). There are a number of high profile international instances of species that have become seriously depleted having their fisheries closed only to fail to recover or rebuild (Walters and Maguire, 1996; Fu *et al*., 2001). An array of explanations have been proposed for the failure of the northern cod fishery to recover but the key finding is that the productivity of the stock has shifted to a different level and the recovery, if it ever happens, is not presently predictable. It is in recognition that there are factors other than fishing that can lead to fish stocks declining that has led to fish stocks found below the soft and hard limits in New Zealand to be referred to as depleted rather than overfished. This is more than a detail or nicety of language as it formally recognizes that there are other factors that may need attention when fish stocks decline. Regime shifts are a reality that cannot be dismissed and Wayte (2012) provides a clear example of the evidence required to demonstrate such events.

In addition to the effects of marine climate and changes in the prevailing environmental conditions affecting stock recruitment relationships it is also possible that some species, particularly when they were fished under a basket species category (e.g. gulper sharks)

may have been reduced to such a low level that the probability of them recovering would become influenced by random events. There is some confusion in the literature concerning the risk of extinction of marine organisms. At a recent conference on the State of the Oceans, that examined extinction risks (Rogers and Laffoley, 2011) there were numerous declarations about overfishing being a marine population stressor but extinctions being referred to were not of commercially fished species (except for the Chinese Bahaba – *Bahaba taipingensis* – which has become extremely valued in the Chinese medicine field). Nevertheless, more evidence has been compiled (Hutchings and Reynolds, 2004) that demonstrates that few populations recover rapidly with few observed populations changing in abundance over 15 year periods.

> *Reductions in fishing pressure, although clearly necessary for population recovery, are often insufficient. Persistence and recovery are also influenced by life history, habitat alteration, changes to species assemblages, genetic responses to exploitation, and reductions to population growth attributable to the Allee effect, also known as depensation. … Unprecedented reductions in abundance and surprisingly low rates of recovery draw attention to scientist's limited understanding of how fish behaviour, habitat, ecology, and evolution affect population growth at low abundance. (Hutchings and Reynolds, 2004, p 297)*

The assumption with most fishery population models is that at low abundance there will be density dependent effects that increase the survivorship of any recruits that are produced. Other density dependent effects are possible but the main one of interest relates to improved recruitment success (not necessarily more recruits, just more surviving; Myers and Barrowman, 1996). This density-dependent effect has been shown to be strong in some species but also weak in others. Where it is weak the species concerned are far more vulnerable to failing to recover if they become depleted (Keith and Hutchings, 2012). It has been 20 years since the northern cod off Newfoundland was recognized as collapsed and there are still no real signs of recovery.

The species that have been identified as highly depleted in Australia were generally depleted well before the introduction of the current HSP. Application of management strategy evaluations (MSE) to test of the effectiveness of an array of harvest strategies in the present HSP were made in the Reducing Uncertainty in Stock Status project (Dowling, 2011; Haddon, 2011; Klaer and Wayte, 2011; Plaganyi *et al*, 2011, 2012). Those MSE harvest strategy tests indicated that the harvest strategies tested should achieve their aims of preventing declines below the LRP and maintain the stock sizes at productive levels. However, of the MSE analyses conducted only Plaganyi *et al.* (2012) who considered the Coral Sea fishery for sea cucumbers analyzed the effect of systematic environmental changes such as climate change. While the MSE conducted on the scallop harvest strategy concluded that the harvest strategy would achieve the intent of the HSP, it could not predict the sudden death of more than 24,000t of scallops in Bass Strait in 2011 (an event mirrored down the east coast of Tasmania). This is an extreme example of where a non-fishery related phenomenon has a large influence on the state of a fishery stock.

# 8 Spatial Management

## 8.1 Introduction

The following issues have been identified:

- *The Review may consider if further guidance is required in relation to how to take into account closed areas and spatial management approaches when designing harvest strategies that are consistent with the Policy objectives.*

There is already a research project underway that is addressing the impact of marine spatial closures on stock assessments and consequently on the harvest strategy policy. There are some species which are relatively data-poor mainly because they are patchily distributed and such patches are heterogeneous in terms of productivity and are often highly variable in abundance due to natural variations (e.g. scallops and squid in the south, and sea cucumbers in the north). Finding limit and target reference points that can be validly applied to such species can be extremely difficult. It is clear that further guidance is required in the HSP with how to deal with such species in a manner considered to be consistent with the intent of the HSP.

## 8.2 Review of Research

### 8.2.1 OVERVIEW

Spatial management (e.g. marine closures and rotational harvesting) may be applied in various contexts within a harvest strategy. It can form the main harvest strategy framework (such as in a system of rotational closures), it can be used to augment a harvest strategy framework, or spatial management measures can be invoked as a control rule (see section below on control rules).

Spatial management is often favoured as a more cost-effective regime and/or in the absence of other information allowing alternative management measures. It can be applied to species for which the concept of an equilibrium biomass has limited meaning as a result of life history. It is also a useful approach for artisanal fisheries where monitoring and compliance limitations make TACs or catch controls impractical and data gathering is more challenging (Pilling *et al.* 2008); compliance with closure boundaries is managed in Australia using a satellite Vessel Monitoring System.

Worm *et al.* (2009) emphasise that conventional management tools used for industrial fisheries are generally unenforceable in small-scale artisanal fisheries when implemented in a top-down manner, and describes a system of co-management to rebuild depleted fish stocks on Kenyan coral reefs via a network of closed areas and the exclusion of beach seines. Worm *et al.* (2009) also cite other examples of successful rebuilding from Latin America, where open-access invertebrate fisheries for valuable invertebrates were transformed by the establishment of spatial management units that had exclusive access by local fishing organizations. Such closures can be successful where conventional management tools are likely to fail but if compliance in remote areas is at all an issue, then closures will also be prevented from being effective.

Spatial management has been successfully implemented in the form of a rotational harvest strategy or temporal pulse fishing frameworks for sessile species that have the propensity to experience 100% local depletion. This has been applied to various scallop fisheries (Dowling *et al.* 2008; Valderrama and Anderson 2007; Myers *et al.* 2001) and to sea cucumber fisheries (Dowling *et al.* 2007; Humble and de la Mare 2008), for which the concept of $B_{MSY}$ or $B_0$ has limited meaning.

The adaptive rotational harvest strategy developed by Humble and de la Mare (2008) closed harvested areas to further fishing until a designated degree of recovery occurred. Instead of a set rotation cycle length, local areas were harvested at a frequency determined by local recovery rates, which may differ over time and by location. Only local density and body mass estimates were required, yet modelling showed that this strategy out-performed one of a constant harvest rate and annual harvest strategy, without requiring estimation of life-history parameters or population abundance on a large scale. Valderrama and Anderson (2007) used an age-structured bio-economic model to demonstrate that economic rents where maximized by engaging in pulse fishing strategies for Atlantic Sea Scallops, whereby fishing only occurs following a multi-year closure period. Closures allowed biomass to accumulate undisturbed for several years in a row, leading to the harvest of premium-size scallops upon reopening of the fishing grounds. Closures also resulted in substantial reductions in operating fishing costs, and the rotational harvesting strategy was found to be robust with respect to a number of assumptions in the model.

Schnute and Richards (2001) agreed that a regulatory scheme that controls fishing mortality with large spatial and temporal fishery closures offers a management strategy more robust to uncertainty than direct control of catch, since only a small component of the stock gets exposed to the fishery. Pitchford *et al.* (2007) used a deliberately simple model, which describes an exploited fishery close to the point where small random perturbations can build up and lead to fishery collapse, to show that closures achieved via marine protected areas (MPAs) can buffer these random effects and alleviate the propensity to collapse. They showed that, compared with harvest control rules based on uncertain estimates of stock size, MPAs can substantially reduce the risk of fisheries collapse for only a very small cost to total yield. It should be remembered, however, that this work used a simple model of a fishery set up at the point of failure.

Rather than imposing a reserve and measuring its effect on profits, Sanchirico *et al.* (2006) examined when no-take zones were economically optimal. Closed areas were an economically optimal solution when the value derived from spillover from the reserve outweighs the value of fishing in the patch. There were circumstances whereby closing low biological productivity areas, and even sometimes low cost areas to fish, can result in greater fishing profits than when both areas are open to fishing.
As opposed to rotational spatial closures or a system of MPAs as the main harvest strategy approach, small, permanent closed areas may be used to augment a harvest strategy in the face of uncertainty (Dowling *et al.* 2008a,b). This is a measure that can be useful when:

- a harvest strategy framework, such as a trigger system, has been formulated, but there remains concern about the extent to which the framework is precautionary, and/or
- the fishery interacts with highly vulnerable species or habitat, and/or

- the fishery is in a developmental state where management should not overly inhibit access and flexibility, and/or
- the fishery may be highly sensitive to small stochastic perturbations (Pitchford *et al.* 2007).

### 8.2.2 CLOSED AREAS/SEASONS

Spatial management measures may be introduced as control rule responses to trigger levels being reached, particularly for highly vulnerable species or species with a high potential for localized depletion. For example, in a trigger harvest strategy framework, spatial closures or short-term provisions for fishing to cease in a given local area may be a possible response to a Level 1 trigger, if analysis shows that the trigger level has been reached as a result of concentrated fishing in a given area. Reuter *et al.* (2010) concur that closed areas, marine refugia or marine protected areas have been suggested as alternative management strategies to quota management, but point out that complications can arise if and when attempting to integrate their effectiveness into traditional stock assessments.

Spatial control rules are particularly useful for artisanal fisheries, where monitoring and enforcement may be difficult. They also lend themselves easily to community management in an artisanal context (Pilling *et al.* 2008). Matic-Skoko *et al.* (2011) described spatial closures being imposed as a control rule in Mediterranean artisanal fisheries, together with gear restrictions. Without compliance by fishers, however, such spatial control rules will fail.

### 8.2.3 MOVE-ON PROVISIONS

Often applied to small-scale fisheries on sessile species, "move on" provisions provide precautionary limits and, like daily catch limits, mitigate against localized depletion. They have been applied to beche-de-mer, lobster and trochus in the Australian Coral Sea hand collectibles fishery (Dowling *et al.* 2008a,b). Move-on provisions are typically defined in terms of a catch obtained within a given spatial region within a given time limit. For example, the Australian beche-de-mer move-on criterion is 5 t of combined species catch from one reef annually per permit; subsequent collection may not continue within a 15 nautical mile anchorage.

As with daily catch limits, move-on provisions are often adjunct control rules within, for example, a broader Total Catch or trigger framework. Move-on provisions require trust among fishers, particularly if the provision applies to some daily catch limit that is unable to be externally monitored.

### 8.2.4 MPAS AS INFORMATION SOURCES FOR MANAGEMENT

This harvest strategy approach involves the comparison of fished and unfished reference sites, typically via the use of Marine Protected Areas (MPAs). With the increasing implementation of MPAs, there is potential for improving decision making in management through comparisons of fished populations with populations in MPAs at spatially explicit scales. This approach is particularly applicable to fisheries targeting, for example, near-shore rocky reef species that exhibit spatial variation in harvest pressure and demographic rates, limiting traditional stock assessment approaches.

McGilliard *et al.* (2011) evaluated the potential use of the ratio of the density of fish outside a marine protected area to that inside it each year (the density ratio, DR) in a control rule to determine the direction and magnitude of change in fishing effort in the next year. Management strategy evaluation was used to evaluate the performance of this DR control rule (DRCR) for a range of movement rates of larvae and adults and other biological scenarios, and determined the parameters of the control rule that maximized cumulative catch (over 95 years) for each scenario.
Wilson *et al.* (2010) used a combination of data-based indicators sampled inside and outside of MPAs as well as model-based reference points for data-poor, sedentary near-shore species in a decision tree model. The model consistently improved total catches while maintaining the biomass and spawning potential ratio at levels within acceptable management thresholds.

The following additional control rules are also applicable in data poor fisheries, noting that these may be used in combination. For example, Welch *et al.* (2005) describes a precautionary approach to management for the data-poor king threadfin fishery taken in the commercial inshore gillnet fishery of northern Queensland, Australia, advocating a phased approach to risk-averse management. Simple assessment of commercial catch and effort data from the fishery did not indicate overexploitation. However, estimation of stock size using models was not possible, and more robust assessments are hampered by limited biological data, an absence of monitoring data, un-validated commercial log-book data, and a creep in fishing effort as technology advances. In such a data poor situation it was recommended that closures be used to protect spawning threadfin aggregations, as well as the use of maximum constant yield (MCY) to set a precautionary limit on annual catches.

### 8.2.5 ROTATIONAL SPATIAL MANAGEMENT

In a spatial management harvest strategy framework, the control rule is whether and which areas to open or close to fishing in a given year or fishing season. The general aim is to maintain some specified level of stock protection and thus indirectly avoid an explicit biomass based limit reference point. Usually this requires some form of pre-season survey to assess biomass or habitat conditions, and possibly the condition of the species (such as for Australian scallops) (Dowling *et al.* 2008a,b).

### 8.2.6 SPATIAL/TEMPORAL INCENTIVES TO AVOID THREATENED, ENDANGERED, OR PROTECTED SPECIES

Incentives relating to allowable catch in respect to location can be imposed as an over-arching regime in a fishery managed under a catch or effort quota system. Such an approach could also form a control rule in response to a reference point or trigger being reached, particularly in a multispecies fishery. Under such an incentive system, catch or effort would be decremented from an individual's quota at a rate relevant to a location or time in which they are operating, leading to a higher rate of consumption of the operator's allocation in areas where the potential impact on the stock would be greatest (Wilcox *et al.* 2010). This is useful if the species of concern is being caught in a specific season or area to which the incentive can be applied.

### 8.2.7 ADJUSTMENT OF SEASON LENGTH (E.G. FROM DEPLETION ANALYSIS)

For highly productive, short-lived species subjected to management by a fishing season of fixed duration or via catch or effort quotas, control rules may be implemented to adjust the season length or the TAC or TAE, according to the most recent information available. For example, if the fraction of the designated TAC/TAE is overshot, then the fishery may be closed or the effort is reduced. Such stocks are typically highly variable and the stock abundance may vary about an order of magnitude inter-annually, depending on the recruitment success in a particular year, although Tuck *et al.* (2001) also describe within-season changes to the TAC for the fishery for the longer-lived, less productive Patagonian toothfish. However, this fishery has few participants.

The Australian Arrow Squid harvest strategy is based on a system of real-time within-season management, where assessment approach is one of undertaking spatial and non-spatial depletion analyses. These project and adjust the cumulative catch for the season with a view to determining either season length or total catch or both for the season, and either may be updated during the season (Dowling *et al.* 2008b). Banana Prawns within the Australian Northern Prawn Fishery are also subject to within-season management (Dichmont *et al.* 2006).

# 9 Other Issues

## 9.1 Over-ride Rules

The discussion paper identified the following issues:

The Guidelines state that '*both the criteria for invoking exceptional circumstances and the response to them need to be clearly specified and agreed ahead of the need to apply them*', but provide little further guidance. In reality, such circumstances are unpredictable in their timing and nature and therefore may not be amenable to management by pre-determined rules.

- *The Review may wish to consider whether additional guidance can/should be developed around the development of 'metarules' to cope with exceptional circumstances.*

Such circumstances might include where assessments have not been completed due to unforeseen circumstances, where there has been an exceptional change in the nature of the fishery or where there has been a change in the ecological environment of the fishery unrelated to impacts of fishing. (HSP, p. 47)

In the previous discussion on meta-rules it was noted that they could be successful in achieving the intent of the policy while finding a practical way to manage complex situations with many interactions occurring at once. As such meta-rules constitute a back-up plan in rare cases of exceptional situations. Therefore it is again simple to conclude that this is an area that requires further detailed exploration and research.

## 9.2 Data related issues

The discussion paper identified the following issues:

Data requirements and availability can impact on the effectiveness of harvest strategies. For example, fisheries data used in assessments can be 12–18 months old by the time those assessments are applied within the harvest strategies, which has led to the application of 'recent catch rate multipliers' in the TAC setting processes (e.g. in the SESSF).

- *The Review may consider whether specific requirements regarding data specification and provision, relevant to harvest strategies, need to be specified within the Policy or Guidelines. This might include consideration of the point at which additional data collection (monitoring and assessment) is required when catches of non-quota bycatch species start significantly increasing (due to targeting or other reasons).*

Previous management strategy evaluations (MSE) of various harvest strategies in the SESSF (Wayte, 2009) have included the time delays in their testing and so such delays between data collection and utilization have received some testing. The use of the TAC adjustment rule based on the most recent CPUE analyses has already been tested with MSE (Wayte *et al*., 2009) and found not to alter the performance of the various harvest strategies procedures within the SESSF in terms of risk to the stock or overall catch levels, although it did significantly increase year-to-year variation in catches.

If the HSP began to require a minimum data requirement to be collected for all key commercial species this would have resource implications that might need to be taken in to account. Without those resources such a requirement could not be met.

# 10 Research Projects of Potential Value

## 10.1 Research Currently Under Way

There are already a number of research projects underway that may have implications for the review of the Harvest Strategy Policy. Unfortunately, given the timetable of the Australian research funding cycle a number of these projects have only recently begun. Nevertheless, they may generate outputs of value to the review committee. There are, for example, three FRDC funded projects currently underway:

### 10.1.1 THE RISK COST CATCH TRADE-OFF.

This work is FRDC project 2012/202, entitled *Operationalising the risk cost catch trade-off*, only started on October 1st 2012 and is due to finish in September 2014. This work will relate directly to the management of all fisheries and assuming the trade-offs can be characterized this work should be especially valuable for the more data-poor species and in making the HSP more internally consistent.

Its objectives are:

1. Extend AtlantisSE to enable the full suite of Commonwealth fishery types (e.g. data poor) to be simulated.
2. Using this modelling platform, define the risk-cost-catch trade-off between target species at different information and Tier levels.
3. In close consultation with managers and industry, develop a set of operational rules and clear quantitative guidelines for assessing the risk-cost-catch trade-off.

### 10.1.2 THE INFLUENCE OF CLOSURES ON THE HSP

This work is FRDC project 2011/032, entitled: *Incorporating the effects of marine spatial closures in risk assessments and fisheries stock assessments*. This project only started In April 2012 and is due to finish in November 2014. With the recent large increase in the number of spatial closures in the marine environment around Australia this has relevance to all Commonwealth fisheries. There is no doubt that various closures have influences fisher behaviour from the Northern prawn fishery, the SESSF, over to the Northwest Shelf trawl fishery. Exactly what influence that has on our perception of the stock status in each case remains unknown

Its objectives are:

4. Develop criteria and procedures for determining whether current methods for incorporating the effects of marine spatial closures in risk assessments and stock assessments are appropriate for all species.
5. Develop a method for incorporating the effects of marine spatial closures in risk assessments and stock assessments for those species where the current approach is not considered effective.
6. Develop a set of rules for determining TACs or catch limits based on the quantity and quality of data available on the species biology, the characteristics of the closure, and the extent of monitoring inside and outside of the closure.

### 10.1.3 THE MANAGEMENT OF BYCATCH SPECIES

This work is FRDC project 2011/028, entitled: *Development of robust methods to estimate acceptable levels of incidental catches of different commercial and byproduct species*. This project only formally started on February 1st 2012 and is due to finish on September 30th 2013. The work is of primary interest to both data-poor species and to those highly depleted species which are now subject to bycatch only TACs. The project stems from a series of FRDC funded workshops in 2011 (Haddon, 2012) that considered the problem of how to Analyse Trends in Abundance for Non-Target Species.

Its objectives are:

1. Develop guidelines and tests to determine if incidental catch levels for any species are likely to be unsustainable or contrary to the principles of the Harvest Strategy Policy, with particular reference to species under rebuilding strategies and provide case examples.
2. Conduct risk assessments to determine acceptable levels of incidental catch TACs for species under rebuilding strategies (e.g. School Shark, Blue Warehou and Gemfish as case studies) within the parameters of the Harvest Strategy Policy.
3. Determine whether any of the methods developed under objectives 1 and 2 can apply to relatively data poor species; develop guidelines for application to species for which there is only catch data.
4. Assess the feasibility of extending the methodology above in objective 1 to develop a practical and workable methodology to estimate acceptable capture limits for rare and TEP species.

## 10.2Research That Would be Useful

### 10.2.1 MULTI-YEAR TACS

Currently there is a growing use of Multi-Year TACs in those fisheries where they can be implemented. However, this strategy and the various means by which is has been and is being implemented have not been subject to formal management strategy evaluation. There are obvious risks of stock depletion if the multi-year TACs are set too high. Part of the implementation, for example, in the SESSF, is the production of breakout rules to aid deciding whether to break out of the sequence of TACs decided upon at the start of their implementation. While some criteria have been drafted for selecting those species deemed suitable for multi-year TACs these have yet to be tested formally using MSE, and in some cases a lack of resources is putting pressure on the RAG outcomes to maintain TACs in the face of uncertainty.

It is simple to conclude that more simulation testing work needs to be conducted to determine the utility of different criteria for selecting species as suitable for multi-year TACs.

### 10.2.2 ALTERNATIVE DATA-POOR HARVEST STRATEGIES

For the major mixed fisheries it would be valuable to conduct research to devise or recommend further data poor stock assessment methods of harvest strategies to improve the defensibility of management selected for such fisheries.

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