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Maximising net economic returns from a multispecies fishery

FRDC Project No 2015-202

Final Report

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Executive Summary

What the report is about

The Australian Commonwealth Harvest Strategy and Policy identifies maximising net economic returns as the primary objective of fisheries management. This has largely been interpreted as maximising the net economic yield (MEY) in fisheries. For multispecies fisheries, this has been based on maximising the net present value of total profit in the fishery over all stocks. The estimation of MEY has largely focused on the benefits to the fishing industry. This may potentially have adverse effects on consumers if achieving MEY results in lower catches and higher prices. At the same time, not all costs are considered when estimating profits to the industry. Costs such as the non-market value of species caught as bycatch are generally ignored. In this report, we examine how including benefits to consumers and the non-market costs associated with bycatch affect the definition of MEY (and how it relates to net economic returns).

Identifying MEY is just one part of the challenge facing fisheries managers. How to achieve MEY in multispecies fisheries is another. A major part of the study was how we might implement MEY in a multispecies fishery. Using a model based on the Southern and Eastern Scalefish and Shark Fishery (SESSF), we examine the effectiveness of different harvest strategies in achieving fishery wide MEY, and also how different approaches to estimating MEY affect the outcomes.

Aims and objectives

The project has two key objectives:

1 Development of a methodology for maximising net economic return to a multispecies fishery as a whole, and with regard to by-catch and discard species;

2 Development of a framework to operationalise the methodology into fisheries management objectives.

Methodology

The study involves reviews of the available literature on identifying and implementing MEY, as well the use of several fishery models. The first of these models is a "generic" model that is used to assess the impact of different assumptions around consumer benefits and bycatch on the optimal level of fishing effort and profits in a multispecies fishery. The second part of the project involves several models. These include a static long run equilibrium model based on the SESSF, a dynamic optimisation model and a series of dynamic models that are used to examine the impact of different harvest strategies on achieving MEY in a multispecies, multi-metier fishery such as the SESSF.

Results and key findings

Achieving fishery MEY may result in a reduction in net economic returns in a broader sense if the loss to consumers exceeds the gain to the industry. Such a loss may occur if supplies to the local market are reduced and prices paid by consumers increase. This results in a transfer of benefits from consumers to producers, which is considered undesirable in itself. However, if the loss to consumers is greater than the gain to producers then overall there is a loss of net economic returns. Similarly, the disutility associated with bycatch in fisheries may also affect our interpretation of "optimal" yields if non-monetary values are assigned. The “generic” multispecies bioeconomic model was used to estimate the impact on target fishing mortality rates of broadening the consideration of net economic returns to include also changes in consumer surplus and inclusion of non-market values associated with bycatch. The model is run stochastically while maximising profit but varying the number of species caught, their biological characteristics and prices, fishing costs, price flexibilities, bycatch rates and values.

The results of the analysis were largely as expect, namely that including consumer benefits into the definition of MEY resulted in a higher optimal level of fishing effort and yield, while including non-market costs associated with discards resulted in a lower optimal level of fishing effort and yield. The degree to which these factors affected the definition of MEY was, unsurprisingly, related to their overall magnitude relative to the benefits to the fishery.

Implementing MEY, once identified, also has several challenges. The study considered a range of harvest control rules, as well as other potential management options. The results of the model analysis suggest that “hockey-stick” harvest control rules in multispecies fisheries may overly restrict the catch of species that are currently above their target biomass. Given the higher abundance, catch of these species is likely to result in increased discarding and lower economic returns than might otherwise be achieved. An alternative harvest control rule that allowed higher than “optimal” fishing mortality rates for species that were above their target biomass resulted in less discarding and higher economic returns.

Having quota on too many species may be counterproductive, as the fishery is largely constrained by the quota for the main species. Imposing quotas also on secondary species can result in a situation where a minor species becomes a “choke” species, restricting the total fishery for little benefit. Reducing the number of species subject to quota constraints to only those that were most important (in terms of revenue) resulted in improved economic performance of the fishery as well as lower levels of discarding. However, in the model changes in targeting ability of the fleet was not considered, so monitoring of fisher behaviour in response to proposed management regimes that only have a few species under quota would be essential.

Introduction

An objective of the Fisheries Management Act 1991 is “maximising the net economic returns to the Australian community from the management of fisheries”, which has been interpreted as achieving the biomass that, on average, produces maximum economic yield (B_{MEY}) in the Commonwealth Fisheries Harvest Strategy (DAFF 2007).

The DAFF Report on the Review of Commonwealth Fisheries Harvest Strategy Policy and Guidelines (hereafter HSP) (DAFF 2013) identified that deriving appropriate economic-based target reference points in multispecies fisheries was a complex problem. While it is recognized in the HSP that achieving a fishery level net economic returns (NER) or maximum economic yield (MEY) may result in some minor commercial species being fished to levels below their individual biomass at MEY (or even biomass at maximum sustainable yield), this increases the risk to these species, and potentially may result in greater costs to the industry if substantial cuts to fishing are required to ensure their future recovery.

The Borthwick (2012) review of Commonwealth fisheries legislation, policy and management also highlighted the need to give greater consideration to issues of bycatch and environmental factors when managing fisheries, including when setting management targets. “Standard” methods for assessing MEY do not account for externalities, particularly in terms of bycatch and discards, which may affect the optimal outcome from a broader societal perspective.

A range of other complications were also identified during a technical review of economic issues (FRDC 2012/225) (Vieira and Pascoe 2013b). One such issue, which has also been raised by others (Hannesson 1993), is the impact of changes in consumer benefits from moving to a fishery profit maximisation target if this also results in higher prices to consumers. In such a case, a broader definition of MEY to include both consumer and producer benefits may be appropriate.

A previous FRDC project (FRDC 2011/200) developed a means to approximate economic-based target reference points in multispecies fisheries (Pascoe *et al.* 2015). These studies, however, did not consider potential constraints on targets to ensure that no species is reduced to levels that may result in high risk of stock collapse, nor the potential future costs of allowing minor stocks to recover to higher levels if necessary. Further, the previous studies have not identified how to monitor the transition to NER, particularly for minor species where data are limited and which also are potentially fished to a low biomass level. Finally, the projects did not consider how to set short-term targets (e.g. TACs) to achieve the long-term biomass targets. The DAFF (2013) report recognised that setting catch targets that are incompatible with the relative catch mixes is likely to result in substantial under-catch of some species and over-catch (and discarding) of others.

Developing harvest strategies that maximise net economic returns is also a different problem to that of identifying targets. The latter is an endpoint while the former is the process to achieve the end point. The purpose and aim of this project is to establish a practical and cost effective methods for managing a multispecies fishery towards maximising net economic returns as a whole, taking into account non-target catches and potential consumer externalities (i.e. impacts on consumer prices).

Such issues are currently relevant for several Commonwealth fisheries, and the Southern and Eastern Scalefish and Shark Fishery (SESSF) (a multispecies and multi-gear fishery) in particular. The SESSF is to be used as the main case study for the project, although it is expected that results will be applicable to other multispecies fisheries also. The SESSF is currently experiencing issues with balancing the catching capacity of the fleet (which is determined by the fleet size, structure and current gear composition) with catch targets.

The report (and project) is structured around two main themes: 1) identifying MEY (and also net economic returns) in multispecies fisheries; and 2) how MEY may be achieved through the use of appropriate harvest strategies (i.e. an implementation framework to achieve MEY). Each theme involves a review component as well as a modelling component. The latter is based around a case-study of the SESSF.

Objectives

The project has two key objectives:

1 Development of a methodology for maximising net economic return to a multispecies fishery as a whole, and with regard to by-catch and discard species;

2 Development of a framework to operationalise the methodology into fisheries management objectives.

Methods

The project involved two main components. The first component involved identifying MEY in multispecies fisheries, and how this relates to the concept of maximising net economic returns, particularly when other sectors (e.g. consumers) and externalities are considered. The second stage involved identifying fishery-wide MEY for a case study fishery, and examining the effectiveness of various management instruments and harvest strategies in achieving this.

What is MEY (and NER) in a multispecies fishery?

There has been considerable debate recently in the fisheries literature about the definition of maximum economic yield (MEY) (e.g. Christensen 2010; Grafton *et al.* 2010a; Norman-López and Pascoe 2011a; Grafton *et al.* 2012; Wang and Wang 2012b; Pascoe *et al.* 2013a), particularly in regard to the scope of the definition of benefits (i.e. to include or exclude regional economic benefits) as well as the way in which they are derived. Further, implications of bycatch and other potential environmental externalities (e.g. habitat damage) have not previously been considered in MEY determinations, although these may impose a (non-market) cost to society. A review of economic issues relating to setting MEY in a range of different types of fisheries (e.g. sequential fisheries, highly fluctuating, jointly managed, etc.) was undertaken by Vieira and Pascoe (2013b).

The first component of the study provides a review of these previous studies, and assesses the implications for the various interpretations for fisheries management in Australia. Multispecies MEY has generally been taken as the level of catch and biomass of each species, and effort of each fleet, that maximises the total economic profits across a fishery. Given that the objective of fisheries management is to maximise net economic returns rather than MEY *per se*, the impact of including broader considerations into a broader "MEY" target is considered through a modelling component. Building on the models developed in the project FRDC 2011/200, implications for multispecies MEY will be considered, taking into account bycatch species and also the potential for bycatch of iconic species. Similarly, the issue of market power is explicitly addressed through considering changes in consumer surplus (the benefits accruing to consumers) as well as fishery profits. The models identified above were modified to include a price-quantity (demand) relationship, and the implications for this on total benefits and appropriate targets will be estimated. Further details of these are presented below.

Review of MEY in multispecies fisheries

The potential economic benefits that fisheries can generate if managed effectively have been long established (Gordon 1954; Scott 1955). However, how these benefits can be achieved is less well established. While enhancing property rights, such as through individual transferable quotas (ITQs) has improved profitability in many fisheries (Dupont *et al.* 2005; Fox *et al.* 2006; Thébaud *et al.* 2014), in others differences in spatial stock structure or fisher location have resulted in the potential benefits not being realised (Costello and Deacon 2007). In other fisheries, implementing measures such as ITQs is not feasible due to large uncertainties in stock abundance and hence an inability to set an appropriate target catch (Buckworth *et al.* 2014), and in some cases such measures are actively resisted by groups opposed to "privatisation" of public resources (Smith and Wilen 2002).

Central to achieving the maximum economic benefits that fisheries are capable of is the need to identify the appropriate target level of catch, biomass and effort that as associated with this. For single species fisheries, identifying these targets has been the subject of considerable previous bioeconomic research, usually assuming a homogeneous fleet, leading to a range of analytical solutions depending on the underlying biological assumptions (Anderson 1975; Clark 1976; Anderson 1986). Variations have also been introduced, for example to take account of the discount rate to derive a value of dynamic maximum economic yield (Clarke *et al.* 1992; Grafton et al. 2010a), non-malleability of capital (Clark *et al.* 1979) or variable prices (Anderson 1973).

With the exception of fairly simple two-species predator-prey systems (e.g. Flaaten 1991; 1998), the extension of analytical measures of maximum economic yield into multispecies fisheries have been less successful, namely due to the large number of technical interactions that need to be simultaneously considered. Instead, models of multispecies fisheries, many of which also contain multiple gear types or fleet segments (e.g. Ulrich *et al.* 2002a; Punt *et al.* 2011), have been numerically based, and usually related to a particular fishery.

Defining maximum economic yields in such multispecies, multi-fleet fisheries is complex, as the optimal effort level, catch and biomass is depend on the relative costs and harvesting capacity of the different sub-fleets. Further, the concept of maximum economic yield implies that returns to the fishers are to be maximised, while other groups may also have an economic vested interest in the fishery (e.g. consumers and conservation groups). As a consequence, there has been considerable discussion recently in the fisheries literature about the definition of MEY (e.g. Christensen 2010; Norman-López and Pascoe 2011b; Grafton et al. 2012; Wang and Wang 2012b; Pascoe et al. 2013a), particularly in regard to the scope of the definition of benefits (i.e. to include or exclude regional economic benefits) as well as the way in which they are derived. Further, implications of bycatch and other potential environmental externalities are not generally considered in MEY determinations, although these may impose a (non-market) cost to society.

The concept of maximising net economic returns has been used in Australian policy to reflect this potential broader economic objective for Commonwealth-managed fisheries (DAFF 2007). While the current policy states that this is operationalised as maximising fishery economic profits over time, this has been recently been reviewed and the need to take into consideration a broader range of sectors has been identified (Borthwick 2012; Vieira and Pascoe 2013a).

The aim of this part of the study was to review the relevant literature around MEY and assess the implications for the various interpretations of MEY for fisheries management. The basic concepts of economic returns and MEY are first explained, followed by the various interpretations of MEY that are not typically captured in MEY estimation, as well as the challenges that may arise when moving from a single-species to multispecies fisheries management. The potential implementation issues in both general and multispecies contexts are then discussed. The outcomes of this review are presented in the results section of the report.

Modelling MEY versus NER – considering bycatch and consumer impacts

The Australian Commonwealth Fisheries Act identifies maximising net economic returns to the community as the over-arching objective of Commonwealth fisheries management. More recently, the Commonwealth Harvest Strategy and Policy Guidelines (DAFF 2007) have interpreted this as maximum economic yield, and specified this as maximising the economic profits in the fishery. A recent review of the policy has identified that this excludes consideration of potential loss of consumer benefits, and also does not consider non-market costs associated with bycatch (Productivity Commission 2016).

Incorporation of these factors into a "MEY" consideration will result in a potentially different estimate of the optimal level of catch and effort. A priori, based on traditional and simple economic supply and demand principles, we would expect several outcomes to occur. Referring to Figure 1, what we might expect is that:

- If price flexibility (the percentage change in the price of a product due to a 1% change in quantity supplied of that product) is zero, then marginal revenue (MR) and average revenue (AR) are the same. Maximising fishing profits by equating marginal revenue to marginal cost (MC) is optimal from a social perspective (Figure 1a);
- If price flexibility < zero, both MR and AR decline with increasing quantity of catch, with MR declining at a faster rate. Maximising profits to producers (MR=MC) may lead to loss of consumer and producer surplus (i.e. deadweight loss) seen as the shaded area in Figure 1b; and
- If other externalities exist, such that the marginal social cost (MSC) is greater than the marginal (fishery) cost, maximising fisheries profits (MR=MC) may result in too much fishing activity from a societal perspective (which is optimised at MR=MSC) (Figure 1c).

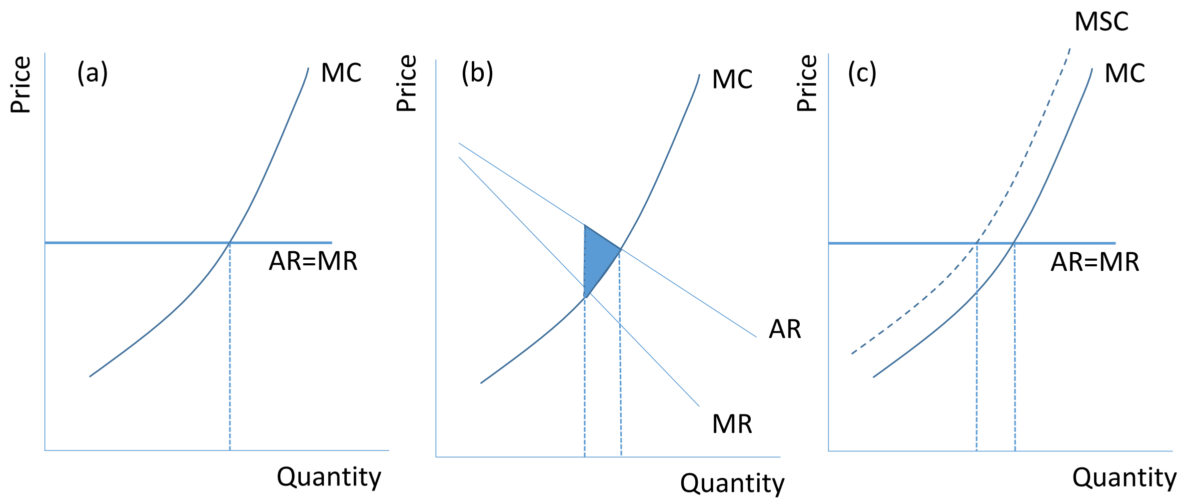


Figure 1. The relationship between price and quantity showing expected marginal cost (MC), average revenue (AR) and marginal revenue (MR) curves and the dead weight loss (shaded area). The marginal social costs (MSC) is also shown in panel (c).

The extent to which maximising fishery profits is equivalent to maximising economic returns to society as a whole will depend on the characteristics of the fishery, including the strength of any price-quantity relationship and the "importance" of bycatch. The aim of the analysis in the next section is to examine how the consideration of consumers and environmental impacts affects the measure of "net economic benefits", and how this compares to a "traditional" view of MEY as well as determining the effort/catch/biomass combination that maximises fisheries profits. The study uses a generic multispecies bio-economic model that includes realistic parameter values (based on existing multispecies fisheries), but does not relate to any fishery in particular. The approach is to use a generic model to estimate traditional measures of MEY, and compare these with measures that also include consumer surplus and also negative externalities, and the effect of these on "optimal" fishing mortality for species within a multispecies fishery.

Model specification

Here, we outline the structure and content of the generic bio-economic model. A key objective of the study was to develop a formal analytical framework on the basis of which a set of rules of thumb could be established, to guide managers in setting appropriate target reference points for species in multispecies fisheries, especially those for which data are limited. Thus, the generic bio-economic model has the functional form that is accepted best practice whereby production is captured by a growth function. Two types of growth models are commonly applied based on different assumptions: the Schaefer model assuming an underlying logistical growth (Schaefer 1954; Schaefer 1957); and the exponential model developed by Fox (1970) based on a Gompertz growth function. Although the logistic model is commonly employed due to its simplicity, the exponential growth model has been found to be more broadly applicable to a wider range of fisheries (Silliman 1971; Halls *et al.* 2006).

In this study we apply the model based on Fox (1970) which has the form:

$$C_i = q_i K_i E \exp(-q_i E / r_i) \tag{1}$$

where r_i is the instantaneous growth rate of species *i*, K_i is the carrying capacity of species *i*, q_i is the catchability coefficient of species *i* and *E* is the level of effort applied to the fishery as a whole.

The model is solved as a non-linear optimisation problem with the base case objective function specified as:

$$\underset{E}{Max}\ \Pi = \sum_i p_i C_i - c(E,C) \tag{2}$$

where Π is total fishery profits, p_i is the price of species *i*, and *c(E,C)* is the total cost of fishing, which is a function of the level of effort and catch.

Additional terms are added to account for the loss of consumer surplus and producer surplus and the penalty associated with catching bycatch such that the objective function for the scenarios is formulated as:

$$\max_E \Pi = \sum_i p_i C_i - c(E,C) - DWL * \left(ProdSloss + ConsumSloss\right) - icon * BycatchP \quad (3)$$

where *ProdSLoss* is the loss component in producer surplus*, ConsumSloss* is the loss component of consumer surplus and *BycatchP* is the non-market cost associated with bycatch. *DWL* and *icon* are 0-1 dummy variables for the scenario modelling that take the value of 1 if the cost is included in the scenario, otherwise a value of 0. These additional cost variables are determined as:

$$\mathrm{ConsumSloss(E)} = \sum_i (0.5 * (\mathrm{MSY_i} - \mathrm{C_i}) * (\mathrm{p_i} - \mathrm{p_i'})) \quad (4)$$

where MSY_i is the catch at MSY and $\mathrm{p_i'}$is the average price at MSY, and

$$\mathrm{ProdSloss(E)} = 0.5\left((\sum_i \mathrm{MSY_i} - \mathrm{C_i}) * \mathrm{p_i} * (1 - \mathrm{crewshare} - \mathrm{freight_i})\right) * \mathrm{c(E - E_{MSY})} \quad (5)$$

where E_{MSY} is the effort (*E*) at MSY and crew share and freight are the share of costs of revenue. If E < $\mathrm{E_{MSY}}$, then the "loss" of producer surplus is negative i.e. a gain to producers.

The rationale for equations (4) and (5) is illustrated in Figure 2. Assuming a linear demand (AR) curve, the loss in consumer surplus is half the area defined by (p-p')*(MSY-MEY). The loss in producer surplus is the area beneath the original price (p') and the supply curve (defined by the marginal cost, MC). This is potentially a non-linear relationship based on the cost of fishing effort and the relationship between fishing effort and catch. The change in costs will be a function of the change in catch (in terms of crew share, freight etc.) and the change in costs due to changed levels of fishing effort.

The transfer of consumer surplus to producer surplus (Figure 2) is captured as part of the fishery profit, and results from the higher prices being received given the lower quantity landed. This is given by:

$$Transfer = (p - p')*Q(MEY) \quad (6)$$

The additional cost associated with bycatch is given by

$$BycatchP = \varphi E * strikerate \quad (7)$$

where *ϕ* is the non-market costs associated with a unit of bycatch (e.g. a seal or dolphin), *E* is the level of fishing effort and *strikerate* is the average rate per unit of effort (such that *E*strikerate* is the number of iconic species that died as a result of fishing).[1]

[1] There is no stock dynamics model of the bycatch species in the bioeconomic model. Bycatch is just assumed to be linearly related to fishing effort.

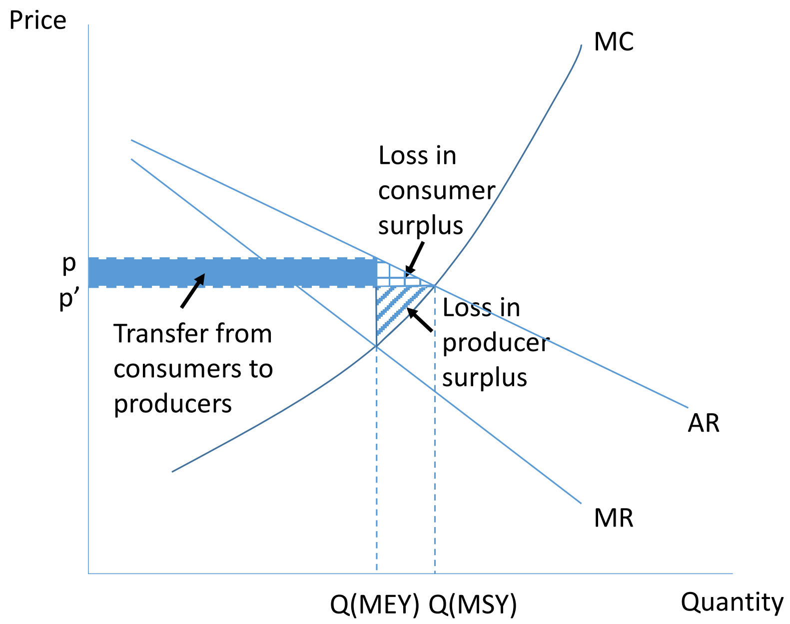


Figure 2. Consumer surplus loss, producer surplus loss and transfers from consumers to producers.

Price is determined from the level of catch per species allowing for flexibility of change with total landings (i.e. a decreasing price with greater volumes landed). It is assumed that the fishery is moving from a position at MSY to one reflecting MEY, and hence price changes take into account the amount of change in catch from MSY to MEY, such that

$$p_i = p_i^{'}\left(1 + flex_i\left(\frac{C_i}{MSY_i} - 1\right)\right) \tag{8}$$

where p_i is the price of species *i*, *flex* is the estimated constant on the price flexibility relationship, which is also species-specific. The average price at MSY is p_i'. Total cost (TC) is dependent on effort and revenue of amount landed as:

$$\mathrm{TC} = \mathrm{c}E + (crewshare + freight) * \sum_i p_i C_i \tag{9}$$

where *c* is the cost per unit of fishing effort (*E*). Crew share and the unit cost of freight per total revenue are predetermined constants. The model determines the level of fishery-wide effort that maximises total fishery profits across all species. The catch of each species at this level of effort is effectively the maximum economic yield of that species. Of interest in this study is the relationship between biomass at MEY and at MSY. The biomass of each species at MEY is given by:

$$B_{MEY_i} = C_i / q_i E \tag{10}$$

whereas B_{MSY} is given by K_i / e for the Fox model where *e* is the exponential function (e^{-1}=0.367879).

Scenarios and simulations

The model was run under four scenarios (Table 1). A base case was also run (Scenario 0) representing the standard assumptions commonly used in bioeconomic model estimates of MEY, namely no price-quantity relationship and an objective of maximising fishery profit only. The other scenarios were compared to the base case to explore the impact of the alternative assumptions regarding the definition of maximum net economic returns.

The first alternative scenario involved introducing a price-quantity relationship, but defining MEY as maximising fishery profits. In the second scenario, both consumer and producer surplus were considered in the definition of MEY, with the loss in each deducted from the total fishery profits. The third scenario introduced non-market values associated with bycatch, but ignored consumer impacts, while the last scenario considered both consumer and bycatch impacts.

Table 1. The scenarios tested in the simulation analysis of the effects of externalities

Name of Scenario	***Explanation of key assumptions and settings. All other parameters as in Table 2.***
Scenario 0 – Base case	No price flexibility Maximise fisheries profits only
Scenario 1 – Fishery profits only	Price flexibility (ranging from -0.035 to -0.75 so fairly inflexible) Maximise fisheries profits only
Scenario 2 – Consider consumers	Price flexibility (ranging from -0.035 to -0.75 so fairly inflexible) Maximise producer and consumer surplus
Scenario 3 – Consider bycatch	Price flexibility (ranging from -0.035 to -0.75 so fairly inflexible) Maximise fisheries profits Introduce non-market value (as a cost) for iconic species bycatch
Scenario 4 – Both bycatch and consumers	Price flexibility (ranging from -0.035 to -0.75 so fairly inflexible) Maximise producer and consumer surplus Include non-market value (as a cost) for iconic species bycatch

The model was run stochastically with 10,000 simulations of each scenarios, each with the same sequence of “random” values. The model was run with varying number of species (from 3 to 20), each with varying biological characteristics (q, K and r) (Table 2). Price flexibilities were also varied randomly within the range estimated by Bose (2004) for key species in the SESSF. Fishing cost data were also drawn from observed ranges for the SESSF (George and New 2013; Skirtun and Green 2015).

There are a wide range of species caught as bycatch, some of which are considered iconic (e.g. turtles, seals, dolphins, seabirds). In some cases, bycatch includes species which are less iconic, but threatened or endangered (e.g. sea-snakes), and their capture would presumably impose a high (non-market) cost on society. For the purposes of this study, we have based the bycatch component of the model on seal bycatch. Bycatch strike rates were not based on empirical estimates of bycatch in any particular fishery, but were consistent with rates observed for some gear types in the SESSF by Goldsworthy and Page (2007) and Tuck *et al.* (2012).

Deriving a value for the iconic bycatch species required several assumptions. Several previous nonmarket valuations studies have valued marine wildlife including seals as a value per household (Langford *et al.* 1998; Kontogianni *et al.* 2012; Lew 2015) rather than a value per seal per se. Bulte *et al.* (2005) found that these values differed based on the threat facing the animals; the value of seeing the animal in the wild was less than the cost of having the animal killed though some human activity. Hutton *et al.* (2010a) assumed a value of AU$1/kg for seals, reflecting the increasing value larger seals over smaller seals. Hannesson *et al.* (2009) assumed a value of US$7/kg for all non-commercial predators (including seals) in a US fishery. Hutton et al. (2010a) assumed a value of AU$1/kg for seals (in general), reflecting the increasing value larger seals over smaller seals. Hannesson et al. (2009) assumed a value of US$7/kg for all non-commercial predators (including seals) in a US fishery. Adult male Australian Sea Lions (*Neophoca cinerea*) weight in excess of 200kg, and up to 105kg for females (Ling 2009). For the purposes of the model analysis we assumed a value of between $100 and $1,000 per individual, representing a range of $1-$5/kg roughly for an "average" seal to test the sensitivity of the optimal yield to the bycatch penalty (Table 2).

Table 2. The parameter value ranges used in the analysis

Parameter	Parameter symbol	Minimum	Maximum
Number of species	*i*	3	20
Catchability	*q*	0.0001	0.05
Carrying capacity	*K*	800	60000
Cost per unit effort	*c*	300	1800
Price flexibility	*flex*	-0.75	-0.035
Freight (share of revenue)	*freight*	0.025	0.1
Crew share of revenue	*crewshare*	0.20	0.45
Bycatch rate (per unit of effort)	*strikerate*	0.05	4
Non-market value of iconic species mortality	*ip*	100	1000

In most cases, a uniform distribution was assumed for each of the parameters. For the intrinsic growth rate (*r*), an inverse relationship between growth (*r*) and carrying capacity (*K*) was assumed, based on previous work by Martell and Froese (2013). The relationship used, given by $r = \exp(1.489207 - 0.319084 * \log(K) + \varepsilon)$ where $\varepsilon = N[0, 0.3958]$, was derived by Pascoe et al. (2015) based on a number of stochastic simulations using similar parameter assumptions for *r* and *K* as used in this model. The fish price at MSY was also derived using a Poisson distribution, given by $p = Poi(2) + 1$ where the +1 was included to avoid zeros (ensuring that the minimum price was 1).[2]

The results from these analyses are presented in the results section.

Operationalising MEY

The target reference points developed in the previous FRDC project (2011/200) are long-run goals, and it is likely that in the transition to MEY, and with natural stock variability, higher or lower biomass levels may be obtained in any one period. For many minor species, information required to derive accurate assessments of relative biomass for monitoring purposes may not be available, so other indicators of stock status may be required.

As with the previous part of the study, this part involved both a review component and a modelling component. The review component considered how multispecies fisheries may be moved closer to MEY, particularly examining at the effectiveness of different governance systems and how they may help move a multispecies fishery towards MEY in the face of imperfect information and potentially incorrect catch targets. These will include how input and output controls work with limited information (e.g. Pearse and Walters 1992; Walters and Pearse 1996; and Walters 1998 suggest a range of strategies), as well as other approaches not currently in use in Australian fisheries (e.g. royalties). The potential role for stakeholder involvement in decision making is also examined (e.g. Pearse and Walters 1992).

In this modelling stage of the project, the case study fishery model developed in FRDC 2011/200 (Pascoe et al. 2015) was redeveloped and updated, and used to test a range of harvest strategies. This included consideration of some of the alternative approaches as well as examination of current harvest strategies.

Development of the fishery specific case study model

Achieving maximum economic yield in fisheries requires the estimation of appropriate target reference points. In Australian Commonwealth fisheries, these have been based on the ratio of the biomass at MSY. For many fisheries, relatively little information is known about the biology and economics underlying the level of harvest and economic performance of the fleet. In such cases, a default target reference point of $B_{MEY} = 1.2B_{MSY}$ has been set under the Commonwealth Fisheries Harvest Strategy Policy and Guidelines (DAFF 2007).

As is recognised in the Commonwealth Fisheries Harvest Strategy Policy and Guidelines, in multispecies fisheries, a single target reference point, common across all species is unrealistic. A generic framework can provide more appropriate proxy target reference points based on limited information on the species.

[2] The basis of the Poisson distribution was the range of prices observed for the species caught in the SESSF (see ABARES 2016 and earlier editions), where most species have a relatively low (and similar) price but a small number of species had substantially higher prices.

In the project FRDC 2011/200 (Pascoe et al. 2015), a multispecies generic fisheries bio-economic model was developed and the optimal biomass for a wide range of species under different combinations of growth, catchability, cost and price assumptions. To test the applicability of the generic framework, a bioeconomic model of a multispecies multi-fleet fishery was also developed, based on the Commonwealth trawl sector of the Southern and Eastern Scalefish and Shark fishery (SESSF). The latter bioeconomic model is not a "true" model of this actual fishery as only a subset of species from the fishery is included in the model, and the associated fleets that target those specific stocks, but the model is used to estimate bioeconomic target reference points for the species considered, allowing various fishing strategies to be defined spatially and by gear (analogous to the metier concept used in many bioeconomic models).

Building on these previous models (FRDC 2011/200), the implications for multispecies MEY were considered taking into account bycatch species and also the potential for discards. The species and fleet activity (series of metiers) included in the models were also expanded (18 species, 11 metiers). As with the previous analysis, the models was run stochastically and the results are summarised herein.

The Southern and Eastern Scalefish and Shark Fishery (SESSF)

As with the previous project (FRDC 2011/200), the application of the model was based on the case of the Southern and Eastern Scalefish and Shark Fishery (SESSF). While the previous project focused on the Commonwealth Trawl Sector (CTS) of the SESSF, the current project also included the Gillnet, Hook and Trap Sector (GHTS) of the SESSF, separating Gillnet from the Hook and trap metier.

The CTS is a diverse sector comprising of two main fleets (trawlers and Danish seiners), and is one of Australia's oldest commercial fisheries. The bulk of the catch in the fishery consists of twenty demersal species or species groups managed by quota, with the main markets being the Sydney and Melbourne fish markets (mostly fresh, some frozen) (Penney *et al.* 2014). The key species landed are Tiger Flathead, Pink Ling, Silver Warehou and Blue Grenadier. The CTS covers an area of the Australian Fishing Zone extending southward from Sandy Cape in southern Queensland, around the New South Wales, Victorian and Tasmanian coastlines to Cape Jervis in South Australia. Otter trawlers generally operate on the continental shelf and upper shelf to around 500 metres, and harvest a range of demersal species on the shelf such as Tiger Flathead, John Dory, Morwong and Silver Trevally, and offshore species such as Gemfish, Silver Warehou, Pink Ling, Ocean Perch and Mirror Dory. In this analysis we include explicitly the trawl metier targeting Blue grenadier. We also include a Tasmanian trawl metier that includes vessels fishing out of ports in Tasmania and targeting trawl species on the East and West coasts of Tasmania. The Danish Seine fleet comprises generally smaller, lower engine power vessels operating in shallower waters and target three main species (Tiger Flathead, School Whiting and Morwong). The total landings of all the species in 2012-13 was 10,724 tonnes. The CTS is financially the largest component of the SESSF. In 2012-13, the CTS had a gross value of production of $50.3 million, representing 62% of the gross value of production of the whole SESSF (Stephan and Hobsbawn 2014).

The Gillnet, Hook and Trap Sector (GHTS) includes the Scalefish Hook Sector (SHS), the Shark Gillnet and Shark Hook sectors (SGSHS), and the Trap Sector. The GHTS extends south from southern

Queensland to the western border of South Australia, and includes waters to the south of Tasmania. Gear types that can be used in the sector include gillnets, droplines, demersal longlines, automatic longlines and, to a lesser extent, traps. A large portion of the catch in the sector is caught by operators who engage in at least some gillnetting (59 per cent by volume in 2012–13). The SHS employs a variety of longline and dropline hook fishing methods, and shares many target species with the CTS. The SGSHS use demersal gillnet and a variety of line methods to target Gummy Shark (*Mustelus antarcticus*). School Shark (*Galeorhinus galeus*) was historically the primary target species in the fishery, but biomass was reduced below the limit reference point around 1990. It remains an important byproduct species and is the second most economically important species in the fishery. In 2012– 13 the total catch of all species in the GHTS was 3,517 tonnes. The gross value of production in the GHTS in 2012–13 was $22.6 million (Skirtun and Green 2015).

Significant spatial closures have been implemented since 2003 to address the bycatch of protected species, in particular Australian Sea Lions and dolphins, and to protect breeding School Shark populations. In 2010, extensive closures to manage bycatch of protected species were implemented around the breeding colonies of Australian Sea Lions to protect sea lions and in the coastal waters of south east South Australia to protect dolphins. The sea lion and dolphin closures prohibit the use of gillnets in the closed areas (AFMA 2015), and have meant that gillnet effort has become more concentrated off the Victorian coast (Skirtun and Green 2015).

Species and fleet structures used in the case study model

A set of criteria was established to determine the most suitable species to include in the study and determine the choice of fleets. First, as MEY has already been assessed for the multispecies Great Australian Bight trawl sector of the fishery (Kompas *et al.* 2012a), this sector was excluded from the model analysis, along with the stocks targeted by this fleet. Second, several species that were highly targetable were excluded as these could be effectively considered single species sub-fisheries (e.g. Royal Red Prawns and Orange Roughy). While also highly targetable, Blue Grenadier was included in the model as this is an important component of the trawl sector. This resulted in eighteen species, nearly all of which are caught using a range of fishing methods (Table 3).

Catches of other species not considered in Table 3 were aggregated into an "other" category. These were included in the model using a simple (aggregated) catch-effort relationship but without a dynamic stock model. Catch-effort relationships for these species were estimated as a quadratic function from fishery data over the period 1983-2012. Implicitly, the analysis assumes that these stocks are all in equilibrium at each level of observed effort.

The trawl component in the model included several fishing strategies, targeting a range of species, which is mixed in nature. Within a mixed fishery, catch composition can only vary through changing either the gear or the area fished, and an economic target reference point needs to take into account where this optimal level of effort is applied across the fishery. The concept of a "metier" is useful in such a context. The use of metiers in bioeconomic models to represent fleet activity is relatively common in European models (Biseau 1998; Pascoe and Mardle 2001; Ulrich et al. 2002a; Ulrich *et al.* 2002b; Ulrich *et al.* 2007; Pelletier *et al.* 2009), and has previously been applied to Australian (Ziegler

2012) and New Zealand (Marchal *et al.* 2009) fisheries. It differs from the concept of a sub-fleet in that a sub-fleet generally consists of a subset of a fleet with similar characteristics, whereas a metier represents a fishing activity that different vessels may participate in.

Table 3. Species/stocks considered in the case study and the main gears that catch them

Species	Main fishing gear			
	Otter Trawl	Danish Seine	Hook, line, trap	Gillnet
Morwong East	X		X	
Morwong West	X			X
School Whiting	X	X		
Eastern Gemfish	X			
Silver Warehou	X			
Tiger Flathead	X	X		
Pink Ling East	X		X	
Pink Ling West		X	X	
Blue Warehou	X			
School Shark	X		X	X
Gummy Shark	X		X	X
Blue Grenadier	X			
Blue eye Trevalla			X	
John Dory	X	X		
Ocean Perch	X		X	
Mirror Dory	X			
Ribaldo	X	X	X	
Silver trevally	X			

In this case study, eleven metiers were identified for inclusion in the model (Table 4). The key unit of fishing effort included in the model is the number of shots in each metier. From the data, many trawlers operated in all five trawl metiers, while some only operated in one or two. Similarly, most Danish seiners operated in both metiers, but tended to operate most in one or the other. Hence, vessels were not assigned to a metier directly, and the model was free to allocate fishing effort to each metier (essentially reflecting changes in vessel fishing activity); depending on the scenario modelled.

Table 4. Métiers included in the model

Metier	Metier code
Trawl metiers:	
• Shelf trawl – Eden to Sydney (NSW)	T_In_NSW
• Shelf trawl – Eastern Bass Strait (EBS)	T_In_EBS
• Offshore - NSW	T_Off_NSW
• Offshore – EBS	T_Off_EBS
• Blue grenadier trawl metier	BGren
• Tasmanian Trawl metier	TasT
Danish seine metiers:	
• Bass Strait (west of Lakes Entrance)	DS_BS
• Eastern Bass Strait (east of Lakes Entrance, Eden to NE Tas) (EBS)	DS_EBS
Gillnet metiers:	
• Gillnet Eastern Bass Strait (EBS)	GillE
• Gillnet Western Bass Strait (WBS)	GillW
Hook and trap metier:	
• Fishery wide	Hook

Model descriptions

As MEY is a long-run equilibrium concept, operationalising MEY in a shorter management time frame (e.g. 10 years) may not guarantee the increase in fisheries profits. To compare short-run versus long-run dynamics, a two-step approach was undertaken.

Firstly, a static equilibrium optimisation model was developed. MEY was estimated as the levels of fishing effort, catch, and biomass that maximise the overall fishery profits. The optimal fishing mortality rate (f_{mey}) for each species and metier was then calculated, which forms the basis for setting total allowable catch (TACs) in the second model; the simulation model.

Secondly, a simulation model, in which TACs were set given a range of potential alternative control rules, was constructed in order to a) simulate how MEY-based TACs under different scenarios (e.g. different quota reallocation assumptions, parameter uncertainties) may affect the levels of biomass, catches, and profitability, and b) to explore the implications for multispecies MEY when taking into account bycatch species and the potential for discards.

Long-run static equilibrium optimisation model

The long-run models are based on a set of equilibrium yield curves for a mixed fishery, with a randomly varying number of species. The models are based on the Fox equilibrium catch-effort model, where the equilibrium catch (C_i) of the key species *i* is given by:

$$C_i = K_i \left(\sum_m q_{i,m} E_m \right) \exp\left(-\sum_m q_{i,m} E_m \Big/ r_i \right) \tag{11}$$

where r_i is the instantaneous growth rate of species *i*, K_i is the carrying capacity of species *i*, $q_{i,m}$ is the catchability coefficient of species *i* in metier *m*, and E_m is the level of effort applied to metier *m*.

Catch of the key species within each metier ($C_{i,m}$) is approximated by:

$$C_{i,m} = \frac{q_{i,m} E_m}{\left(\sum_m q_{i,m} E_m \right)} C_i \tag{12}$$

The above equations only consider the major species identified in Table 3. Catches of all other species ($C_{i=o,m}$) were aggregated into an "other" category that were included in the model as a non-linear catch-effort equation, but without a specific dynamic stock model for each species, given by:

$$C_{i=o,m} = \alpha_m E_m - \beta_m E_m^{\ 2} \tag{13}$$

where α_m, β_m are metier specific parameters estimated from catch and effort data.

The model is solved as a non-linear optimisation problem with the objective function:

$$\underset{E}{Max}\ \Pi = \sum_{i,m} p_i(1 - cw_m - mkt_m)C_{i,m} - \sum_m (fuel_m + o_m)E_m - \sum_m (f_m + v_m)V_m \tag{14}$$

where Π is total fishery profits, p_i is the price of species *i* (assumed constant, and includes a value for "other" species), cw_m is the crew share of revenue paid by vessels operating in metier *m*, mkt_m is the marketing cost as a share of revenue paid by vessels operating in metier *m*, $fuel_m$ are the fuel cost per shot by a vessels operating in metier *m*, o_m are other running costs per day by vessels operating in metier *m*, f_m is the annual fixed costs associated with a boat operating in metier *m*, v_m is the user cost of capital (here defined as total vessel capital times a depreciation rate of 2.6% plus an opportunity cost of capital of 5%), and V_m is the number of vessels operating in metier *m* (estimated from the level of fishing effort E_m and the average number of shots per vessel). As vessels may operate in more than one metier, fractional vessel numbers are permitted in the model (e.g. a trawl boat may operate in potentially six different metiers, although in practice this is not likely to be the case).

The model determines the level of effort in each metier that maximises total fishery profits (E_{MEY}) across all species. The catch of each species at this level of effort is effectively the maximum economic yield of that species.

The exploitation rate associated with each species at MEY (f_{MEY}) is given by

$$f_{MEY_{i,m}} = q_i E_m \tag{15}$$

at the metier level ($f_{MEYi,m}$) and

$$f_{MEY_i} = \sum_m q_{i,m} E_m = \sum_m f_{MEY_i,m} \tag{16}$$

at the species level, where E_m is the optimal level of fishing effort in metier *m*. The full matrix of catchability coefficients is given in the Appendix A.

Simulation model (recursive optimisation model)

The "simulation" model was developed as a recursive optimisation model. That is, fishing effort was allocated between metiers in order to maximise the profits in that year given the level of available quota for each species, prices and costs. The stock biomasses in the following year were derived from the difference between the surplus production and the catch taken in that year, which in turn determined the following year's TACs (and so on). It is simulation in that the TACs were set based on a given harvest control rule (described in later parts of the report), and that the optimisation part of the model was only to allocate fishing effort between the different metiers based on the incentives and constraints faced (reflecting fisher behaviour).

For a given metier *m*, the catch of species *i* during time *t* ($C_{i,m,t}$) in the simulation model is given by:

$$C_{i,m,t} = q_{i,m} E_{m,t} B_{i,t} \tag{17}$$

where $q_{i,m}$ is the catchability coefficient relating to species *i* in metier *m* , $E_{m,t}$ is the level of effort expended in each metier *m* during year *t* and $B_{i,t}$ is the level of biomass of species *i* in time *t*. Catch of "other" species are considered in the same way as in the equilibrium model (i.e. Equation 13). In the simulation model, effort in each metier is constrained to change no more than 20 per cent from the previous level (i.e. $0.8E_{m,t-1} \le E_{m,t} \le 1.2E_{m,t-1}$ with the level in the initial year based on the observed level of effort in the base year of the model. The aim of this was to reflect a degree of inertia in the fishery and to prevent large an unrealistic changes in effort allocation year to year. Studies of fleet behaviour suggest that habits are a major factor influencing effort allocation even in the face of changes in economic incentives (Holland and Sutinen 2000; Hutton *et al.* 2004; Pascoe *et al.* 2013b).

A TAC is set for each species at the level of catch associated with f_{MEY} derived from the equilibrium optimisation model, such that the TAC for species *i* in year *t* ($TAC_{i,t}$) is given by:

$$TAC_{i,t} = f_{MEY_i} B_{i,t} \tag{18}$$

The metier level TAC for species *i* in year *t* ($TAC_{i,m,t}$) is correspondingly given by:

$$TAC_{i,m,t} = f_{MEY_{i,m}} B_{i,t} \tag{19}$$

Fishing mortality targets at the metier level are updated in the simulations allowing perfect quota transferability, such that $f_{MEY_{i,m}} = f_{MEY_i} * C_{i,m,t-1} \Big/ \sum_m C_{i,m,t-1}$, with allocation in the base year based on the observed catches in each metier. It is "perfect" in the sense that it is assumed that those fishers (metiers) which needed additional quota in the previous year were able to obtain this quota and retain the quota for start of the next year. For simulations of imperfect quota transferability, allocations to each metier are fixed (so imperfect quota transferability allows for quota to transfer between fishers in a metier, but not between metiers). These reflect possible extreme ends of the transferability spectrum, with true transferability most likely somewhere in between. To some extent, the imperfect quota scenarios represent the "worst case" outcomes.

The model allows for discards and quota undercatch to occur when the TAC does not perfectly align with the catch composition. Discard of each species *i* in each metier *m* during year *t* ($D_{i,m,t}$) is the difference between the pre-defined TAC and catch, while undercatch is the difference between catch and TAC where the former is less than the latter. These are given by

$$D_{i,m,t} = \max\left[0, (C_{i,m,t} - TAC_{i,m,t})\right] \tag{20a}$$

$$U_{i,m,t} = \max\left[0, (TAC_{i,m,t} - C_{i,m,t})\right] \tag{20b}$$

From equation 20a, discards are positive if catch exceeds the TAC, otherwise they take the value of zero. Discards of "other" species are given a value of zero as no explicit TAC is set in the model. Conversely, from equation 20b, under-catch is positive when the TAC exceeds the catch.

Revenue is estimated initially at the metier level, before aggregation to total fishery level. This is because some costs are related to revenue, with this relationship varying by metier (which in term is determined by the gear used in that metier). Metier level revenue ($R_{m,t}$) in time *t* is given by:

$$R_{m,t} = \sum_i p_i (C_{i,m,t} - D_{i,m,t}) \tag{21}$$

where p_i is the price of species *i* (including "others"), assumed to be constant in real terms over the duration of the simulation. Total fishery revenue ($R_{m,t}$) is given by $R_t = \sum_m R_{m,t}$.

Fishing cost are assumed to include both fixed and variable costs. If we assume a constant average number of shots per vessel, and that changes in effort levels represent changes in boat numbers, then all costs can be considered variable (i.e. a function of total fishing effort) for simplicity. Some costs, such as crew and marketing costs vary based on revenue, while other costs vary with fishing effort directly. We estimate fishing costs in each time period for each metier ($CST_{m,t}$) by

$$CST_{m,t} = \sum_i (cw_m + mkt_m) p_i C_{i,m,t} + (fuel_m + o_m) E_{m,t} + (f_m + v_m) V_m \tag{22}$$

where the parameter values are those previously defined in equation 14. The number of vessels in each metier are derived from the level of fishing effort (e.g. shots) divided by the average shots per boat. This effectively implies fractions of boats within each metier, but as vessels can operate across more than one metier (determined by its main gear), and as costs are gear-specific, this does not create an unrealistic distortions in the analysis.

Fishery profit at the metier level in time *t* ($\Pi_{m,t}$) is given by

$$\pi_t = R_t - CST_t \tag{23}$$

As noted previously, the model is used to estimate the effort allocation within each year that maximises the profits within that year based on the prevailing biological (stock) and economic conditions. Fishing effort levels within each metier are constrained to only vary within ±20% of the previous year's value of *Emey* (lower and upper bounds) given the level of pre-defined TAC (based on *Fmey*) and simulated biomass. This is to avoid unrealistic changes in effort from year to year that would imply substantial changes in fleet composition.

The stock dynamics is represented by the dynamic form of the Fox (1970) model, given by:

$$B_{i,t+1} = B_{i,t} + B_{i,t} r_i \ln\left(\frac{K_i}{B_{i,t}}\right) - \sum_m C_{i,m,t} \tag{24}$$

where r_i is the instantaneous growth rate and K_i is the carrying capacity of species *i*. The biomass in the first year ($B_{i,0}$) is the estimated biomass in 2015.

Finding the optimum – use of genetic algorithms

Both the long-run optimisation model and the short-run effort allocation models (given existing biomass and catch limitations) are highly non-linear. Traditional optimisation approaches often rely on steepest slope search and/or differentiation approaches. As such, they are subject to problems of local optima and also non-optimal outcomes using traditional non-linear programming techniques. For example, in Figure 3, depending on where the solution process started, the traditional non-linear optimisation procedure may identify any one of five points as optimal, even though one is infeasible (i.e. not actually an optimal, but the best outcome the search procedure could find), and three of the other produce an outcome that is less than the true optimum outcome.

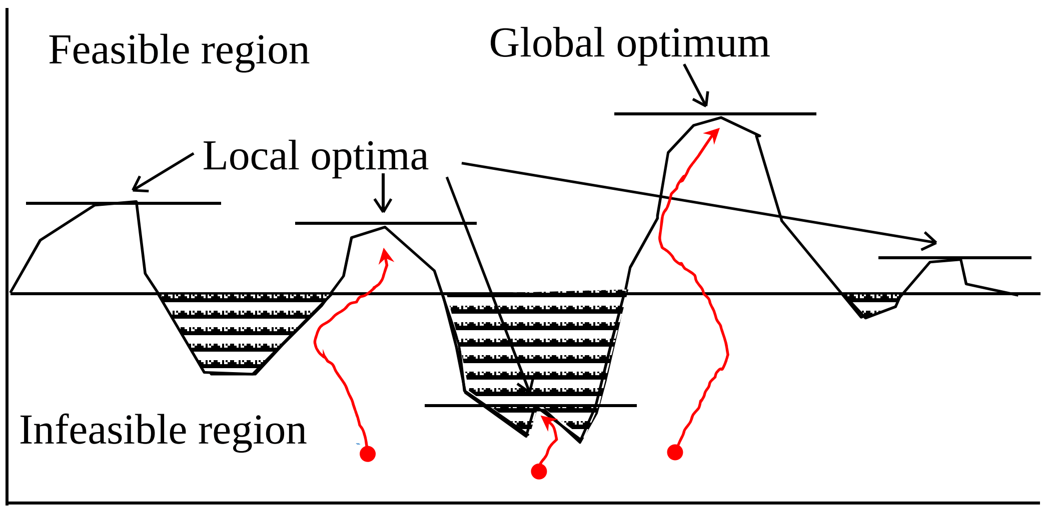


Figure 3. Problems of local optima and infeasible solutions in non-linear optimisation problems

Initial analyses found these problems were common in most model runs, resulting in unreliable results. As a result, evolutionary optimisation approaches were used. These involve the consideration of hundreds of potential input combinations, from which an outcome is estimated for each. The best half of the combinations are maintained, the bottom half are discarded. The remaining "organisms" cross breed to produce new combinations, with some level of mutations also introduced to bring in additional combinations not in the current population. This process continues for a predefined number of generations, with each successive generation gradually converging towards the optimal combination that results in the global optima. Unlike traditional approaches, however, a terminal condition is not imposed as convergence is not formally achieved. However, lack of improvement in the outcome after multiple generations is an indicator that the global maximum has been reached (and that no further improvement in outcome is likely). Initial analyses suggested that a population of 200 organisms run over 200 generations would achieve a close to global optimum. An example of the evolution of the solution is given in Appendix B. While these approaches are useful in overcoming problems experienced with traditional non-linear optimisation approaches, they are relatively slow at reaching the optimum. However, the positive side is that the final outcome is more likely to be a global optimum than would be derived using traditional optimisation methods under these circumstances. These approaches have been used previously in fisheries bioeconomic modelling where a large number of non-linearities exist (Mardle and Pascoe 2000; Mardle *et al.* 2000b).

The simulations were conducted over a time frame of 20 years. A common criticism of optimisation models is that they estimate what should happen rather than what would happen. To limit this, the model was run as a series of short-run optimisations (each year separately), linked by the stock dynamics in equation 24, rather than as a single optimisation over the full 20 years. In this case, the virtual fishers seek to optimise their short term (annual) profits based on the underlying conditions in that time period, making it more operationally realistic. As a consequence, the analysis is part optimisation and part simulation.

Scenarios

A range of scenarios was explored using the model. These are provided in more detail in the results section. Two quota transferability assumptions were considered: the best case scenario maximises total fishery profits assuming the "perfect" transferability of quota (within and across metiers). The second scenario maximises the total fishery profits assuming imperfect transfer of quota, that is quota transfer is permitted only within a metier but not across metiers (potentially a worst case scenario). This was simulated by having a total TAC for each species in the first instance (assuming that reallocation between metiers can take place), and a metier-specific TAC for each species (equation 19) in the latter case (assuming that quota cannot move between metiers). This has an impact on the level of discarding.

In addition, parameter uncertainties were introduced through adding random error to the TAC setting process (i.e. assuming errors in the f_{MEY} estimates e.g. $TAC_{i,t} = f_{MEY_i} B_{i,t} \varepsilon_{i,t}$ where $\varepsilon_{i,t}$ is normally distributed $N[1,0.1]$), and stochasticity in biomass levels between years (i.e. a multiplicative error term applied to the $B_{i,t+1}$ estimate to reflect influences of environmental fluctuations, given by $B_{i,t+1} = (B_{i,t} + B_{i,t} r_i \ln(K_i / B_{i,t}) - \sum_m C_{i,m,t}) \varepsilon_{i,t}$).

The effects of limiting quotas to a subset of species only was examined by setting TACs for non-key species to an arbitrary high level (so effectively not binding).

Model parameters

The species-related parameters currently employed in the model are given in Table 5. The key biological parameters (growth (*r*) and carrying capacity (*K*)) were derived from logbook data covering the period 1913-2009 for some species and from 1985-2009 for all species (Tuck 2010). Biomass estimates and catchability coefficients are based on data for 2015. Values for growth (*r*) and carrying capacity (*K*) were previously derived by fitting the Fox (1970) model to annual total biomass and estimated total catch from the most recent available SESSF Tier 1 stock assessments. Where the Fox model did not provide a value for *K* consistent with the initial biomass estimated by the stock assessment, the assessed initial biomass value was used as a fixed value for *K*. For species without available stock estimates, the model was fitted to the available standardised fishery CPUE series. Thus, the time series of biomass produced by the Fox model are largely consistent with the biomass from available stock assessments, or the commercial CPUE and catch for species with no stock assessment.

For some species, notably Gemfish, the parameters used in the model are not directly consistent with stock assessment information. The most recent stock assessment suggest that Eastern Gemfish stocks are currently at around 8.5% of their unfished level (Helidoniotis *et al.* 2017), substantially lower than the values used in the model analysis. At such a low level, the Fox surplus production model would estimate a rapid increase in biomass. However, in reality, the stocks have failed to increase despite low catch limits. Despite some evidence of a regime shift in the stock (and a potentially lower carrying capacity than the very original unfished biomass estimate), the management reference points have not changed (Klaer *et al.* 2015). As our model is illustrative to a large degree, and based on relatively simple stock dynamics, some of the model parameters appear to deviate from their "true" values in order to better replicate changes in the fishery.

Prices were derived from ABARES fisheries statistic and are specific to the SESSF (ABARES 2015).

Table 5. Species level parameters used in the analysis (*r*, *K* and *p*)

Species	**Species Code**	**Growth** **r**	**Carrying capacity** **K**	**Biomass (t)** **B2015**	**Price** **p**
Blue Warehou	TRT	0.069	16086	3303	3.11
Flathead	FLT	0.153	44566	22419	6.34
Gemfish	GEM	0.208	40000	19539	2.93
John Dory	DOJ	0.044	5431	1810	7.21
Ling_East	LIG_E	0.215	11960	7306	6.33
Ling_West	LIG_W	0.262	15566	13299	6.33
Mirror Dory	DOM	0.614	13389	12849	2.64
Morwong_East	MOW_E	0.128	30231	14275	3.95
Morwong_West	MOW_W	0.151	4447	3880	3.95
Ocean Perch	REG	0.311	4657	4014	4.08
Ribaldo	RBD	0.288	1077	493	2.47
Silver Trevally	TRE	0.197	10654	5919	6.5
Silver Warehou	TRS	0.204	38577	28146	2.2
Whiting	WHS	0.42	13586	11848	3.61
Blue Grenadier	GRE	0.181	118926	84326	3.87
Gummy Shark	SHG	0.385	17369	13148	6.91
School Shark	SHS	0.076	36074	6943	5.53
Blue-Eye Trevalla	TBE	0.109	9319	1663	9.27
Other species					2.75

Vessel level cost parameters used in the model are given in Table 7. Cost parameters were derived from the ABARES fisheries survey report for the year 2013-14 (Skirtun and Green 2015).[3] Separate costs were identified for the Danish seine and trawl fleets, but these did not distinguish between fishing area. Fuel and other vessel costs were estimated on a per-shot basis. Vessel costs included fixed costs on the basis that all costs were variable in the longer term. This differs to a degree from the way in which costs are treated in some other Australian bio-economic models e.g. Punt et al. (2011). However, it allows for the effects of varying fleet size without the need to explicitly separate shots per boat and number of boats and is a practical approach applied to other bio-economic models in particular circumstances.

Table 6. Metier level cost parameters used in analysis

Metier	Fuel cost ($'000/x)	Vessel cost (other) ($'000/x)	Crew share (% rev)	Freight and marketing (% rev)	Fixed costs (per boat-annual) ($'000s)	Capital Costs (per boat annual) ($'000s)
Shelf trawl NSW	0.836	0.836	0.229	0.155	134.80	456.53
Offshore trawl NSW	0.836	0.836	0.229	0.155	134.80	456.53
Shelf trawl EBS	0.836	0.836	0.229	0.155	134.80	456.53
Offshore trawl EBS	0.836	0.836	0.229	0.155	134.80	456.53
Tasmanian Trawl	0.836	0.836	0.229	0.155	134.80	456.53
Danish Seine Bass Strait	0.087	0.087	0.403	0.183	80.33	276.0
Danish Seine EBS	0.087	0.087	0.403	0.183	80.33	276.0
Blue grenadier trawl	0.836	0.836	0.229	0.155	134.80	456.53
Gillnet EBS	0.356	0.353	0.428	0.039	110.62	319.94
Gillnet WBS	0.356	0.353	0.428	0.039	110.62	319.94
Hook and trap	0.356	0.353	0.428	0.039	110.62	319.94

[3] Additional cost information was also provide by ABARES splitting the trawl component into otter trawl and Danish seiners.

Results

What is multispecies MEY?

Much of the earlier fisheries literature focused on the generation of resource rent (Andersen 1983), the theoretical return to the resource after returns to all other inputs (labour and capital) had been accounted for. Economic rent is often assumed equivalent to the commercial fisheries economic profits, determined by the revenue earned from fishing less the costs of fishing. This revenue reflects the quantity of fish caught across the fishery multiplied by the average market price received for that catch. Similarly, the costs of fishing reflect the quantities of inputs (e.g. labour, fuel and capital) used in the fishery multiplied by the market prices paid to use or employ those inputs. Costs of fishing also include non-cash costs such as depreciation of capital, as well as opportunity cost of capital and labour. The opportunity cost is the foregone earnings that could have been realised if an input such as capital was put to its next best alternative use.

In a perfectly homogeneous fishery (in terms of vessel characteristics, efficiency and cost structure), fishery economic profits would be equivalent to resource rent, as this is the returns from the fishery once the full cost of labour, capital and management had been taken into account. However, in fisheries with more heterogeneity in the fleet, some of this profit represents other "rents", such as the return to (business rather than fishery) management, skipper skill or other individual vessel characteristics (e.g. their home port may be closer to the main fishery than other vessels, such that a location rent is applicable). Separating this intra-marginal rent from the measure of net economic returns (or fishery profits) to estimate resource rent is complex (Coglan and Pascoe 1999), and is usually not undertaken. Hence, the key measures of interest to fisheries managers and industry tend to focus on total fishery economic profits.

Net economic returns also takes into account other costs usually not incurred by the fishers themselves. For example, the cost of fisheries management. In some fisheries, some or all of these costs may be included in the measure of fisher cost if cost recovery is undertaken, while in others management costs may be fully borne by the general public.

The net economic return has a linkage to community benefit, i.e. through markets. Market prices received by the fishery for its output (if supplied to the domestic market) and paid by the fishery for its inputs reflect their value to the community. For example, the market price for fish is determined by the willingness of consumers to pay, in conjunction with the market supply of fish. A consumer's willingness to pay for a good reflects the benefit they expect to derive from consuming it, and they will consume it if their willingness to pay is equal to or higher than the market price. Community benefit that accrues to consumers, known as consumer surplus, is represented by the difference between their willingness to pay and the market price. Similarly, the market price paid for an input is determined by the current supply of that input and its current demand, the latter being a function of the productive benefits that can be generated from that input for the community.

The net economic return provides an indication of the level of inputs that should be devoted to fishing. For example, if net economic returns becomes negative, it indicates that fewer inputs should be devoted to producing catch in the fishery. Furthermore, it suggests that those inputs have a more beneficial (or productive) use for the community if employed in another sector of the economy. The link between net economic return and community benefit is, however, not always clear-cut. For example, net economic returns in export-focused fisheries partially reflect benefits that accrue to non-domestic consumers. Additionally, net economic returns typically do not capture costs associated with the environmental impacts of fishing. Such impacts, therefore, require the use of other indicators to measure overall fishery management performance. While such issues exist, net economic return still provides an accessible and easily understood indicator of community benefit (Vieira and Pascoe 2013a).

Maximum economic yield as a management target

A common objective in fishery management, both internationally and in Australia, has been to maximise the sustainable catch of a fishery, or maximum sustainable yield (MSY). While this target maximises the gross value of production for a fishery, it does not ensure that the fishery is maximising economic returns. If the management objective is to maximise the economic benefit from the fishery, the optimum rate of exploitation is defined by the maximum economic yield (MEY), a point associated with a conjointly occurring level of sustainable catch, fishing effort and stock biomass (Figure 4). The concept was introduced by Warming (1911) (subsequently translated by Andersen (1983)), Gordon (1954), and Scott (1955). Clark (1973) and Clark and Munro (1975) made significant contributions to developing the concept further, by specifying steady state expressions for dynamic formulations of the problem.

For profits to be maximised it must also be the case that the fishery applies a level of boat capital and other resources in combinations that minimise the costs of harvest at the MEY catch level. The fishery, in other words, cannot be overcapitalized and vessels must use the right combinations of such inputs as gear, engine power, fuel, hull size, and crew to minimise the cost of a given harvest (Kompas 2005). Hence, 'fishery-level efficiency' being achieved by managing a fishery at MEY, although achieving full economic efficiency also requires 'vessel-level efficiency' (vessels harvest in a profit maximising manner) and 'management efficiency' (required management services are provided at least cost) (Kompas *et al.* 2011).

Achieving MEY involves a trade-off between higher revenues (through higher catches) and lower harvesting costs (through lower effort and more abundant stocks, which allows fish to be caught more easily, reducing the unit cost of capture). The latter 'stock effect' is the fundamental reason that MEY is associated with a more conservative (higher) level of biomass relative to MSY (Grafton *et al.* 2007a; b). In being more conservative, MEY is also advantageous in that it ensures that stocks will be more resilient to negative environmental shocks. Similarly, higher profitability at MEY means that industry will be more resilient to negative changes in economic conditions.

The static equilibrium MEY concept is presented in Figure 4, which illustrates a simple surplus-production model for a fishery as a whole based on the traditional concepts of Gordon (1954) and Schaefer (1954). On the right hand side of the figure, the vertical axis is dollar amounts of revenue and cost and the horizontal measures fishing effort (e.g. days fished). The total revenue curve is derived from a biological surplus production function, and depicts the relationship between effort and equilibrium levels of catch at that effort level in monetary amounts. Every point along this curve (sustainable yield-effort curve) represents an effort and yield combination that is sustainable. The total cost curve is taken as the total cost of fishing. This is traditionally assumed to be increasing with effort, and is usually represented as a linear relationship in most models (although non-linear variants exist).

On the left hand side of Figure 4, the relationship between sustainable yield and biomass is illustrated, with biomass increasing from right to left, from zero to the biological carrying capacity (K). When the biomass is at B_{MSY}, the catch that can be sustainable taken is maximised (i.e. MSY or maximum sustainable yield).

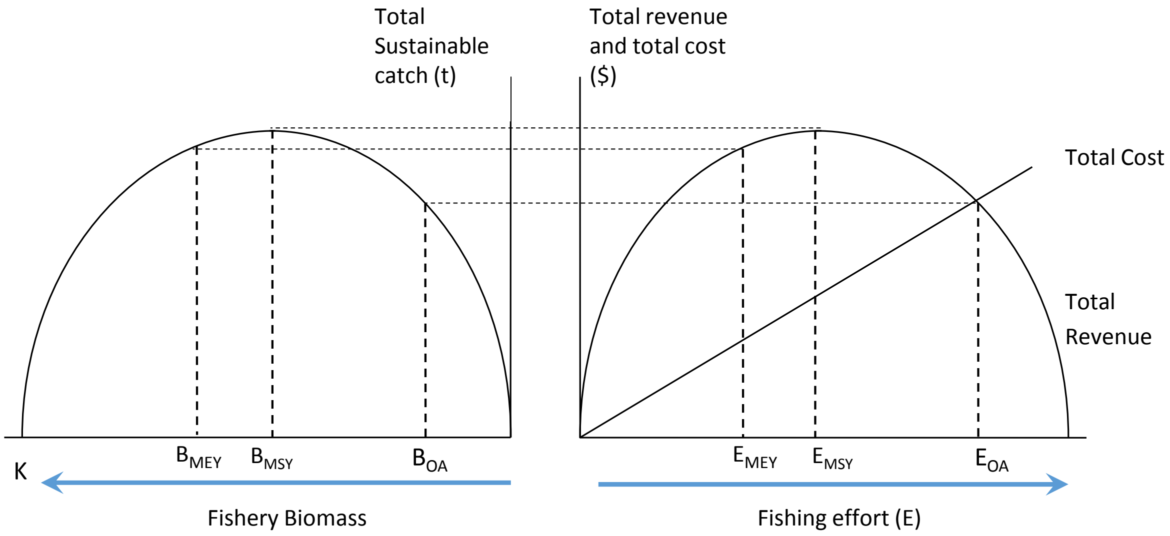


Figure 4. Traditional bioeconomic model illustrating concepts of maximum sustainable yield and maximum economic yield

MEY occurs at the effort level E_{MEY} and corresponding value of effort that creates the largest difference between the total revenue and total cost of fishing, thus maximising profits. In the absence of any management controls or fully delineated property rights, and with positive harvesting costs, the bionomic equilibrium (the point at which total revenue equals total cost) of fishing effort, E_{OA}, exceeds both E_{MEY}, and the effort at maximum sustainable yield (E_{MSY}). In the static equilibrium model, with no discount rate, MEY occurs at effort levels less than effort at MSY and thus at stock levels that are larger than those associated with MSY and the bionomic equilibrium (i.e. $B_{MEY}>B_{MSY}>B_{OA}$). In general, and whenever the discount rate is ignored, if harvesting costs rise as population size declines, a profit maximising policy will automatically lead to biological conservation, with an equilibrium population in excess of the population corresponding to MSY, although there are some exceptions (e.g. Clark 1973; Clark *et al.* 2010a).

The figure also illustrates that targeting MSY or above generates smaller profits than targeting MEY (i.e. the difference between the revenue and cost curve decreases as effort moves beyond E_{MEY}). With a small increase in the cost of fishing, profits could easily go to zero — if so, this would replicate a common property or open access equilibrium even though a management regime was in place and operating (Kompas 2005).

While the static Gordon-Schaefer model provides insights into equilibrium conditions, it does not consider the "optimal path" to get to the profit maximising levels of biomass, nor does it consider the consequences of discounting (Grafton *et al.* 2004). Fisheries in the real world are dynamic systems in which individual fishers respond to changes in parameters such as the cost of fishing and price of fish. In the 1970s, economists began to question the adequacy of the static model. Capital theory was integrated into the economics of fisheries by the mid-1970s (Clark and Munro 1975; Clark 1976).

The setting of a fishery's harvest levels is equivalent to an investment decision about how many fish should be conserved to contribute to future stock size and catches (Clark and Munro 1975). The expected values of future revenues and costs from fishing need to be considered in this context by accounting for the fact that a dollar earned today will typically be valued more than a dollar earned in the future. This is because a dollar earned today is immediately available to generate further economic returns. This is particularly relevant for fisheries that are already depleted, and require rebuilding of the stock to achieve the static MEY level of output. In such cases, future gains in terms of discounted economic profits may not exceed the short term cost to achieve them, and an alternative dynamic optimal level of yield, effort and biomass may exist.

This dynamic treatment of MEY has implications for the optimal MEY as well as the path that should be taken to this optimal point, and is synonymous with maximising the flow of the present value of economic profits over time. It generally results in a higher level of catch and effort and a lower level of biomass than the static MEY levels. The divergence between the static and dynamic MEY levels depends on the discount rate used (Clark 1976), relative to the stock effect, or sensitivity of the cost of fishing to changes in stock level and lower the discount rate, the closer the two MEY points (i.e. static and dynamic estimate). At higher discount rates, the closer the dynamic MEY level of effort approaches that associated with MSY. Consequently, in most cases, $E_{MEY}<E_{DMEY}<E_{MSY}$, where E_{DMEY} is the level of effort at the dynamic MEY.[4] Kompas *et al.* (2011) provide more detail on these topics. The role of discount rate in fisheries management is generally considered small as it is often dominated by the stock effect on fishing costs in the determination of MEY (e.g. Grafton et al. 2007).

A key challenge in achieving MEY is determining the actual harvest target itself. MEY is more than just a catch target—it also relates to a stock size and level of fishing effort that enables the catch to be taken. The complexity of estimating MEY as an actual management target for a real fishery (rather than a conceptual or theoretical exercise) was highlighted by Dichmont *et al.* (2010). A review of issues relating to setting MEY in a range of different types of fisheries was undertaken by Vieira and Pascoe (2013a). Estimating MEY requires some form of a bioeconomic model, which in turn requires detailed information on the biology of the species, technical interactions between fishing gears and catches (especially in mixed fisheries), cost structures of the fishing fleet and market conditions. In many cases, information on one or more of the required model components is not available, with the implication that the utility of specifying an empirical-based bioeconomic model can be potentially low,

[4] There are some exceptions to this situation, as detailed in the sections below.

given the high cost of obtaining all the data and updating it regularly. If an investment has already been made in data collection, then the utility becomes greater.

Alternative interpretations of MEY

As MEY has gained increasing attention by policy makers as a management target, the issues of how we define MEY and what need be accounted for in estimating MEY has recently generated considerable debate in the fishery literature. The debate is generally associated with three themes: (1) whether fishery profitability should the main focus of management or whether other factors such as employment, processing and retail sectors need to be also taken into account; (2) whether the biomass at MEY (B_{MEY}) is always greater than the biomass that gives MSY (B_{MSY}); and (3) how MEY is defined and evaluated within a multi-species context.

Social considerations

In cases where social considerations, such as generation of employment opportunities and improvement of income distribution is important, modification of the MEY is required to account for those social aspects. Introduction of social considerations may limit the efficacy with which management measures are introduced, or it may justify a more intensive rate of fishing than is justified on purely economic grounds. Overall, opponents argue that management strategies based on maximising the net profit for the fishery alone, would lead to undesirable social consequences when potential opportunities for other relevant sectors are ignored. Christensen (2010) argued that MEY falls short of capturing the benefits for society as a whole, as MEY only accounts for the fish harvesting sector and excludes full value-chain impacts, and therefore, MSY is more appropriate target level.

Others have argued that the estimation of costs needs to take into account “social benefits” when undertaking any bioeconomic assessment, including estimation of MEY. For example, Wang and Wang (2012b) carried out a retrospective analysis of the buyback program in the Australian Northern Prawn Fishery aimed at moving the fishery to MEY and concluded that greater economic benefits ($17 million between 2006 and 2009) would have been achieved if the buyback had not taken place. The basis of this assumption was that crew payments, rather than being a cost of fishing, was in fact a benefit of the fishery, and should be treated as such. This effectively assumes that the opportunity cost of labour was zero. In this case, the argument by Wang and Wang (2012b) was refuted by Pascoe *et al.* (2013a) who used corrected economic parameters to re-estimate economic profits from the same fisheries and showed that the industry profits during the same period would have been $22-25 million lower if the buyback had not taken place.

Others have argued that MEY ignores the economic benefits generated in other sectors both upstream (e.g. industries supplying fishing) and downstream (e.g. retail industries selling fish product). Norman-Lopez and Pascoe’s (2011b) contributed to this debate by conducting an input-output analysis to estimate net economic effects of achieving MEY for several Australian fisheries and showed that although losses occur across sectors in the short term with a move to MEY, a net economic benefit to society results in the long term.

Grafton et al. (2012) further contributed to the debate by examining the trade-off between fisheries profit and employment in a case study on the Western and Central Pacific Tuna Fisheries. They showed that the long-term gains per job lost from pursuing a MEY target were worth several times the value of the average GDP per capita of Pacific Island countries. They also showed that including a processing and retail sectors along with the measures of consumer benefit lowers B_{MEY}, but in general, it is greater than B_{MSY}.

The conditions under which B_{MEY} may be close to or exceed B_{MSY} have been investigated by various authors (e.g. Clark 1973; Clark et al. 2010a; b; Grafton et al. 2010b; Sumaila and Hannesson 2010; Grafton et al. 2012; Squires and Vestergaard 2016) and remains open to further debate. The two key influencing factors are the growth rate of the fish stock and the discount rate that is applied to future revenues and costs—a slow growth rate and a high discount rate will move B_{MEY} closer to B_{MSY}, and, in some cases, potentially beyond B_{MSY}.

Non-market public benefits

Another important aspect of social benefits that is not typically captured in MEY is consideration of public goods that do not have market values. The non-market values include non-use values, reflecting existence values (e.g. people feel good about knowing a fish species exists), option values that reflect the values of keeping the option open of using the resource in the future (Weisbrod 1964), and bequest values that reflect the values that the current generation places on ensuring the availability of biodiversity or ecosystem functioning to future generations (Krutilla 1967), as well as non-consumptive use values (e.g. viewing wildlife). More detailed classification of the components of total economic value can be found in Pearce *et al.* (1989).

A review of social and economic valuation methods for fisheries have been undertaken by Vieira *et al.* (2009). Although there are a number of studies that quantify non-market values of marine resources, the studies that examine their implications for the calculation of MEY is limited. Very few studies have taken into account non-market values of fishery resources in bioeconomic models to maximise social welfare. Nevertheless, when the public benefits from leaving fish in the water (for biodiversity, ecosystem services) are explicitly incorporated into the dynamic model, B_{MEY} increases relative to B_{MSY} (e.g. Bulte *et al.* 1998; Kasperski and Wieland 2009; Kellner *et al.* 2011).

Whale species, for instance, are mixed goods involving elements of both private or consumptive good and a public good (Tisdell 1991; Kuronuma and Tisdell 1993) as they are commercially harvested, at the same time society value their continued existence for themselves and for future generations. Bulte *et al*. (1998) and Bulte and van Kooten (1999) estimated B_{MEY} for the Minke Whale stock in the Northeast Atlantic by incorporating non-use (existence) values of the whale in terms of annual household willingness to pay for the whale stock conservation in Europe, and concluded that the equilibrium stock size of a mixed good will be larger than the equilibrium stock of a pure private good. Incorporating the values of ecosystem service provided by oysters in terms of improving water quality, Kasperski and Wieland (2010) found that the optimal harvest rate in the Northern Chesapeake Bay oyster fishery was substantially lower than the optimal harvest rate without those benefits.

Other than target species, bycatch or/and prey species and marine habitats also provide non-market benefits, although those are overlooked in most existing bioeconomic models. A number of theoretical studies exist that had investigated the implications of non-market values of prey species on optimal harvesting of commercial (predators) species (e.g. Ragozin and Brown Jr 1985; Flaaten 1991; Ströbele and Wacker 1995; Flaaten and Stollery 1996; Flaaten 1998; Hoekstra and van den Bergh 2005), although empirical applications are limited. A review of bioeconomic models that consider habitat-fisheries interactions was provided by Foley *et al*. (2012).

Kellner *et al*. (2011) examined how non-market values of a predator (grouper for their contribution to tourism and biodiversity), and prey fish (parrotfish and snapper for ecosystem function) in the Caribbean may affect the optimal harvesting rates, and conclude that higher non-market values relative to market price can result in temporary or permanent fishing moratoriums. Recent work by Armstrong *et al*. (2015) provides an example of inclusion of non-market values for marine habitat in a fishery's MEY. They investigated how inclusion of non-use values for a habitat would affect optimal fisheries management, using the cold water corals (CWCs) habitat and the North-East Arctic Cod Fishery in Norwegian waters as a case study. The non-use value of CWCs was estimated in terms of the Norwegian population's willingness to pay for the protection of CWCs in addition to current measures. They found that the inclusion of a non-use value reduces the optimal level of the cod stock, but increases the optimal CWC stock, as one would expect. The effect of the non-use value was relatively small in magnitude when considering Norwegian households alone, increasing optimal coral habitat by just under 6%, while decreasing the optimal fish stock size by 2%.

There are various ways in which managers can take account of non-market values. Where the non-market value is associated with the target species, a stock larger than the MEY level could be maintained by reducing catch quotas below the MEY catch level. Where the non-market value is associated with a by-catch species, fishery managers can endeavour to protect these stocks by means of regulation of fishing gear and practices, such as the use of "environmentally friendly" scallop dredges, the use of turtle excluding devices in trawl nets, and devices to reduce the ability of seabirds to take longline bait (Campbell *et al.* 1997).

Multispecies fisheries and MEY

MEY is primarily a single-species concept. The mixed-species nature of most of the world's commercial fisheries further complicate issues of estimating MEY in practice. Within a single fishery, for instance, the same fishing gear may catch several species simultaneously. In other cases, different fisheries are spatially overlayed, using different gears and catching different combinations of the same sets of species. When such technical interactions occur, deriving estimates of MEY requires taking into account the impacts of one fishery on the other, as well as the effects of a given level of effort on the sustainable yields of all species caught (Anderson 1975). A result of this is that each species' biomass at fishery MEY will be differ to its individual B_{MEY} level if each was caught independent of the others (Duarte 1992; Vieira and Pascoe 2013b; Pascoe et al. 2015).

This is illustrated for a four-species fishery in Figure 5 (Vieira and Pascoe 2013b). The upper panel shows a fishery's revenue earned from four individual species and its total costs for different effort levels. The lower panel depicts total revenue (summed across the four species), total costs and total profit. For each effort level, each species will be associated with a given biomass level (with effort and biomass being inversely related). The level of fishing effort that maximises total sustainable fishery profits is around six units (shown by the dark green vertical line). At this level of effort, each species is associated with a given biomass that achieves fishery-wide MEY (denoted B_{FMEY}). For example, species 1 is fished beyond its MSY such that its $B_{FMEY} < B_{MSY}$ on a 'single species' basis, species 2 is close to its B_{MSY} (such that B_{FMEY} approx. = B_{MSY}), and B_{FMEY} for species 3 and 4 are below B_{MSY} and close to what may be considered their single species B_{MEY}. In this example, profits are also maximised at a level close to maximum sustainable revenue, although this is not always the case.

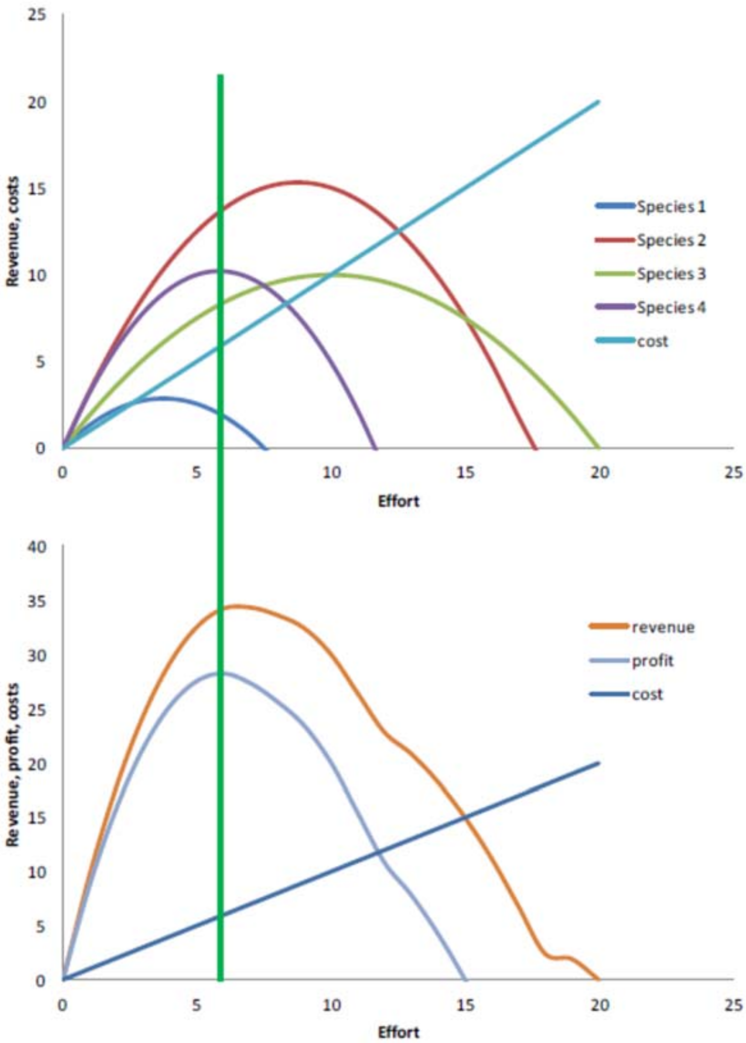


Figure 5. Conceptual multi-species equilibrium bioeconomic model

Deriving general analytical models to identify conditions for MEY in multispecies fisheries has been described as a formidable, if not impossible task (Silvert and Smith 1977; Chaudhuri 1986). Most attempts to estimate MEY in multispecies fisheries have been empirically based, using bioeconomic models to estimate MEY across the set of species in the catch (e.g. Placenti *et al.* 1992; Ward 1994; Sandberg *et al.* 1998; Holland and Maguire 2003).

In Europe, Holland and Maguire (2003) showed that managing fish stocks individually in a multispecies fishery where joint production cannot be avoided may lead to significant reductions in overall revenues from fishing, and increase in the variability of revenues. Da Rocha *et al* (2012) used a three-species model of the European Northern Hake Fishery, and identified a fishery-wide MEY for the fishery to be superior to single-species management, leading to higher discounted profits and larger long-term spawning stocks for all the species with the ratio of B_{MEY} to B_{MSY} ranging from 1.19 to 1.42. Guillen *et al*. (2013) considered a three species multi-gear demersal fishery in the Bay of Biscay, and found that the ratio of B_{MEY} to B_{MSY} ranged from 1.04 to 2.17.

In Australia, multispecies bioeconomic models have been developed for several fisheries and used to provide management advice and estimates of fishery-level MEY (e.g. Punt *et al.* 2002; Kompas and Che 2006; Kompas *et al.* 2009; Punt et al. 2011). Grafton *et al.* (2012) used a generic bioeconomic model that considered the producer surplus for the catching and processing sector, as well as consumer surplus, in a discussion of the benefits of MEY in fisheries such as the Western and Central Pacific tuna fisheries. Their model used hypothetical calibration data derived from a single-species modelling approach. In another study of this same fishery, Kompas *et al*. (2010) used a multispecies dynamic bioeconomic model with multiple fishing fleets to estimate MEY and the associated optimal allocation of fishing effort across fleets and species. Based on their results, the authors estimated that adopting a multispecies version of B_{MEY} as a target would result in increased fish stock sizes of the three main target species (up to a ratio of between 1.19 and 2.47 of the estimated single-species B_{MSY}), with gains exceeding US$3 billion. Wang *et al.* (2015) investigated optimal effort allocation in a multispecies fisheries, using the case study of the Moreton Bay Prawn Trawl Fishery in Queensland. They found that E_{MEY} decreases with increasing daily fishing cost, with the ratio of E_{MEY} to E_{MSY} ranging from 0.20 to 0.62 (0.37-0.43 when a daily cost of $750, which most closely represents the actual present fishing cost, was assumed).

The development of bioeconomic models requires considerable biological information on each individual species, which is often unavailable. In some data poor fisheries where only catch and effort data are available (plus some indicative economic variables), aggregated yield functions have been used. That is, total catch of all species is aggregated and modelled as a function of total effort. These have been deployed largely in developing countries (e.g. Lorenzen *et al.* 2006), but have also been used in more developed countries where fisheries are based on a large number of species, each contributing a relatively small proportion to revenue (e.g. Chae and Pascoe 2005; Jin *et al.* 2012). In other cases, attempts to estimate proxy economic target reference points have been made based on the characteristics of the species and fisheries (Zhou *et al.* 2012; Pascoe *et al.* 2014b; Pascoe et al. 2015).

Accounting for a large number of species and gear types in bioeconomic analysis has also been proven to be challenging. Vieira and Pascoe (2013a) note that modelling work in the SESSF has been less successful than Northern Prawn Fisheries (NPF) due to the large number of species in the fishery, and the number of different gears that catch these species in differing combinations. A substantial proportion of most quota species in the SESSF are caught as byproduct when targeting other species (Klaer and Smith 2012), and only a relatively small proportion of the key species have appropriate biological parameters available for bioeconomic analysis. This limits the usefulness of the model as a management tool, especially in relation to estimating target reference points (Vieira and Pascoe 2013a).

Biological (ecological) interactions

A fishery may also affect multiple species indirectly, through the biological interactions between the species directly impacted from fishing, and their predators, prey or competitors. These lead to similar challenges in determining MEY, although most models that account for biological interactions assume that the species can be separately targeted (e.g. Anderson 1975; Silvert and Smith 1977; May *et al.* 1979).

A number of theoretical bioeconomic analyses of prey-predator and competing species have been undertaken, including Silvert and Smith (1977), May et al. (1979), Hannesson (1983), Flaaten (1991), Agar and Sutinen (2004), and Singh and Weninger (2009). Applied studies on interacting species are undertaken on competing tuna species (Conrad and Adu-Asamoah 1986), on plankton feeders-fish-sea mammals interactions (Flaaten 1988), lobster-herring (Ryan *et al.* 2010) and on cod-herring-sprat interactions (Nieminen *et al.* 2012). The focus of the existing studies vary from identifying optimal exploitation rates to the issues of discarding for non-target species. A range of end-to-end whole of system models exist including a fleet bio-economic model component (e.g. Fulton *et al.* 2011a); however the large number of uncertainties involved with the trophic level interactions make these models more appropriate for strategic analysis rather than tactical target setting.

Multiple objective target yield

Fisheries management globally is characterised by multiple objectives. Studies of management objectives around the world have identified three core areas, namely economic, social and environmental sustainability (e.g. Mardle *et al.* 2002; Cheung and Sumaila 2008; Ward and Kelly 2009; Cowx and Van Anrooy 2010; Péreau *et al.* 2012). Others have considered additional objectives, namely political (Crutchfield 1973; Hilborn 2007), food security and income generation (Charles 1989). These latter objectives are often considered components of broader social objectives, although in other cases are considered separately.

In Australia, as elsewhere, there is increasing interest in developing approaches to assess management strategies taking into consideration a broad range of objectives. The Australian Commonwealth (i.e. Federal Government) were amongst the first fisheries jurisdictions to adopt maximising net economic returns as a primary management target as a means to capture both economic efficiency and ecological sustainability. Many Australian States have also identified social as well as economic objectives of management as important, while consideration of social objectives is also being undertaken at the Commonwealth level (Brooks *et al.* 2015). For some Australian fisheries, cultural fishing is an important component, while in many northern States encouraging participation by indigenous fishers is also seen as an important social objective (van Putten *et al.* 2013; Pascoe *et al.*

2014a). Studies in Australia have identified commonality in the broad objective sets, but differences in priorities among stakeholder groups, and jurisdictions (Jennings *et al.* 2016).

Concurrently, models and decision making processes are being developed to support fisheries management in a multi-objective framework. These range from qualitative approaches based on expert opinion (e.g. Dichmont *et al.* 2013b) to complex composite models (Dichmont *et al.* 2013a; Plagányi *et al.* 2013; Fulton *et al.* 2014). These models have been used to assess management strategies taking into account social, economic and ecological interactions rather than to estimate multi-objective target reference points. An overview of the multi-objective models developed and used in Australia is presented in Pascoe *et al.* (2017).

Elsewhere, multi-objective models have been used to estimate optimal fleet, effort and catch combinations in multispecies fisheries (e.g. Charles 1989; Mardle *et al.* 2000a; Pascoe and Mardle 2001; Mardle and Pascoe 2002). More recently, Rindorf *et al.* (2017) introduced the concept of 'pretty good yield' that identified combinations of fishing mortality in multispecies fisheries that are consistent with a range of objectives with some flexibility. These took into account economic, social, ecosystem and stock biological objectives. This concept is similar in some regards to viability analysis, in which a viable range of fishing mortality (or other target measures) is identified consistent with minimum acceptable level of each objective measure (Eisenack *et al.* 2006; Rapaport *et al.* 2006; Doyen *et al.* 2012; Gourguet *et al.* 2013; Sinclair 2014; Gourguet *et al.* 2015).

Implementation of MEY in a multispecies context

While the estimation of MEY has its own set of challenges, implementation of the policy in Commonwealth fisheries has also demonstrated further challenges associated with operationalising MEY as a management target. A synopsis of the general challenges that arise when implementing MEY is provided by Dichmont *et al.* (2010), who drew on experiences in the Northern Prawn Fisheries (NPF) and identified six key challenges:

- **Specifying the model**—Dichmont *et al*. (2010) note that modelling MEY is complicated by the many factors that affect it. They point out that a key factor that is often not well captured is fleet dynamics, particularly in terms of fleet responses to regulatory change and, for multispecies fisheries, targeting behaviour changes.
- **Defining the boundaries**—MEY optimises economic returns to the fishery and excludes sectors linked to the fishery such as the processing sector. If MEY is achieved over time with reductions in a fishery's catch, it also reduces economic activity in these downstream sectors. Although the result of such action is that resources previously consumed in fishing are freed up to be used more productively in other sectors, doing so can be politically difficult (Dichmont et al. 2010).
- **The best model outcome may not always be practical**—bioeconomic models can produce a result that is optimal in the "model world", but may be unacceptable in real life (Dichmont et al. 2010) as they may not capture factors relevant to the interests of industry or the community. This implies that careful design of the model is required, to include relevant constraints to account for these factors and/or careful interpretation of its outputs.

- **The need for accurate economic data**—economic parameters (such as output and input prices) will be a key determinant of MEY results but are highly variable and can lead to high levels of uncertainty regarding optimal harvest paths. Additionally, once economic cost data are obtained, costs need to be appropriately incorporated into the model. Decisions such as how to separate fixed and variable costs are not necessarily straightforward (Pascoe *et al.* 2014c). These data issues mean that regular revision of MEY results and management advice may be required.
- **A good target is not enough**—changes in things such as fisher behaviour, cost structure, stock biology and the regulatory environment will mean that the MEY path and target will need to be re-estimated regularly to allow optimal performance to be approximated.
- **Implementation in a co-management arena**— in this context Dichmont *et al*. 2010 point out the importance of using an adaptive management framework and 'that operationalising MEY is not simply a matter of estimating the numbers but requires strong industry commitment and involvement' (Dichmont *et al*. 2010 pg. 1), which requires a balanced combination of education and consultation.

MEY versus NER – including bycatch and consumers

The generic model was used to determine how the optimal level of fishing effort and fishery profit changes when the objective of management was changed from maximising fishery profits to maximising total economic returns, taking into account the impacts on consumers (through potentially higher prices) and also the impacts of environmental externalities (i.e. bycatch of iconic species). The objective function used in each case was given by Equation 3, where the dummy variables *DWL* and *icon* were set to zero if consumer benefits and bycatch were not considered in the optimisation, and given a value of 1 if they were.

The results from the simulations were compared with those of the base run to determine the effect of the assumptions in the scenario on the definition of "MEY". The base run involved the assumptions of zero price flexibilities (i.e. no price-quantity relationship) and hence zero consumer surplus, and a zero price associated with bycatch. Each set of simulations was run with the same set of "randomly" derived parameters (i.e. based on the same fixed seed value) so that differences in outcome were due to differences in assumptions (rather than differences in random parameter values).

Impacts on fishing effort

When prices vary with the level of catch (i.e. a price flexibility less than zero), optimal effort levels are less than those if prices were constant (Figure 6a). Cost saving from reduced fishing effort and the higher price with the lower catch results in higher fishery profits at lower levels of fishing effort. In contrast, considering consumer surplus results in higher levels of fishing effort than in the base case (Figure 6b). Higher catches result in lower prices for consumers, and the gain in consumer surplus offsets (to an extent) the reduced fishery profits.

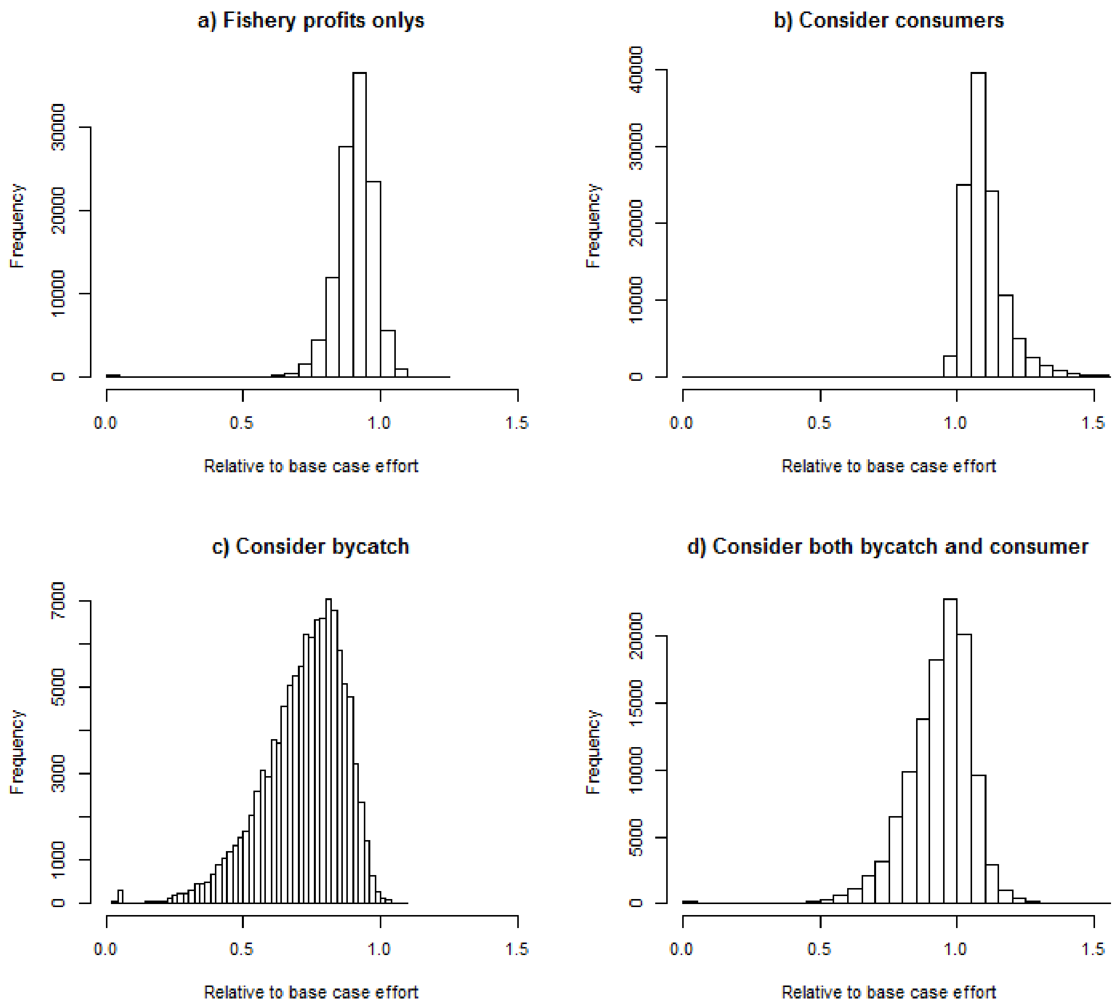


Figure 6. The relative effect on fishing effort for the four scenarios

Including the non-market cost of bycatch (Figure 6c) results in lower fishing effort than both the base case and also the case where profits are maximised with variable prices (Figure 6a). In this case, the higher cost per unit of effort results in a lower optimal fishing effort, which is compounded by the higher price received at lower levels of catch. Combining all components (consumers, bycatch and fishery profits with variable prices) may result in either higher or lower levels of fishing effort (Figure 6d) depending on which component dominates.

Impacts on fishery profits

Fishery profits with variable prices were consistently greater than the base run profits assuming a constant price (Figure 7a), although the increase was generally fairly small (less than 5%). When both consumer surplus and fishery profits were maximised, fishery profits tended to be lower than in the base run (Figure 7b). Including the explicit cost of bycatch into the profit function also resulted in a reduction in fleet profitability most of the time, but in some cases the higher prices at the lower catch level offset the additional costs, resulting in higher fleet profits (Figure 7c). Including both a cost of bycatch and consumer surplus into the profit function resulted in total fishery profits being within 5% of the base run level, suggesting such considerations largely offset each other (Figure 7d).

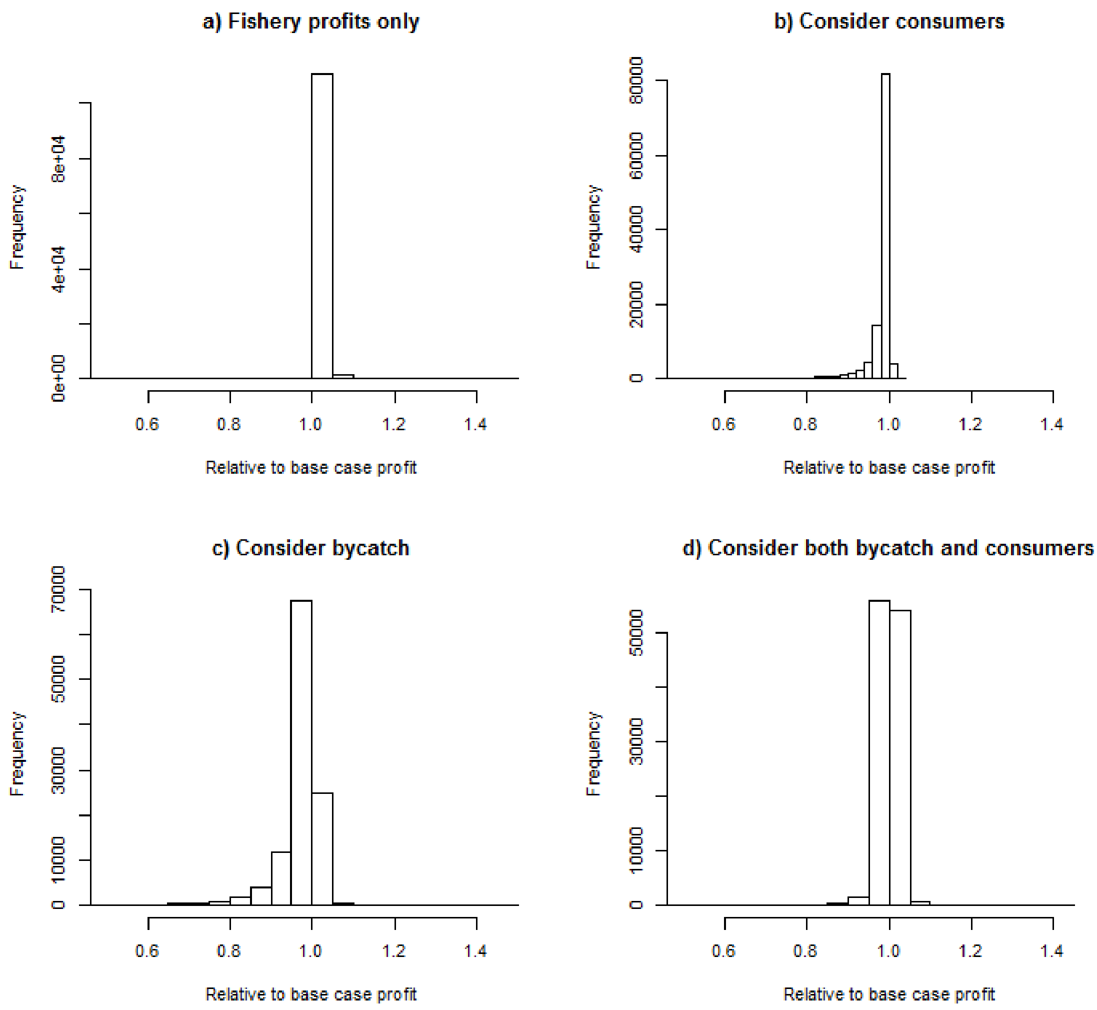


Figure 7. The relative effect on total fleet profit for the four scenarios

Transfers of producer and consumer surplus

The changes in fishery profit in Figure 7b and c largely reflect transfers of benefits between consumers and producers (fishers). With a price-quantity relationship, fishers have the potential to increase total profits by reducing supply, with the higher price more than offsetting the loss due to the reduced catch. As shown in Figure 8, this involves a transfer of part of the consumer surplus (the benefits to consumers) to producers (derived from Equations 4 and 5). This same result was seen in the model analysis, with the gain in fishery profits seen in Figure 8a also resulting in a net transfer of benefits from consumers to fishers. Conversely, if we include consumers into the objective function, then the optimal level of output results in some transfer of benefits from producers to consumers (Figure 8b).

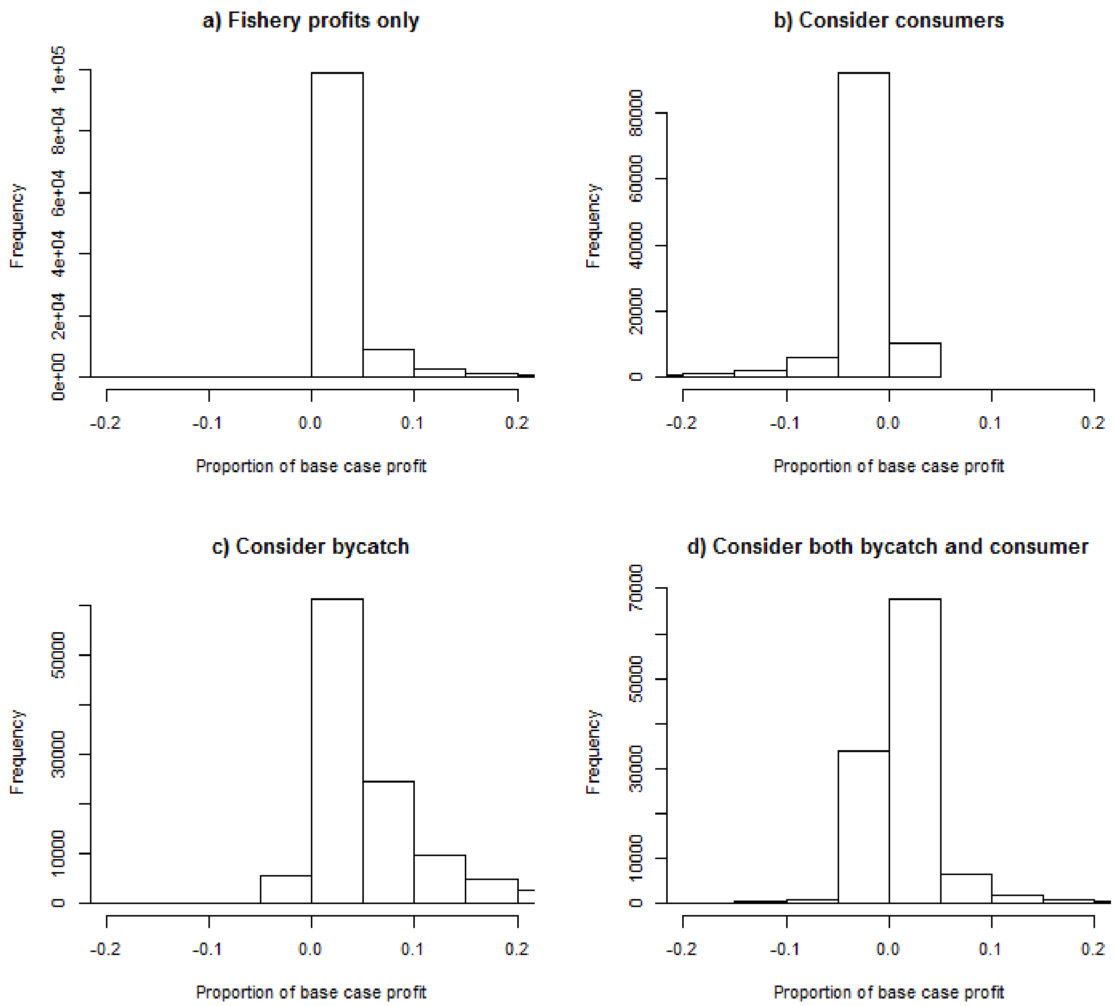


Figure 8. The relative effect on transfers from consumers to producers for the four scenarios

Imposing a bycatch penalty results in a decrease in catch and correspondingly higher prices if a price-quantity relationship exists. Maximising fishery profit under this scenario again results in a transfer from consumers to producers in most cases (Figure 8c). Considering both consumers and bycatch in the optimisation results in the potential for a transfer of benefits in either direction, depending on the specific combination of price flexibility and bycatch penalty (Figure 8d).

Target fishing mortality rates

The relationship between the optimal fishing mortality rates under the different scenarios can be seen in Figure 9. Bringing in additional considerations into the definition of MEY increases the divergence in optimal fishing mortality rates from the base case (f0). The target fishing mortality rates, however, seem to be most affected by the bycatch consideration (f3). This is moderated slightly when consumer surplus is also considered with bycatch in the objective function (f4).

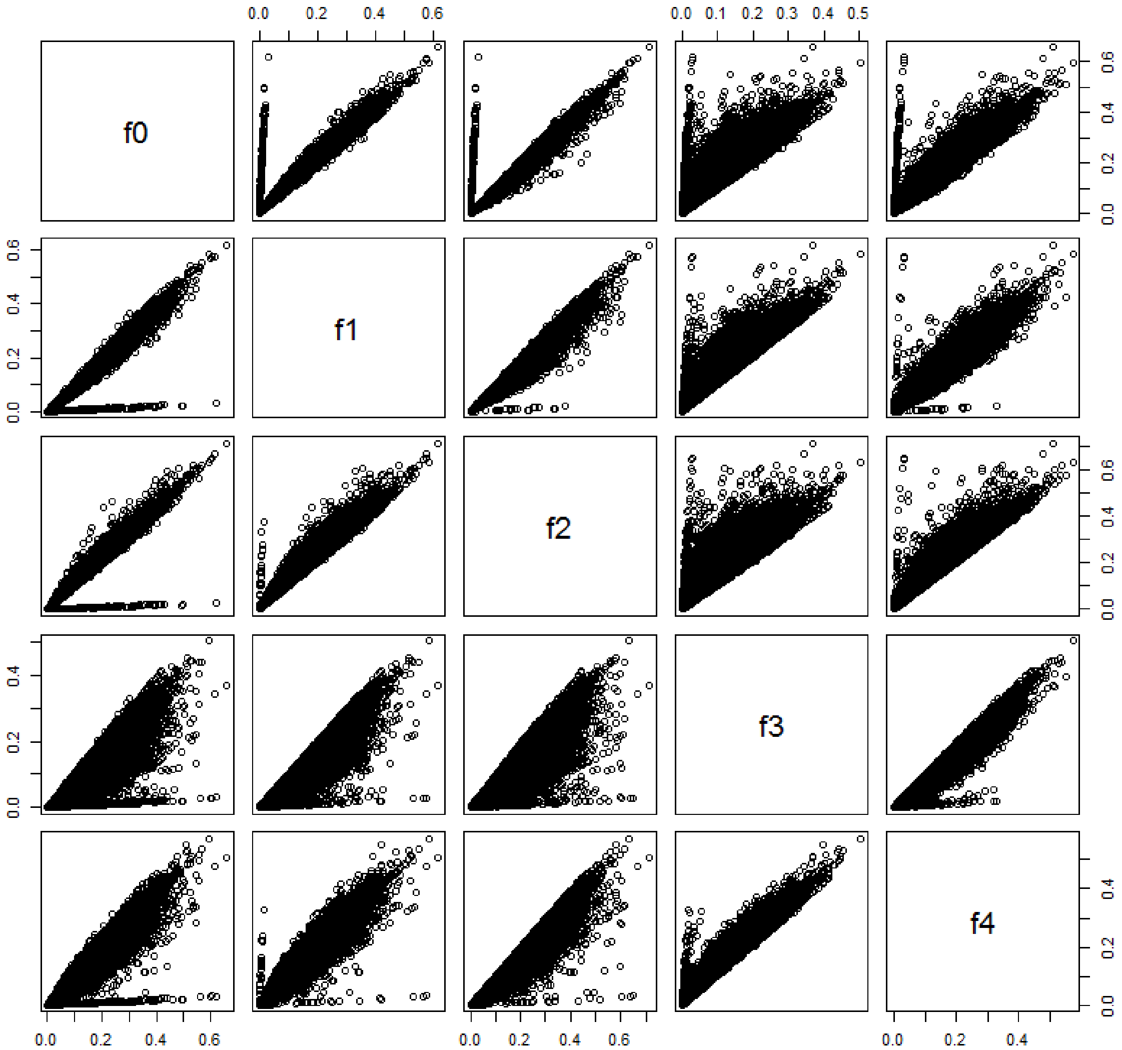


Figure 9. The correlation between optimal fishing mortality *(f)* levels for the base case and the four alternative scenarios derived from 10,000 simulations of each

Regressing the optimal fishing mortality with all components included (f4) against the different characteristics of the fishery and species within the fishery provides a potential predictive model for target reference points in multispecies fisheries (Table 7). While standard errors and t-statistics are presented also in Table 7, these are to some extent an artefact of the large number of derived data points. However, the relative size of each coefficient provides an indication of its importance in determining optimal fishing mortality.

As all the data in the model are in log form, the parameter estimates in Table 7 also represent elasticities. That is, the percentage chance in optimal fishing mortality due to a one percent change in the variable value. From this, optimal fishing mortality is relatively linearly related to catchability as the elasticity is close to 1. Optimal fishing mortality increases with the species-specific factors such as price, growth rate, and carrying capacity, decreases with its revenue share (i.e. share of the total fishing revenue) and the absolute value of the price flexibility[5]. Fishery level characteristics also affect the optimal individual species fishing mortality, increasing with the total number of species in the fishery, and decreasing with fishing cost, bycatch strike rate and bycatch penalty.

Table 7. Log-regression results of factors affecting the optimal fishing mortality

	Estimate	Std. Error	t value	Sig.
(Intercept)	1.145	0.024	48.5	***
log(p)	0.119	0.002	49.9	***
log(r)	0.146	0.002	61.2	***
log(q)	1.044	0.001	753.9	***
log(K)	0.129	0.002	58.9	***
log(c)	-0.112	0.002	-67.2	***
log(num)	0.256	0.003	93.6	***
log(revshare)	-0.096	0.002	-50.5	***
log(abs(flex))	-0.004	0.001	-3.8	***
log(strikerate)	-0.123	0.001	-84.1	***
log(ip)	-0.116	0.001	-85.7	***
Adj R2	0.93			
F	146600			***

Potential for different management instruments to achieve fishery-wide MEY in multispecies fisheries: a brief review

Achieving maximum economic yield (MEY) in fisheries requires two key aspects to be considered: (1) identifying MEY; and (2) a management instrument and harvest strategy that will achieve MEY. Identifying MEY in multispecies fisheries itself presents a number of challenges (Hoshino *et al.* 2017). Even though we are able to define MEY reasonably well, it may not necessary result in MEY being achieved if the management instrument is not appropriate. Here we discuss how multispecies fisheries may be moved closer to MEY when information is limited, particularly the availability of comprehensive and accurate bioeconomic models. A general consideration will be given to different governance systems and how they may help move a multispecies fishery towards MEY in the face of imperfect information and potentially incorrect targets.

Two common types of governance systems (hereafter management instruments) are input controls, and output controls. Input controls limit the level of effort and capacity invested into fishing activities, thereby limiting the overall harvesting power (ability to catch fish) of a fishery. Output controls limit the amount of catch that can be taken in a fishery for a given season. Input and output controls are not mutually exclusive and a combination of both can be used. In addition, there is another type of management instrument such as royalties or taxes, where direct control is not provided but rather provide fishers with financial incentives to modify their behaviour in attempt to indirectly reduce fishing mortality for certain species. In this section, we review advantages and disadvantages of each management instrument to achieve fishery-wide MEY in multispecies fisheries.

[5] Absolute values were required as price flexibilities are negative.

Input controls

Input controls are aimed at reducing fishing mortality of target species and limiting the external impact of fishing on by-catch species or broader ecosystems through restricting harvesting capacity in fisheries. Input controls have been applied to a wide range of fisheries globally and take various forms, including restrictions on: the number of boats and fishers allowed to operate in a fishery (i.e., limited entry or licence restrictions); the amount, size and type of gear allowed to use (i.e., gear restrictions); the size or power of vessels; the time allowed to fish through seasonal closures or days-at-sea restrictions; and areas where fishing is permitted. Input controls may be applied either at a fishery level by establishing a total allowable effort (TAE) or at an individual level, where a share of the TAE is allocated between individuals as a form of individual effort quota (IEQ). In an input controlled fishery, fishers are provided with individual transferable effort quota (ITEQ) where the transferability of IEQ is allowed so that quota can be bought, sold or leased among quota owners.

Setting a TAE requires a reliable estimate of fishing morality at the target level (e.g., MEY) for each species (F_{targ}) and a parameter known as the catchability coefficient for each unit of fishing effort (q). The level of fishing effort or TAE that is compatible with the target fishing mortality can be calculated as $E^* = F_{targ}/q$. In the context of multispecies multi-gear fisheries, E^* is an $k \times 1$ vector of fishing effort for k types of gear, F_{targ} is an $n \times 1$ vector of fishing mortality for n species, and q is an $n \times k$ vector of the catchability coefficient.

Advantages

In an attempt to control fishing mortality, input controls do not require a reliable estimate of exploitable fish biomass every year, as the level of fishing mortality is directly restrained as in equation (1). This means that the TAE remains constant irrespective of fluctuations in stock size while the catch realised from the constant effort varies with the stock fluctuation (Shepherd 2003). Squires *et al.* (2016) argue that effort controls are effective at managing fishing mortality when: there is clear and direct link between effort and fishing mortality with minimal uncertainty or stochastic variation in q; there is considerable unavailability or low quality of data that relatively affects estimation of biomass than q; and there are difficulties in conducting rapid and within-season stock assessments for short-lived species such as squid, prawn and small pelagic species. Therefore, in fisheries where effort controls are preferred management instruments, monitoring and enforcement costs tend to be less expensive than catch quota system as there is lower need to accurately record landings, to prevent under-reporting and misreporting (Shepherd 2003; Cotter 2010). The overall costs of monitoring and surveillance are an important consideration particularly for multispecies fisheries where close monitoring of all catches, including both target and by-catch species, can be prohibitively costly. In contrast, various effort measures, such as the number of vessels, fishing days and gear used, are easier to monitor.

Disadvantages

The implementation of input controls alone has been rejected by many studies as an effective means to manage fisheries. The problems associated with input controls have been well documented for many years (e.g. Wilen 1979; Wilen and Hannesson 1989; Townsend 1990; Dupont 1991). For example, in fisheries where fishers can adjust their input mix flexibly, input substitutions occur when restrictions on the use of one set of inputs provide an incentive for fishers to use other unrestricted inputs more intensely to circumvent the controls (Wilen 1979). Empirical evidence of such behaviour and the costs associated with it is widely available. Pascoe and Robinson (1998) show that fishers in the English Channel Beam Trawl Fishery changed their fishing location in response to restrictions on engine power, resulting a higher cost of production. Similarly, increased use of unrestricted inputs (i.e., labour and gear) occurred in the British Columbia Salmon Fishery where the number of vessels and the size of each vessel were restricted (Dupont, 1990). Kompas *et al.* (2004) also show that input substitution in the Australian Northern Prawn Fishery has resulted in lower technical efficiency and higher than optimal levels of fishing effort.

Furthermore, limiting the number of fishing units provide incentives for the remaining fishers to extend their fishing power to increase their share of the catch by replacing their boats with bigger, more powerful vessels, and equipped them with more advanced equipment for finding, catching, and handling fish (Pearse 1992). This phenomena referred to as "effort creep" or "technological creep" limit the ability of input controls to reduce fishing mortality as 'effective' effort can grow due to technological progress and investment even where the 'nominal' units of effort (e.g. days, number of pots) may remain constant (Shepherd, 2003). Dupont (1991) show that, in addition to input substitution, the existence of redundant capacity and inefficient vessels are the major factors contributing to the amount of rent dissipated in the input-controlled Salmon Fishery in British Columbia.

In multispecies fisheries, the issues above become more complex and difficult to be overcome. It is rarely possible or desirable to maintain a constant target fishing mortality rate for each individual species. Moreover, setting fishing effort to maximize sustainable revenues for overall fishery will almost certainly result in overfishing of some stocks in some years (Holland and Maguire 2003). Even when effort rights are assigned to individuals and the transferability of effort rights is ensured, input controls provide few incentive for individuals to adjust harvesting capacity autonomously, to avoid overfished species, and to achieve the optimal composition of species. The transferability of effort right allows more profitable fishers to catch more. However, this means that effective fishing effort or the productivity for each unit of effort is expected to increase as a result of the exchange of effort rights, resulting in increased catches and fishing mortality of more valuable but overfished species in the fishery.

Output controls

Output controls are intended to control fishing mortality by limiting the amount of catch that a fishery can take for a given period of time. In multispecies fisheries, output controls can be applied by setting a total allowable catch (TAC) for each individual species or group of species. Fishers may be provided with individual quota (IQ) and vessel catch limits where a share of the TAC is allocated between fishing units. If IQs can be bought, sold, and leased by quota owners, they are referred to as individual transferable quotas (ITQs). In contrast to the case of input controls, setting a TAC requires a reliable estimate of the current biomass (B) as well as the target level of fishing morality (F_{targ}). Given the estimate of biomass, the total allowable catch that is compatible with the target is calculated as $H^* = F_{targ} \times B$

Advantages

Since output controls are direct limits on catches, there is a clear link between the control and target catches. Output controls in the form of ITQ has a potential for autonomous adjustment of fleet size, as it provides incentives for inefficient fishers to exit the fishery (Boyce 1992; Grafton 1996). Similarly to the case of input controls, the transferability of individual harvesting rights allows a greater share of the total catch is taken by efficient fishers. However, unlike ITEQ, an introduction of ITQs do not create incentives for individuals to increase input use, increase the productivity for wasteful competition or to target overfished species (Peacey, 2002). Studies of ITQ systems in operation around the world show an increased value of landed product, improvements in cost efficiency, and a voluntary exit of less efficient vessels, all of which contributed to gains in economic efficiency in the fishery after the implementation of ITQ programmes (e.g. Grafton 1996; Arnason 1997; Dewees 1998; Kompas and Che 2005b). In addition to various economic gains, there is empirical evidence for the conservation benefits of ITQs, in terms of an increase in the abundance of target species (Costello et al. 2008, Branch, 2009, Chu, 2009) However, the literature on the impact of ITQs on broader ecosystems as well as non-target species is relatively underdeveloped.

Disadvantages

Because output controls intend to restrict the amount of catch to a safe proportion of the exploitable fish stock biomass, setting a TAC requires reasonably accurate estimates of exploitable stock biomass for each species and total catch of that species, including commercial and recreational catches, discards, and subsistence use. Estimating absolute size of exploitable fish stock biomass is not only difficult and expensive (Pope 2009), but also may result in TACs set at too high or too low if the biomass estimates are biased due to an inaccurate stock assessment. In such environment, an introduction of ITQs may not have the desired outcomes or may even lead to adverse consequences. For example, the ITQ-managed Tasmanian rock lobster fishery experienced increased incentives for race to fish during a period of non-binding TAC and excess capacity remained in the fishery more than 10 years after the introduction of the ITQ system (Emery et al. 2014, Rust et al., 2017). Moreover, Costello and Deacon (2007) show that efficiency gains from an ITQ system are limited when stocks are heterogeneous in terms of their density, location and unit value (as in multispecies fisheries).

Another main problem of catch-based controls, particularly in the form of ITQ is that they increase the risk of discarding and highgrading (discarding lower valued species or smaller size fish in favour of higher valued fish) (Copes 1986). The implementation of ITQ system encourage quota owners to fill their quotas with the most valuable fish and such behaviour may increase the monitoring and enforcement costs and may undermine the conservation benefits of the management system. Theoretical studies suggest that ITQ management increases incentives to discard and highgrade (e.g. Copes 1986; Boyd and Dewees 1992; Anderson 1994; Arnason 1994; Sampson 1994; Vestergaard 1996; Pascoe 1997; Turner 1997), although empirical evidence are mixed (Grafton 1996; Branch and Hilborn 2008). In attempt to combat issues of discards from multispecies fisheries under quota management, European Union adopted landing obligation (bans on discards) in 2015 where all catches of fish under quota management must be landed, although the effectiveness of the policy has been questioned (e.g. Condie *et al.* 2014; Borges 2015; Villasante *et al.* 2016). Similar attempts were made in Norway, New Zealand and US with mixed results (Pascoe 1997).

These problems become more pronounced in multispecies fisheries, where there are complex technical and biological interactions among species (Copes 1986; Squires *et al.* 1998). Issues of joint production is particularly a problem where some species may be overfished and others may be underfished. The limited degree to which individual species can be targeted has been confirmed in different multispecies fisheries around the world (Pascoe et al. 2007, Pascoe et al. 2012, Scheld and Anderson, 2017). For example, Scheld and Anderson (2017) use data from the New England multispecies groundfish fishery and show that it is difficult in substituting production across groundfish species. A limited ability to target specific species may constrain fishers' profitability if total harvests of target species are restricted to prevent the overexploitation of vulnerable bycatch species (Squires and Kirkley 1996). If quotas are imposed on a subset of species, fishing pressure on non-ITQ species can be increased, which in turn results in an extension of the ITQ program over additional species. Such proliferation of quota may increase the difficulty of administration and monitoring to the public sector, thereby undermining the potential economic gains associated with the use of ITQ systems (Dupont and Grafton 2000).

There are also important social impacts of ITQs to be weighed against their potential economic benefits. For example, fishing communities may lose out when quota is sold to "outsiders" (Eythórsson 2000), employment for crew members may decrease (Copes and Charles 2004), and the high price of quota will make it too expensive for young fishers to enter ITQ fisheries (Dewees 1998). An introduction of ITQ management may also result in concentration of fishing power towards a few large fishing companies or particular fishers (van Putten and Gardner 2010), raising potential social concerns, such as the development of monopoly power in the fishery, increased inequality in the distribution of fishery profits, and destruction of social norms and cultural heritage held by the fishing community (Davis, 1996, McCay 1995, Sumaila, 2010).

Price-based instruments: taxes, royalties and fees

Output and input controls are considered as a quantity-based instrument as they directly limit the amount of fish that can be taken for a given season or the level of effort that is allowed in the fishery. In contrast, fisheries regulation on the basis of price-based instruments, such as taxes, royalties and fee, are aimed at changing the economic incentives faced by fishers to land or discard their fish (Pascoe 1997). A tax has long been proposed as a solution to address issues of negative environmental externalities such as pollution (Pigou 1920; Baumol 1972). In fisheries, royalties are often used to support cost recovery schemes (Schranket al. 2003). The use of price-based mechanisms has also been proposed as an effective means to limit the catch of bycatch species, discarding and over-quota catch of target species (e.g. Anderson 1994; Pascoe 1997; Hutton *et al.* 2010b). For example, a bycatch tax system can be used to place explicit prices for bycatch, so that fishers are responsible for paying their own bycatch and this provides fishers with the incentives to avoid the bycatch. However, despite its potential benefits, a practical implementation of such a system to reduce the level of bycatch is limited.

Another instrument similar to taxes is a “deemed value payment” system under which fishers are charged a fee for the catch above their quota or allowed to surrender or discard catch they cannot match with quota (Sanchirico *et al.* 2006). Deemed value system provide fishers with the incentive to reduce discarding of over-quota catch by providing fisher incentive to land without providing incentives for further catching the fish (Pascoe 1997; Hutton et al. 2010b). The deemed value system is incorporated in the Quota Management System in New Zealand where the deemed value (i.e., penalty) is set above the lease price of quota and below the price of fish so that there is sufficient incentive to avoid the over-quota catch (Pascoe et al. 2010).

In multispecies fisheries, a deemed value system may offer potential benefits when TACs are set incorrectly (i.e. due to unexpected and random fluctuations in the stock). In such a case, a “hard” quota may result in one species effectively stopping the catch of other species, or it will be discarded if it is still economical to fish. A deemed value will provide some revenue from the over-quota catch without providing an incentive to continue fishing for the species specifically.

Advantage

The economic efficiency of landing taxes as an alternative to ITQ systems has been demonstrated by a number of theoretical studies (e.g. Weitzman 2002; Jensen and Vestergaard 2003; Hannesson and Kennedy 2005; Hansen *et al.* 2008). Despite a large body of theoretical studies, however, empirical investigations on the relative performance of price-based instruments against other management systems are limited. Hannesson (2007) also shows that when crew is paid a share of the catch value instead of a parametric wage, ITQ systems can lead to overinvestment, but such distortion can be corrected with a tax on fish landings. Moreover, Weitzman (2002) and Jensen and Vestergaard (2003) show that tax regulation performs better than ITQs when there is imperfect information about the stock-recruitment relationship. Hansen *et al.* (2008) show that taxes further become a preferred option over ITQs when the regulator is uncertain about the extent of noncompliance in the fishery. Conversely, Hannesson and Kennedy (2005) show that a quota-based control results in a higher profit than tax regulation does when the uncertainty in the current stock size and the unit price of fish is relatively large. Therefore, different types of regulator uncertainty must be taken into account when choosing between price-based and quantity-based regulatory instruments.

Disadvantage

Imposing a tax or royalty is politically unpopular, regardless of whether a fishery is single species or multispecies (Androkovich and Stollery 1989). Setting the appropriate levels of tax or royalty is also difficult, especially if fish prices fluctuate. The New Zealand deemed values experience suggest that when the market price is less than the base price (which is fixed for a period at a time based on the previous year market conditions) then the incentive price will be less than the landing costs and fishers will still have an incentive to discard (Pascoe 1997). Conversely, when prices are high, fishers have an incentive to continue fishing and pay the tax, resulting in higher than planned catches. (However in the latter case it could also be argues that with a higher price the target catch would also most likely be higher).

Discussion

In multispecies fisheries, issues of discarding, overfishing and underfishing of some species arise for both output and input controls at different degree. In both management instruments, some stocks may be overexploited and others under-exploited. Effort quota may require supplementary regulations, including area and real-time season restrictions and stock or gear-specific measures to insure some control of fishing mortality across species and area (Squires et al. 2016). For output controls, institutional designs (Hilborn *et al.* 2005; Grafton *et al.* 2006) and flexibility in the production of managed species (Copes 1986; Squires et al. 1998) affect successful application of output controls in multispecies fisheries (Scheld and Anderson 2017). "Catch-quota balancing" is a major issue under catch quota systems but fishery managers have addressed this difficulty by allowing "retrospective balancing" or trades after landings are made to allow a fisher to cover overharvest of quota, while others use non-trading mechanisms such as "deemed value payments" (Sanchirico et al. 2006). All of these mechanisms introduce flexibility. Fishers can also adjust their fishing behaviour to match quota with catch by changing i.e. fishing location (Branch and Hilborn 2008).

Uncertainty regarding stock abundance and stock-recruit relationship continue to be the major limitation for successful implementation of output controls, while uncertainty regarding catchability is the major issue for input controls. The choice of management instruments is most likely to be case specific. Using the data from the Northern Prawn fishery (NPF), Kompas *et al.* (2008) show that TAC management is preferable over TAE management because there is greater variation in the estimated catch per unit of effort relationship than in the stock recruitment relationship. Using the Faroe Islands fisheries as a case study, Baudron *et al.* (2010) show that when stocks are considered in isolation, a TAE system does not necessarily perform better than TAC, but when the stocks are considered together in mixed fisheries, TAE management seems to be more appropriate. Yamazaki *et al.* (2009) also compare TAC and TAE under uncertainty and noted that neither instrument is always preferred in a world of uncertainty. Factors such as the manager's risk aversion, their importance weighting in terms of expected net profits and biomass, and the trade-off in terms of expected values and variance under each management instrument determine instrument choice. Based on operational multispecies ITQ fisheries in Iceland, New Zealand, Australia and Canada, Sanchirico et al. (2006) show that combinations of approaches are most likely to be successful.

Others suggest that aiming for an optimum outcome is not appropriate nor feasible in highly variable multiple-species fisheries. For example, Wilson (1982) suggested that the long-term management problem in such fisheries should not be the attainment of either maximum biological or economic yields (MSY or MEY) but rather the adaptation of the fishery target in response to the natural, uncontrollable variations in relative species abundance.

One approach that enables such an adaptive management under uncertainty is harvest strategies. Harvest strategies do not rely on conventional stock assessments in setting up target levels of catch or effort, but can use empirical-based harvest control rules (e.g. indices of stock abundance such as standardized catch per unit effort) to automatically adjust the levels of target when environmental and economic conditions change or when new biological and economic information becomes available. Likely performance of different harvest strategies to achieve specific management goals can be tested in a simulation framework, termed management strategy evaluation (MSE) under alternative hypothesis about e.g. recruitment numbers, forms of stock-recruitment relationship, stochastic variation of stocks, errors in catchability, different prices etc., hereby more precautious than traditional management approaches (Smith 1994; Smith *et al.* 2007). With harvest strategies, one can set a target level of catch or effort that corresponds to MEY at first instance, then adjust it based on harvest control rules (e.g. Hoshino *et al.* 2012).

Bioeconomic assessment of management options to achieve MEY

The aim of this part of the project was to assess the ability of different management options to achieve maximum economic yield, assuming that the optimal target reference points were known. To this end, the multispecies multi-fleet model based on the SESSF was used to first estimate the level of effort, biomass and exploitation rate that maximised long term sustainable fishery profits. Given these targets, a range of scenarios was examined, applying different approaches to achieve MEY. These were run dynamically over a 20-year period, with key indicators being the level of fisheries profit generated over time, the number of boats, the level of biomass and the level discards. Initial baseline model runs also considered other measures, such as levels of under-caught quota. A 20 year period was chosen as this represents two management plan cycles (where a management plan is generally of 10 years duration). From the perspective of both managers and industry, a longer time period would result in little benefits.

Long run static equilibrium model results – finding B_{MEY} and F_{MEY}

A static equilibrium version of the model was run to determine the level of fishing mortality, fleet structure and biomass level that maximises sustainable fisheries profits. This was a static equilibrium model run that does not take into account the initial state of the stocks nor the time taken to reach this equilibrium levels of stock size and profits (which would be determined under a dynamic equilibrium framework). However, it provides a point of reference as to what the fishery may be able to achieve in the long term.

The long term sustainable maximum profits was estimated to be $13.9 million a year. The level of fishing effort (in terms of total shots) required to achieve this is shown in Figure 10, and is compared with the current level of fishing effort in each metier. From Figure 10, effort is currently higher than the optimal level in most trawl metiers, with Danish seine effort lower than optimal in Bass Strait, but close to optimal in eastern Bass Strait. Western gillnet effort is close to the optimal level, while other GHT metiers are currently above their optimal level.

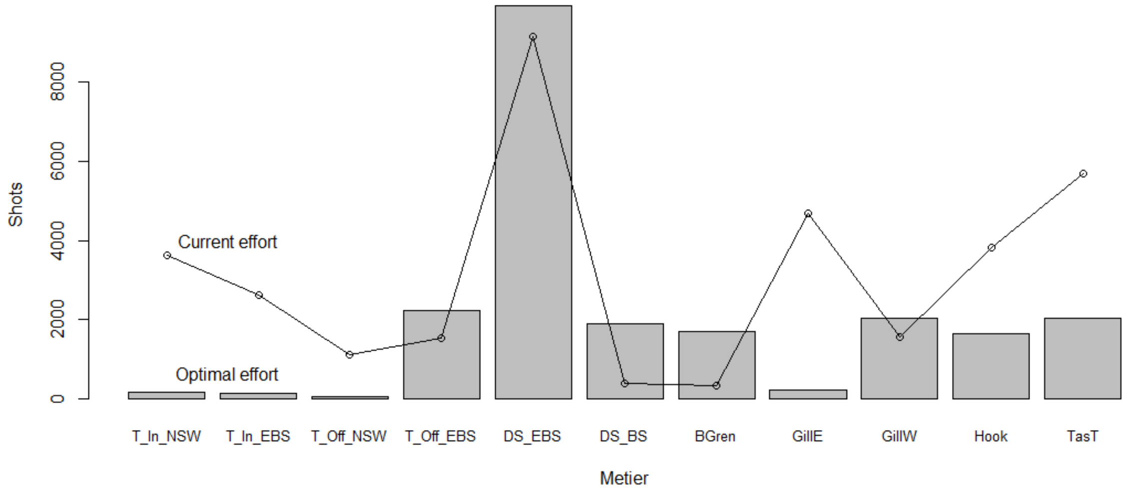


Figure 10. Long run static profit maximising level of fishing effort by métier

The associated optimal fleet configuration is shown in Table 8. The model optimises fishing effort at the metier level, and vessel numbers are derived based on this effort and the current average effort per vessel. As a result, the optimal effort involves fractions of vessels rather than whole vessels (e.g. 10.6 east coast trawlers). In reality, while whole boats would be required, although potentially some vessels could move between fleets as currently defined for part of the season (e.g. some east coast trawlers could operate part of the year in the Tasmanian trawl).

The optimal fleet involves a larger number of Danish seiners than in the 2015 fleet, although all other fleet segments have fewer boats than in 2015. Overall, the total optimal fleet is around half the 2015 fleet (Table 8).

Table 8. The 2015 and long run optimal fleet configuration

Fleet segment	2015 boat numbers	Optimal Boat numbers
East coast trawlers[a]	23	10.6
Tasmanian trawl	14	5.1
Danish seiners	14	17.4
Gillnet, Hook and Trap	72	27.9
Total	123	61.0

a) Includes Grenadier Trawl

The estimated optimal level of biomass at MEY (B_{MEY}) relative to the unexploited level of biomass (K) is compared with the estimated 2015 level (B2015, based on the model parameters) in Figure 11. This suggests that, for many species, their 2015 biomass was close to the optimal level. For some species (e.g. School Shark, SHS), their current biomass is estimated to be substantially lower than the level associated with MEY. In summary, any species below the line are economically overfished.

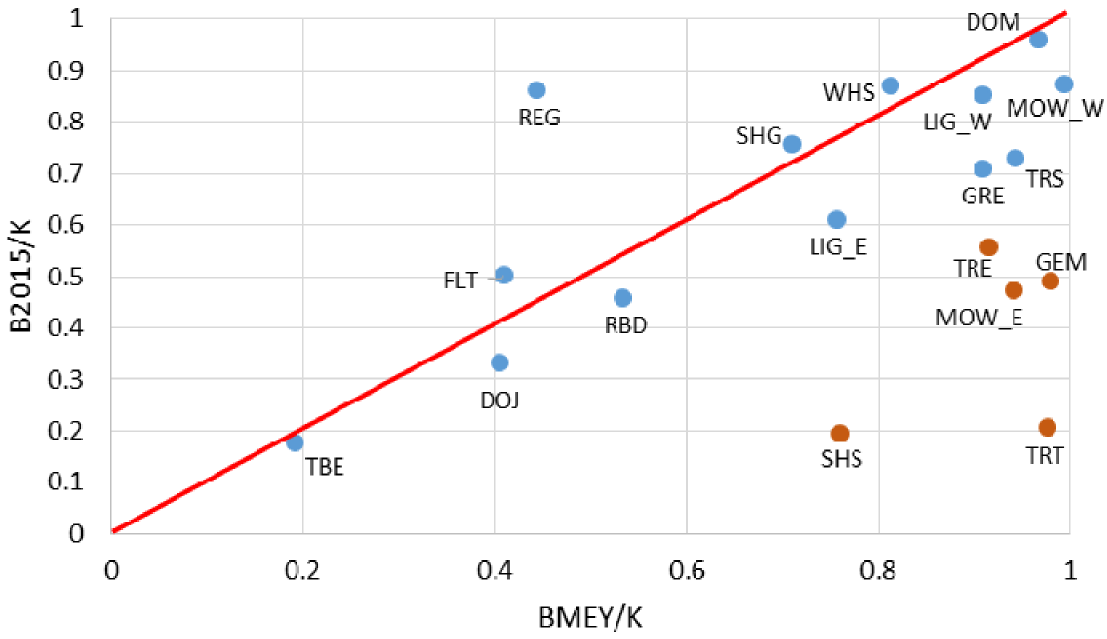


Figure 11. Comparison of 2015 stock status versus model-estimated fishery level B_{MEY}.
Both measures are relative to unfished biomass. The blue colour indicates that the biomass of the species (B2015) in 2015 is close to BMEY, and red line indicates the optimal level of biomass at MEY. The orange colour indicates that the biomass of the species in 2015 is significantly below BMEY.

The results in Figure 11 also highlight the difficulty in assigning a common proxy target reference point for all species in a multispecies fishery. For some species (e.g. Mirror Dory, DOM, in the example model), their optimal level is close to an unexploited level, whereas for others (e.g. Blue Eye Trevalla, TBE) their optimal level is close to the limit reference point.[6]

The model-estimated optimal fishing mortality that produced MEY in the long run (F_{MEY}) and the associated biomass (B_{MEY}) is presented in Table 9. For most, but not all, species, the optimal level of biomass is greater than that in 2015. To achieve MEY this, then, requires rebuilding some stocks while reducing others.

[6] The biological models underlying the bioeconomic model are relatively simplistic, so these results are indicative only. For example, the current assessment of blue eye trevalla (TDE), based on a tier 4 cpue-based assessment, suggests that the stocks are well above their limit reference point (AFMA 2016). Recent analysis also suggests that there may be several stocks of blue eye trevalla rather than a single stock (Williams *et al.* 2017).

Table 9. Estimated optimal fishing mortality (F_{MEY}), optimal biomass (B_{MEY}) and 2015 biomass

Species code	Species name	F_{MEY}	B_{MEY} (t)	B_{2015} (t)
TRT	Blue Warehou	0.024	15719	3303
FLT	Flathead	0.087	18247	22419
GEM	Gemfish	0.014	39201	19539
DOJ	John Dory	0.039	2198	1810
LIG_E	Ling_East	0.122	9055	7306
LIG_W	Ling_West	0.057	14148	13299
DOM	Mirror Dory	0.095	12950	12850
MOW_E	Morwong_East	0.022	28456	14276
MOW_W	Morwong_West	0.001	4422	3880
REG	Ocean Perch	0.001	2069	4014
RBD	Ribaldo	0.526	575	493
TRE	Silver Trevally	0.031	9767	5919
TRS	Silver Warehou	0.028	36351	28146
WHS	Whiting	0.022	11033	11848
GRE	Blue Grenadier	0.114	108122	84326
SHG	Gummy Shark	0.179	12322	13148
SHS	School Shark	0.062	27418	6943
TBE	Blue-Eye Trevalla	0.296	1784	1663

Achieving MEY - Dynamic simulations

The model was run under a range of different assumptions about quota transferability and errors in the stock assessment process. Initial baseline runs included a comparison of different assumptions of quota transferability, namely perfect transferability – where quota could readily move between metiers and fleet segments, and imperfect quota transferability – where quota could move between boats within a metier, but could not move between metiers. The latter are potentially overly restrictive assumptions, while the former is most likely over optimistic. Kompas and Che (2005a) found considerable trade between trawlers and Danish seiners in the SESSF, mostly through leasing rather than permanent transfers. However, studies in other southern Australian fisheries (e.g. van Putten *et al.* 2011) found that trade can be restricted by a more limited range of personal networks. However, van Putten *et al.* (2011) also found that the importance of these networks (and their restrictions on trade) decreased as the quota market evolved, with more non-fishers (e.g. investors, processors etc.) holding a higher share of the total quota.

A range of harvest control rules (HCRs) were considered for the quota-based scenarios (Figure 12). These included a constant fishing mortality (set equal to the fishing mortality estimated in the long-run equilibrium model that maximised long run fleet economic profits over the projection period of 20 years); the "traditional" hockey-stick HCR, where the long run optimal fishing mortality was imposed if the biomass was greater than or equal to the target biomass, otherwise a reduced fishing mortality was imposed based on the where the biomass was in any year relative to the target and limit biomass; and an "alternative HCR" where the fishing mortality was increased above the estimated long run f_{MEY} if the biomass was above the target biomass level.

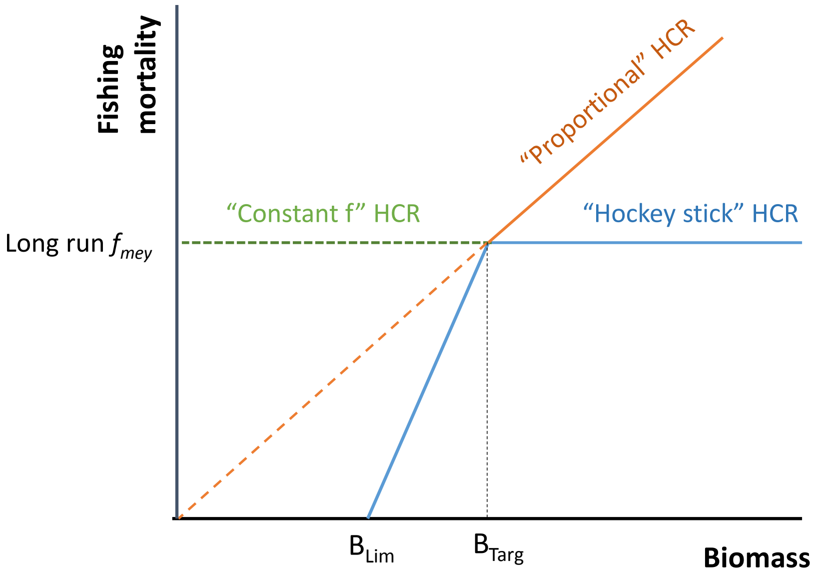


Figure 12. Alternative harvest control rules applied in the simulations

The full range of scenarios examined, described previously, is re-presented in Table 10, with the scenarios re-grouped into similar themes, namely deterministic baseline scenarios (M1, M2), stochastic baseline scenarios (M3-M6), stochastic quotas based scenarios with HCRs (M7-M16), royalty effort control-based scenarios (M17, M18) and effort control based scenarios (M19, M20). Each scenario was run over a 20 year period. With the exception of M1 and M2, the models were run 50 times introducing error into either the estimated biomass or the target value of F_{MEY}. Ideally, more stochastic simulations would have been preferable, although each set of 50 runs took between 24 and 48 hours due to the nature of the genetic algorithm used to find the optimal effort allocation given the available TACs each year.[7]

[7] For example, the genetic algorithm was run for 200 generations (based on earlier trials which found that this was required to ensure convergence), for each of the 20 years optimisations and for each of the 50 sets of simulations, resulting in some 200,000 generations for each model scenario.

Table 10. Summary of dynamic scenarios applied in the final analysis

	Description
Baseline scenarios	
•	*Deterministic, constant fishing mortality (*long run equilibrium model-estimated f_{MEY}*)*
M1	Maximises total fishery profits assuming "perfect" transfer of quota
M2	Maximises total fishery profits assuming imperfect transfer of quota, that is quota transfer is permitted only within a metier but not across metiers.
•	*Stochastic, constant fishing mortality*
M3	Perfect transferability with errors in F_{MEY} estimates
M4	Perfect transferability with error in biomass estimates
M5	Imperfect transferability with errors in F_{MEY} estimates
M6	Imperfect transferability with error in biomass estimates
Harvest control rules	
•	*Imperfect quota market, biomass assessment error, hockey stick HCR*
M7	F varies each year based on hockey stick HCR; long run equilibrium model-estimated f_{MEY}
M8	F varies every THREE years based on hockey stick HCR; model estimated f_{MEY}
M9	F varies each year based on hockey stick HCR; proxy $f_{MEY}=0.8f_{MSY}$ (all quota species)
M10	F varies each year based on hockey stick HCR; proxy $f_{MEY}=0.8f_{MSY}$ for primary species; f_{MSY} for others
•	*Imperfect quota market, biomass assessment error, alternative HCR (f increases above F_{Targ})*
M11	F varies each year based on alternative HCR; based on model estimated f_{MEY}
M12	F varies each year based on alternative HCR; based on proxy $f_{MEY}=0.8f_{MSY}$ (all quota species)
•	*Perfect quota market, biomass assessment error, alternative HCR*
M13	F varies each year based on alternative HCR; based on model estimated f_{MEY}
Quota on only a few species;	
•	*Imperfect quota market, biomass assessment error, constant fishing mortality*
M14	*FMEY*-based TAC set for only the most important species for each gear type (based on contribution to GVP). Selected species were Flathead, Gummy shark and Blue grenadier.
M15	F_{MEY}-based TAC set for only the top 3 important species for each gear type (based on contribution to GVP). Selected species include the species included in above plus Blue-eye Trevalla and Silver Warehou.
M16	F_{MEY}-based TAC set for only the top 1 important species for each gear type (based on contribution to GVP).
Alternative management	
•	*Royalties (i.e. "tax" on landings as a proportion of revenue)*
M17	10% royalty
M18	20% royalty
•	*Effort controls*
M19	Effort not limited other than existing inertia constraints
M20	Effort control where total allowable effort based on hockey-stick style HCR

Baseline scenarios (Models M1 to M6) – holding f constant at the long run f_{MEY} level

The aim of the baseline runs was to determine the potential trajectory of the key fishery indicators in the absence or presence of stochastic error and assuming the optimal fishing mortality (long run f_{MEY}) was applied in each year. The difference between models M1 and M2 reflect how quota could be transferred between metiers. In M1, quota for a particular species could be transferred between metiers (e.g. flathead quota allocated to a trawler metier could be transferred to Danish seiners in subsequent years). In M2, transfers were restricted to boats within the metier. M2 is more restrictive than would be likely to occur in reality, while M1 represents a “better than likely” scenario of perfect transferability.

As expected, fleet profits with the more flexible quota transferability assumptions were generally higher than if restricted (Figure 13a). The net present value (NPV) of profits from M1 was estimated to be $97.4 million compared to a NPV of $84.3 million with the more restricted quota transferability assumption. This suggests that ensuring quota markets are efficient can provide substantial benefits to the fishery. Profits under both scenarios did not reach the level estimated from the long run model ($13.9 million a year) over the 20-year period, but were still increasing. This suggests that achieving the long-run maximum economic yield through holding fishing mortality constant at the “optimal” level will take considerably longer than the 20 years of the modelled scenarios. A simple extrapolation of the profit trajectory suggests that the maximum sustainable profits may not be achieved until as much as 110 years have passed given this strategy.

MEY is a combination of catch rates, biomass and fleet structure that maximises profits. While the number of vessels decreased in the model scenarios (75 and 71 boats respectively, Figure 13b), they did not decrease to the level suggested by the long run model (61 boats) over the period of the simulation. Total biomass increased under both scenarios by a similar magnitude (Figure 13c).

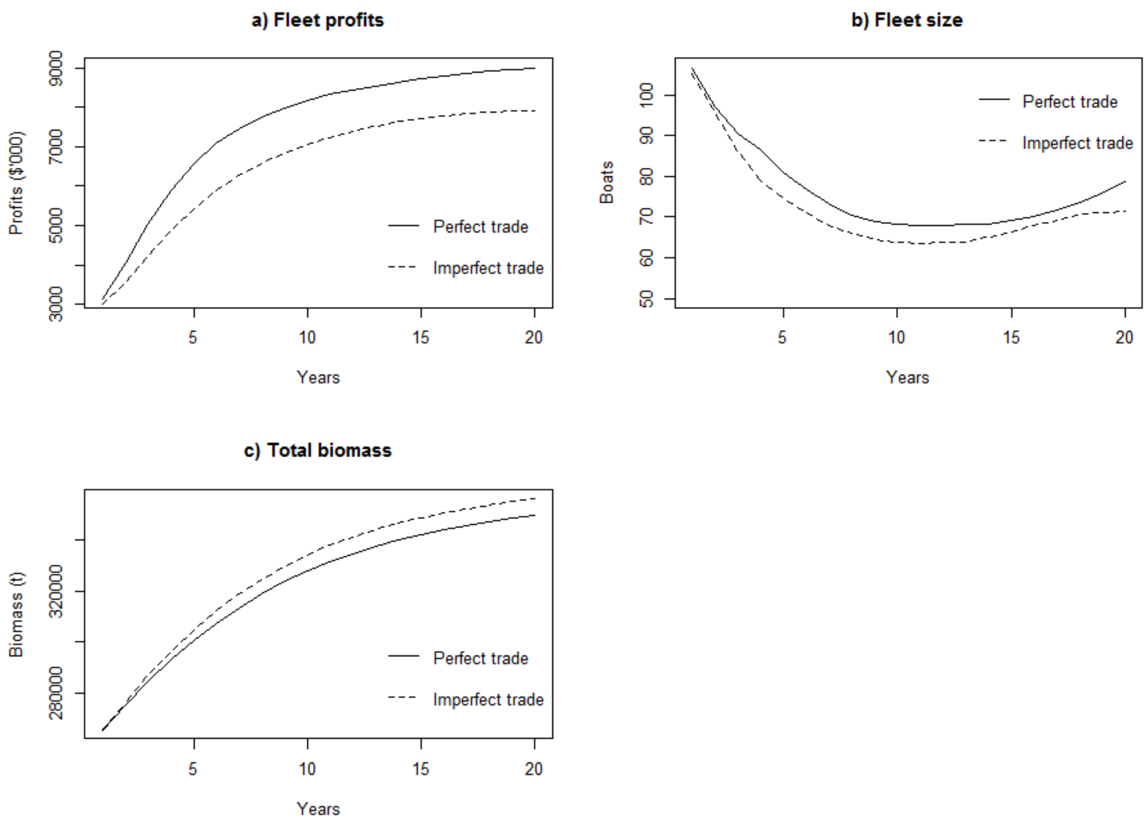


Figure 13. Deterministic baseline model results: profits, fleet size and biomass

Counter to expectations, discards were lower in the scenario with the more restrictive quota trading assumption (Figure 14a) than under more flexible trading assumptions. The inability (in the model) to increase quota in other metiers resulted in less effort transfer and hence less discarding. It also resulted in a greater quantity of quota under-catch than when quota was more readily transferable (Figure 14b). The existence of choke species in the quota mix also contributed to the lower profits earned in the fishery (Figure 13a).

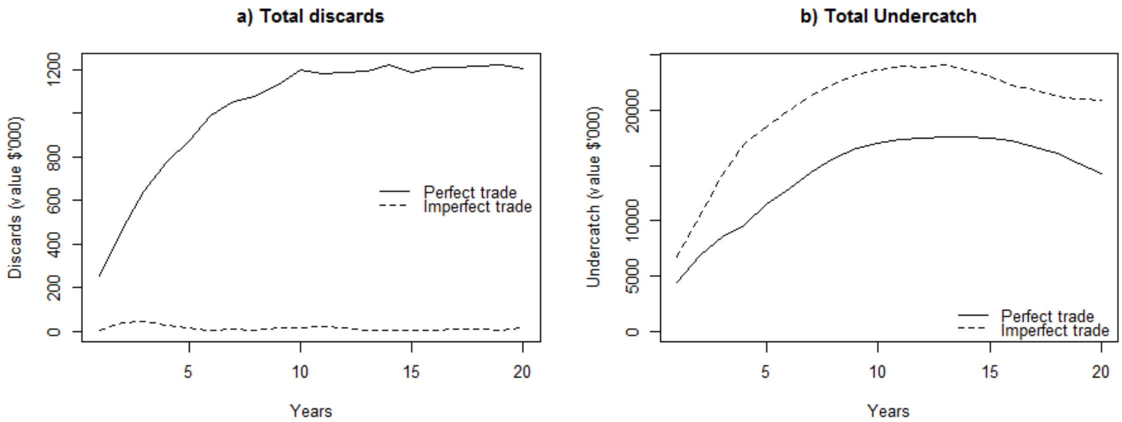


Figure 14. Deterministic baseline model results: discards and under-caught quota

At the species level, discards are dominated by whiting (Figure 15). Discards of all species are relatively low (less than 1% of the TAC) in the model with imperfect quota trading. While more species have a higher proportional under-catch (i.e. % of TAC) in the analysis assuming perfect quota trading, these are dominated by the lower valued species. With imperfect quota trading, a higher proportion of high valued species being under-caught (e.g. sharks, flathead) result in in the total value of the under-caught quota being higher in this scenario (Figure 14b) even though the proportional under-catch is lower for most species (Figure 15d).

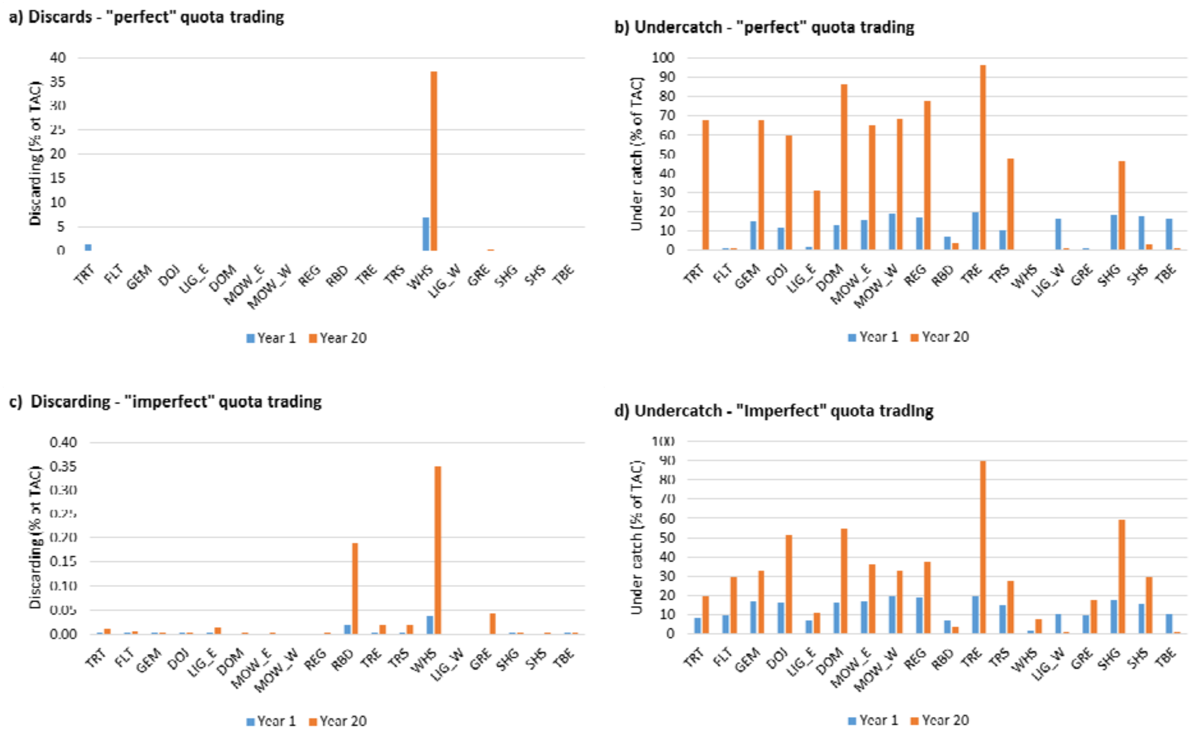


Figure 15. Discards and under-caught quota by species under different quota market assumptions

Models M3 to M6 considered the impacts of errors in the assessment process and the impacts of this over the 20 year simulation period. Models M3 and M5 were based on the model with perfect quota transferability (M1), with random error in the estimate of fishing mortality in M3 and random error in the estimated biomass (used to determine the TAC in the following year) in M5. Similarly, models M4 and M6 considered these errors, but under conditions of imperfect quota trade (i.e. based on model M2). The median net present values for all models were slightly higher than those derived from the purely deterministic case, although this may be an artefact of the limited number of stochastic simulations (i.e. 50). Errors in the estimation of the appropriate f_{MEY} had a greater impact on the variability of fishery profits than errors in the assumed stock size (Figure 16).

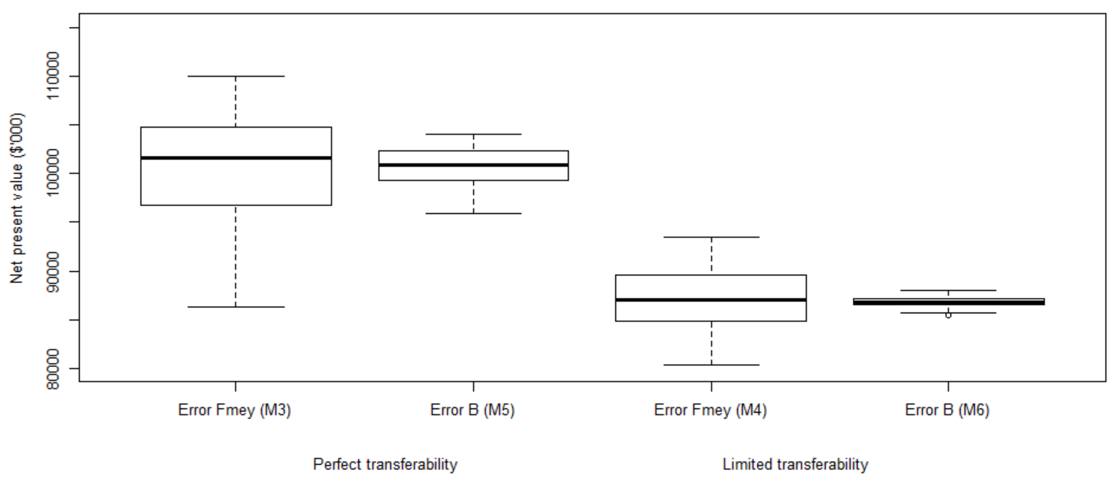


Figure 16. Stochastic baseline model results with errors in f_{MEY} (M3, M4) and estimated biomass (M5,M6)

Harvest control rules

Scenarios M7 to M13 involved a range of harvest strategies based around fixed harvest control rules (HCR). M7 to M10 involved the use of a "hockey stick" – style HCR; while M11 to M13 involved an alternative HCR which allowed a higher fishing mortality rate if the stock was above its target level (see Figure 12). All models were run assuming random error in the estimate of the actual biomass used in quota setting.[8]

Scenarios M7 to M10 were also based on the model with limited quota transferability (M6). M7 and M8 were based on the model-estimated f_{MEY} as the upper limit to fishing mortality, with lower levels of f being imposed if the stock was below the target. M8 assumed a multi-year quota, with the TACs updated every three years based on the estimated biomass in the years that the TACs were updated. The results of these scenarios are presented in Figure 17. The results for scenario M6 is also re-presented for comparison.

From Figure 17, the use of a hockey-stock HCR based on the model estimated f_{MEY} (M7) resulted in a lower net present value of profits on average than the use of a constant f_{MEY} rate (M6). The use of a multi-year quota system (M8) resulted in similar levels of profit to the scenario with annual assessments (M7), although it had a greater variability in the NPV over the set of stochastic simulations (Figure 17).

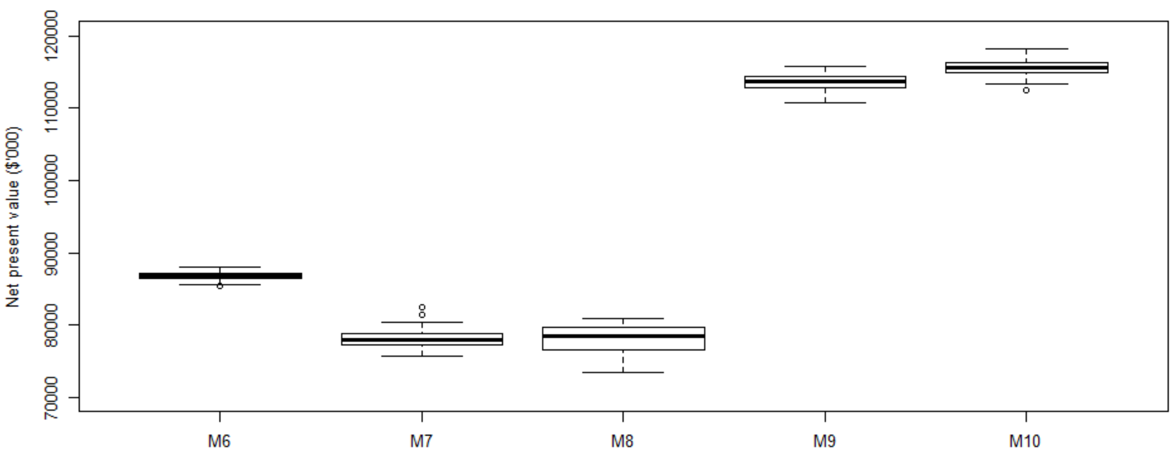


Figure 17. Stochastic model results with errors in estimated biomass and using a hockey-stick HCR

[8] From the previous set of model runs, we can assume that the median value of the estimated profits and other outputs is similar to those where we add random error to the estimate of F_{MEY}, although the variability is less.

Using the default proxy target fishing mortality ($\tilde{f}_{MEY} = 0.8 f_{MSY}$)[9] for all species (M9) resulted in a higher NPV than using the model estimated f_{MEY}, while using the proxy value for the main target species (top 5 species by value) and f_{MSY} for the remainder (M10) resulted in a higher average NPV again. This is a reflection of the (generally) higher fishing mortality rate for each species based on the proxy compared with those derived from the long run optimisation model (Figure 18). This reduces the impact of choke species on the ability of the fishers to land catch, resulting in a higher revenue and level of profits over the 20 years of the simulations.

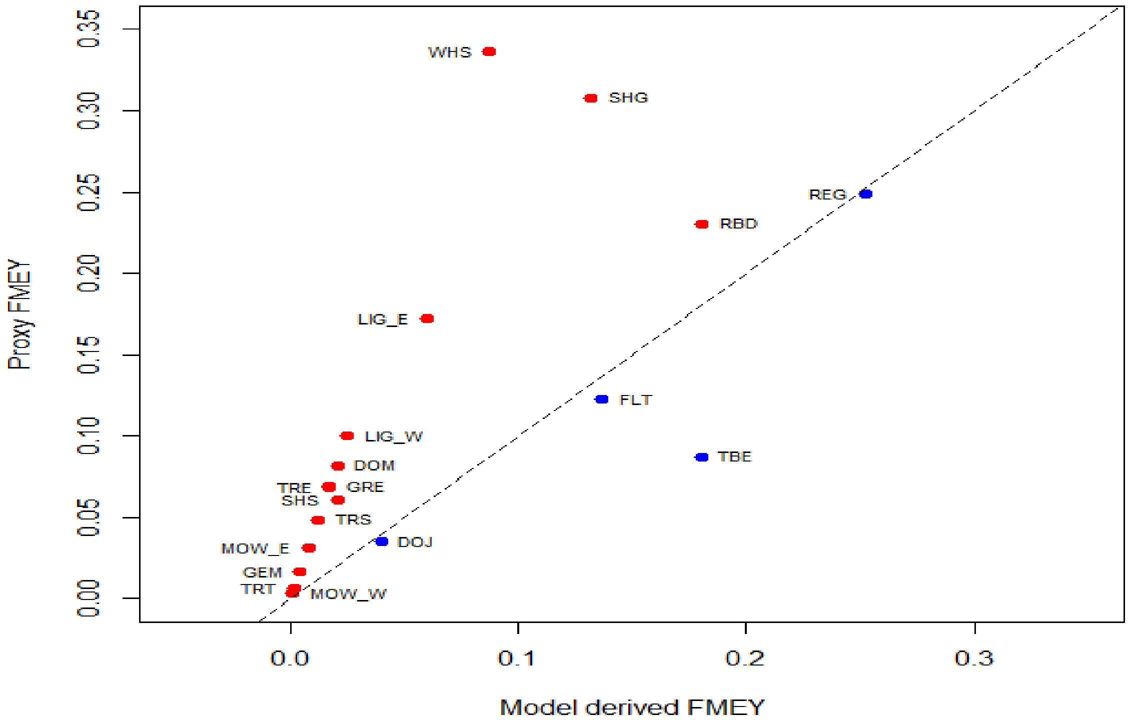


Figure 18. Comparison of proxy fishing mortality rates ($\tilde{f}_{MEY} = 0.8 f_{MSY}$) versus model derived rates. Red dots indicates that the proxy fishing mortality for the species is greater than the model-derived level. The blue dots indicate that the proxy fishing mortality for the species is lower than the model-derived level.

While producing a higher NPV of profits over the 20 years of the simulations, the proxy fishing mortality also resulted in a substantial under-catch (i.e. the TACs derived from the proxy fishing mortality were substantially higher than the ability of the fleet to take them, given also the constraints imposed by other TACs) (Figure 19). Under-catch (and discards) are aggregated in value terms (i.e. price times quantity discarded/under-caught) to represent the potential forgone revenue. From the simulations, under-caught quota was four to five times greater (in value terms) when the proxy MEY fishing mortalities were applied than when the model estimated MEY fishing mortalities were used to set the TACs. Discards were also higher when the proxy fishing mortality rates were used, mainly because the combination of TACs were less proportional to the potential catch composition of the

[9] The Commonwealth harvest strategy policy specifies a proxy target biomass of $\tilde{B}_{MEY} = 1.2 B_{MSY}$. From this, it can be shown that, using the Gompertz growth model underlying the Fox model, $F_{MEY}=0.817F_{MSY}$. This has been rounded down to $0.8F_{MSY}$.

fleet. Using a constant fishing mortality to set the TACs resulted in the least discarding in the simulations as the catch composition was better aligned to the set of TACs. However, undercatch of quota was still estimated to occur.

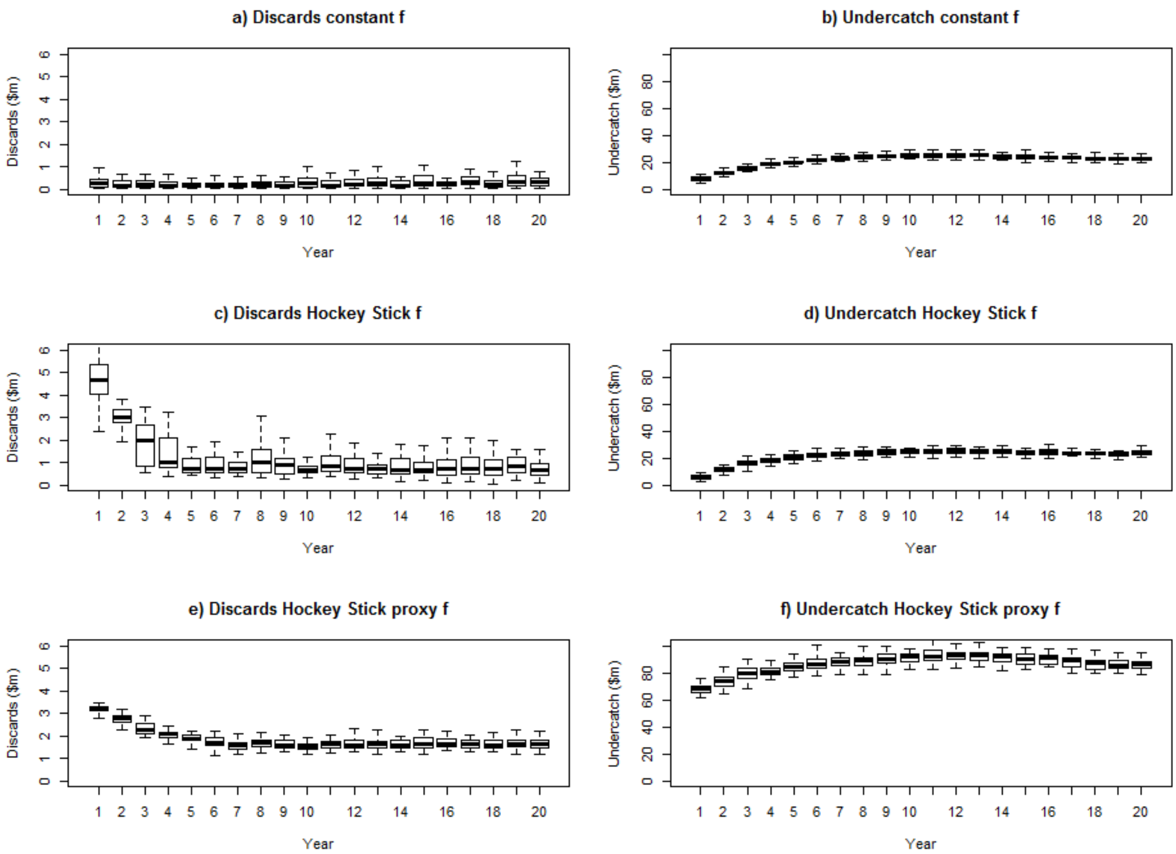


Figure 19. Discards and under caught quota, hockey stick HCR. Panels (a,b) = M6. Panels (c,d) = M7. Panels (e,f) = M9.

The total fishery biomass was estimated to increase similarly under all hockey stick HCR scenarios (Figure 20), although the biomass in the final year (year 20) was higher in the scenarios using the model estimated f_{MEY} compared with those using the proxy $\tilde{f}_{MEY}$ (Figure 21). Boat numbers were also higher in the scenarios using the proxy $\tilde{f}_{MEY}$ (Figure 21), although boat numbers in all scenarios were higher than the optimal fleet size at the long run equilibrium (the red line in Figure 21a).

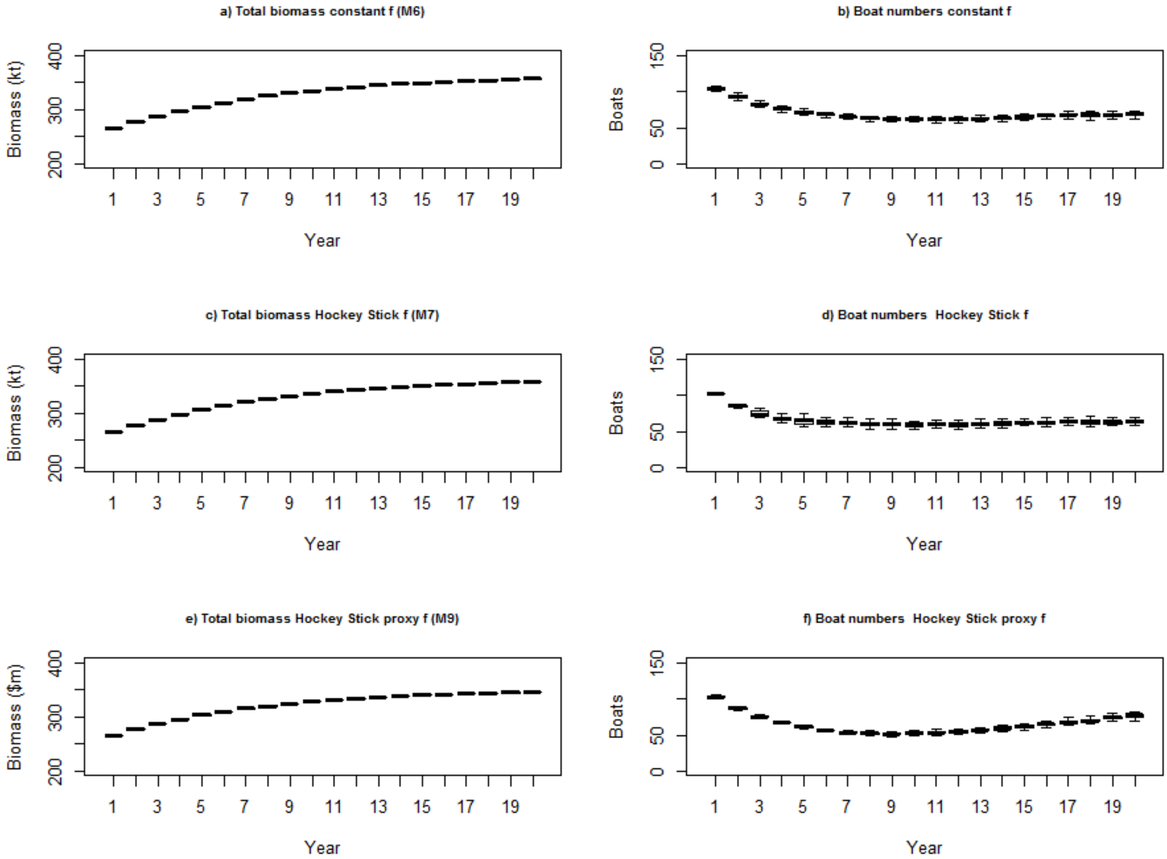


Figure 20. Biomass and fleet size, hockey stick HCR

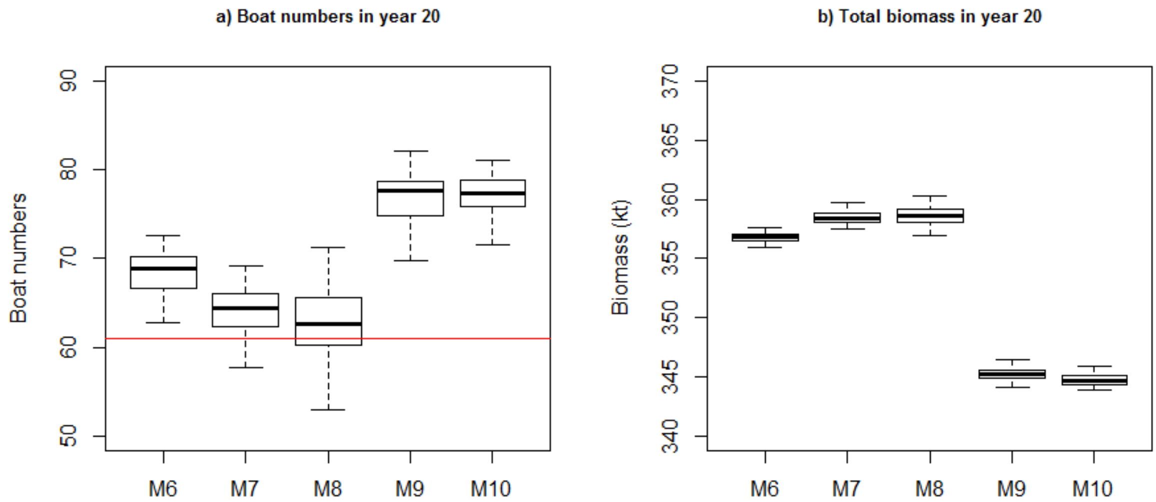


Figure 21. Biomass and fleet size in year 20, hockey stick HCR

The hockey-stick HCR places an upper limit on the level of fishing mortality (Figure 12). This places an additional constraint on the TACs (and catch) when the biomass is greater than the target reference level. A modified (proportional) HCR was considered in which the target fishing mortality rate was increased if the biomass exceeded the target biomass (i.e. $f = f_{MEY}\left(B / B_{MEY}\right)$), otherwise it was reduced in line with the standard hockey-stock HCR (i.e. such that $f=0$ if $B<B_{LIM}$).

The alternative HCR provided relatively few additional benefits over the standard HCR when using the model estimated f_{MEY} (i.e. M11 versus M7) and when quota transferability is limited (Figure 22). Improving quota transferability results in a substantial increase in economic profits (M13). In contrast, when the proxy $\tilde{f}_{MEY}$ is used, the alternative HCR results in a substantial increase in the net present value of economic profits (M12 versus M9).

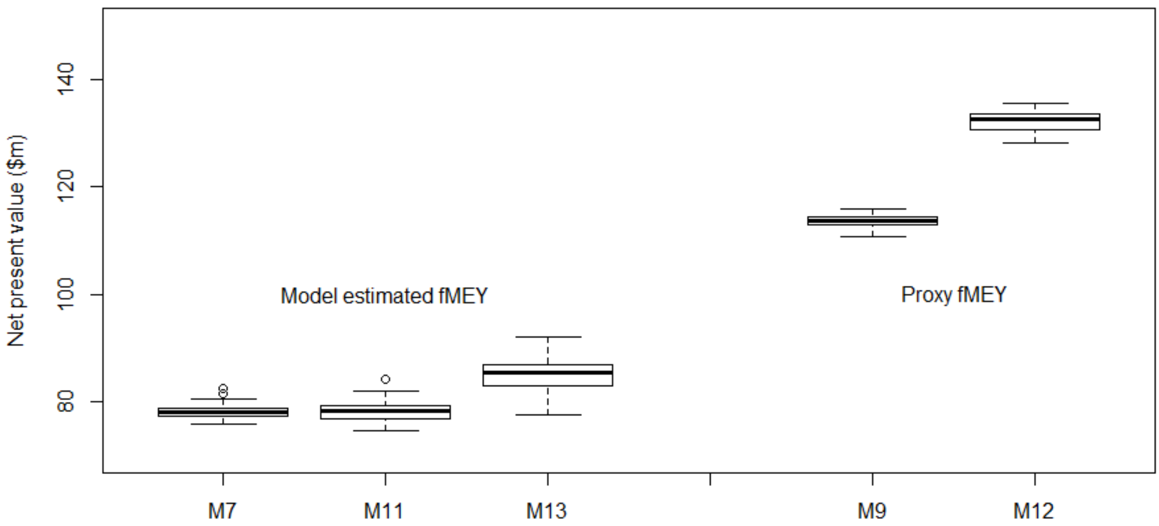


Figure 22. Stochastic model results with errors in estimated biomass and using an alternative HCR

Alternative approaches: Royalties

Royalties or some other form of landings charge have been proposed as an alternative means of generating MEY in fisheries (e.g. Ganguly and Chaudhuri 1995; Grafton 1995; Hannesson 2007). A key benefit of the use of such a system is that the economic rents are captured for use by the broader community (Grafton 1995). A variable royalty system can also provide incentives for fishers to try and avoid some species by reducing the effective revenue gained by their capture. A downside of such a system, however, is that it reduces the profitability of the industry itself, reducing the acceptability of the management system. As no quotas are in place, there are no incentives for discarding or under catch to occur.

Determining an optimum set of royalties is complex, and is beyond the scope of this study. Instead, two common royalty rates were examined – 10% and 20% - which were applied to all landed catch.

The results of the stochastic simulations are presented in Figure 23. The objective function for the short term (annual) model was again maximising fishery profits on the basis that this still drives fishery behaviour, although revenues were reduced by the royalty rate. The ITQ scenario with a constant fishing mortality (M6) is presented also for comparison.

From Figure 23, a 10% royalty on landings results in a lower profit to the industry (upper panel of figure), but an overall gain when the value of the royalties collected is taken into consideration (lower panel of figure). In comparison, a 20% royalty results in all economic rent being extracted, with the industry making a small economic loss on average. The total net economic rent is similar to that generated under the base run scenario, with the difference is that the former accrues all to the rent-collector (i.e. the agency that receives the royalty) and the latter accrues all to the industry.

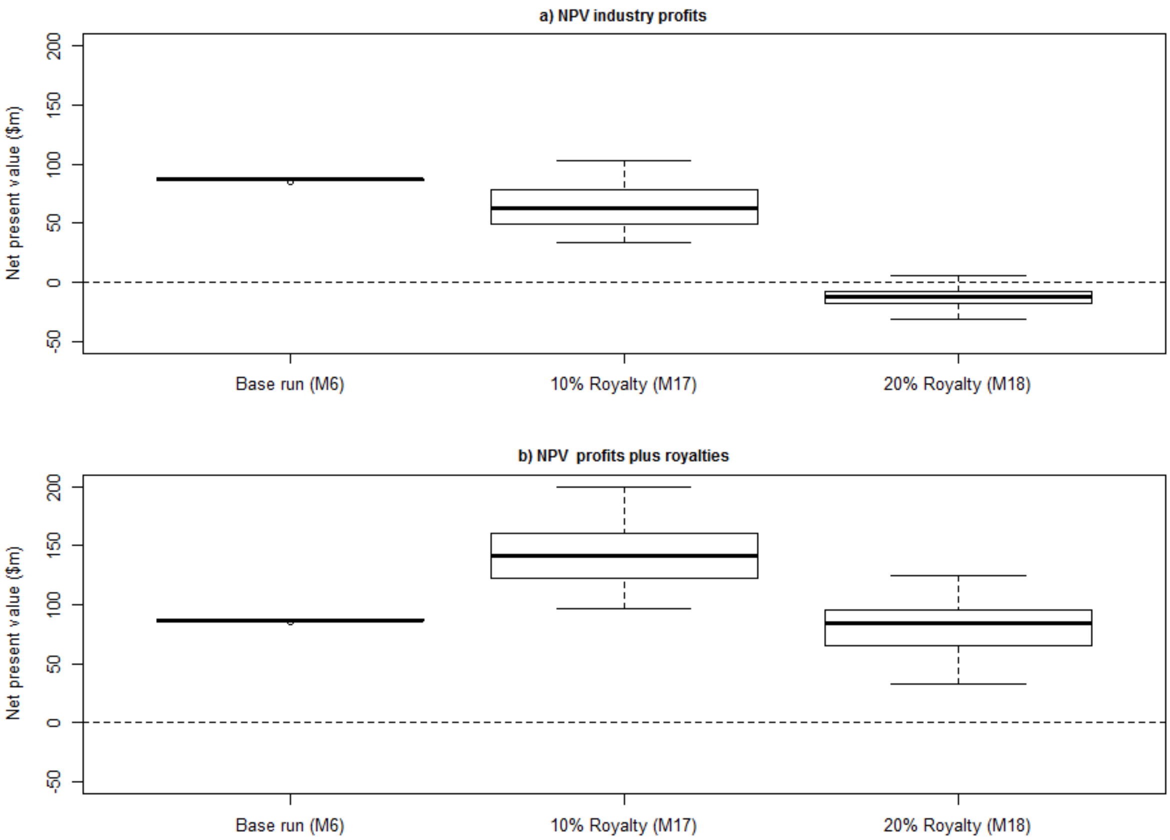


Figure 23. NPV of industry profits and total royalties under different royalty scenarios

The introduction of a royalty system also has a substantial negative impact on fishery profits in the short term. Both a 10% and 20% royalty result in negative economic profits being achieved by the industry in the first few years of its implementation (Figure 24). Under a 20% royalty, even though the NPV was negative, the industry does achieve positive economic profits but not until after 9 years of stock recovery.

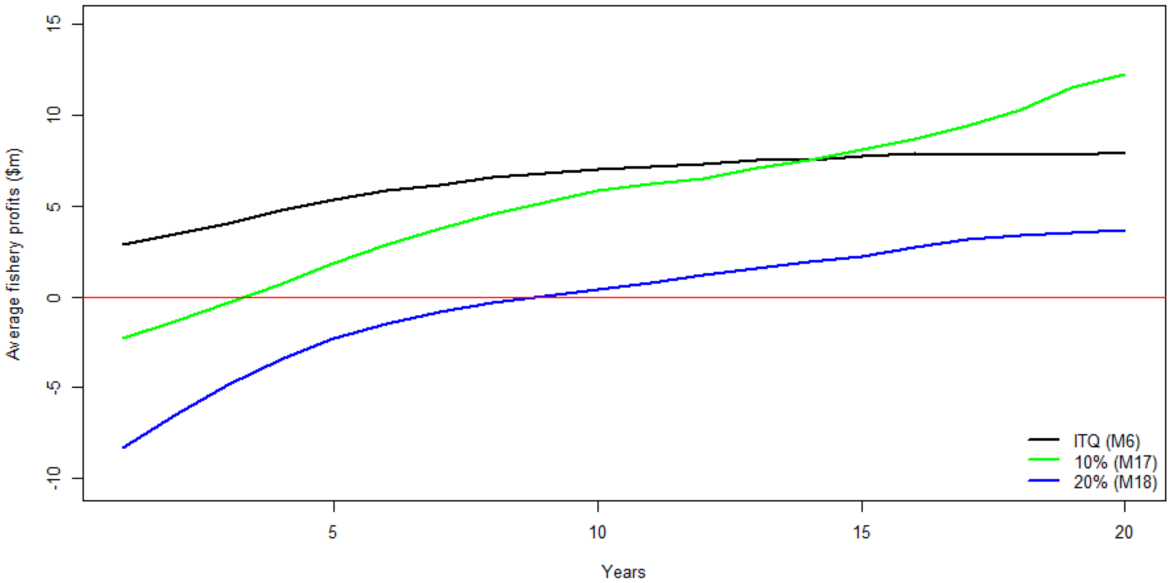


Figure 24. Industry profits over time under different royalty scenarios

The biomass under each management option over the 20 year period is presented in Figure 25, while the change in fleet size is presented in Figure 26. A 10% royalty results in an initial decrease in fleet size, but boat numbers subsequently increase as the total biomass increases. This increase in fishing effort results in the total biomass being lower than under the other scenarios, and decreasing towards the end of the simulation period (Figure 25b). In contrast, the removal of economic profits from the fleet with a 20% royalty result in a substantial decline in vessel numbers – below that of the long run equilibrium optimal level – and a subsequent increase in biomass to levels higher than that estimated under the baseline ITQ scenario.

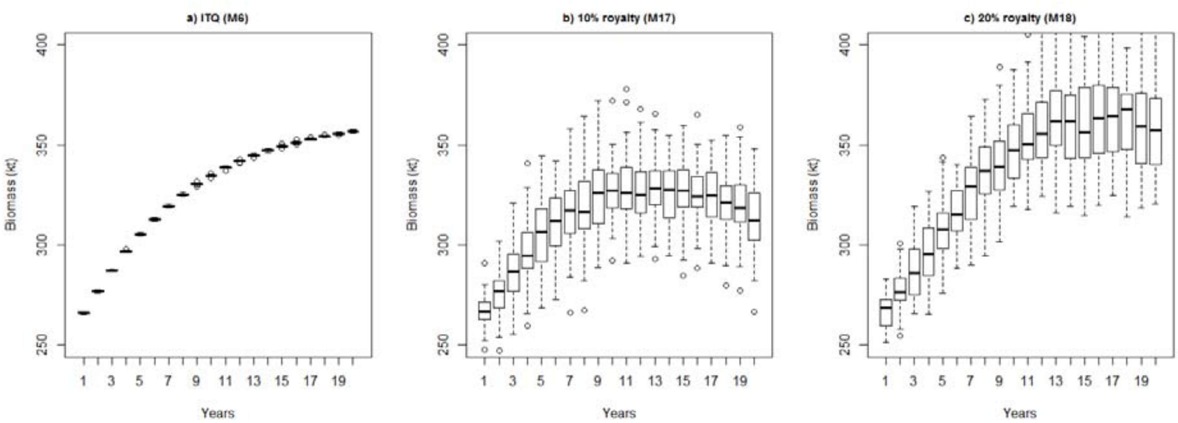


Figure 25. Biomass under different royalty scenarios

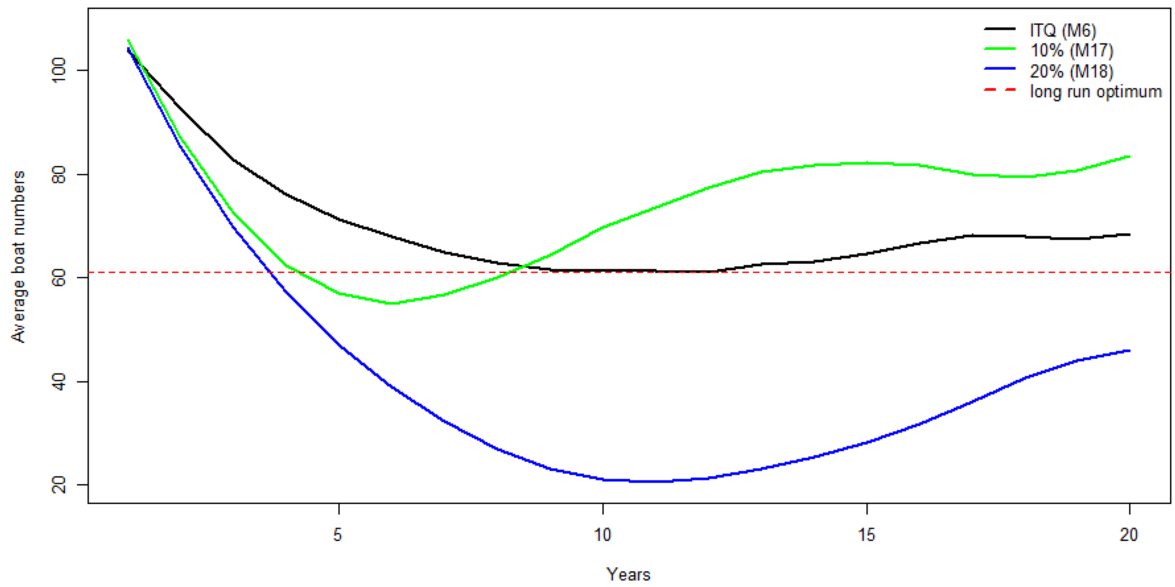


Figure 26. Fleet size under different royalty scenarios

Alternative approaches: Effort controls

The model was also run removing the current quota controls, but retaining restrictions on vessel movement between metiers. Two alternative scenarios were considered: one with just the current assumed limits on effort reallocation (linked to inertia assumptions in the fishery), and the other where the annual total allowable effort limits were modified based on a HCR analogous to that used for the TAC setting (i.e. based on the average of the relationship between target and current biomass of the set of species). Again, as no quotas are in place, there are no incentives for discarding and under catch to occur.

As would be expected, removal of quota constraints results in a substantial increase in the NPV of fishery profits over the 20 year period (Figure 27), with profits increasing at a greater rate over time than under the ITQ scenario (Figure 28). However, profits were estimated to start to decline towards the end of the 20 year period.

Fleet size stayed high under both effort control scenarios, increasing towards the end of the period (Figure 29). The effect of this was to slow the growth in biomass, with biomass declining from midway through the time period examined (Figure 30). This declining biomass contributed to the decline in profits observed in the simulation output at the end of the simulation period. Given the decline in biomass, it is likely that profits beyond the 20 years would continue to decrease.

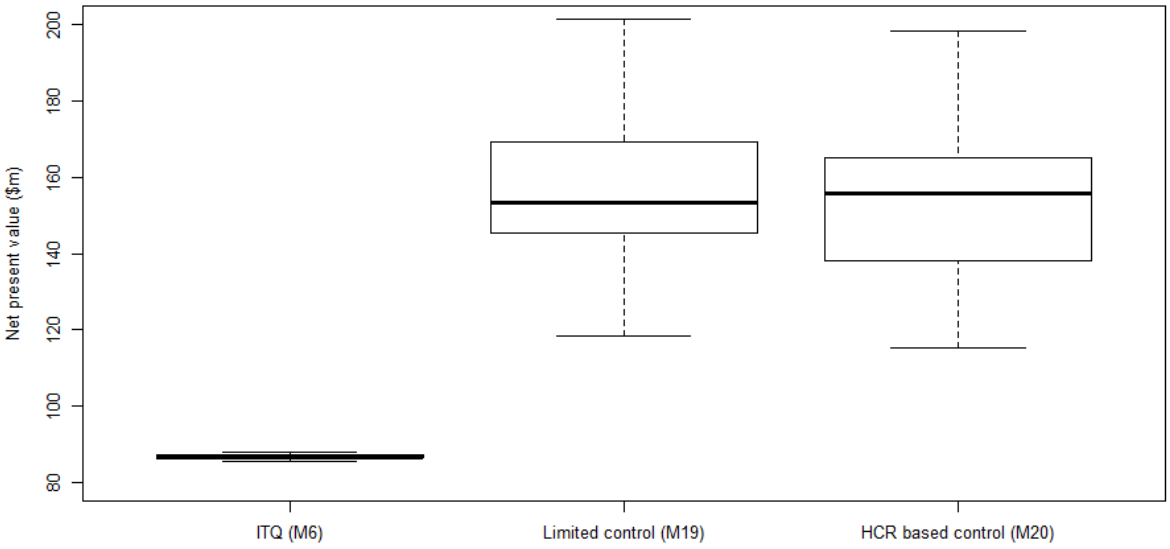


Figure 27. NPV of fishery profits under different effort control scenarios

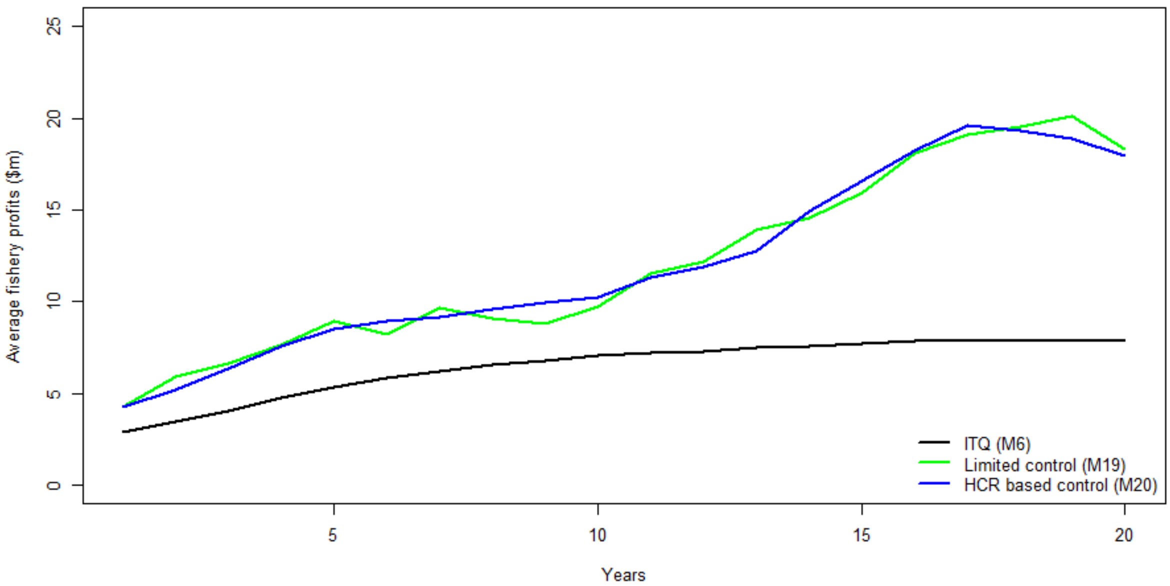


Figure 28. Flow of fishery profits over time under different effort control scenarios

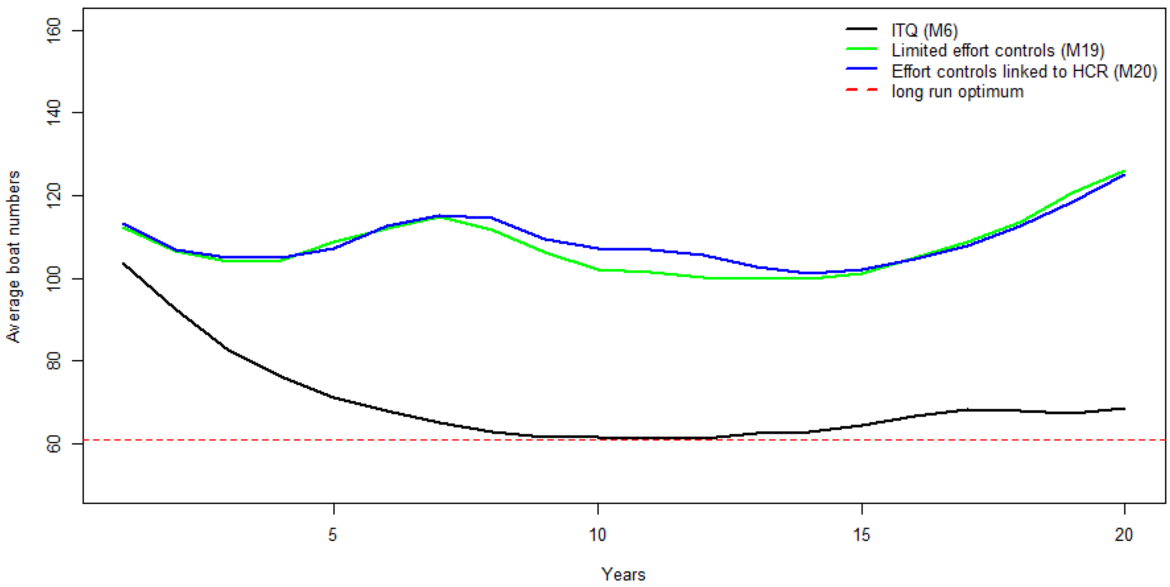


Figure 29. Fleet size over time under different effort control scenarios

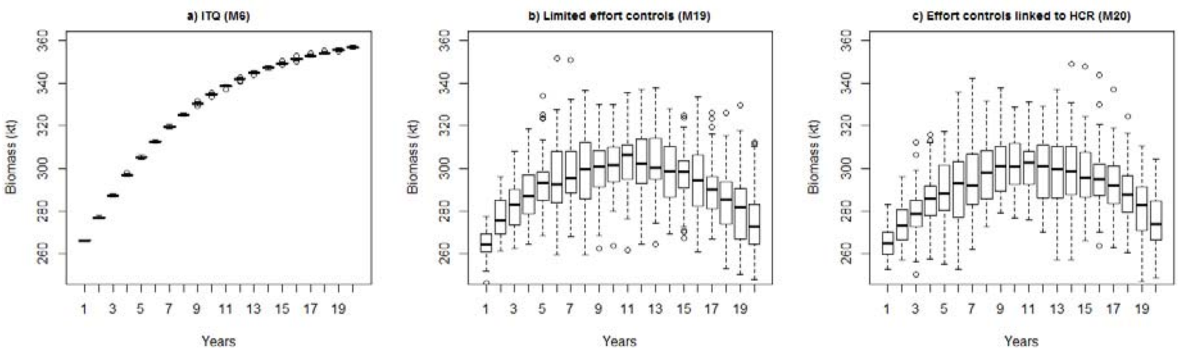


Figure 30. Biomass over time under different effort control scenarios

Dynamic MEY – finding DB_{MEY} and DF_{MEY}

Basing harvest strategies on the long run equilibrium "optimal" fishing mortality generally resulted in lower values of economic profits over the time period of the simulations compared with alternative approaches. Some of these approaches, such as the effort control scenarios, resulted in a higher net present value of profits over the 20 year period, but substantially lower (and declining) levels of biomass, suggesting that profits beyond this period would eventually decrease. Further, stock levels will not be at a level that sustains maximum net economic returns in the future.

From extrapolating the results of the model based on the long run equilibrium "optimal" fishing mortality, this equilibrium may not be achieved for over 100 years. While not tested,[10] it is expected

[10] Model 6 was also run for a 50 year period, over which time profits were still increasing (at a slow rate) and stocks had not reached their equilibrium state (although were fairly close). A longer time period was not examined.

that this scenario would result in the greatest economic benefits over time with a zero discount rate *and* an (effectively) infinite time horizon. Such a time horizon is not realistic from a management perspective. Similarly, discount rates are non-zero in reality.

The determination of dynamic MEY in a single species fishery is well understood (Grafton et al. 2010a). An approximation for dynamic MEY in multispecies fisheries has been the harvest rate that maximises the net present value of fishery profits over time (Silvert and Smith 1977). This approach is used in the Northern Prawn Fishery bioeconomic model to set an MEY-based effort level (Punt et al. 2011), although this is a "pragmatic" version of dynamic MEY – it assumes the fleet size is fixed and effort levels are varied to achieve an equilibrium yield/biomass over a seven year period.

Hence, the determination of the dynamic MEY fishing mortality and biomass is a function also of the assumptions and constraints used in the analysis. The dynamic version of the model was run with two sets of assumptions. First, the constant level of fishing mortality for each species that would maximise discounted profits over time was estimated, assuming the level of profits in year 20 would continue indefinitely. This assumes an immediate reducing in fleet size in year 1 required to achieve the appropriate fishing mortality. Second, a "sliding scale" was imposed, where fishing mortality was reduced gradually over the first five years, with fishing mortality fixed from year 6. Again, it was assumed that profits in year 20 would continue indefinitely. As the model is attempting to determine the optimal fishing mortality over time, there is assumed to be no discarding and no under-catch (i.e. the quotas are set and allocated perfectly in each time period – an unrealistic assumption, but consistent with the aim of determining an optimal set of fishing mortality rates).

Fishing mortality rates that maximise the net present value of future profits

The fishing mortality rates that maximised the net present value of fishery profits over time were generally lower than those derived from the equilibrium model (Figure 31). This is counter to initial expectations, but given the estimated long time period required for the fishery to reach the long run equilibrium, there are benefits in recovering the stocks quicker once a non-zero discount rate is included in the analysis, even if this requires a short run decrease in harvest. This result is largely driven by the higher flow of profits from year 20: the NPV of profits over the first 20 years under this scenario is $140 million, while the NPV of profits after year 20 is an additional $157 million. Hence, the long run outcome is still highly influential on the shorter term optimal fishing mortality given the discount rate used (4%) even in a dynamic optimisation.

The optimal fleet activity in the dynamic MEY model also differs slightly from that of the long run equilibrium (Figure 32). For example, the dynamic optimal fishing effort in the Tasmanian trawl metier was substantially higher than that estimated using the static model. For some metiers, optimal fishing effort was higher than in the base year. The potential increase was constrained in the optimisation procedure on the basis that the fleet size in any metier could not substantially increase in the short term (although could decrease). The total fleet size under this scenario was 63 boats.

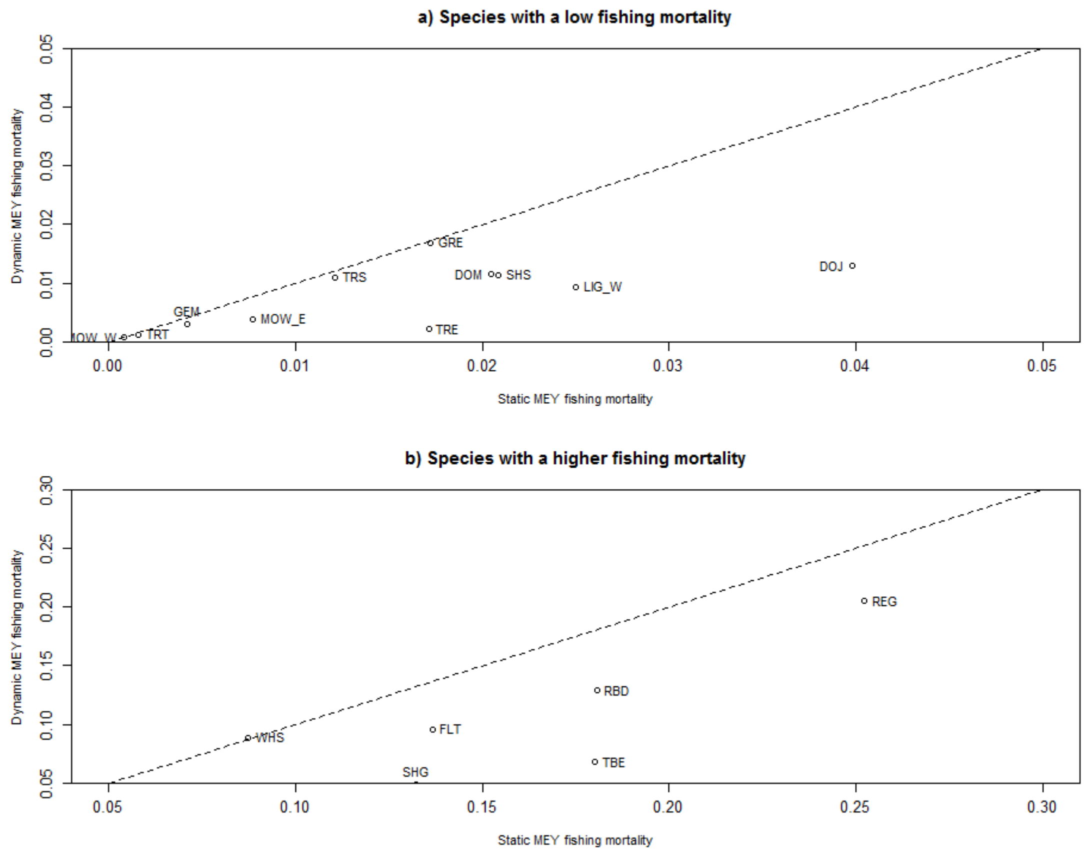


Figure 31. Comparison of optimal fishing mortality rates from the static and dynamic equilibrium models

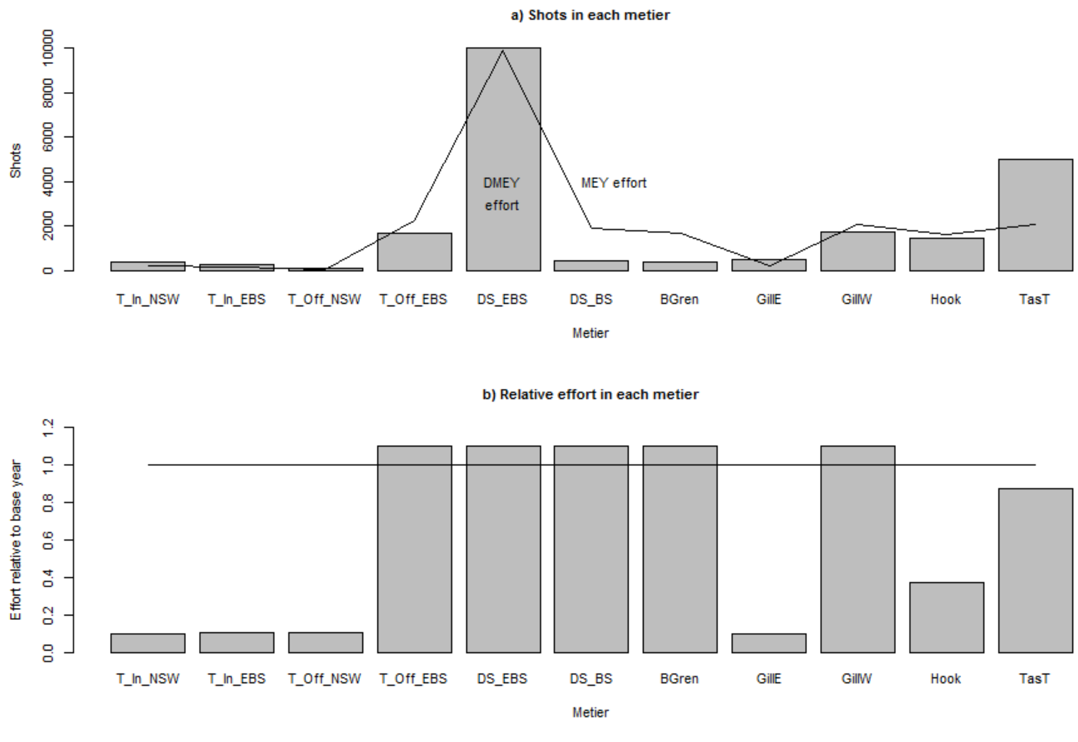


Figure 32. Optimal fishing effort configuration, dynamic MEY

Fishing mortality rates that maximise the net present value of future profits with effort decreasing more gradually over the first five years

The previous optimisation assumed that fleet effort could decrease immediately, with a constant fishing mortality from year 1 that maximised the net present value over time. An alternative scenario was developed in which the fishing mortality decreased by three percent a year over the first five years, after which it would remain constant (at 85% of the first year's level).

The higher fishing mortality in the first few years resulted in higher profits over the first 20 years (Figure 33), and an overall higher NPV of profits over this period ($145 million) while also resulting in a higher level of discounted future profits beyond year 20 ($165 million).

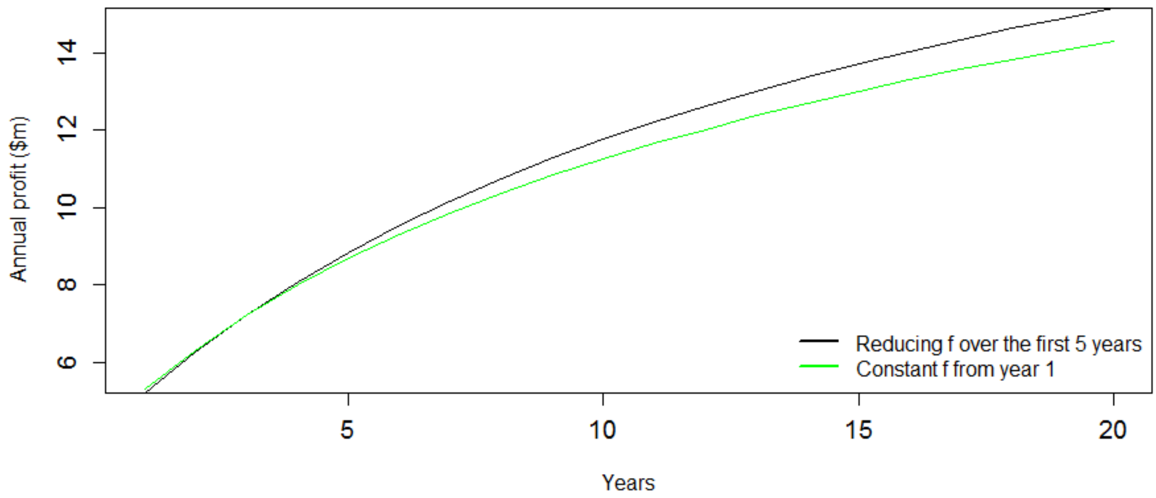


Figure 33. Flow of profits over the first 20 years; constant fishing mortality from year 1 compared with reducing fishing mortality over the first five years

Again, the fishing mortality rates that maximised the net present value of fishery profits over time were generally lower than those derived from the equilibrium model (Figure 34). Two levels of fishing mortality are presented in Figure 34: the fishing mortality in year 1 and the fishing mortality from year 6.

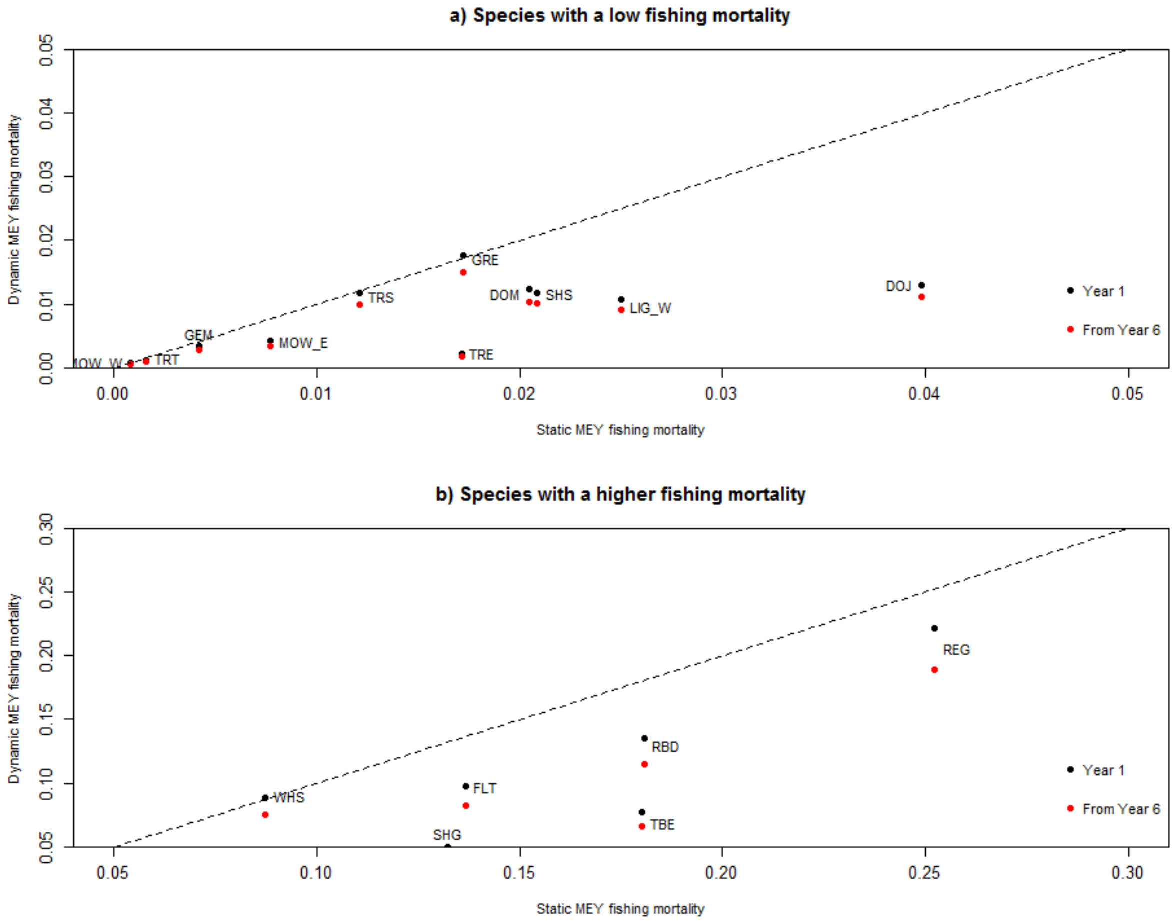


Figure 34. Comparison of optimal fishing mortality rates from the static and the modified dynamic equilibrium models

Comparison of "optimal" fishing mortality rates under the different assumptions

A comparison of the different "optimal" fishing mortality rates under the different assumptions is presented in Table 11. For most species, the proxy MEY fishing mortality was substantially higher than that estimated using either the static or dynamic models. This contributed to the higher NPV over the 20 years of the simulations, although longer term economic benefits are likely to be lower than those of the static or dynamic equilibrium models.

The fishing mortality rates that maximised the net present value of profits over time (the dynamic MEY) were generally lower than those associated with the static fishery MEY, counter to expectations. This is potentially an artefact of the low discount rate assumed (4% in the simulations), as the longer term stream of future benefits had a substantial impact on the outcome. With a higher discount rate, it is expected that the optimal fishing mortality rates would be also higher. Allowing the fishing mortality rates to decrease gradually over a 5 year period resulted in fishing mortality rates in year 1 being higher than those in the constant *f* optimisation, and lower rates being observed from year 6. This strategy did allow for a higher net present value of profits that the constant *f* optimisation.

The proxy f_{MEY} rates were also higher in general than those estimated from the static equilibrium and dynamic models. These also resulted in a higher NPV of profits over the 20 year period of the simulations, but as biomass levels were lower in the 20th year, it is expected that the NPV over a longer term would be less using the proxy rates than those derived through the optimisation models. Nevertheless, from a management perspective, and given the potential for changes in technology, costs and prices over time, a 20 year time horizon may be more suitable to based management decisions than a much longer time frame.

Table 11. Comparison of "optimal" fishing mortality rates under the different assumptions relative to F_{MSY}

Species		f_{MSY}	Alternative f_{MEY} estimates relative to f_{MSY}				
			Constant f			Decreasing f	
			Static equilibrium	Proxy (0.8 f_{MSY})	Dynamic MEY	Dynamic Year 1	Dynamic Year 6+
TRT	Blue Warehou	0.0079	0.20	0.80	0.15	0.15	0.14
FLT	Flathead	0.1530	0.89	0.80	0.63	0.64	0.54
GEM	Gemfish	0.0208	0.20	0.80	0.15	0.16	0.14
DOJ	John Dory	0.0440	0.90	0.80	0.29	0.30	0.25
LIG_E	Ling_East	0.2150	0.28	0.80	0.19	0.19	0.17
LIG_W	Ling_West	0.1252	0.20	0.80	0.08	0.09	0.07
DOM	Mirror Dory	0.1024	0.20	0.80	0.11	0.12	0.10
MOW_E	Morwong_East	0.0387	0.20	0.80	0.10	0.11	0.09
MOW_W	Morwong_West	0.0043	0.21	0.80	0.19	0.19	0.16
REG	Ocean Perch	0.3110	0.81	0.80	0.66	0.71	0.61
RBD	Ribaldo	0.2880	0.63	0.80	0.45	0.47	0.40
TRE	Silver Trevally	0.0854	0.20	0.80	0.02	0.02	0.02
TRS	Silver Warehou	0.0606	0.20	0.80	0.18	0.19	0.16
WHS	Whiting	0.4200	0.21	0.80	0.21	0.21	0.18
GRE	Blue Grenadier	0.0862	0.20	0.80	0.20	0.20	0.17
SHG	Gummy Shark	0.3850	0.34	0.80	0.13	0.13	0.11
SHS	School Shark	0.0760	0.28	0.80	0.15	0.16	0.13
TBE	Blue-Eye Trevalla	0.1090	1.65	0.80	0.62	0.71	0.61

Discussion

Since the introduction of the Australian Commonwealth Harvest Strategy Policy (HSP) in 2007 (DAFF 2007), only two fisheries have developed bioeconomic models for assessing economic targets for its key species: the Northern Prawn Fishery (Punt *et al.* 2011; Dichmont *et al.* 2016); and the Great Australian Bight Trawl Sector (GABTS) of the Southern and Eastern Scalefish and Shark Fishery (SESSF) (Kompas *et al.* 2012b). While these are both multi-species fisheries, they are generally limited in terms of the number of species considered and the number of fleets that are harvesting them.

In this study, we have developed a number of models of multispecies fisheries. The first model was a generic multispecies equilibrium model, which was used to assess the impact of including consumers and the bycatch of iconic species on the definition of maximising net economic returns. The second model, based on the SESSF, was used to examine the impact of different assumptions about optimal fishing mortality and different harvest strategies on the generation of economic benefits as well as discards. The model contained 18 species which were caught across 11 different metiers.

The models, and associated literature reviews, provided several insights into the management of multispecies fisheries.

Defining MEY – role of models and proxy values

The revised Commonwealth Harvest Strategy again identifies maximising the net economic returns from fishing as the main objective of Commonwealth fisheries management. While it recognises that maximising economic profits in the fishery (i.e. the fishery maximum economic yield) does not necessarily equate to maximum net economic returns, for practical purposes, estimating MEY is considered the most appropriate means for determining target reference points. Proxy target reference points of 0.48 B_0 (i.e. 48 per cent of unfished biomass) or 1.2 B_{MSY} for MEY have also been recommended by the harvest strategy policy where bioeconomic models are not available.

Previous work on target proxy reference points in multispecies fisheries (Pascoe et al. 2015) suggests that achieving these targets simultaneously is both impossible, and also undesirable, as individual species target reference points in multispecies fisheries vary substantially. From the model analysis in this study, for a fishery such as the SESSF determining an appropriate target even using a bioeconomic model is complex. While such targets can be identified, the recovery time required to reach these targets is substantial and beyond the likely practical time frame of fishery managers. In the model simulations, the use of proxy target reference points resulted in higher economic benefits in the short to medium term compared with “optimal” longer term target reference points estimated from the bioeconomic model. However, they also resulted in higher levels of discarding and under-caught quota; the former potentially creating non-monetary costs to the broader community and the latter seen as forgone revenue to the industry (even if it is not currently profitable to harvest the under-caught quota).

The model results suggest that the operational definition of MEY needs to be re-defined to take into account the practical time period of the managers (i.e. maximise profits over what time period) and the effects of the current fleet structure and stock structure also has on what can be reasonably achieved from the fishery. This may require MEY to be considered a moving target that is constantly re-assessed, and the fishery gradually guided towards a long run position (which itself will change with changes in costs and prices).[11]

Once we move from a theoretical optimum to a dynamic and constrained system, our definition of what is MEY and the associated target reference points become highly dependent on the assumptions used. For example, the rate at which we trade-off future profits against current profits (i.e. the discount rate), the length of the period that we consider optimising over and any terminal constraints (e.g. minimum biomass at the end of the period) will all affect the definition of the target biomass and optimal fishing mortality.

Beyond maximum profits

The results of the model analyses were largely as expected. Maximising fishery profits when a price-quantity relationship exists results in a transfer of benefits from consumers to producers. Conversely, maximising both producer and consumer surplus results in higher effort, lower fishery profits, and transfer of benefits from industry to consumers. Ignoring price quantity relationships in models used to estimate MEY may result in an outcome that neither optimises benefits to either consumers or producers. Even relatively low price flexibilities, as used in the analysis and representative of those seen in a key Commonwealth multispecies fishery, can have an impact on the outcome depending on how MEY is defined (i.e. maximise fishery profits or maximise benefits to both fishers and consumers). Hence, how policy makers consider this broader definition of MEY is important to the outcome.

Including a cost associated with bycatch results in lower effort, higher profits and transfer from consumers to industry than if bycatch was costless. This is again consistent with a priori expectations based on the simple demand and supply curve analysis and previous studies (Pascoe *et al.* 2010a; Innes *et al.* 2015). While bycatch catch rates may be known for some species and fleets, the appropriate penalty to apply to bycatch (i.e. the non-market value of the externality) is not known. Relatively few studies have attempted to place values on such bycatch species (Lew 2015). Given the sensitivity of the optimal fishing mortality to this value, investment in non-market valuation studies will be required if the definition of net economic returns was also to include the cost of bycatch.

When both bycatch costs and consumer benefits are considered, there was no clear consistent direction of change in the optimal fishing mortality. The optimal outcome then depends on the individual characteristics of the species and also the general characteristics of the fishery.

While other studies have suggested that the definition of MEY also include flow-on effects related to production (Christensen 2010; Wang and Wang 2012a), we have ignored these effects in this study. Although these impacts can be substantial (Pascoe *et al.* 2016), the effect of moving to MEY on these flow-on effects is less obvious. In some fisheries, income-induced effects (i.e. expenditure by crew and owners given the income they receive from fishing) can dominate the flow-on effects (Pascoe *et al.*

[11] Such an approach has been adopted for the Northern Prawn Fishery, where the path to MEY is regularly re-assessed, and the time period to achieve target reference points is also specified.

2016) and a move to MEY may result in higher profits and wages, resulting in a net increase in flow-on effects (Norman-López and Pascoe 2011a).

Reasonable proxy values can be estimated using some of the meta-results developed as part of this study, in particular the regression analysis. Extrapolating these results to real fisheries, however, would benefit through testing in using management strategy evaluation (MSE) first.

Maximising net economic benefits is not the same as maximising fishery profits, particularly when price-quantity relationships exist and the fishery generates other negative externalities that are not captured in fisheries profits. Identifying appropriate adjustments without the aid of a bioeconomic model is difficult. When bycatch and consumer surplus were considered, the optimal fishing mortality may be either greater than or less than that based on fisheries profits alone, depending on the characteristics of the species and the fishery.

Given this, fisheries managers and advisors need to pay attention to market relationships and also consider non-market values of environmental externalities when setting catch targets.

Implementing MEY

The ability of harvest strategies to achieve MEY was also tested in the study. The results suggest that "hockey-stick" style harvest control rules may be less appropriate in multi-species fisheries, particularly where some species are above their target biomass level. The lower catch on these species imposed through the upper limit on fishing mortality can act as a constraint on the catch of other species, resulting in lower profits, increased discarding and higher levels of under-caught quota.

An earlier study (Pascoe et al. 2015) noted that attempting to impose a target reference point on all species may not be necessary (nor feasible) in multispecies fisheries. Instead, imposing a target on the dominant species (in terms of revenue share) for each sub-fleet results in outcomes close to optimal, and reduces conflicts in catches where target reference points are not perfectly aligned. The results of this study also suggest that there are benefits in imposing target reference points on only the major species in the fishery, both in terms of increased industry profitability and reduced discards.

Another complexity in implementing MEY in multispecies fisheries arise as species caught are rarely fixed in proportions and catch composition can change through gear change, changes in seasonal or spatial fishing patterns. Previous studies examining fishers' ability to control output mix in a fishery (Squires 1987; Pascoe *et al.* 2007; Pascoe *et al.* 2010b) found that the ability to target some individual species may be limited, but not impossible. However, some studies of fisher behaviour also suggest that apparent targeting behaviour (or a lack of) may be an artefact of the management schemes, and changing management may change this relationship as fishers respond to the new incentives created (Christensen and Raakjær 2006). For example, Woods *et al*. (2015) found that short-term profit maximising behaviour of fishers can affect the utility of the catch–quota balancing regulations used in the Icelandic ITQ system for the multispecies demersal fishery.

Such fishers' behavioural changes in response to economic, biological and regulatory conditions can be incorporated into a modelling framework, although there is generally a lack of realistic representation of resource users' behaviour in most existing models (Bunnefeld *et al.* 2011; Fulton *et al.* 2011b). In this study, fishers were assumed to respond to short term economic incentives through adjusting effort levels in the different metiers given the available quota mix. Constraints on the rate of adjustment were assumed (no more than a 20 per cent increase or decrease each year), which would also have affected the outcome. Studies on fleet behaviour in multi-metier fisheries are limited. Research into how effort adjusts internally in a fishery in response to changing economic conditions is an area of future importance. If the results from this study are to be "operationalised" in the actual SESSF fishery; then due consideration (with extensive stakeholder input) must be given to future potential realistic changes in effort per metier (which will meet targets and expectations) in constantly adaptive framework given the ever changing biological and economic environment.

Conclusions

The project had two key objectives:

1 Development of a methodology for maximising net economic return to a multispecies fishery as a whole, and with regard to by-catch and discard species;

2 Development of a framework to operationalise the methodology into fisheries management objectives.

With regards to the first objective, the results of the study have indicated that maximising net economic benefits is not the same as maximising fishery profits, particularly when price-quantity relationships exist and the fishery generates other negative externalities that are not captured in fisheries profits. Identifying appropriate adjustments without the aid of a bioeconomic model is difficult. When bycatch and consumer surplus were considered, the optimal fishing mortality may be either greater than or less than that based on fisheries profits alone, depending on the characteristics of the species and the fishery.

Given this, fisheries managers and advisors need to pay attention to market relationships and also consider non-market values of environmental externalities when setting catch targets. From the model results, the effects of these factors on the definition of MEY will vary, being dependent on the relative size of the influence of consumer surplus and discard externalities on the optimal outcome. These factors can be readily incorporated into a bioeconomic model of the fishery to assess their impacts, as shown in the model analysis. For the majority of fisheries without a bioeconomic model, however, managers and scientists will need to apply value judgements as to how these factors influence the fishery targets.

Operationalising any set of targets in a multispecies problem (objective two) was also complex. While some trends in outcomes were observed from the multiple modelling exercises undertaken, developing a framework that was more than a set of principles was not feasible. However, these principles can provide guidance as to how best to manage these fisheries in the future:

1. For practical purposes, there is no such thing as a clear-cut definitive estimate of "MEY" in a multispecies, multi-fleet fishery that can be used to guide management. While an equilibrium set of biomass, catch and fleet values can be identified that is "optimal" in the long run, this may not be achievable in practical terms. In the case study model used, achieving this state required over 100 years – well beyond the practical time frame of any management administration. Over a more realistic time frame, the "optimal" trajectory resulted in a lower NPV of profits than that obtained using simple proxy measures of optimal fishing mortality.
2. Even if an optimal fishing mortality trajectory can be determined, achieving this in the case of a multi-metier fishery is also difficult. Fishers respond to the prevailing set of incentives that they face. With random variability in stocks, and also divergence between current and "optimal" stock combinations in the fishery, fishing effort may move in directions counter to achieving the longer term goals.

3. Translating fishing mortality targets to actual fishing mortality rates is also complicated in multi-metier fisheries with differing combinations of catchability. Even if catch quotas can be set consistent with the targets, the differences in catchability of the different sub-fleets will result in discarding and under-caught quota, which will further influence the dynamics of the stocks away from the optimal path.
4. In quota fisheries, a well-functioning quota market is essential. The simulations with limited quota trading resulted in substantially higher levels of discarding and under-caught quota as well as lower profits than the simulations with “perfect” transferability. However, even with a perfect quota market, some discarding and under-caught quota was observed.

The ability of a management plan to generate economic benefits is also dependent on the management system used. The study considered a range of harvest control rules, as well as other potential management options. Again, some general trends were observed:

1. In multi-species fisheries, “hockey-stick” harvest control rules may overly restrict the catch of (mostly secondary) species that are currently above their target biomass. Given their higher abundance, catch of these species is likely to result in increased discarding and lower economic returns than might otherwise be achieved. An alternative harvest control rule that allowed higher than “optimal” fishing mortality rates for species that were above their target biomass resulted in less discarding and higher economic returns.
2. Having quota on too many species may be counterproductive. Reducing the number of species subject to quota constraints to only those that were most important (in terms of revenue) resulted in improved economic performance of the fishery as well as lower levels of discarding. However, in the model changes in targeting ability of the fleet was not considered, so monitoring of fisher behaviour in response to few quota species would be essential to ensure there were no longer term consequences for biomass.
3. The MEY proxy values of the fishing mortality rate, being generally higher than those that were “optimal” in the longer term, can result in improved economic benefits if used in a harvest strategy in the short- to medium-term. However, as these are not developed taking into account the technical interactions within the fishery, they also result in substantially higher levels of discarding and under-caught quota.
4. Alternative management systems such as the use of royalties can eliminate problems of discarding and under-caught quota as well as achieve overall high economic benefits. However, these benefits accrue largely to the receiver of the royalty, with the fishing industry incurring costs for several years before stock recoveries occur sufficiently to return them to a normal level of profits.

The focus of the study had largely been to identify appropriate target fishing mortality rates and how they may be used to achieve MEY in multi-species fisheries. A confounding issue that emerged in most modelled scenarios was the way in which the fleet adjusted in response to the incentives generated. While fleet adjustment took place in each scenario, the degree to which the fleet moved to the optimal long run configuration varied.

The model analyses also did not consider changes in technology, prices or costs, nor longer term potential environmental influences on stocks nor short term natural fluctuations in stock abundance. Changes in economic and stock conditions (which in some cases could also result from the previous response of fishers to the previous targets) requires the optimal trajectory to be constantly re-assessed based on the prevailing stock and fleet conditions. Such regular re-assessment is currently undertaken in the Northern Prawn Fishery. Wilson (1982) noted that, in effect, the long-term management problem in complex, highly variable multiple-species fisheries is not the attainment of MEY, but rather the adaptation of the fishery as completely as possible to its natural, uncontrollable variations in relative species abundance.

Dissemination Activities

Presentations

National conferences

AARES 2016:

- Trevor Hutton, Sean Pascoe, James Innes, Satoshi Yamazaki and Tom Kompas. **Maximising net economic returns in mixed fisheries: how many species do we need to control?** The 60th Annual AARES Conference, 2nd – 5th February 2016, Hyatt Hotel, Canberra. Australian Agricultural and Resource Economics Society.

AARES 2017:

- Australian Agricultural and Resource Economics Society Conference. 61st Annual Conference, Brisbane Convention and Entertainment Centre, Brisbane, Queensland, 7-10 February 2017
- Mini-symposium: Maximising net economic returns from Australian Fisheries (Sean Pascoe and Trevor Hutton, CSIRO): with papers on:
  1. Current overview of the revision of the Commonwealth harvest strategy policy
  2. Review of definitions of MEY and net economic returns
  3. Modelling analysis including cost of bycatch of iconic species and consumer surplus in "MEY"
  4. Modelling analysis of potential harvest strategies in an example multispecies fishery

AARES 2018:

- *Estimating Multi-species Maximum Economic Yield in a Fishery: evaluating various practical considerations for operationalisation*

International conferences

IIFET 2016 (Abeerdeen, Scotland):

- Sean Pascoe, Trevor Hutton and Eriko Hoshino. **Offsetting externalities in estimating MEY in multispecies fisheries.** International Institute of Fisheries Economics and Trade (IIFET) Conference 2016. Aberdeen, Scotland. Monday 11 - Friday 15 July 2016.

IIFET 2018 (Seattle, Washington, USA):

- **In pursuit of achieving MEY in multi species, multi metier fisheries.** Accepted presentation based on this study.

Other

AFMA/CSIRO/ABARES Science workshop, Hobart, 16-18 March, 2016

- **Maximising net economic returns in mixed fisheries**

Papers published

Pascoe, S., Hutton, T. and Hoshino, E. (2018). Offsetting Externalities in Estimating MEY in Multispecies Fisheries, *Ecological Economics* 146, 304-311.

Hoshino, E., Pascoe, S., Hutton, T., Kompas, T. and Yamazaki, S. (2017). Estimating maximum economic yield in multispecies fisheries: a review, *Reviews in Fish Biology and Fisheries,* https://doi.org/10.1007/s11160-017-9508-8.

Appendix A: Catchability coefficient matrix

Species		Metier										
		T_In_NSW	T_In_EBS	T_Off_NSW	T_Off_EBS	DS_EBS	DS_BS	BGren	GillE	GillW	Hook	T_Tas
Blue Warehou	TRT	0.0222	0.1475	0.0011	0.4561	0.0227	0.0241	0.0134	0.0013	0.0008	0.0003	0.0336
Flathead	FLT	6.0163	7.7422	1.7979	1.3332	6.4304	1.1155	0.2092	0.0010	0.0011	0.0156	2.3304
Gemfish	GEM	0.0121	0.0168	0.3416	0.2491	0.0009	0.0003	0.1259	0.0000	0.0000	0.1395	0.4838
John Dory	DOJ	3.9811	6.1456	0.0972	0.1686	0.8603	0.2052	0.0033	0.0066	0.0009	0.0001	0.1526
Ling_East	LIG_E	0.1774	0.6238	4.0473	13.4781	0.0219	0.0018	12.5372	0.0007	0.0000	5.8622	0.0000
Ling_West	LIG_W	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0004	5.9450	0.0000
Mirror Dory	DOM	0.1238	0.1899	6.8106	2.0895	0.0141	0.0000	0.4323	0.0000	0.0000	0.0000	1.1500
Morwong East	MOW_E	0.2456	0.7457	0.0104	0.1506	0.0405	0.0000	0.0114	0.0007	0.0000	0.2893	0.4325
Morwong West	MOW_W	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0281	0.0000	0.1280
Ocean Perch	REG	0.1342	0.0818	14.4952	6.7488	0.0003	0.0000	0.8068	0.0004	0.0000	0.9133	29.9648
Ribaldo	RBD	0.1073	0.0170	0.0081	13.7916	2.3214	53.6301	34.1717	0.0000	0.0000	23.5717	2.7705
Silver Trevally	TRE	3.5392	0.0019	0.0024	0.0050	0.0142	0.4213	0.0000	0.0000	0.0000	0.0015	0.0052
Silver Warehou	TRS	0.0324	0.0335	0.1215	1.0592	0.0027	0.0565	3.8125	0.0000	0.0000	0.0002	1.5471
Whiting	WHS	1.2635	0.0899	0.0000	0.0006	4.4529	33.8165	0.0000	0.0000	0.0000	0.0000	0.0071
Blue Grenadier	GRE	0.0004	0.0080	0.0780	2.6528	0.0000	0.0000	13.7344	0.0000	0.0000	0.0186	1.4620
Gummy Shark	SHG	0.0880	0.5145	0.0786	0.2841	0.1690	0.0926	0.2488	14.6471	15.5385	4.7579	0.0000
School Shark	SHS	0.0060	0.0129	0.0040	0.0242	0.0098	0.0000	1.6961	1.3286	4.3099	1.8698	0.0000
Blue-Eye Trevalla	TBE	0.0554	0.0764	0.0212	0.5513	0.0013	0.0032	0.1053	0.0000	0.0000	42.4538	0.0000

The catchability coefficients were estimated from observed catch and effort data for the base year, and the estimated biomass for each species in the base year (i.e. $q_{i,m} = C_{i,m} \Big/ \left(E_m \hat{B}_i \right)$ where $\hat{B}_i$ is the estimated biomass of the species/stock in the base year. The relationship between these catchability coefficients and metier are also depicted in Figure 35.

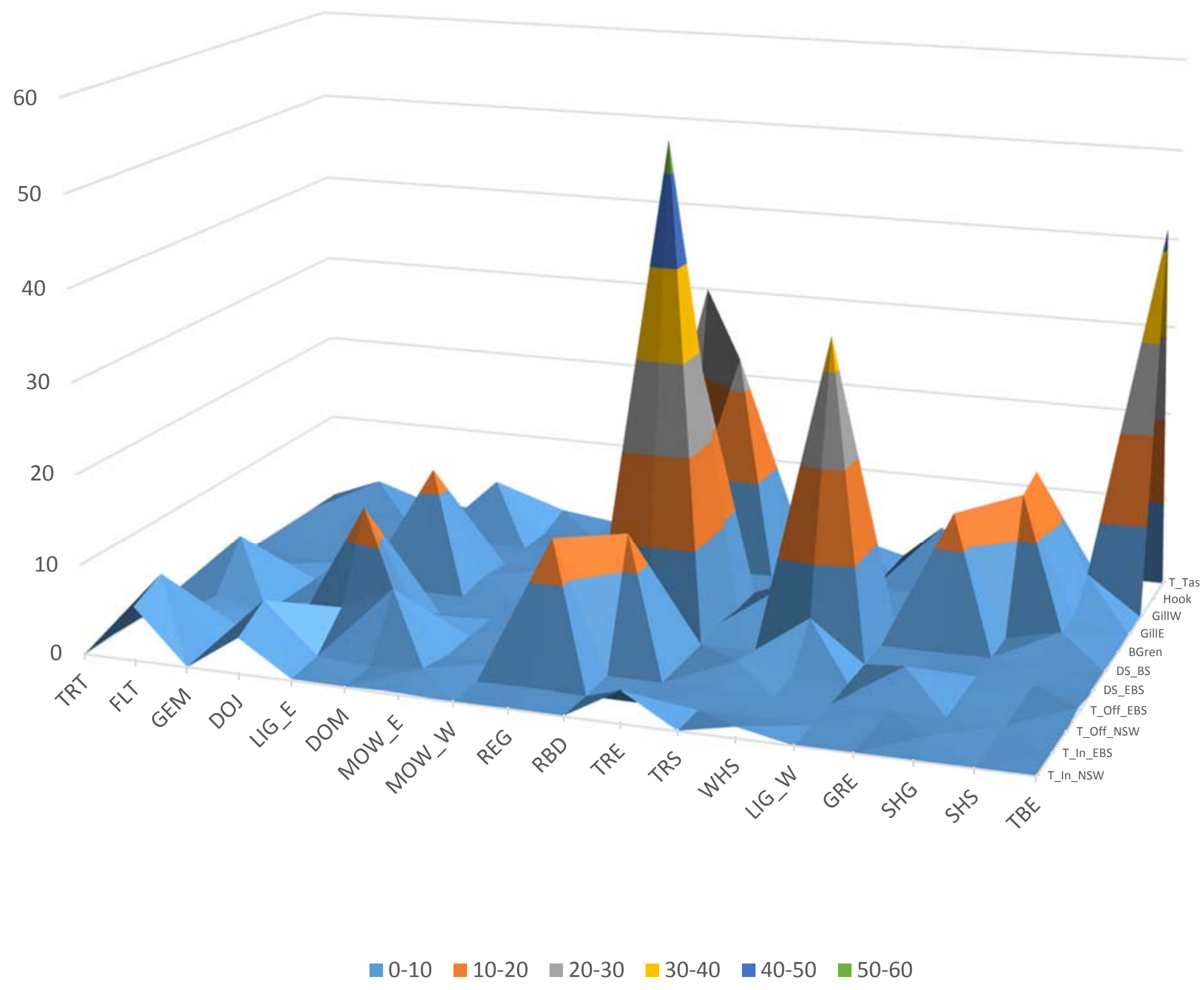
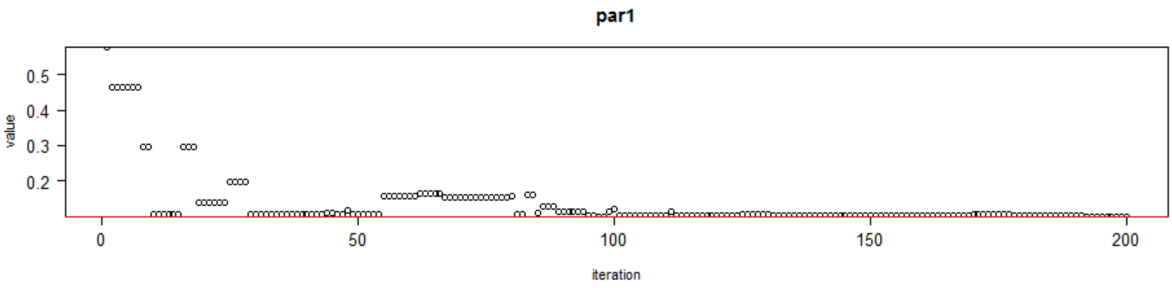
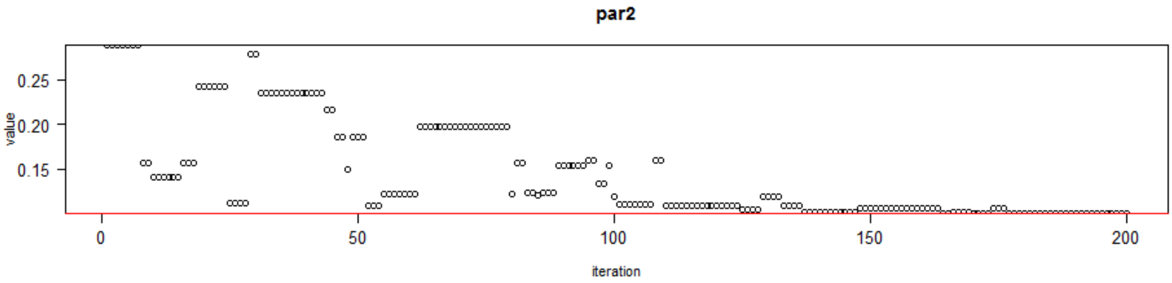


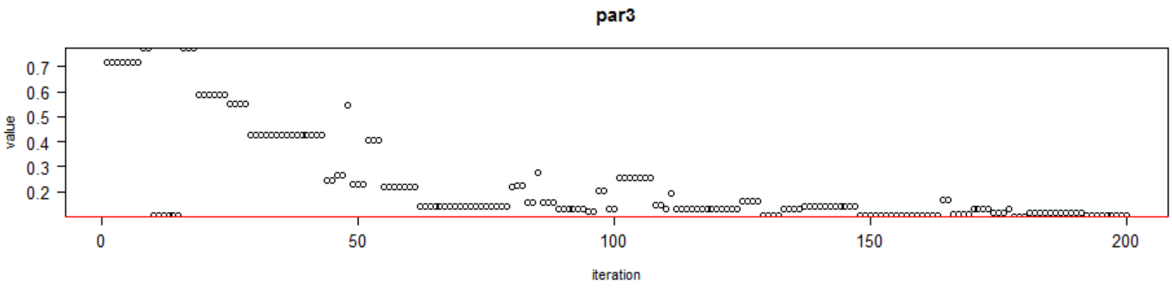
Figure 35. Catchability by métier

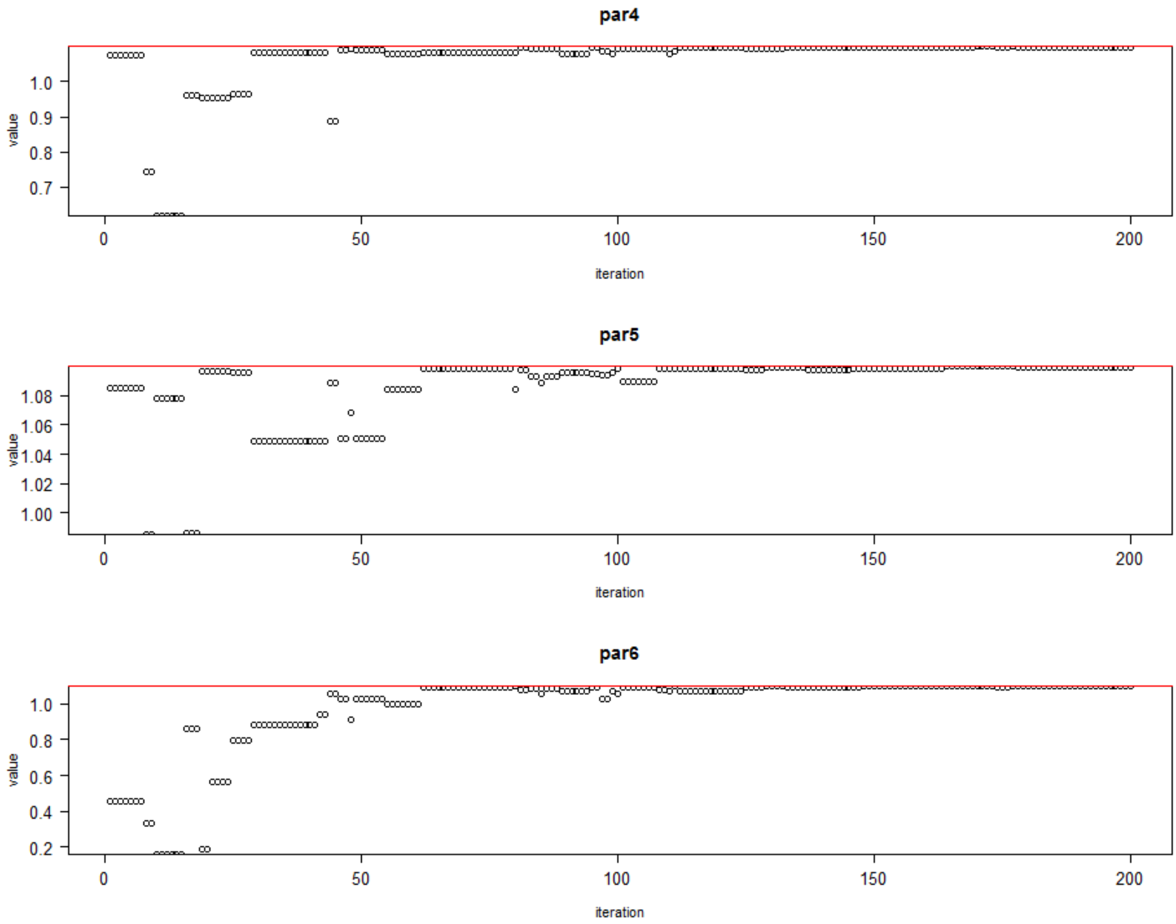
Appendix B: Example of convergence in the genetic algorithm

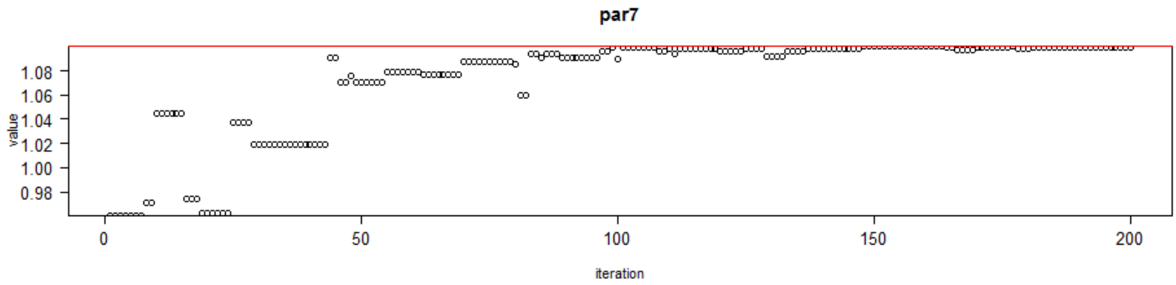
The evolution of fishing effort in each metier (described as par1 to par11 in the following graphs) can be seen below. The graphs show the change in fishing effort over one of the dynamic optimisation runs. This change was bounded from zero to 1.2, so that effort could be fully removed from a metier, but could only increase by 20% at most. This was to reflect the assumptions that effort changes would largely result from vessels leaving the fishery, with relatively new investment in new vessels.

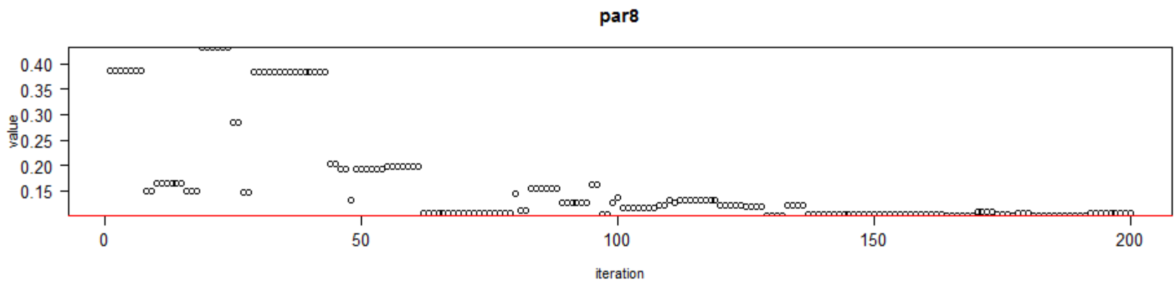


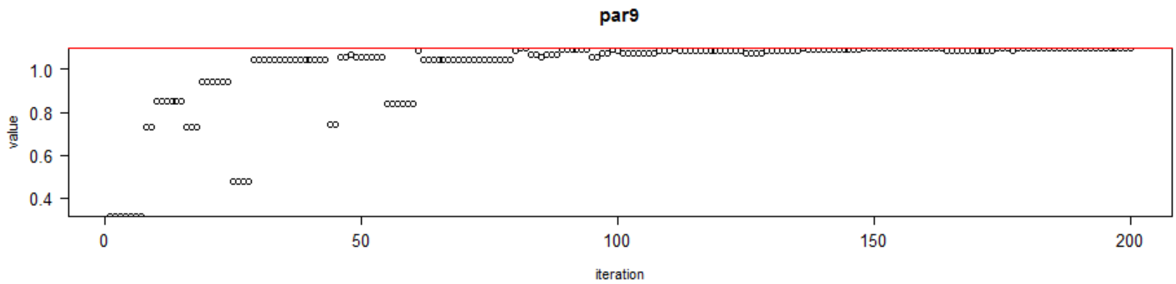


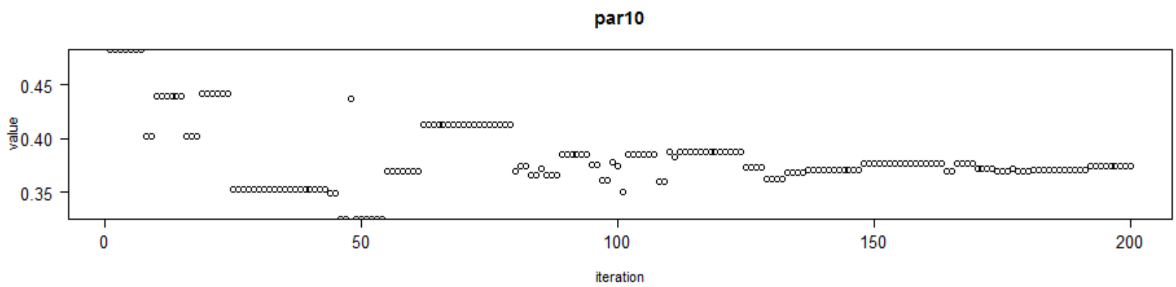


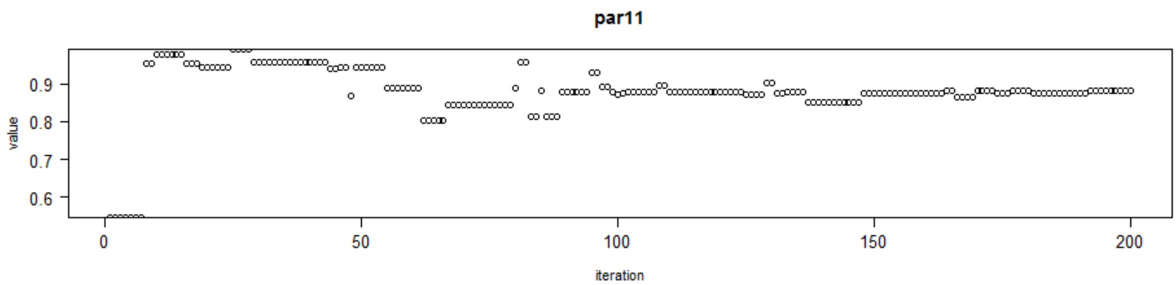

par4
value
1.0
0.9
0.8
0.7
0
50
100
150
200
iteration
par5
value
1.08
1.06
1.04
1.02
1.00
0
50
100
150
200
iteration
par6
value
1.0
0.8
0.6
0.4
0.2
0
50
100
150
200
iteration



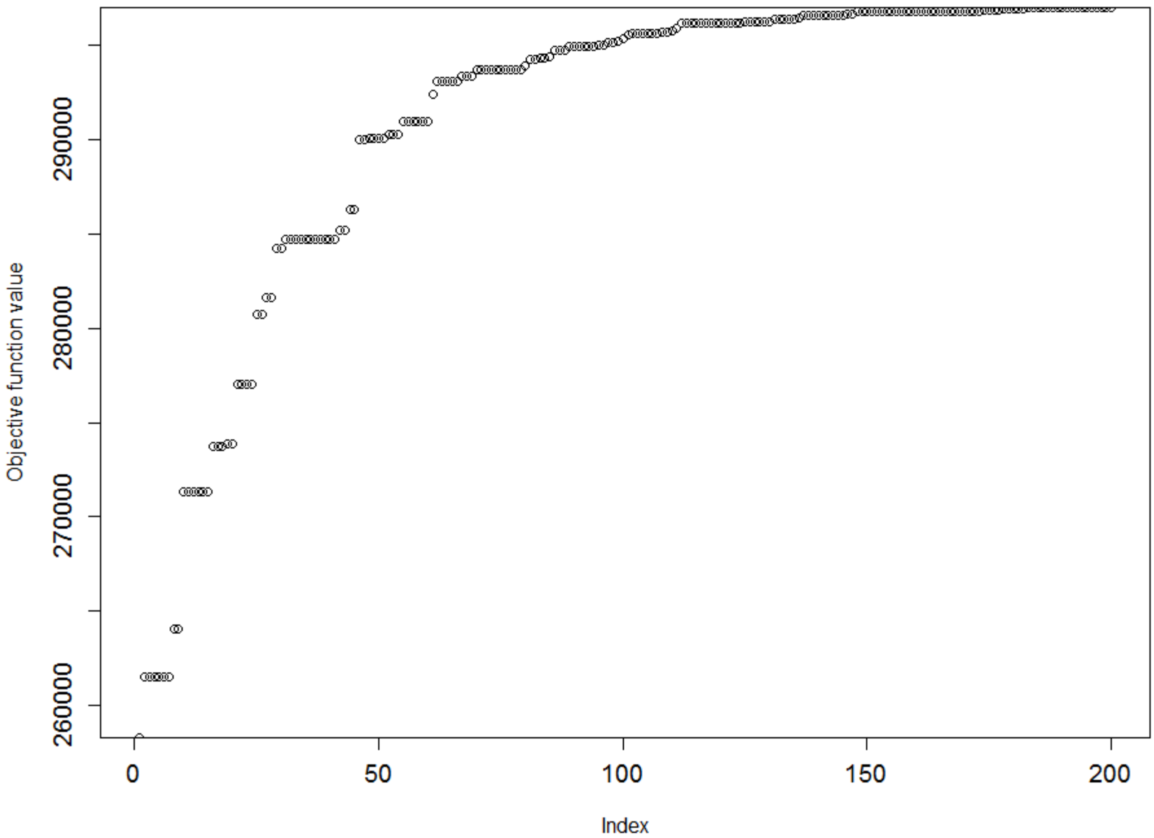








The corresponding effect of the effort changes after each generation (iteration) on the objective function (the net present value of fishery profits) can be seen in the following graph. Unlike traditional optimisation models, a genetic algorithm will continue until a specified number of generations have been achieved. While it is possible to include a “convergence” condition, where the algorithm stops if no improvement is made, this does not necessarily result in a global optimum being achieved. For example, in the below graph, there was no improvement in the objective function between generations 40 to 50, at which point a “convergence” condition may have stopped the estimation procedure continuing. However, after generation 50, substantial improvements in the objective function were found. Very little improvement was observed after generation 150, and a decision was made to limit the number of generations to 200.



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