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# Optimising harvest strategies over multiple objectives and stakeholder preferences

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## Abstract

Natural resource management has long recognised that the multi-objective nature of management is important, but has struggled to operationalise this into quantitative, measurable objectives for functional use in management. Operationalising broader ecological and social objectives has been particularly problematic. In fisheries management, the focus has mainly been on target species sustainability and, in the past few decades, on profitability. However, multi-objective management is now essential as fisheries have become recognised as complex social-ecological-systems.

Policy and legislation demand a move towards quantitative approaches for reconciling multiple objectives and operationalising these within harvest strategies. We present a quantitative, non-commensurable-unit approach, via a multi-indicator value function with explicit objective preference weights. We use a simulation to set Total Allowable Catches (TACs) for three main species groups in a reef line fishery in Australia's Great Barrier Reef. Our method enables stakeholders to consider a richer range of tradeoffs than is possible with bio-economic models. Moreover, it allows the formal evaluation of performance across alternative stakeholder group preferences, providing an impartial way to obtain an overall optimum TAC. The simulation requires extensive fishery data and requires the performance indicators associated with each objective to be quantitatively and defensibly defined. Thus, our approach provides a pathway forward that forces managers and stakeholders to confront the associated data requirements.

## Keywords

Triple bottom line; harvest strategy development; reef line fishery; stakeholder preference weightings; multi-objective trade-offs; simulation

# 1 Introduction

Maintaining healthy ecosystems and healthy human communities that depend on them is increasingly recognised as important to natural resource management, including fisheries (Asche et al., 2018; Berkes, 2000; Charles, 1995; De Young, 2008; FAO, 2009; Marshall et al., 2017; Voss et al., 2014). Elkington (1998) conceived the Triple Bottom Line (TBL) – encompassing economic, ecological and social objectives – as a tool for influencing a single decision maker to explicitly value non-financial objectives by optimising over the three different objectives. Halpern et al. (2013) note that maximising conservation goals and achieving equity in social outcomes, while minimising overall costs, is the ideal TBL outcome. In a fisheries context, Stephenson et al. (2017) proposed four "pillars of sustainability" that includes institutional aspects in addition to economical, ecological and social "pillars". Pascoe et al. (2013b) also considered institutional or managerial objectives of "simplifying and improving management structures".

In fisheries, several jurisdictions have legislated the consideration of multiple objectives. For example, the United States Magnuson-Stevens Fishery Conservation and Management Act (1996) mandates consideration of economic and social outcomes in addition to environmental outcomes in National Standard 8. In Australia, the Fisheries Management Act 1991 requires the effective integration of long-term and short-term economic, environmental, social and equity considerations into policy development for Commonwealth-managed fisheries (Department of Agriculture and Water Resources, 2018), while the Productivity Commission Inquiry into Marine Fisheries and Aquaculture also reinforced the need to include social, economic and environmental considerations into fisheries policy and management (Productivity Commission, 2016).

Concurrent with the recognition of the need to include multiple objectives into fisheries management has been the increased development and adoption of harvest strategies to assist in management decision making. Harvest strategies comprise pre-agreed monitoring and performance indicators (usually obtained from a stock assessment), and decision or harvest control rules invoked in response to the assessment, that are collectively used to control fishing mortality on the target species (Butterworth and Punt, 2003; Punt et al., 2002; Sainsbury et al., 2000). In fisheries management, harvest strategies are used for tactical fisheries management to set control variables such as the Total Allowable Catch (TAC) or limit recreational catch through daily bag limits per person (Garcia et al., 2003). Concomitant with the development of harvest strategies has been the development of quantitative tools to assess potential harvest strategies. In particular, Management Strategy Evaluation (MSE) has developed as a formalised approach to pre-test different harvest strategies via simulation before their implementation (Punt et al., 2016; Smith, 1994; Smith et al., 1999).

Although the recognition of the importance of consideration of TBL (and in some cases the extended fourth pillar relating to governance) outcomes in fisheries management has occurred concurrently with the recognised benefits of the use of harvest strategies to aid management decision making, the implementation of TBL has not been operationalised within fishery harvest strategies (Mangel and Dowling, 2016) nor MSE. Indeed, Elkington (2018) sought to recall and rethink the TBL concept, stating that it has "failed to bury the single bottom line [economic] paradigm".

In this paper, we present a quantitative, non-commensurable-unit approach, via a multi-indicator objective function to set TACs for three main species groups in the Queensland reef line fishery on Australia's Great Barrier Reef. The fishery is complex in that it i) comprises several sectors with disparate motivations, including commercial, charter and recreation; ii) targets multiple important reef species; and iii) is undertaken in a World Heritage Area facing significant pressures ranging in scale from local to global (Great Barrier Reef Marine Park Authority, 2019). The Queensland Government's Sustainable Fisheries Strategy 2017–2027 states that TBL objectives should be considered in the development of harvest strategies for all major fisheries that fall within their jurisdiction (State of Queensland, 2017). We use simulation with explicit objective preference weights. We focus the requisite methodology for explicitly incorporating all objectives as quantifiable and comparable through the development of a scaled performance indicator for each objective.

Our approach is consistent with the "efficiency frontier" (Halpern et al., 2013), which is a curve or surface on which optimal solutions lie, different solutions representing different weights given to conservation versus equity goals. We consider the objective weighting profile for different stakeholder groups as part of an integrated value function that is optimised across a suite of catch levels (cf. Rindorf et al. (2017) who progressively refine a suite of fishing mortalities corresponding to sustainable yield). Moreover, our approach provides a means to reconcile alternate stakeholder objective preferences. That is, we present a formal way by which to trade off the objectives across the various sets of weightings, where these show a lack of agreement among stakeholders. This demonstrates a rational approach to "mutually disagreeing".

# 2 Background

## 2.1 Incorporating multiple objectives into fisheries management decision making

To date, consideration of the TBL and governance objectives has been largely limited to conceptual treatment (Stephenson et al., 2017) or intuitive forecasting methods using expert opinion (Bernstein and Cetron, 1969; Dichmont et al., 2012; Dichmont et al., 2014; Pascoe et al., 2019). For example, Pascoe et al. (2009) presented a qualitative framework that aids in the analysis of alternative spatial management options in coastal fisheries. The framework combined expert opinion and the Analytic Hierarchy Process (Saaty, 1980) to determine which options performed best, taking into account the multiple objectives inherent in fisheries management. Read and West (2010) used a qualitative Ecological Risk Assessment to assess the effectiveness of managed-use zones in six multiple-use marine parks located in New South Wales. Dichmont et al., (2012, 2016) employed an expert group to qualitatively develop different governance "strawmen" (or management strategies). These were assessed by a group of industry stakeholders and experts using multi-criteria decision analysis techniques against the different objectives; one strawman clearly provided the best overall set of outcomes given the multiple objectives.

Development of quantitative models, such as those underlying "standard" MSE, to assess multi-objective outcomes of harvest strategies has been complicated by the abstract nature of some of the objectives, particularly social objectives. A major problem is that arbitrary increases or decreases in catch or effort have often become a proxy for socio-economic considerations (Mangel and Dowling, 2016). Dichmont et al. (2010) illustrate that this is a fraught assumption. While maximum economic yield (MEY) has been identified as a primary management objective for Australian fisheries, first attempts at estimating MEY as an actual management target for an actual fishery (rather than a conceptual or theoretical exercise) highlighted some substantial complexities generally unconsidered by theoretical fisheries economists. Using a bioeconomic model of an Australian fishery for which MEY is the management target, Dichmont et al. (2010) showed that unconstrained optimisation may result in effort trajectories that would not be acceptable to industry or managers. For example, while in theory it may be economically optimal to reduce fishing effort in the short term, most bio-economic models did not account for the costs associated with effort reduction or fishery closure, nor may it be possible for fishers to survive a short-term period of negative profits, because vessels still need to cover their fixed costs (see Mangel (2006) pg 218 for a simple example). Additionally, in the case of recreational fishing, economic value extends to non-catch aspects (such as catch rates, available fishing days, and season length), as well as the trade-offs between attributes that are trip-based and those that measure opportunity over a season (Young et al., 2019). Clearly, catch and effort are not socio-economic proxies, so that both short-and long-term social objectives need to be considered explicitly within any formal evaluation framework that is used to operationalise the TBL.

Benson and Stephenson (2018) reviewed TBL methods and found that two of seven proposed tools to support decision making in the management system could provide tactical advice, but only Management Strategy Evaluation (MSE) provided advice that was consistent with their criteria for generation, transmission, and use of scientific information in management advisory processes. Even MSE (e.g., Plagányi et al., (2012, 2013)) is conditioned on how TBL objectives are weighted, and there is no means to formally make recommendations that reconcile different interest groups.

Stephenson et al. (2017) identified three key impediments to embracing TBL and governance objectives in a full quantitative analysis: the lack of explicit social, economic and institutional objectives; the lack of a process for routine integration of all four pillars of sustainability; and a bias towards biological considerations. Incorporating social relationships, together with economic and ecological sustainability objectives into models to provide management advice is challenging, particularly when this advice requires complex trade-offs between objectives (Pascoe and Dichmont, 2017). The process is further complicated by differences in quality and quantity of data across fisheries and difficulties in quantifying social objectives and outcomes.

Quantitative attempts to address the TBL have been made using bioeconomic modelling, but social objectives have generally been downplayed, and the treatment has largely been theoretical as opposed to operational (Pascoe et al., 2017). Plagányi et al. (2012) and Plagányi et al. (2013) used a suite of integrated models to capture multiple objectives, aimed at assessing TBL outcomes of different allocations between islander and non-islander fishers of the Torres Strait Rock Lobster Fishery, as well as different management strategy outcomes. These included a Bayesian Network model to assess how the islander sector might respond to different management strategies and allocations (van Putten et al., 2013), and a model of non-islander fleet adjustment under different quota allocations (Pascoe et al., 2013a). The economic implications of the fleets' effort levels were assessed using a bioeconomic model (Plagányi et al., 2012).

Where social objectives have been explicitly included in quantitative models, these have often been limited to metrics that can be readily linked to catch or effort levels, such as employment. For example, multi-objective goal programming models included economic (profits), social (employment) and environmental (stocks size, discards etc) objectives as specific targets, and estimate the fleet structure and catches required to optimise the fishery performance across these objectives given different objective weights (e.g. Charles, 1989; Mardle et al., 2000; Pascoe and Mardle, 2001). More recently, bioeconomic models based on co-viability analysis have been developed to assess management strategies that achieve at least minimum levels of outcome under each TBL objective (e.g. Gourguet et al., 2016).

More commonly, bioeconomic models have been applied to address just the economic and environmental TBL pillars. Zimmermann and Yamazaki (2017) modelled a multi-stock fishery to study how biological and economic management objectives were affected by stock interactions. Punt et al. (2010) modelled the Australian Northern Prawn Fishery, focusing on MEY and the level of effort in each of two fishing strategies to maximise the net present value of fishery profits. Gaichas et al. (2017) used a length-structured, multispecies, multi-fleet model to illustrate trade-offs between objectives of yield, biomass, species diversity and revenue, under changing environmental conditions. Guillen et al. (2013) estimated MSY and MEY in multi-species and multi-fleet fisheries, and analysed the resulting impacts on the optimal effort allocation between fleets that had different economic structures. Griffin and Woodward (2011) analysed a wide range of recreational management strategies and their impacts on red snapper yield, economic surplus and fish stock. Dichmont et al. (2013) used an MSE that included a bio-economic and ecosystem model to evaluated marine spatial closures with conflicting fisheries and conservation objectives.

Pascoe et al. (2013b) showed the importance of stakeholder preferences in TBL management by assessing the relative importance of the different objectives to different stakeholder groups in the Queensland East Coast Otter Trawl Fishery, Australia. Across stakeholder interest groups, preference weightings showed a 4-fold difference in economic outcomes, 2-fold difference in social outcomes, and almost 2-fold difference in environmental outcomes. This motivates the need to reconcile weightings, and TBL harvest strategies, across interest groups.

To be sure, operationalising the triple bottom line, beyond a simple conceptualisation is complex. Embedding the TBL in formal management requires each of the TBL objectives needs to be operational (quantifiable) as a performance indicator, and objectives need to be weighted according to individual preferences, which will naturally vary across the fishery's stakeholders. Objectives need to be evaluated in the context of a formal harvest strategy, and preference weightings need to be reconciled among and between stakeholder groups. Finally, for quantitative evaluations, operational objectives need to be direct or indirect functions of the management mechanism used within the harvest strategy.

Despite these challenges, legislative mandates require TACs to be set based on TBL objectives and their associated performance indicators. The challenges need to be met in a quantitative manner. The question remains as to how to optimise a TBL value function, given a set of weightings, across a range of scenarios and a range of stakeholder interest groups. Richerson et al. (2010) showed that, by using relative quantities, triple bottom line performance metrics that were otherwise incompatible could be made commensurate. Mangel and Dowling (2016) demonstrated a more fundamental way of interpreting weightings for various stakeholder groups, in the form of a single TBL value function. Our simulation approach builds on and extends this previous work.

## 2.2 Case study fishery: the Queensland Coral Reef Finfish Fishery

The Queensland Coral Reef Finfish Fishery ranges from Cape York (10˚41'S) in the north, to Bundaberg (24˚30'S) in the south, operating mostly within the Great Barrier Reef Marine Park. The commercial sector mainly targets several species of coral trout (*Plectropomus* and *Variola spp.*, CT), of which *P. leopardus* is predominantly landed as live fish and exported to Asia; red-throat emperor (*Lethrinus miniatus*, RTE); and over 100 other reef-associated fish species (OS) including groupers (mainly *Serranidae*), emperors (*Lethrinidae*) and tropical snappers (mainly *Lutjanidae*), landed as dead whole fish (Thébaud et al., 2014). In addition, there is a large, valuable and iconic recreational fishery, a regional charter fishery, and a small indigenous fishery.

Commercial operators use hand-held lines with baited hooks, with vessels ranging from single, small vessels that take short (12–48 hour) trips, to small fishing dories (tender boats) operating from larger mother vessels that undertake trips of up to three weeks. Commercial fishers employ various targeting strategies: some boats are fully dedicated to live CT capture, while others actively target a broader range of species. The commercial fishery is subject to a range of input and output controls, including limited entry, a commercial total allowable catch, allocated via individual transferable quota (ITQ) units, tradability of input and output entitlements, and seasonal spawning closures. The recreational and charter fishery is controlled through control of inputs such as daily limits per species group per fisher, and seasonal spawning closures. Within the Great Barrier Reef Marine Park there are also no-take areas that apply to this fishery.

The fishery has a Working Group consisting of stakeholders from the commercial, recreational and charter sectors, a conservation sector representative, fisheries and marine park managers, and scientists. The Working Group provides advice to Fisheries Queensland on the operational aspects of the management of the fishery, including the development of a harvest strategy for the fishery.

# 3 Methods

## 3.1 Objectives, performance indicators and preference weightings

Previous studies of fisheries management objectives in Australian fisheries (Brooks et al., 2015; Farmery et al., 2019; Jennings et al., 2016; Pascoe et al., 2014; Pascoe et al., 2013b) identified 75 different potential objectives, each of which fell in one of the following categories: ecological/environmental, economic, social and institutional/management. With these as a starting point, a series of workshops held with members of the Working Group (approximately 20 different individuals were involved in the discussions) allowed us to iteratively identified 22 objectives of most relevance to the fishery (Table 1). One objective (4.2.2) was considered to be outside of the mandate of fisheries managers and therefore the control of a harvest strategy. As a result, only the remaining 21 objectives were considered in the simulation.

We translated each conceptual objective into an operational objective. To be operational, an objective had to be measurable and simulation-achievable, with quantitative performance indicators against which it could be assessed (Table 1, Table SI 1).

| **Overarching objective** | **Sub-objectives** | **Specific objectives** |
|---|---|---|
| 1. Ensure ecological sustainability | 1.1. Ensure resource biomass sustainability | 1.1.1 As per the Queensland Sustainable Fisheries Strategy, Policy achieve $B_{MEY}$ (biomass at maximum economic yield) (~60% unfished biomass), or defensible proxy, by 2027 (if below biomass at maximum sustainable yield, $B_{MSY}$, aim to achieve $B_{MSY}$ (~40-50% $B_0$) by 2020), for the main commercial, charter and recreational species (coral trout, RTE and key other species yet to be identified) |
| | | 1.1.2 Minimise risk to Other Species (that are harvested, per the “Other Species” list) in the fishery which are not included in 1.1.1. above |
| | 1.2 Ensure ecosystem resilience | 1.2.1 Minimise risk to bycatch species |
| | | 1.2.2 Minimise discard mortality (of undersized target species, or from high-grading of target species) |

| | | |
|---|---|---|
| | | 1.2.3 Minimise broader ecological risks |
| | | 1.2.4 Minimise risk to threatened, endangered and protected species (TEPS) |
| | 1.3. Minimise risk of localised depletion | 1.3.1. Due to fishing |
| | | 1.3.2. In response to environmental event (e.g. cyclone, climate change) |
| 2. Enhance fishery economic performance | 2.1 Maximise commercial economic benefits, as combined totals for each of the following sectors | 2.1.1 Commercial fishing industry profits |
| | | 2.1.2 Charter sector profits |
| | | 2.1.3 Indigenous commercial benefits |
| | 2.2. Maximise value of recreational fishers and charter experience (direct to participant) | |
| | 2.3 Maximise flow-on economic benefits to local communities (from all sectors) | |
| | 2.4 Minimise short term (inter-annual) economic risk | |
| | 2.5 Minimise costs of management associated with the harvest strategy: monitoring, undertaking assessments, adjusting management controls | |
| 3. Enhance management performance | 3.1 Maximise willingness to comply with the harvest strategy | |
| 4. Maximise social outcomes | 4.1 Maximise equity between recreational, charter, indigenous and commercial fishing | 4.1.1 Increase equitable access to the resource |
| | 4.2 Improve social perceptions of the fishery (social licence to operate) (rec, commercial, charter, indigenous) | 4.2.1. Through sound fishing practices, minimise adverse public perception around discard mortality (compliance with size limits, environmental sustainability, and waste) |
| | | 4.2.2. Maximise utilisation of the retained catch of target species |
| | | 4.2.3 Through achievement of objectives 1.1 and 2.3, maximise the potential for fishing to be perceived as a positive activity with benefits to the community (commercial, recreational, and charter) |
| | 4.3 Enhance the net social value to the local community from use of the resource | 4.3.1 Increase access to local seafood (all species) |
| | | 4.3.2 Maximise spatial equity between regions or local communities |

**Table 1:** Summary of the 22 fishery objectives identified by the Working Group.

We used a modified version of the Analytical Hierarchical Process (Pascoe et al. 2019) through an online survey of 110 fishery stakeholders to elicit preference weights. The approach used comparisons of each set of objectives at each level of the hierarchy (i.e. the overarching objectives, sub-objectives and specific objectives in Table 1) and produced relative weights by stakeholder group at each level. Pascoe et al. (2019) fully describe the approach taken to weight the objectives and details of the resultant weights associated with each of the objectives.

## 3.2 Simulation model

To more quantitatively evaluate TBL and governance objectives, we developed a simulation model, approximating the three main species groups in the fishery: coral trout (CT), red-throat emperor (RTE), and other species (OS). The simulation is not fitted to data and is based on the assumption of perfect information: it contains neither a stock assessment nor a sampling model to estimate underlying biomass. However, to give the simulation model more fidelity to nature, we calibrated species' biomass levels and trends using stock assessment models (Leigh et al., 2006, 2014; O'Neill et al., 2011) and the historical catch data for the different sectors (described in detail below).

We simplified the fishery to two latitudinal regions (north and south), noting that, longitudinally, all commercial fishers concentrate their effort on the mid-shelf along an essentially north-south coastline. We chose the boundary between regions at latitude 18.1°S to allow for both lower fishing intensity and greatly decreased abundance of red-throat emperor north of this latitude, as presently occurs. We assumed no fish movement between regions, and region-specific recruitment. In the projections, we assumed that the charter and recreational fishing mortality were equally distributed between regions. We distributed the commercial fishing mortality as per equation (13) in Supplementary Material 1.5 (Little et al., 2007).

In a 31-year historical period of the simulation, we calculated fishing mortality based on the species-, sector- and region-specific historical catches for the two regions, after which we used the optimisation to determine a total allowable catch for each species group, allocated to one or more sectors, for a subsequent 25 years. The TACs also had the option of being region-specific. In Supplementary Material 1, we provide a full description of the population dynamics.

We optimised, over a range of possible TAC levels, a value function for each of a given set of stakeholder group weightings. This approach allowed us to test any harvest strategy decision rule, but here we limited our treatment to determining optimal species-specific, and, for some scenarios, region-specific, TACs across the operational objectives. We assumed that the optimised TACs were fully realised, with no over- or under-catch.

Following Richerson et al. (2010) and Munch et al. (2017), we defined a quantitative performance indicator for each of the 21 operational objectives, which had to be a function (directly or indirectly) of the management control, in this case, the TAC. Defining these operational objectives required strong assumptions about the relationship between the resource, fishery and control rule, particularly for the social objectives (Table SI 1, Supplementary Material 1). In general, the objectives are denominated in different units, so were normalised from 0 to 1 (with 0 being the "worst" performance, and 1 the "best"), to make the performance metrics commensurate (Richerson et al., 2010).

In setting functional forms for the performance indicators (i.e. determining the relationship between the performance indicator and the TAC), and associated target and limit reference points, we had to ensure that the logic remained as consistent as possible throughout, to avoid nonsensical or uninformative zones along the solution surface. Specifically, we: i) avoided uninformative "plateaus" to the extent possible. That is, we avoided "hockey stick" style relationships where the value of the performance indicator remained at 1 above the target reference point, and rather penalised the performance indicator as a function of its distance from the target; ii) detected and removed "impossible conflicts" that compromised the fitting process (for example, if the target reference points for the relative biomass of each species are such that OS relative biomass is greater than its target reference point, while CT and RTE relative biomasses are less than theirs, it is very difficult to optimise the TACs when different species are being driven in different directions); and iii) ran the simulation using single, or subsets of, performance indicators only, to ensure that each was behaving as anticipated. The functional forms of each performance indicator are illustrated in supplemental figure 1.8.

Having defined the 21 quantitative performance indicators, we then applied a corresponding stakeholder preference weighting to each performance indicator and summed to obtain an overall value. The value function in year *y* for any set of stakeholder group *g*'s objective preference weightings is

$$V_{g,g,y} = \sum_{j=1}^{21} PI_{j,y} \cdot Wt_{j,g} \tag{1}$$

where $PI_{j,y}$ is the value of performance indicator *j* in year *y*, and $Wt_{j,g}$ is the weighting of performance indicator *j* by stakeholder group *g*. In each year *y* of the simulation projection, we optimised to find the species-specific TACs that maximised $V_{g,g,y}$ (Mangel and Dowling 2016).

To ensure that the global minimum was achieved when optimising across a rugged likelihood profile, we initialised ("peppered") the model using 64 different parameter combinations of initial TAC values (for those scenarios for which TACs were also region-specific, one-third of the species' initial TAC value was assigned to the northern region, and two-thirds to the southern region). That is, initial values for each species' TAC were set at 300t, 1000t, 2000t or 3000t (4 sets of values for each of 3 species = 4x4x4 = 64 initial parameter value combinations). These values were initial guesses for the TAC parameters based on the historical catch levels, and used for each year of the projections, that were then changed through estimation by the optimisation process.

Given the optimum TACs for each stakeholder group's weightings, we calculated the value function using the weightings of every other stakeholder group. For each year, this gives a matrix of values according to each set of stakeholder group weightings, calculated using the performance indicators derived from the optimal strategy (TAC) for each stakeholder group. We write this as a matrix in which each row represents one stakeholder group's optimal strategy, which is applied to each stakeholder group's preference weighting, by column. Thus, for *n* stakeholder groups, we have a matrix of the form

$$\begin{bmatrix} V_{1,1,y} & V_{1,2,y} & \cdots & V_{1,g,y} & \cdots & V_{1,n,y} \\ V_{2,1,y} & \ddots & & & & V_{2,n,y} \\ \vdots & & \ddots & & & \vdots \\ V_{g,1,y} & & & V_{g,g,y} & & \vdots \\ \vdots & & & & \ddots & \vdots \\ V_{n,1,y} & V_{n,2,y} & \cdots & V_{n,g,y} & \cdots & V_{n,n,y} \end{bmatrix}$$

Each column of the matrix is standardised relative to the value for that column's stakeholder group for which the strategy is optimal, so that the diagonal elements are equal to 1).

We used two alternative criteria to select the overall optimal TAC: i) the highest average value across all stakeholder weightings (i.e., the row of the matrix that has the highest average, indicating that the strategy is overall optimal across all preference groups), and ii) the highest minimum value across all stakeholder weightings (the "maximin" criterion; the row of the matrix that has the highest minimum value across, indicating that this strategy results in the “minimum whinge” across all preference groups).

## 3.3 Input data

The historical harvest and effort data for each of the three species groups, for each of the commercial, charter and recreational sectors, span the 31 years from the beginning of the Queensland commercial logbook database in 1988 to 2018. Specific species targeting information was generally not available. The commercial sector focuses strongly on coral trout, so that we could quantify effort from commercial vessels equipped for live CT, but we could not delineate activity directed at dead CT, RTE and OS.

Commercial and charter harvest and effort came from the logbook database that has been compulsory for commercial fishers since 1988 and for charter fishers since 1996. We extrapolated charter data back to 1988 by assuming that they were constant over the period 1988–1996.

Recreational harvest and effort came primarily from the Australia-wide National Recreational and Indigenous Fishing Survey in 2000, and Queensland’s Statewide Recreational Fishing Surveys in 2011 and 2014 (Henry and Lyle, 2003; Taylor et al., 2012; Webley, 2015). Information in some other years (1997, 1999, 2002 and 2005) came from Queensland surveys that used different methodology. The latter surveys were used only as a trend and their overall estimates were scaled to match that from the 2000 survey. We interpolated data loglinearly for the years between 1997 and 2014 in which surveys were not carried out and assumed recreational harvest and effort were constant from 1988 to 1997, and from 2014 to 2018. We subtracted charter records from the recreational surveys in order to avoid double-counting of charter data: we regarded the charter logbook database as more accurate and it also included data from guests who did not live in Queensland.

We defined effort for the commercial and charter sectors respectively as the number of commercial-dory days or charter-guest days on which any fish were caught. Reliable data were not available on any finer time scale such as hours fished, or on days on which no fish were caught. For the recreational sector, we defined effort as the number of person-days on which fishing took place, including zero catches. Such measures of effort are particularly suited to TBL inputs such as costs of fishing, quality of fishing experience and impacts on non-target species. Their associated catch per unit effort (CPUE) ratios were less accurate indices of abundance of fish than would have been produced by, for example, standardisation by generalised linear models.

In Appendix Table A1, we summarise the general model and biological input parameters. They were derived from stock assessments of CT (Leigh et al., 2014), RTE (Leigh et al., 2006), and parameters for tropical snappers *Lutjanus* spp. (O’Neill et al., 2011). *Lutjanus* spp. constitute a substantial proportion of the OS catch, and many of them are long-lived, thereby providing contrast with CT and RTE, and providing a precautionary slant to the analysis. For the OS group, we used growth and weight-at-length for crimson snapper *L. erythropterus*, which are typical of the size of species in the OS category. We chose OS values of 0.15 $yr^{-1}$ for the natural mortality rate $M$ and 8 years as the age at maturity as typical for tropical red snappers. The value of the initial population-size parameter (see SM for details) for OS is a conservative educated guess to produce exploitable biomass approximately three times that for coral trout, bearing in mind that the OS category covers a multitude of species. The proportional splits of recruit numbers into regions was based on historical catch sizes, adjusted for the lesser intensity of commercial and charter fishing in the northern region (see SM for further details).

The number of age classes (20) was sufficient to embrace the lifespans of CT and RTE. Some of the OS species such as *Lutjanus* spp. live to more than 40 years but are still adequately covered by 20 age classes because they grow relatively quickly. Moreover, the final age class is a “plus group” containing all fish aged 19 years or more.

## 3.4 Alternative TAC specifications

### 3.4.1 Commercial TAC only

We began by applying a dynamic TAC only to the commercial sector. Currently, the charter and recreational sectors have no TAC, and the historical data for the charter and recreational sectors show a relatively constant catch over recent time (Figure 1). Thus, we fixed catch for these sectors, based on the average catch for each species group over the final three years of the historical time series.

Unless stated otherwise, in this and all other scenarios used the highest average, to obtain the “winning” stakeholder group preferences.

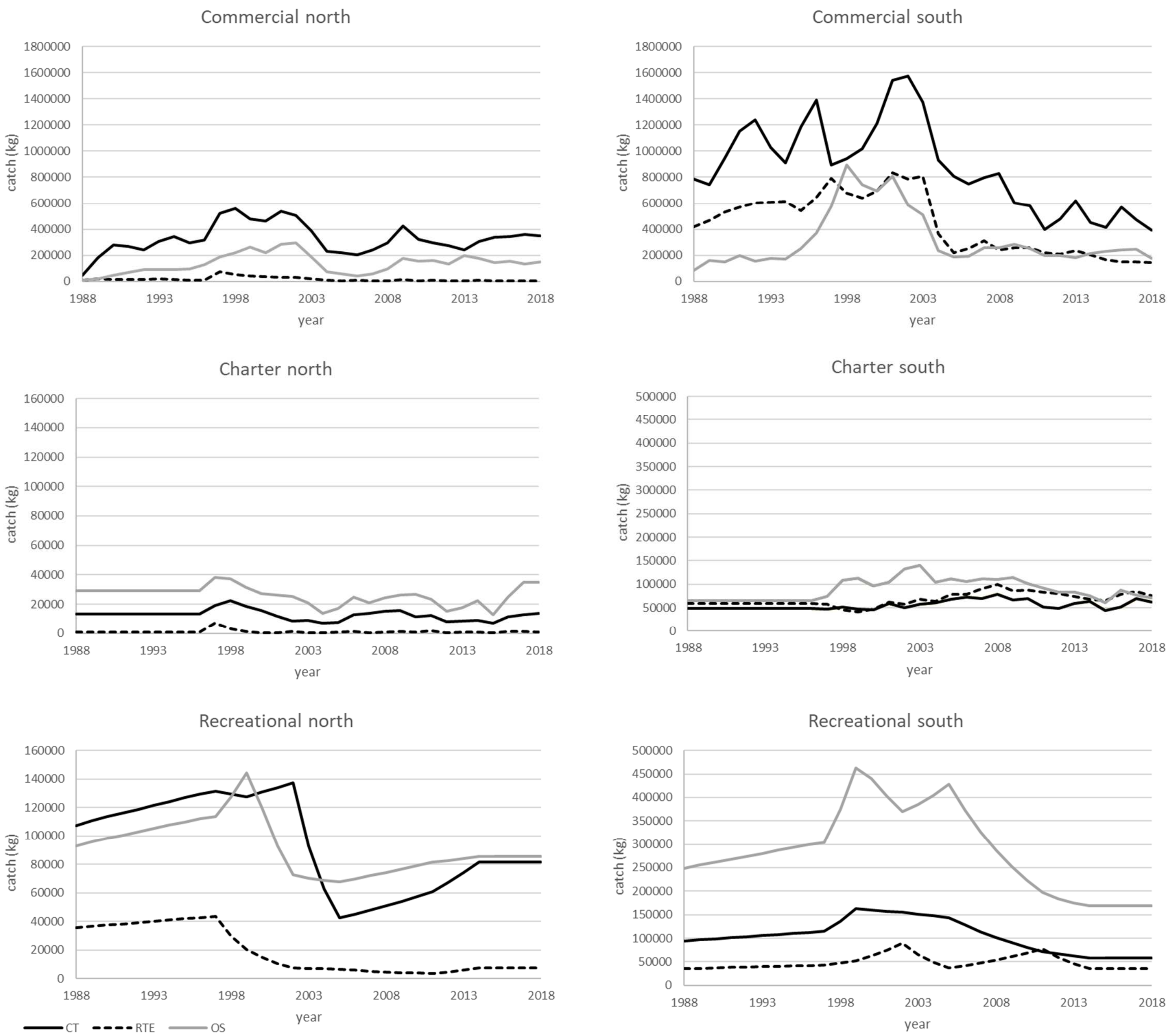

**Figure 1:** Reconstituted or actual historical time series of commercial, charter and recreational catch in the Coral Reef Finfish Fishery, by species group and region.

*3.4.1.1* *Commercial TAC optimised with "Maximin" criteria*

When determining the overall optimal TAC across stakeholder groups, we took as the default the highest average value across all stakeholder weightings. In this scenario, the TAC was assigned to the commercial sector TAC only, but using the “maximin” criteria, as opposed to using the highest average, to obtain the “winning” stakeholder group preferences. That is, the “maximin” approach takes the highest minimum value across all stakeholder weightings, indicating that this strategy results in the minimum loss of value across all preference groups.

3.4.2 Commercial and charter TAC

*3.4.2.1* *Base 2-TAC and 1 area*

One of the alternative harvest strategy options proposed by the fishery Working Group was for the charter sector to have its own TAC. For this scenario, we divided the modelled TAC as a fixed proportion (based on historical precedence) between the commercial and charter sectors. The recreational projected catch remained a fixed catch as described above.

This commercial and charter TAC scenario formed the basis for several additional scenarios including simulating the effect of environmental perturbations and climate change.

The reasons for building from this 2-sector alternative scenario rather than a commercial only TAC is because the former scenario conferred greater flexibility across the fishery through enabling the majority of the catch to be dynamically modelled and it was a key scenario considered in the Pascoe et al. (2019) study of the same fishery.

*3.4.2.2* *Cyclone (“acute” event) and climate change (chronic regime shift)*

To consider the effect of key environmental influences, we simulated acute and chronic environmental change in a simple way. Although these are rudimentary, they allow us to acknowledge the importance of such external forces to the fishery (Hughes et al., 2018; Kim et al., 2019) and to illustrate how their impacts might be considered.

Tropical cyclones are semi-regular events that correlate with major falls in fishery catch rates of the primary target species group coral trout (CT) in the southern region of the fishery, with simultaneous increases in red-throat emperor (RTE) catch rates (Bureau of Meteorology, 2019; Courtney et al., 2015; Queensland Government, 2019). We simulated a single cyclone event in the 5$^{th}$ year of the projection period, by reducing the availability of the CT species group by 40% and increasing availability of the RTE species group by 20% in the southern region for years 5–8. That is, we assume no impact on the underlying biomass, but rather on the availability of these species groups to the fishery.

We modeled climate change as a 1% per year migration of all species from the northern to the southern region, as well as an overall reduction of abundance of all species by 0.7% per year. These figures were chosen as levels that made a substantial difference but not enough to cause a complete fishery collapse.

*3.4.2.3* *Over-exploited resource*

To acknowledge that the level of historical fishing pressure was not high for all species, particularly for RTE and OS species groups, we considered a scenario where the stock was heavily fished for an additional 10 years before the projections, with constant catches by each fleet in each region of 1.6 times, 100 times, and 4 times that of the final historical year for CT, RTE and OS, respectively. These multipliers were chosen to give catch levels that would drive each species toward the limit reference point of 20% of the initial biomass by the end of the additional 10 years. In the case of RTE, the population biology was so resilient that even 100 times the final year catch only drove the stock level down to 47% of the initial stock size. For the CT and OS species groups, any heavier fishing than 1.6 or 4.0 times the final historical year would drive older age classes to extinction.

#### 3.4.3 Area-specific TAC scenario

We also ran an additional simulation in which TACs were set by region (thus 6 TACs per annum). We used the fleet dynamics models developed in previous studies of the fishery (Little et al., 2007; Little et al., 2016) to distribute fishing mortality by area.

#### 3.4.4 Commercial, charter and recreational TAC

In an additional scenario, we assigned all sectors fixed proportions of the modelled TAC. For each of these scenarios, the species-group-specific TACs were for the whole fishery with all regions combined (3 TACs per annum). We used the previously developed fleet dynamics models (Little et al., 2007; Little et al., 2016) to distribute fishing mortality. It should be noted that an annual (non-charter) recreational TAC is not practicable for the fishery, as there is no mechanism to record recreational harvest in close to real time. This case is modelled but only as a single scenario.

## 3.5 Model uncertainties and sensitivity analysis

Because the emphasis of this paper is a simulation that operationalises a multi-objective (TBL and governance objectives) harvest strategy, and there are multiple levels of unknowns and assumptions, the results should be interpreted with caution. The underlying operating model incorporates assumptions around the groupings of species, the fleet dynamics, and fish movement and recruitment patterns and these are assumed known. We also simplified the spatial regions and the characteristics of the commercial fleet (in combining “live” and “dead” CT fishers, dedicated RTE and OS fishers), as well as various inferences to approximate the historical catch and effort for the recreational sector.

Furthermore, translating each conceptual objective into a quantifiable operational objective (performance indicator) that is some function of the catch or effort requires assumptions concerning the form of the relationship for each performance indicator, the values of any associated reference points, and tolerance thresholds (Table SI 1). One way to have reduced the associated uncertainty would have been to have used higher-order (hence, fewer) objectives, but we did not do so because these were too vague in their articulation and contained too much inherent (hidden) detail to be sufficient for purpose.

Consequently, we undertook simple sensitivity analyses wherein we fixed the form of the relationship of each performance indicator and considered only one alternative parameter specification. The form of each sensitivity test is described in Appendix Table A2. We found that the performance indicators related to target species sustainability and commercial profitability resulted in the strongest changes (increases or reductions) in interannual variability in species-group-specific catch, and across the suite of performance indicators. The latter is unsurprising, since most of the performance indicators are functions of catch and biomass.

In general, the indicator values that were most strongly affected within sensitivity tests were those to which the change in specification was being applied. However, other performance indicators were affected by changes in the parameter values of any one performance indicator, typically with an increase in variability about their mean, if not a change in their mean values. Generally, across all the indicator-specific scenarios considered, the most sensitive indicators were the ecological indicators pertaining to minimising risk to bycatch species (1.2.1) and discarding (1.2.2), and the related social perception of the fishery (4.2.1). The former two are functions of effort and size structure, respectively, which were more affected by the sensitivity tests than overall catch and biomass.

# 4 Results

## 4.1 Historical catch data

Across both the north and south regions, catches generally increased to a peak in about 1998, before stabilising or declining from around 2003 when there was a major fishery restructure through the introduction of ITQs and no take areas were increased (Figure 1). Catches were much higher in the southern region partly due to higher human population numbers, and also due to regional differences in species distribution. Coral trout dominated the commercial catch, while the "other species" group dominated the charter and recreational catches, particularly the recreational sector in the south. The charter sector had the lowest catches of the three sectors.

In terms of modelled relative biomass, by the end of the 31-year historical time series, CT was recovering from being reduced to ~30% $B_0$ at around year 22, to be at ~40% $B_0$. RTE relative biomass was reduced to ~75% $B_0$ by year 17, but then increased to be above 90% $B_0$ by the end of the historical time series. OS biomass was at ~80% $B_0$ by year 31, up from ~73% $B_0$ in year 17.

## 4.2 Key scenarios

For each scenario, we present time series of total catch (Figure 2) (species-specific catch time series are also provided in Figure A1), total final biomass (Figure 3) (biomass time series are also provided in Figure A2) for each species group, as well as the mean of each of the 21 performance indicators, taken across the 25 projection years (Figure 4) (means with standard deviations are also provided in Figure A3).

Keeping the charter and recreational catches constant constrained the commercial TAC setting: total catch for each species showed very little variation from the final historical year (Figure 2). CT and OS biomasses continued to increase to over 60% and 80% $B_0$, respectively, while RTE biomass stabilised at over 90% $B_0$ (Figure 3). This optimised economic benefits of minimising interannual variability in profit (objective 2.4) and costs of management (objective 2.5), and the social objective of maximising equity between sectors (Figure 4). However, this was at the expense of the maximum economic yield not being reached (per lower values of profitability performance indicators relating to objectives 2.1.1-2.1.3), with stocks not being fished to $B_{MEY}$. To have achieved this would have required an extreme increase in commercial TAC that would have compromised other performance indicators, such as discarding (a function of effort) the equity between sectors (objective 4.2.1), and interannual variability in profit (objective 2.4).

Assigning TAC to the commercial sector only, but using the "maximin" criteria, as opposed to using the highest average, to obtain the "winning" stakeholder group preferences, increased RTE catch (Figure 2) such that RTE biomass achieved its target (Figure 3). This shows the sensitivity to, and hence the importance of, the criteria used to determine the "winning" set of stakeholder group preference weightings in each year. Using the "maximin" criterion, the most predominant winning stakeholder groups were quota owners and commercial fishers and processors/buyers/wholesaler, while the charter and recreational, and “other” group categories were the predominant winners using the “highest average” criterion. The most marked differences between these sets of groups was that the former strongly favoured commercial (and the directly related indigenous) profits (objective 2.1) (driving increased catches in RTE), and assigned less weighting to equity across the fishing sectors (objective 4.1) (such that the increased RTE catch for the commercial sector relative to the others was less important).

For brevity, the results presented below are based only on the “highest average” criterion.

The Working Group’s proposed scenario of allowing both commercial and charter sectors to have a dynamic TAC gave greater flexibility to the model. The catches of each species (combined across sectors) showed strong interannual oscillations, that were highest in magnitude in the first 5 years of the projection, but that ultimately fluctuated around an average (Figure 2). There was an approximately 20x overall increase in RTE catch to average around 6000t, a slight overall increase in average OS catch to average around 1000t, and CT catch averaged around 1000t. The increases in RTE and OS catch drove their respective relative biomasses down, such that all species stabilised around their targets of (for CT and RTE) between 0.4-0.6 $B_0$, and (for OS) 0.4 $B_0$ (Figure 3). We emphasise that we were careful to align the target reference points of all performance indicators, and that when these were misaligned, the oscillations lead to chaotic time series with inconsistent magnitudes with no discernible average.

When including performance indicators sequentially into the simulation (results not shown), it became clear that the commercial and charter profitability performance indicators were primarily responsible for the observed oscillations in catch. When the catches of all species were combined, the total catch across species resulted in a relatively stable time series. Essentially, CT and RTE catches were inversely correlated, suggesting there were multiple optimal states (combinations of species-specific catch) for which profit is optimal.

In terms of the performance indicators for this scenario, the target species sustainability indicators (relating to objectives 1.1.1, 1.1.2, 1.3.2), the profitability (objectives 1.1.1-1.1.3), recreational value (objective 2.2) and flow-on economic benefits (objective 2.3) were all optimal for this scenario (Figure 4). The cost of management, specified as a function of catch, also increased, such that the objective to minimise this was compromised (objective 2.5), as was (obviously, given the high variability in the early years especially) the objective minimising interannual variability in profit (objective 2.4). Willingness to comply with the harvest strategy (due to increased management complexity (objective 3) was also slightly compromised.

The performance indicators were at zero, indicating poorest possible performance, for the objectives of minimising broader ecological risk, and risk to Threatened, Endangered and Protected (TEP) species. Risk to bycatch species was also high (i.e. low value of objective 1.2.1) (Figure 4). These performance indicators were specified as functions of effort, with targets and limits set at fractions of the historical value. With the increase in effort associated with the higher catches of RTE in particular, the performance of these objectives was compromised. Performance was also poor for discard mortality risk (objective 1.2.2), indicating the proportion of small-sized fish in the catch increased. As a result, performance associated with the public perception risk associated with discards and TEP species (objective 4.2.1) was also low. Finally, equity between sectors (objective 4.1) and regions (objective 4.3.2) was compromised. Since the targets were based on historical precedent, and RTE catch in particular broke that precedent, the targets may need to be revised, leading to a paradigm shift in the fishery management rule.

When all three sectors received TAC, the catch trajectories again showed strong fluctuations in the first 5 years of the projections (Figure 2), but thereafter were stable and smooth at levels that maintained the relative biomass at target levels (with the exception of a slight decrease in OS biomass at the end of the projected time series, albeit one still within the 10% tolerance about the target reference point of 40% B0) (Figure 3). Relative to TAC being allocated to only the commercial and charter sectors, the main trade off in terms of performance indicators was the charter sector profit, since the TAC allocation that had previously been assigned to this sector was now being shared with the non-charter recreational sector (Figure 4). The performance indicator relating to objective 2.2 (maximise value of recreational fishers and charter experience (direct to participant)) was optimal for both scenarios, because this is determined across both the charter and recreational sectors. Despite the stable total catch trajectory, there was an increased interannual variability in commercial and charter profit (and so a lower value for the performance indicator relating to objective 2.4), indicating higher interannual variability in how the catch is shared between sectors, likely due to multiple uniform states across the likelihood profile across various relative TAC proportions. Willingness to comply with the harvest strategy (due to further increased management complexity (objective 3)) was also slightly compromised.

When TACs were set for the commercial and charter sectors separately for each of the two regions, the increased flexibility had the result that the total catches for each species did not show the same strong interannual oscillations, and particularly, the overshooting in the first 5 years of the projection, though, for CT, the longer-term interannual oscillations in catch were stronger in magnitude than for the non-region-specific-TAC scenario (Figure 2). RTE catch again increased by approximately 20 times, and the average projected catches of all three species were ultimately similar to the non-region-specific-TAC scenario. Consequently, the relative biomass trajectories were also similar to the non-region-specific-TAC scenario, with the biomasses of all three species being driven to their target values (Figure 3). The CT biomass also was more stable than that for the non-region-specific-TAC scenario, which continued to increase throughout the projection. The stability is again likely due to the greater flexibility afforded by assigning TAC by region and thereby being able to more directly achieve the sustainability objectives.

In terms of the performance indicators, there was little difference between the region-specific and non-region-specific TAC scenarios (Figure 4). The main gains over non-area-specific TACs were small, and were mostly in terms of three objectives. The first two were i) the reduced discarding of undersize fish (objective 1.2.2), presumably because the TACs were now being directed towards to the regions of higher relative abundance, and ii) the related improved public perception that is partly related to discarding practices (objective 4.2.1). The third was slight improvement in the perception of equitable access by region (objective 4.3.2), possibly because, despite the increase in RTE catch, the relative regional TAC assignment may be more consistent with past relative catch patterns on which the target was based.

The cost of this improvement in performance indicators was in terms of the management “willingness to comply” objective (objective 3), which is directly related to the increased number of management controls (TACs). Despite the reduction in high-magnitude oscillations in catch at the start of the time series, there was no change to the average interannual variability in the performance indicator (objective 2.4) relative to TACs being non-region-specific, likely because the total catches across all species for both scenarios showed relatively small interannual changes beyond the first projection year.

The scenarios with environmental change resulted in very little medium- to long-term changes in catch and biomass (Figure 2, 3). Recall that we simulated a cyclone in the 5$^{th}$ year of the projection period by reducing the availability (but not the actual abundance) of the CT species group by 40% and increasing availability of the RTE species group by 20% in the southern region for years 5–8. Relative to the scenario with no environmental perturbations, this was reflected by a short-term reduction in CT catch from years 5-7 of the projection period (years 36-38). However, catch quickly recovered (since the underlying abundance was assumed to be unaffected) to its long-term stable state. In the same years, a short-term increase in RTE catch occurred (Figure 2).

Given that all modelled species biomasses were well above their target reference points, the effect of the simulated climate change was due more to the 1% per year migration of all species from the northern to the southern region, than to the overall reduction of abundance of all species by 0.7% per year (Figure 3). There was no effect on overall catch or biomass, nor most of the performance indicators (Figure 4). There was a slight relative increase in discarding (a reduction in performance indicator relating to objective 1.2.2, as well as a worsening of the associated social perception indicator relating to objective 4.2.1) as a result of increased relative proportions of undersized fish in the catch, possibly as a result of the reduction in abundance. Across all performance indicators, the main difference was a reduction in the charter sector profitability. This appears incongruous given that commercial profitability was unaffected, but as opposed to commercial profitability, charter profitability is simulated as a function of effort. There is relatively higher charter catch in the southern region than the north. Total catches, and the performance indicators pertaining to equitable access between sectors and regions indicated no significant sector- or region-specific differences in catch. Since we simulated effort for each sector in each year as the catch divided by the product of the catchability and the fishable biomass, an increasing fishable biomass in the south led to a reduction in effort in the predominantly fished southern region, and hence, a reduction in charter sector profitability.

Populations recovered to sustainable target levels when the biomass was historically more heavily fished down towards the limit reference point. As with the earlier scenarios, changes to the TAC were greatest within the first 5 years of the projections (Figure 2) (with large interannual changes in TAC that compromised the performance indicator pertaining to interannual variability in profit (objective 2.4). In this time period, CT and OS TACs were consistently very low, while RTE continually declined. CT and RTE total catches were stable thereafter, with the exception of one inversely correlated year. OS catches increased over the final 8 years of the projection, as a result of higher catches in the north.

For RTE, the projected catch did not increase substantively in the northern region; thus, most of the biomass increase occurred in the north. The opposite was the case for OS. There was more overall biomass in the southern region for both species groups, but the total RTE biomass was within its target ranges after being “fished down”, meaning the catch in the more abundant southern region did not significantly change. Total OS biomass, however, was at its limit of 20% $B_0$ after being “fished down”, with very low relative biomass in the northern region. As such, much of the recovery of this species group was driven by low catch the southern region. The northern region OS catches actually increased, keeping the biomass in this region low, presumably because the relative contribution of the northern region to the recovery of the total OS biomass was so low as to be negligible.

The depletion associated with “fished down” stocks affected the oldest age classes most strongly, and hence the performance indicators related to discarding (objectives 1.2.2, 4.2.1) were minimal (Figure 4) as a result of the increased relative proportion of undersize fish in the catch. The OS sustainability performance indicator (relating to objective 1.1.2) was also compromised due to this species group being the most heavily fished down. The reductions in commercial and charter TAC while recreational catch levels were kept constant also minimised the performance indicator pertaining to equity between sectors (objective 4.3.2).

We note that the model does not consider the ratios of TACs between species. However, it is unlikely that effort could be targeted to achieve species-group-specific catch limits, particularly if these vary significantly from the historically achieved ratios. Discarding is therefore a risk around implementing unrealistic TAC ratios. Similarly, it is highly unlikely that 100 times the historical catch of RTE would occur concomitant with small increases in CT and OS catch, as was simulated here for the “fished down” scenario.

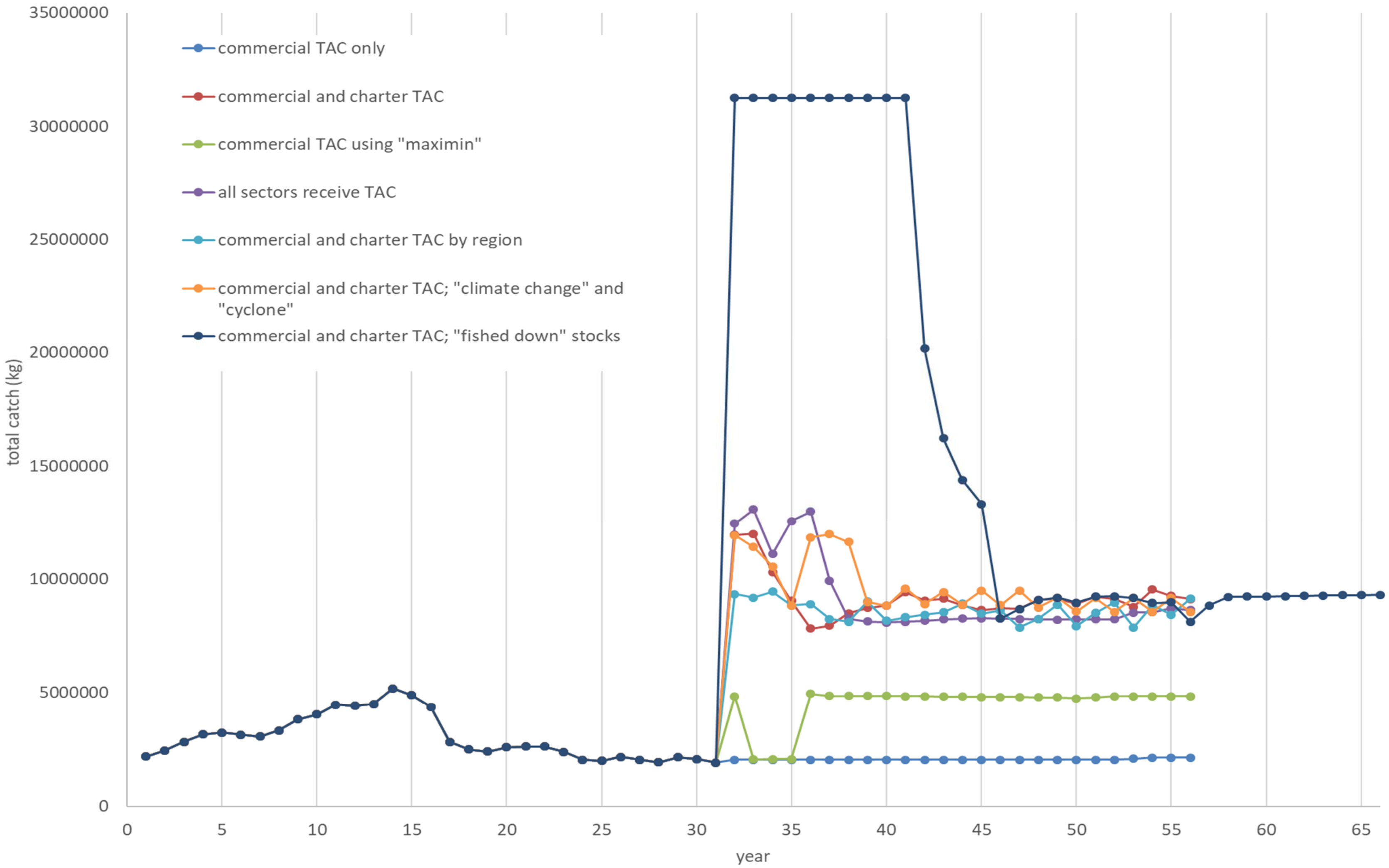


**Figure 2:** Time series of total catch (kg) summed across each species group, for each scenario considered.

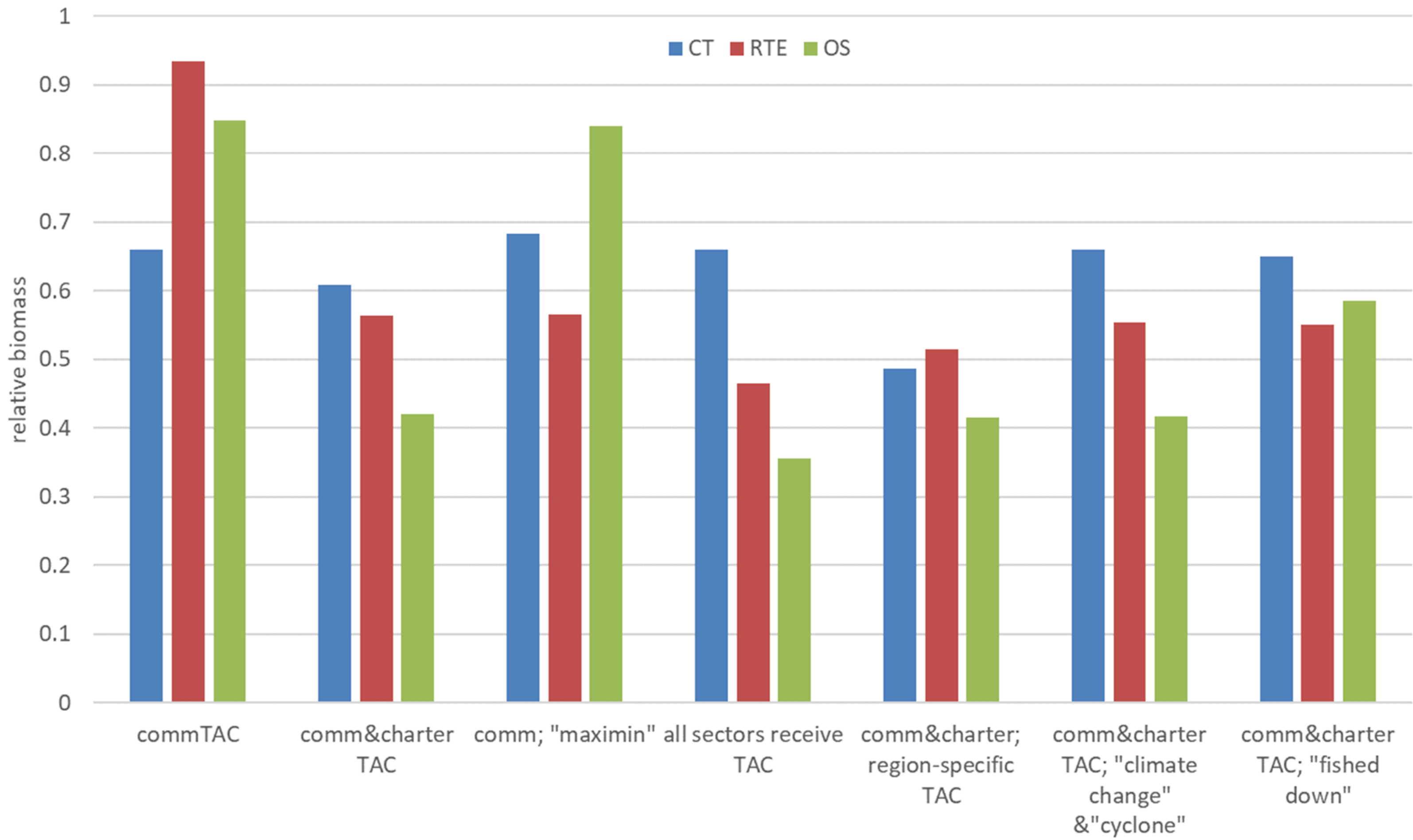


**Figure 3:** Barplot of final year biomass, relative to the initial year, for each species group and scenario considered.

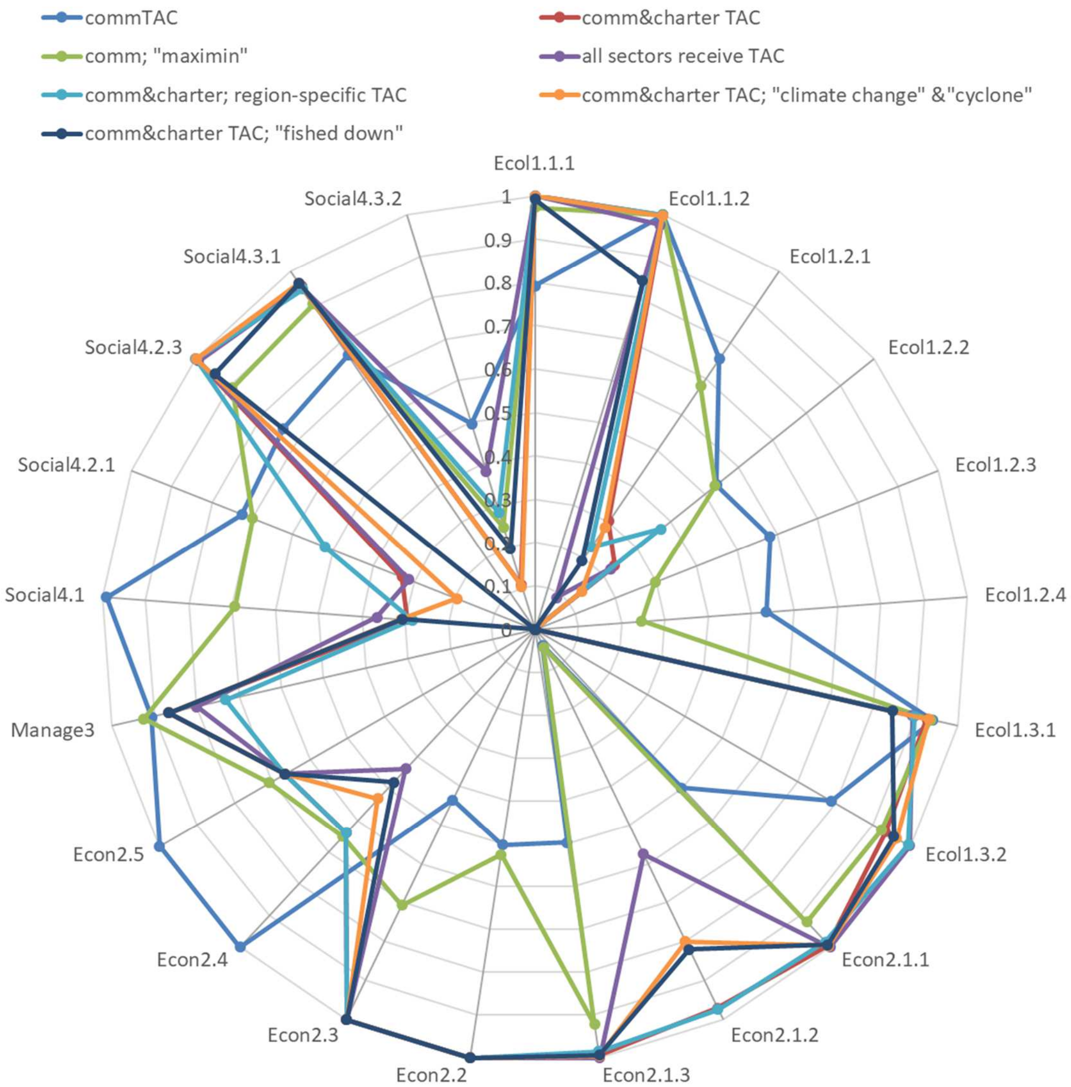


**Figure 4:** Radar plot of mean value across the projection years, for each of the 21 performance indicators, for each scenario examined.

# 5 Discussion

Our goal is to provide a tool for managers, fishery management councils, scientists, and stakeholders to consider a richer range of tradeoffs than possible with bio-economic models only. Consistent with policy and legislative requirements, the model we developed provides a quantitative means to explicitly evaluate the four pillars (TBL and governance) and their tradeoffs in terms of clearly defined stakeholder objectives. In addition, it allows for formal evaluation of performance of the four pillars across alternative stakeholder group preferences, providing an impartial means to obtaining an overall optimum harvest strategy (here, a set of species-group-specific TACs). As opposed to semi-quantitative/expert judgement approaches that rank or rate alternative harvest strategy specifications, our approach leads to both quantified alternative harvest strategy options, and the optimal values for the management controls.

Our model is less complex than many current ecosystem models. It is relatively easy to implement and by placing all the indicators on the same scale, disparate indicators can be compared. Importantly, implementing it requires detailed discussions with stakeholders on objectives and their relative weights. Different stakeholder opinions (in the form of weights) on importance are overtly considered. This linkage between a discussion on objectives (without restriction to the model's needs) was initially seen as a benefit, but in hindsight has delivered some of the difficulties with the model.

While the model is conceptually not complex, parameterising and optimising it was fraught with technical challenges. Given the number of objectives and performance indicators that came out of the stakeholder process, the model is information hungry. This leads to having to define several indicator's functional forms and their targets, many of which are unknown to stakeholders and scientists alike, and produced a likelihood function that was complex and resulted in a sensitive (in an estimation sense) model. The formulation of separate performance indicators for each of the objectives estimated annually meant the model had "no sense of consequence" for an optimisation in following years. Finally, as for many mathematical models, stakeholder engagement is more restricted given the technical content of the model. Below we expand on these issues and then discuss possible solutions.

Multi-sector, multi-species fisheries such as the Coral Reef Finfish Fishery need to address the TBL. However, the quantity and quality of data are often mixed, many reference points are uninformed, and performance indicators vary in their quality of information: broader environmental, economic, and, particularly, social information is often limited. As data collection programs expand over time, this difficulty will become less important but is unlikely to disappear. Had data been available – for example, for social performance indicators in the form of a survey – we could at least have tuned the model to these in addition to stock status. Additionally, while we were able to move beyond an abstract specification of objectives, the information hungry nature of the model meant that many of the operational objectives (performance indicators) were still ultimately specified in terms of catch and effort as; that is, catch and effort were used as proxies for socio-economic considerations. As highlighted by (Mangel and Dowling, 2016) and Dichmont et al. (2010), these can be fraught assumptions.

As with all models, a range of factors determine the nature of the results. These include specification of the performance indicators, the choice of values for (depending on the indicator's specification) target or limit reference point values, weightings, penalties, or parameters. Several of the performance indicators were extremely difficult to quantify, especially those in the social objective arena, and drove much of the model's sensitivity and (initial) instability. This has also been found by others (Brooks et al., 2015; Pascoe et al., 2017; Symes and Phillipson, 2009; Triantafillos et al., 2014; Vieira et al., 2009). We addressed this issue head on by developing performance indicators and associated parameters as a function of a single management control (TACs). The sensitivity of the model to the scenarios, as well as to the functional form of the performance indicators and their reference point values, showed the risk of using many detailed performance indicators to obtain meaningful management advice. We had to carefully construct the performance indicator specifications to ensure that these were aligned across objectives, and we had to "pepper" the starting parameter values to avoid local minima in what was still a rugged solution surface. Separate objectives (e.g. profitability and final biomass) competed unless their targets were consistent and optimal for both, e.g., the maximum economic yield and the biomass corresponding to maximum economic yield. With 21 performance indicators, ensuring such consistency was a challenge.

The projected time series of most of the model scenarios showed at least some years of interannual oscillation in the sector- and species-specific TAC values, particularly in the early years of the projection. For RTE and OS, historical catch levels had been well below those corresponding to target reference points (most notably, maximum economic yield). However, TACs oscillated rather than ramping up during projection years. This occurred because, by undertaking optimisation within each year, the model has no sense of medium- to long-term consequences.

Another issue contributing to inter-annual oscillations in the sector- and species-specific TAC was the inverse correlation of CT and RTE catch in many of the scenarios. While catches of these species, and any dependent performance indicators, showed interannual fluctuations, the projected catch totaled across both species was relatively stable. When examining performance indicators by incrementally including each, the projected catch time series only became strongly interannually fluctuating with the inclusion of commercial and charter profitability performance indicators, themselves direct functions of the CT and RTE catch. This speaks to alternate states of CT and RTE relative catch that are equally profitable. Future work should optimise over the medium- to longer-term, rather than annually.

Because of such complexities, we had had less direct stakeholder involvement, other than objective identification and weighting, than more conceptually-based semi-quantitative approaches. The results are also more technically challenging to interpret, as both input and output are demanding of information. This may mean that stakeholder buy-in to the model will remain low until the method matures and absorbs some of the solutions discussed below.

One option for reducing the uncertainty and complexity of the simulation is to include fewer operational objectives and performance indicators. Katsikopoulos et al. (2018) suggest that under such conditions, simple models may be more appropriate than more complex models for decision making, particularly in the case of repeated operational decisions such as required when implementing a harvest strategy. A high number of objectives is may be excessive in a practical sense. However, reducing the number of objectives will require reconsideration of how to translate broader objectives into quantitative performance indicators. One way this may be achieved could be to subsume many of the correlated performance indicators into single metrics; for example profitability and target biomass could be combined as one does in a standard bio-economic model. Reducing or subsuming the number of objectives and performance indicators may also help overcome the problem of roughly similar weightings across the different stakeholder groups (see also Pascoe et al., 2019). The similar weightings across stakeholder groups may be an artefact of the "dilution" effect of distributing higher level objective weights over many sub-objectives. An alternative way to define some of the objectives could be to use a Bayesian Belief Network (BBN) to capture non-quantitative objectives. The outputs of the operating model would then feed into the BBN model to quantify the social components.

Clearly, a multi-year forward optimisation process would have been preferable. Longer-term expectations should be captured by the value at which the target reference points are set, and if these are established correctly then the projections should eventually achieve them. The forward optimisation can then also be constrained if needed by for example, a smoothing term.

Two alternatives to the model described here are viability and frontier analyses. Gourguet et al. (2016) developed viability analysis for Australia's Northern Prawn Fishery. With this approach, one does not aim to identify an optimal outcome, but instead aims to ensure at least a minimal acceptable level for each of the objectives. It is thus analogous to Simon's notion of satisficing, e.g. Simon et al. (1950). In frontier analysis (Halpern et al., 2013), the frontier consists of TBL solutions, where one can optimise conservation goals and equity while minimising costs. The frontier does not prescribe a single solution but instead presents the range of options, all optimal, that represent the trade-off between stated goals. The choice of the optimal solution by a decision maker will be based on their relative importance weights for each objective. While potentially less transparent than the use of pre-determined weights, decisions are made with an explicit recognition of what is being given up. The policy frontier thereby complements the decision-making process without aiming to replace it (Sylvia and Enríquez, 1994).

On the contrary, our approach keeps harvest strategies in mind and leads to a recommended TAC, optimised across all multiple (TBL plus governance) objectives, and acknowledges the alternative preference weightings of stakeholder groups and is suitable for embedding in an MSE. Neither viability nor frontier analysis allows for this. Our approach also showed sensitivity to the criteria used to identify the "winning" set of stakeholder group preferences, or weightings, in each year: the "highest average" approach gave markedly different outcomes to when the "maximum minimum" value criterion was utilised.

Even with the sensitivities, inherent assumptions, and simplification, our model illustrates the trade-offs between multiple objectives and different stakeholder group preferences, and the value of region- and sector-specific TACs in different environmental contexts. The next step would be to reduce the number of objectives so as to reduce the inherent uncertainties and data requirements, and the complexity of the solution surface, and to optimise across the longer term.

Policy and legislation demand that fishery management moves towards a quantitative approach to TBL objectives and operationalising these defensibly within harvest strategies. We developed a model whose likelihood surface was proved highly complex and sensitive to inputs and assumptions, which will force managers and stakeholders to confront extensive data requirements.

To advance TBL/four pillar fishery management, a high level of involvement of stakeholders is required in determining fishery objectives and their weightings. An appreciation by management agencies of the data requirements of multi-objective fishery management, and a commitment to implement a quantitative approach that sets precise values for management controls, is also recommended. At the same time, this should be tempered given data limitations and the need for a manageable number of objectives across the four pillars.

More broadly, quantitative ways to operationalise multi-objective harvest strategies are likely to have relevance for other renewable resource industries where the TBL matters, provided these have management controls that can be changed. Our approach has provided a stepping-stone towards this goal and a basis for further modification and has highlighted the technical pitfalls of using simulations to optimise across multiple objectives in complex fisheries.

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# Appendix A1: Additional Figures and Tables

| Input parameter | Abbreviation | | Value | | |
|---|---|---|---|---|---|
| | | | CT | RTE | OS |
| Number of historical years | Nhist | 31 | | | |
| Number of years to project | Nproj | 25 | | | |
| Number of areas | Narea | 2 | | | |
| Number of fleets | Nfleet | 3 | | | |
| Number of species (groups) | Nspecies | 3 | | | |
| Number of age classes (for each species group) | Nage | | 20 | 20 | 20 |
| Maximum age (for each species group) | MaxAge | | 19 | 19 | 19 |
| Number of sets of preference weightings | NsetsWts | 8 | | | |
| Weight-at-length (WtL) parameters a,b | a | | 6.8500E-06 | 1.3778E-05 | 2.4400E-05 |
| (for each species group) | b | | 3.19640 | 3.06507 | 2.87000 |
| von Bertalanffy (vonB) growth parameters | Linf | | 66.33 | 51.68 | 58.45 |
| | k | | 0.1005 | 0.24146 | 0.3922 |
| | t0 | | -5.256 | -1.243 | 0.1768 |
| Natural mortality at age (for each species group) (assumed age-independent) | NatM | | 0.4656 | 0.5117 | 0.15 |
| Selectivity-at-age | SelAge | Age | | | |
| | | 0 | 0 | 0 | 0 |
| | | 1 | 0.5 | 0 | 0 |
| | | 2 | 0.66 | 0 | 0.05 |
| | | 3 | 0.78 | 0.3 | 0.1 |
| | | 4 | 0.86 | 0.8 | 0.2 |
| | | 5 | 0.9 | 1 | 0.35 |
| | | 6 | 0.93 | 1 | 0.5 |
| | | 7 | 0.95 | 1 | 0.65 |
| | | 8 | 1 | 1 | 0.8 |
| | | 9 | 1 | 1 | 0.9 |
| | | 10 | 1 | 1 | 0.95 |
| | | 11 | 1 | 1 | 1 |
| | | 12 | 1 | 1 | 1 |
| | | 13 | 1 | 1 | 1 |
| | | 14 | 1 | 1 | 1 |
| | | 15 | 1 | 1 | 1 |
| | | 16 | 1 | 1 | 1 |
| | | 17 | 1 | 1 | 1 |
| | | 18 | 1 | 1 | 1 |
| | | 19 | 1 | 1 | 1 |
| Steepness (by species group) | Steep | | 0.5 | 0.8 | 0.7 |
| Age at maturity (by species group) | AgeMat | | 3 | 3 | 8 |
| Initial number seed (numbers) (by species group) | RoInit | | 16800575 | 15466824 | 2787694 |
| Fixed allocation proportion of TAC between sectors (commercial, charter, recreational) | PropFfleet | | | | |
| commercial | | | 0.85 | 0.50 | 0.50 |
| charter | | | 0.05 | 0.30 | 0.25 |
| recreational | | | 0.10 | 0.20 | 0.25 |
| Fixed relative spatial distribution (for recruits) | Frac | | | | |
| region 1 | | | 0.3 | 0.2 | 0.3 |

| | | | |
|---|---|---|---|
| region 2 | 0.7 | 0.8 | 0.7 |

**Table A1**. Summary of model and biological input parameters

**Table A2:** Descriptive summary of conceptual objectives together with their translation into operational objectives, or performance indicators, the assumptions made in the specification and parameterisation of the operational objectives, and the sensitivity tests undertaken on each.

| Overarching objective | Sub-objectives | Specific objectives | Operational objective (descriptive; full equations in Supplementary Information) | Assumptions | Sensitivity analysis |
|---|---|---|---|---|---|
| **1. Ensure ecological sustainability** | **1.1. Ensure resource biomass sustainability** | **1.1.1 As per the Queensland Sustainable Fisheries Strategy, Policy achieve $B_{MEY}$ (biomass at maximum economic yield) (~60% unfished biomass), or defensible proxy, by 2027 (if below biomass at maximum sustainable yield, $B_{MSY}$, aim to achieve $B_{MSY}$ (~40-50% $B_0$) by 2020), for the main commercial, charter and recreational species (coral trout, RTE and key other species yet to be identified)** | We use a dome-shaped specification (Figure S1.8.1). If the relative biomass is within 10% of the target range, the score for that species is 1.Below the limit of 20% of the unfished biomass, the score for that species is 0. Between the lower end of the 10% tolerance around the lower target value, and the limit of 0.2, the score tracks linearly with relative biomass. Above the upper target value + 10%, the score decreases linearly from the target reference point to virgin, down to a minimum of (currently) (set as variable) 0.5 (i.e. we're half as happy as at target). If the relative biomass of any one species is below the limit reference point, then the overall PI is zero. Otherwise, for each of the alternative specifications, the overall PI is taken as the average values across both species. | The target reference point is assumed to range from 40%-60%, while the limit reference point is 20%, of the unfished biomass. The broad target, or plateau for the dome, encompasses the range from biomass at maximum sustainable yield (traditionally assumed to be 0.4B) and biomass at maximum economic yield (traditionally assumed to be 0.48B0), as well as the Queensland specified target of 0.6B0. From a conservation standpoint, these targets may be higher (trialled in sensitivity analysis). | 1) For 1.1.1, Target biomass range changed from 0.4 to 0.6, to 0.6 to 0.85. For 1.1.2, target biomass increased from 0.4 to 0.6 and limit biomass increased from 0.2 to 0.3 (i.e. more conservative reference points) |
| | | **1.1.2 Minimise risk to Other Species (that are harvested, per the "Other Species" list) in the fishery which are not included in 1.1.1. above** | The performance indicator follows a hockey-stick rule, being 1 above a relative biomass of 0.4, 0 below a relative biomass of 0.2, and tracking linearly with relative biomass between these values | The target reference point is 0.4 of the unfished biomass, as a proxy for MSY. From a conservation standpoint, a target of 0.6 and a limit of 0.3 may be more aligned with this objective (trialled in sensitivity analysis). | See 1) above |
| | **1.2 Ensure ecosystem resilience** | **1.2.1 Minimise risk to bycatch species** | This performance indicator is assumed to scale as a linear function of effort, normalized to some multiple of the maximum historical effort (here, 1.5). For each target species, fleet and area, the effort is calculated relative to the historical high, and | This refers to generic bycatch, as opposed to specific species. It is not inclusive of undersize discarding, or high grading, as these are covered in separate | 2) Changed effort threshold to 3x historical high, as opposed to 1.5x |

| | | | | | |
|---|---|---|---|---|---|
| | | | set to 1 if the effort is greater than 1.5 times the historical high. These values are then averaged to yield an overall value. We then subtract this mean value from 1 to give the final performance indicator. | performance indicators below. At the same time, it is noted that almost all catch is sold in the fishery, and that the gears are relatively clean, so that bycatch is not a critical issue in the fishery. | |
| | | **1.2.2 Minimise discard mortality (of undersized target species, or from high-grading of target species)** | The total proportion of discards by fleet, species, area and year, is calculated by standardizing the undersize catch relative to the total (legal and undersize) take. The minimum legal length for each species group is taken to be that corresponding to the age at maturity. The average is taken over fleet, species and area to yield a mean overall discard. The discard percentage is then normalized according to the worst possible expected discard percentage. | The worst possible discard percentage is assumed to be 0.5. We assume zero high grading for this fishery (moreover, high-grading is irrelevant in the context of a value function unless it is assumed to be a direct or indirect function of the TAC). | 3) Change worst discard percentage to 0.2 |
| | | **1.2.3 Minimise broader ecological risks** | The broader ecological risk is assumed to be a function of effort. We set the PI to 1 when effort is 0, and to linearly decrease to 0.8 between 0 and a target effort level. The PI value then linearly decreases from 0.8 to 0 between the target and limit effort values and is set to 0 when effort exceeds the limit. | Half of the effort, averaged over the last 5 years, is the most desirable (target), while the historical high effort is the least (limit) | 4) For 1.2.3 and 1.2.4, change to 30% of average effort being most desirable and 80% of historical high the least |
| | | **1.2.4 Minimise risk to TEPS** | The TEP risk is formulated in a similar manner to 1.2.3, except that, between the target and limit effort, the PI value is a weak inverse exponential function of effort. | Half of the effort, averaged over the last 5 years, is the most desirable (target), while the historical high effort is the least (limit) | See 4) above |
| | **1.3. Minimise risk of localised depletion** | **1.3.1. Due to fishing** | Applies only to CT and RTE. The performance indicator is set as 1 above a relative area-specific biomass of 0.5, 0 below a relative area-specific biomass of 0.2, and is assumed to track linearly with relative biomass between these values. The performance indicator is the minimum across the | Target and limit relative biomass reference points are set at 0.5 and 0.2. | 5) Target and limit reference points are changed to 0.6 and 0.3 |

| | | | | | |
|---|---|---|---|---|---|
| | | | species and areas. | | |
| | | **1.3.2. In response to environmental event (e.g. cyclone, climate change)** | Cyclones and climate change are considered using separate model scenarios. However, this performance indicator needs to reflect the need to be conservative and precautionary given these perturbations. As such, we and apply a 20% penalty to the target and limit reference relative biomasses used in PI 1.1.1, by dividing these by 0.8. We then use a dome specification as for performance indicator 1.1.1, with the penalized targets. The final performance indicator value is the mean across the species groups. | Target and limit relative biomass reference points are set at 0.5-0.75, and 0.25. | 6) Penalty = 0.6 as opposed to 0.8 |
| **2. Enhance fishery economic performance** | **2.1 Maximise commercial economic benefits, as combined totals for each of the following sectors** | **2.1.1 Commercial fishing industry profits** | This is calculated as price multiplied by catch, minus costs. Costs are a function of fuel, gear (which are functions of effort) and catch. Commercial profit is then catch multiplied by price, minus the costs. The PI is calculated by taking the ratio of profit to that at MEY, where the latter was approximated by taking the simulated historical high profit for the commercial sector, noting that these corresponded approximately to 0.6B0 for the CT species group. If the current profit exceeds the approximation for profit at MEY, the performance indicator reduces linearly until it reaches zero at 1.5 times the profit at MEY. If the current profit exceeds 1.5 time the approximation for profit at MEY, the performance indicator is set to zero. Concurrently, if the biomass of any one species is less than the limit reference point of 0.2B0, the PI = 0. | Unit costs of fuel, gear and effort have all been assumed. Profit at MEY was approximated by taking the historical high profits for each fishing sector, noting that these corresponded approximately to 0.6B0 for CT. | 7) Costs are multiplied by 1.5 AND ProfitMEY by 1.2, both for this and 2.1.2 below |
| | | **2.1.2 Charter sector profits** | Gross profit for charter operators is assumed to be the product of effort in days (as a proxy for the number of people fishing per day), multiplied by the charter price per day. Costs, profit and the performance indicator then are calculated in the | Unit costs of fuel, gear and effort have all been assumed. Profit at MEY was approximated by taking the historical high profits for each fishing sector, noting that | As for 7) above |

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| | | | same manner as for the commercial sector. | these corresponded approximately to 0.6B0 for CT. | |
| | | **2.1.3 Indigenous commercial benefits** | In the absence of a better understanding, we assume that indigenous commercial benefits scale with commercial profit, and as such, we specify this as an additional weighting on the commercial profit performance indicator. | The assumption of a direct correlation with commercial profit is a gross oversimplification in the absence of data. | N/A |
| | **2.2. Maximise value of recreational fishers and charter experience (direct to participant)** | | We assume the value of recreational fishing and charter experiences, direct to the participants, is some weighted function of charter and recreational catch, catch-per-unit-effort (CPUE), and effort. Each area's utility is, in turn, weighted according to the proportion of recreational effort in that region. The average is taken over all regions, and the performance indicator is calculated by standardising this average by the maximum historical recreational utility. | We assume the same weightings between the charter and recreational fleets, since we are considering the same recreational participants (i.e. the fishers, rather than the charter boat operators). Weights on each of catch, CPUE and effort are assumed, as are the weights assigned to each species group. The maximum historical high catch, CPUE are effort are those averaged over area. | 8) Changed catch, CPUE and effort weights from (0.4,0.3,0.3) to (0.7,0.25,0.05) to emphasis catch and CPUE, and changed species group weightings from (0.4,0.3,0.3) to (0.6, 0.3, 0.1) to emphasise CT catch |
| | **2.3 Maximise flow-on economic benefits to local communities (from all sectors)** | | Average benefit (across areas) is the sum of the commercial and charter profits (from 2.1.1 and 2.1.2), and an assumed unit dollar value applied to the recreational effort. The performance indicator is obtained by normalising relative to the historical maximum. | The recreational dollar scalar, and the historical maximum as the reference, are both assumed. | 9) Changed recreational dollar scalar from 10 to 100 |
| | **2.4 Minimise short term (inter-annual) economic risk** | | We approximate short-term risk as the interannual percent variability in commercial and charter profit. We take the coefficient of variation in profit for each fleet over the past 10 years. We assume a "hockey stick" relationship between the CV and PI score for each fleet, where a variation of +/- 10% CV is | The target and limit reference values are assumed. | 10) Changed target from +/-10%CV to +/- 5%CV, and limit from +/-25%CV to +/-20%CV |

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| | | | optimal and equates to a PI value of 1, and that +/- 25% is the limit below which the PI score value is 0. If the CV for any one fleet is below the LRP, then whole score for this objective is zero. Otherwise, the performance indicator is the mean of the CV scores across the commercial and charter fleets. | | |
| | **2.5 Minimise costs of management associated with the harvest strategy: monitoring, undertaking assessments, adjusting management controls** | | For now, we simply assume that if the TAC for each species group exceeds 1.5x the historical high catch, management costs increase. The species group score is 0 if the TAC is under the threshold and 1 is the threshold is exceeded. The performance indicator is the average of the species group scores. | The assumption of an increase in management costs above a threshold is a grossly oversimplified assumption in the absence of information. | 11) Changed threshold from 1.5x to 1.0x historical high catch |
| **3. Enhance management performance** | **3.1 Maximise willingness to comply with the harvest strategy** | | We assume that willingness to comply with the harvest inversely scales with management complexity; that is, the more management controls (here, the number of TACs by species, region, and sector), the higher the lack of compliance. The relative "complexity fail" score is the ratio of the number of management controls to the maximum possible. We also consider the lack of compliance because of people actively disagreeing with the harvest strategy, and assume this is normally distributed about a target combined (across all species) TAC. That is, the further the TAC is from the target, the lack of compliance increases. The performance indicator is calculated by taking a weighted average of these two terms and subtracting from 1. | We assume a target combined TAC of 4,500t and a standard deviation of 1000. It is currently assumed that the "complexity fail" and the "disagree fail" terms are weighted 0.4 and 0.6, respectively. The former pertains to inadvertent mistakes; the latter is an active disregard due to disagreeing. | 12) Weighting on "disagree fail" term changed from 0.6 to 0.7 (i.e. "complexity fail" term weighting changed from 0.4 to 0.3) |

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| **4. Maximise social outcomes** | **4.1 Maximise equity between recreational, charter, indigenous and commercial fishing** | **4.1.1 Increase equitable access to the resource** | Equitable access is approximated as the extent to which the catch proportion by sector and species conformed to the specified (fixed) allocation fraction. The deviation from equitable access is defined using a "hockey stick" relationship, with a deviation threshold above which the fleets are dissatisfied, set at 20% (deviation above this = 1), and a deviation tolerance below which the fleets are satisfied, set at 2% (deviation below this = 0). The performance indicator is one minus the average deviation across species groups and sectors. | The allocation fraction, and the deviation tolerances, are assumed and are fixed through time. Given that the TAC is divided according to these allocation fractions, and that there is currently no error in the model, there should not be deviations at least for the commercial sector. | 13) Deviation threshold changed to 10% and tolerance to 1% |
| | **4.2 Improve social perceptions of the fishery (social licence to operate) (rec, commercial, charter, indigenous)** | **4.2.1. Through sound fishing practices, minimise adverse public perception around discard mortality (compliance with size limits, environmental sustainability, and waste)** | We already have indicators of discarding (1.2.2) and TEPS (1.2.4). We recast these performance indicators so that the higher their value, the lower the risk. For the TEP risk, the perception is 0 when the risk is 0, and rises linearly with risk to be 0.2 when the risk is 10%. At and above a risk of 10%, the perception again linearly increases, from 0.2 to 1.0 at 50% risk. Above 50% risk, the TEPS "perception score" is 1.0. For the discarding risk, we assume a "saturation" relationship, where there is no concern below 50% risk, with a linear increase in perception (concern) above this. We then take a weighted mean of the two perceptions and subtract this from 1 to obtain the performance indicator. | The nature of the perception relationships, together with their threshold/asymptotic values, are assumed. The perceptions around discarding and TEPS are weighted 0.7 and 0.3, respectively. The stronger weighting on discarding is due to a greater public awareness of this relative to any awareness of the fishery interacting with TEPS. | 14) Changed TEP risk threshold and limit to 0.3 and 0.05 (from 0.5 and 0.1), respectively, and discard risk asymptote to 0.3 (from 0.5) |
| | | **4.2.2. Maximise utilisation of the retained catch of target species** | It was agreed that this objective is outside of the mandate, and control, of a harvest strategy. We moved this to a broader "management regime objective" as opposed to a harvest strategy objective and renormalised the objective preference weightings to exclude this objective. | | |
| | | **4.2.3 Through achievement of objectives 1.1 and 2.3, maximise the potential for** | The concept here is that if the fishery is sustainable, with positive flow-on community benefits, public perception will be high. We assume the potential for | Each of the three contributing performance indicators is currently equally weighted. | 15) Changed weights from equal, to 0.5 CT & RTE, 0.3 |

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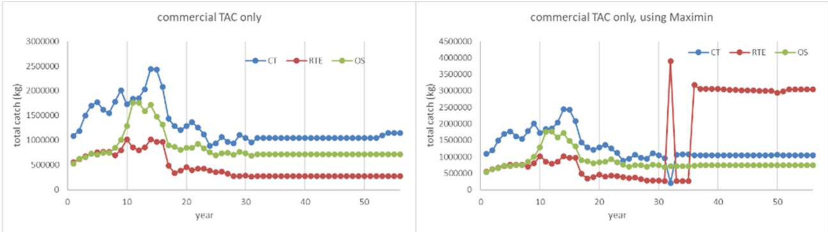
| | | **fishing to be perceived as a positive activity with benefits to the community (commercial, rec, and charter)** | fishing to be perceived as a positive activity scales directly with objectives 1.1.1 (CT and RTE sustainability), 1.1.2 (OS sustainability), and 2.3 (flow-on economic benefits), and take an average across them. | | OS, 0.2 flow-on economic benefits |
| | **4.3 Enhance the net social value to the local community from use of the resource** | **4.3.1 Increase access to local seafood (all species)** | This is a function of the non-exported commercial and charter landings (= dead CT, plus all RTE and OS catch). We assume some fixed proportion of live to dead CT (currently, that 10% of CT catch is non-live). We assume the performance indicator value is 0 if the local available domestic percentage is <20%, and 1 if the local available domestic percentage achieves that from the past, assumed to be equal to 0.5. We assume a "hockey stick" relationship between these two thresholds. | The nature of the relationship, together with their threshold values, are assumed, as is the percentage of dead CT. | 16) Changed to assume 30% dead CT (rather than 10%), a past local availability of 0.7 (rather than 0.5), and the threshold local availability to be 0.4 (rather than 0.2) |
| | | **4.3.2 Maximise spatial equity between regions or local communities** | We assume the equitable proportions of catch (by weight) by area are those of the relative average biomass across species groups. We compare relative regional catches to the equitable proportions using a distance function. The deviation threshold, above which the area is “unhappy”, is set at 20%. The deviation tolerance, below which the area is “happy”, is set at 5%. The absolute percent difference between the relative catch by area and the equitable proportion is calculated, and a "hockey stick" relationship is assumed between the two thresholds. If at least one region yields no catch, then the performance indicator value is 0. Otherwise, the performance indicator is one minus the region-averaged spatial allocation deviation. | The definition of spatial equity, the nature of the relationship, and the threshold values, are assumed. | 17) Changed equitable spatial allocation from being directly proportional to relative abundance to 40%/60% for the northern/southern regions; changed deviation tolerance from 5% to 2% and threshold from 20% to 10% |



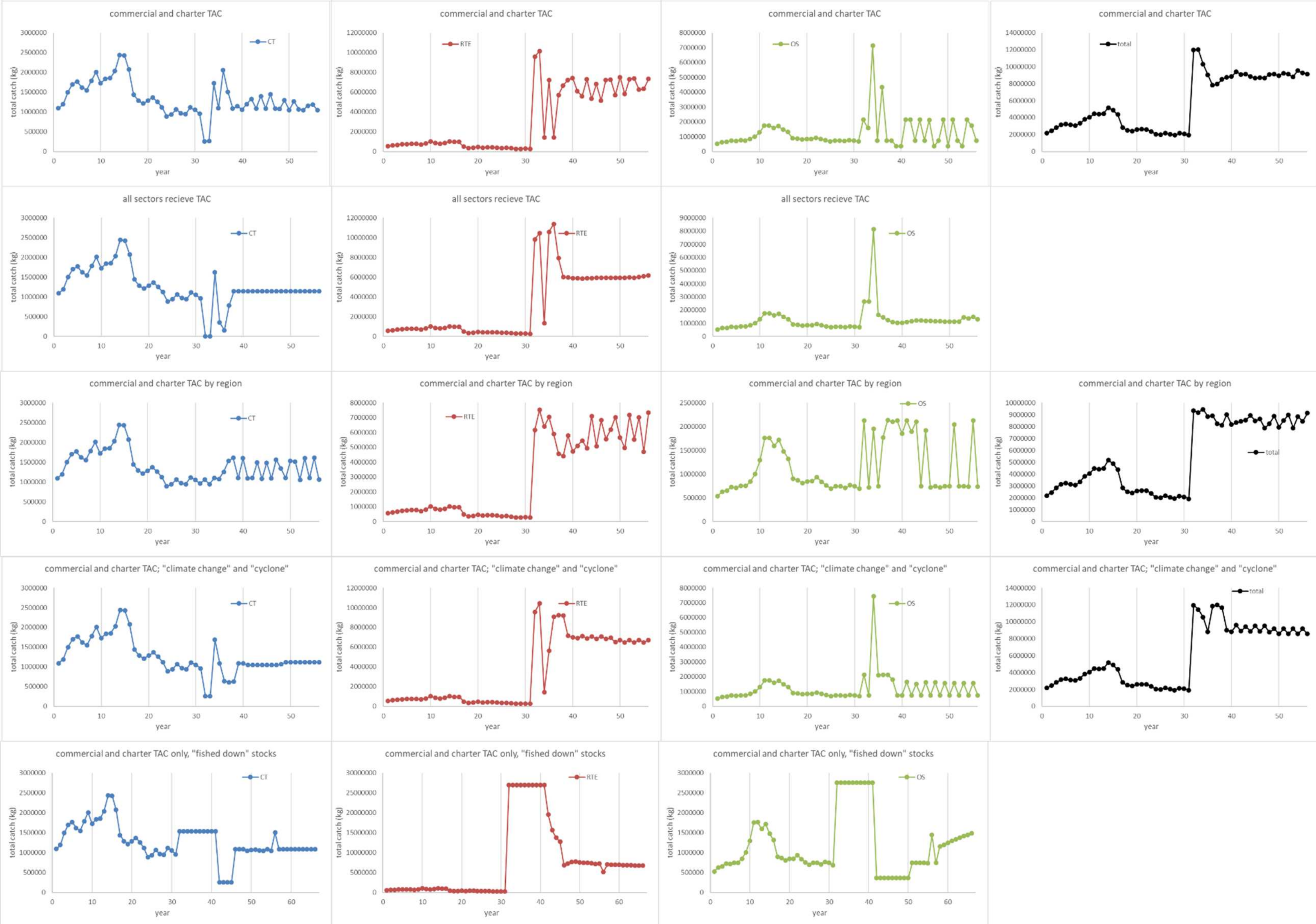
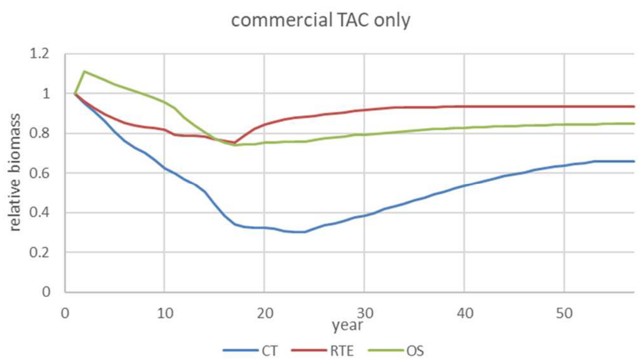
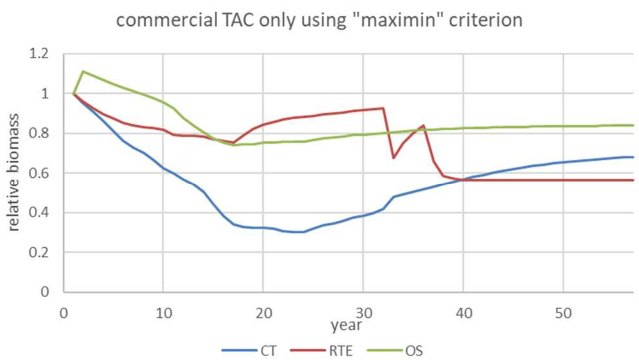
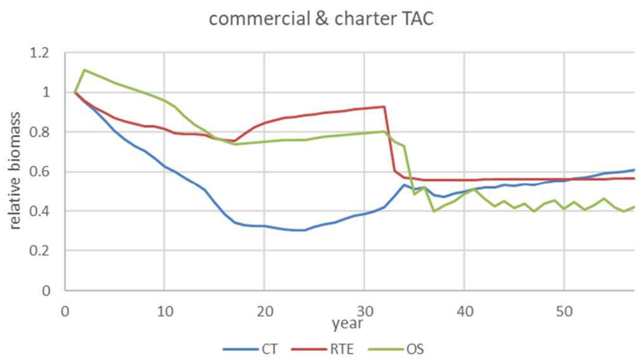
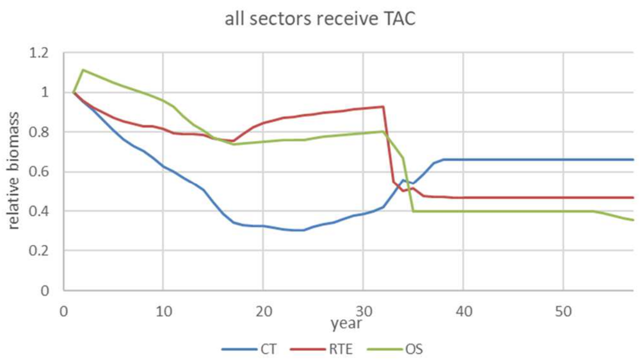



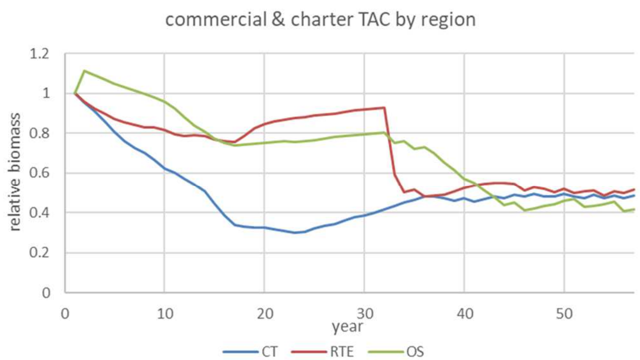

Figure A1: Time series of total catch (kg) for each species group (columns) and scenario (two scenarios in first row; one scenario per row thereafter) considered. For some scenarios, the time series are presented in individual panels for each species, due to differences in magnitude precluding ease of reading if these were overlaid.

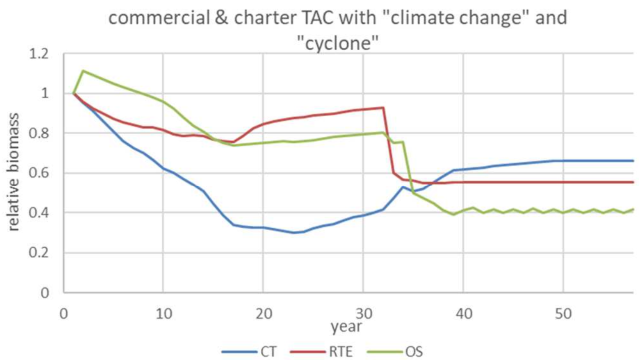

commercial TAC only
relative biomass
year
CT
RTE
OS


commercial TAC only using "maximin" criterion
relative biomass
year
CT
RTE
OS


commercial & charter TAC
relative biomass
year
CT
RTE
OS


all sectors receive TAC
relative biomass
year
CT
RTE
OS


commercial & charter TAC by region
relative biomass
year
CT
RTE
OS


commercial & charter TAC with "climate change" and "cyclone"
relative biomass
year
CT
RTE
OS

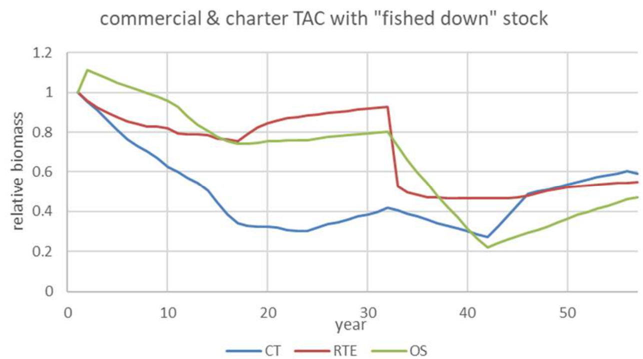

commercial & charter TAC with "fished down" stock
relative biomass
year
CT
RTE
OS

Figure A2: Time series of biomass, relative to the initial year, for each species group and scenario considered.

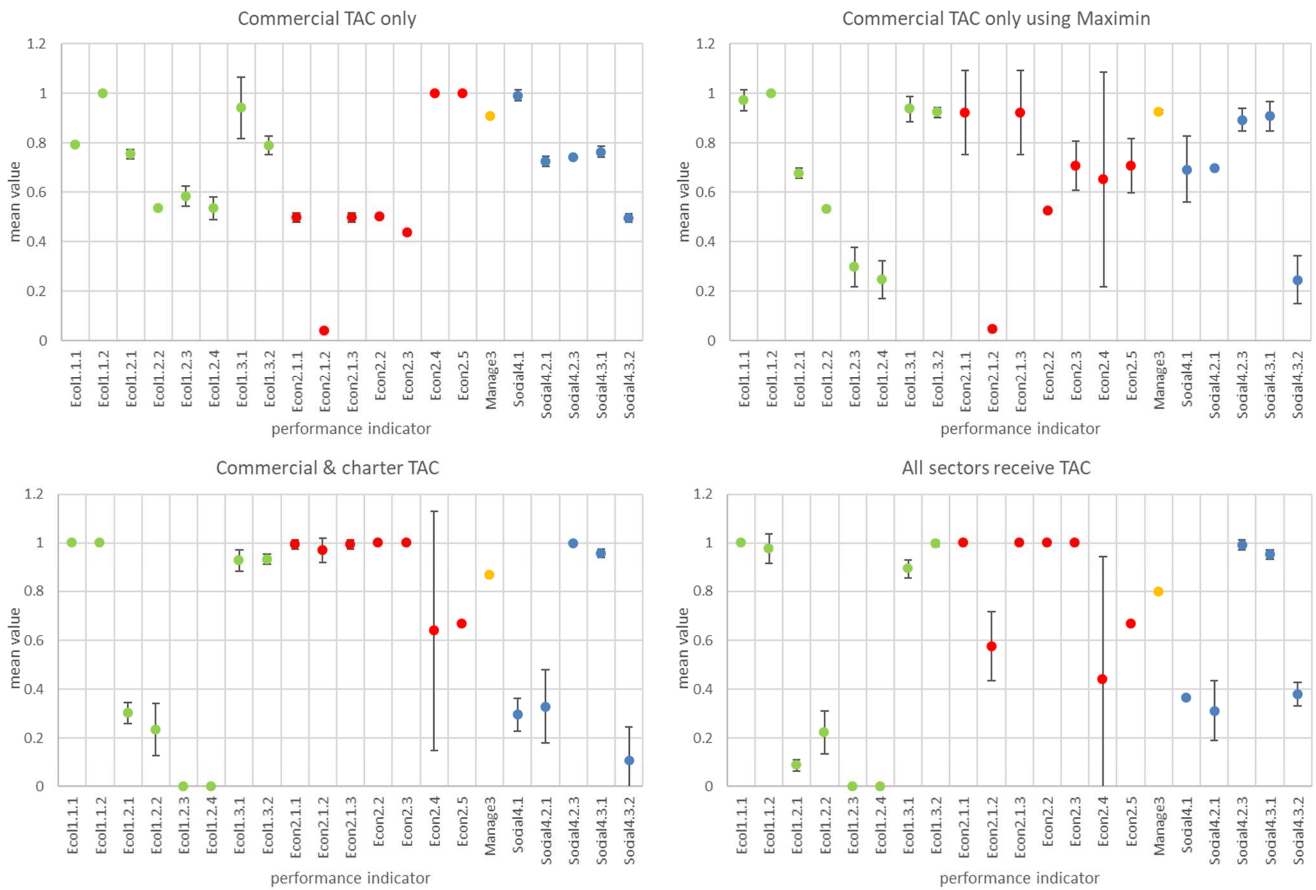



Figure A3: Mean, plus and minus one standard deviation, of each of the 21 performance indicators, for each scenario examined