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Operationalizing triple bottom line harvest strategies

Catherine M. Dichmont [1]*, Natalie A. Dowling[2], Sean Pascoe [3], Toni Cannard [3], Rachel J. Pears[4], Sian Breen[5], Tom Roberts[5], George M. Leigh[5], and Marc Mangel[6,7]

[1]*Cathy Dichmont Consulting, Bribie Island 4507, Australia*
[2]*CSIRO Oceans and Atmosphere, Hobart, Tasmania 7000, Australia*
[3]*CSIRO Oceans and Atmosphere, Queensland Biosciences Precinct, 306 Carmody Rd, St Lucia, Brisbane, Queensland 4067, Australia*
[4]*Great Barrier Reef Marine Park Authority, PO Box 1379, Townsville, Queensland 4810, Australia*
[5]*Department of Agriculture and Fisheries, Fisheries Queensland, GPO Box 46, Brisbane, Queensland 4001, Australia*
[6]*Department of Applied Mathematics and Statistics, University of California, Santa Cruz, California, CA 95060, USA*
[7]*Theoretical Ecology Group, University of Bergen, Bergen 9020, Norway*

**Corresponding author: tel: +61 7 34086407; e-mail: cathydichmont@gmail.com.*

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Over the past 50 years, the diversity of fisheries types being actively managed has changed from mainly data-rich, industrial sectors to more socially, economically, and environmentally complex multispecies and multisector fisheries. Accompanying this change has been a broadening of management objectives to include social and economic considerations with traditional resource sustainability objectives, the so-called triple bottom line, and the need to include these considerations into harvest strategies. The case of a line fishery in Australia's Great Barrier Reef is used as a demonstration of the first steps in implementing triple bottom line harvest strategies. This fishery has several disparate sectors including commercial, tourism, and recreation; targets multiple but important reef species; and is undertaken in a World Heritage Site. This work highlights the need for a much-expanded set of objectives elicited from stakeholders that are either included in the trade-off analyses of the different harvest strategies or directly in an optimization. Both options demonstrated that a paradigm shift is required to emphasize representative participatory management systems that assemble teams from quite different backgrounds and viewpoints; use much broader set of objectives; and modify tools and (especially) the data collected within revised monitoring programmes to underpin these tools.

Keywords: harvest strategies, Hjort, social, triple bottom line

Drivers of change

Over the past 50 years, the diversity of fisheries types being actively managed has changed from mainly data-rich, industrial sectors to more socially, economically, and environmentally complex multispecies and multisector fisheries. Fisheries science and formal management grew out of applications to industrial sectors that focused on one, or only a few, target species in data-rich environments and applied to only a small fraction of the tens of thousands of fisheries worldwide (Hilborn and Ovando, 2014; Costello *et al.*, 2016). Scientific theory concentrated on sustainability and understanding the biology of key species and how these interact with their environment. This is demonstrated in Johan Hjort's work, which has been a large influence on global biological theory (Hjort, 1914) that relates the environment, recruitment, and adult survival of fish species. Fisheries management, largely driven by scientific advice, focused on resource sustainability and maximizing potential yields from the resource (Hilborn, 2007).

The increased complexity of fisheries has been further exacerbated by evidence that the changing climate is impacting our fisheries' environment. In Queensland, extensive coral bleaching and loss of coral habitat of the Great Barrier Reef (GBR) was recorded in 2016 and 2017 (Hughes *et al.*, 2018). In addition, a study of cyclone impacts on reef fisheries examined the impacts from three "severe" Tropical Cyclones in Queensland in successive years (Cyclones Hamish in 2009, Ului in 2010, and Yasi in 2011) and one less-severe cyclone (Cyclone Dylan in 2014), which produced significant wave heights of over 3 m combined with peak-energy wave periods of over 10 s on the GBR. These cyclones were followed by major falls in fishery catch rates of the primary target species group coral trout (CT), in contrast to the relatively benign previous 12-year period (1997–2009) during which no cyclone had a major effect on the fishery (Courtney *et al.*, 2015; Bureau of Meteorology, 2019; Queensland Government, 2019). Major cyclones damage the coral habitat of reef fishes; however, CT catch rates recovered strongly within 1 or 2 years, implying that the acute effects of these cyclones were on availability or catchability rather than causing large mortality of CT. Babcock *et al.* (2019) provided empirical evidence that "extreme climate events" like marine heatwaves and floods in Australia have been attributed to climate change, and that these have profound impacts on habitat forming species such as corals, whereas the East Australia Current, based on climate model projections, is projected to transport greater volumes of water southward (Hobday and Lough, 2011) with resultant shifts in some species' distributions.

The focus of fishery stock assessment has traditionally been on estimating stock status and on understanding the species biology. Assessments have been geared to long-term sustainability; that is to an environmental objective—an important aspect of successful fisheries management. Broader ecological, economic, and social objectives have generally been secondary considerations. However, over time the drivers in this objective space have changed through stakeholder and thus management demands. For example, the Queensland Government Department of Agriculture and Fisheries' Sustainable Fisheries Strategy, released in 2017, mandates that "Queensland will use a harvest strategy to set out pre-determined management actions in a fishery for defined species (at the stock or management unit level) necessary to achieve the agreed ecological, economic and/or social management objectives. Fishery objectives should set out the direction and aspirations for the fishery. These should include ecological, social and economic objectives (often referred to as triple bottom line objectives)" (Department of Agriculture and Fisheries, 2017).

In the United States, the Magnuson-Stevens Fishery Conservation and Management Act (1996) mandates that "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry". (National Standard 1) and "Conservation and management measures shall, consistent with…conservation requirements…(including the prevention of overfishing and rebuilding of overfished stocks), take into account the importance of fishery resources to fishing communities by utilizing economic and social data…in order to a) provide for the sustained participation of such communities, and b) to the extent practicable, minimize adverse economic impacts on such communities" (National Standard 8).

Therefore, whether by policy mandate or stakeholder consensus, there is a growing need for fishery management to expand beyond the biology of target species, to take a whole-of-environment/ecosystem approach [e.g. Ecosystem Based Fishery Management (EBFM)], and to address economic and social factors (Rindorf *et al.*, 2017; Stephenson *et al.*, 2017). But, how can one operationalize "triple bottom line" (TBL) (Elkington, 2006) management in a changing paradigm from single-species stock assessment and management focused around sustainability, to the environmental, economic, and social management of complex, often data-limited multispecies/multisector fisheries that are robust to environmental climate drivers and social demands? One such way is to expand the single-species harvest strategy (HS) approach to those focused on TBL objectives, even though there are few cases of where TBL HSs have been actively applied (Mangel and Dowling, 2016).

Some definitions

HSs (aka management procedures) have been applied to data-rich and valuable fisheries for several decades in countries such as South Africa and Australia. HSs include pre-agreed monitoring, performance indicators (which are 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 (Sainsbury *et al.*, 2000; Punt *et al.*, 2002; Butterworth and Punt, 2003). HSs have been developed and advocated more globally, including in ICES (e.g. Kell *et al.*, 2005; Froese *et al.*, 2011). HSs are best tested in simulation using approaches such as management strategy evaluation (MSE) (Sainsbury *et al.*, 2000; Punt *et al.*, 2016).

The concept of the TBL is attributed to Elkington (1997) and pertains to reconciling economic, ecological (environmental), and social objectives and reporting performance in this context. A single decision maker explicitly values, and (purportedly) optimizes over, the different objectives. In practice, Stephenson *et al.* (2017) and Pascoe *et al.* (2013) describe four "pillars of sustainability", with the fourth being institutional, or managerial, objectives. These typically equate to simplifying and improving management structures.

Possible operational approaches

There are many tools applied within the environmental and other research domains that could have, and have been, applied in fisheries. In an operational context, Benson and Stephenson (2018)'s review of TBL methods found that two of seven proposed tools to support decision making in the management system could provide tactical advice, but only MSE provided advice that was consistent with their criteria for generation, transmission, and use of scientific information in management advisory processes. Furthermore, formal methods that acknowledge the TBL result in discrete strategies do not consider stakeholder's weightings (preferences) and provide no formal means of determining the optimal solution given these weightings. Even in MSEs that aim to include more than just sustainability issues (e.g. Plagányi *et al.*, 2012), there is no means to formally make recommendations that reconcile different stakeholder groups.

The importance of stakeholder preferences was illustrated by Pascoe *et al.* (2013)'s assessment of relative importance of the different objectives to various stakeholder groups in the Queensland Eastern Trawl Fishery, Australia, using the analytic hierarchy process (AHP). AHP uses a hierarchy of performance indicators and a set of questionnaire responses to calculate respondents' preference weightings for the different indicators. A different weightings vector was derived for each stakeholder group. Across stakeholder interest groups, preference weightings showed a fourfold difference in economic outcomes, twofold in social outcomes, and almost twofold in environmental outcomes. This motivates the need to reconcile weightings (priorities), and therefore, TBL HSs, across interest groups.

TBL HS evaluation may be either qualitative or quantitative. Qualitative risk assessments can take the form of multicriteria decision analysis (MCDA), qualitative models, such as Bayesian

Belief Networks, or intuitive forecasting methods, including the MCDA approach (a polling technique for systematic solicitation of expert opinion). Ecosystem risk assessments (ERA) may be used to determine whether proposed management tools, such as marine parks, may achieve the desired objectives, see for example Read and West (2010). Dichmont *et al.* (2013) used an expert group to develop different governance strawmen (or HSs), which were assessed by a group of industry stakeholders and experts using MCDA techniques against the different objectives. Read and West (2010) assessed the effectiveness of managed-use zones in six multiple-use marine parks located within NSW using qualitative ERA. Pascoe *et al.* (2009) present a qualitative framework that aids in the analysis of alternative spatial management options in coastal fisheries. The framework combines expert opinion and the AHP to determine which options perform best, taking into account the multiple objectives inherent in fisheries management.

Quantitative approaches that may be taken to evaluate TBL HSs require some unit of measurement. For example, all outputs may be expressed in dollar terms such as a cost benefit analysis (e.g. Freese *et al.*, 1995), or in utility terms such as multi-attribute utility analysis (e.g. Healey, 1984). Alternatively, non-commensurable units can be scaled with explicit objective weights, for example using a goal programming bioeconomic model (e.g. Charles, 1989; Pascoe and Mardle, 2001); separate outcomes can be generated for each objective in a hybrid model or simulation approach (e.g. Mapstone *et al.*, 2008; Little *et al.*, 2016); or coviability analysis can be employed (Gourguet *et al.*, 2013, 2016).

However, despite the extensive availability of tools, the process of operationalizing TBL HSs, beyond a simple conceptualization, remains complex. To embed the TBL in formal management, each of the TBL objectives need 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 HS, 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 lever used within the HS, for example catch or effort.

Methods

The case of a line fishery on Queensland's GBR is used as a demonstration of the first steps in implementing TBL HSs. This fishery has several sectors with disparate commercial, tourism, and recreational objectives and values related to fishing. The fishery targets multiple important reef species and is undertaken in a World Heritage Site.

The article highlights two alternative approaches: a semi-quantitative "MCDA" approach to compare broad management options or reforms and a quantitative, non-commensurable units simulation model, via a multi-indicator objection function, with explicit objective weights to set total allowable catches (TACs) for the three main species groups (the "objective-function" approach).

TBL objectives are included either in the MCDA trade-off analyses of the different HSs using stakeholder input or in the objective-function approach, directly in an optimization model. The emphasis of the article is not on the approaches themselves, which are described in detail elsewhere (Pascoe *et al.*, 2019; Dowling *et al.*, Submitted), but on comparing these two approaches to illustrate the challenges of operationalizing TBL HSs, the extra insights obtained by using the two approaches in parallel, and to demonstrate the need for a paradigm shift in fisheries management. The MCDA approach is best described using a flow diagram (Figure 1). The objective-function approach utilizes the same information in the first four steps of the MCDA approach and only then departs by combining steps 5–7 within a simulation model (see Supplementary Material S1 for details on the model).

The Queensland Coral Reef Finfish Fishery operates mostly within the GBR Marine Park, spanning a broad latitudinal range from the tip of Cape York to Bundaberg (24°30′S) in the south. In order of decreasing value, the commercial sector mainly targets several species of CT (*Plectropomus* and *Variola* spp, "CT"), of which *Plectropomus 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 other cods (mainly Serranidae), other emperors (Lethrinidae), and tropical snappers (mainly Lutjanidae), landed as dead whole fish (Thébaud *et al.*, 2014). In addition to the commercial sector, there is a large, valuable, and iconic recreational fishery, a charter fishery for tourists and locals, and a small indigenous fishery.

Commercial fishery operations use hand-held lines with baited hooks and range from single, small vessels that take short (12–48 h) trips to small fishing dories (or tender boats) operating from a larger mother vessel that undertake trips of up to 2.5 weeks. A range of targeting strategies are deployed, with some boats fully dedicated to live CT capture, whereas 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 TAC allocated via individual transferable quota units, tradability of input and output entitlements, and seasonal spawning closures. A fishery specific Working Group, formed to help implement a new HS for the fishery that aligns with the new Queensland Policy, consisted of stakeholders from the commercial, recreational, and charter industries, conservation groups, managers, and scientists.

Identification of objectives

Reviews of objectives previously applied in Australian fisheries (Pascoe *et al.*, 2013, 2014; Brooks *et al.*, 2015; Jennings *et al.*, 2016; Farmery *et al.*, 2019) identified a wide range of different potential objectives, each of which fell in one of the four TBL categories: ecological/environmental, economic, social, and institutional/management. A series of workshops was held with members of the Working Group to iteratively identify a final set of 21 TBL objectives that were of most relevance to the fishery. Objectives that were removed by the Working Group fell into one of two categories. First, some objectives were not relevant to the specific fishery or could be subsumed into another (more relevant) objective and second, some objectives were important to fisheries management, but would usually not be controlled through a HS. An example of the former was "Minimize human induced changes in water flow regimes" and the latter "Ensure fisheries collected data is available in a timely and publicly accessible manner". Approximately 20 different individuals in total were involved in the discussions, the majority of whom attended all sessions. Each conceptual objective was translated into an operational objective. To be considered operational, the objectives needed to be measurable and simulation-achievable, with quantitative performance indicators against which each objective could be assessed.

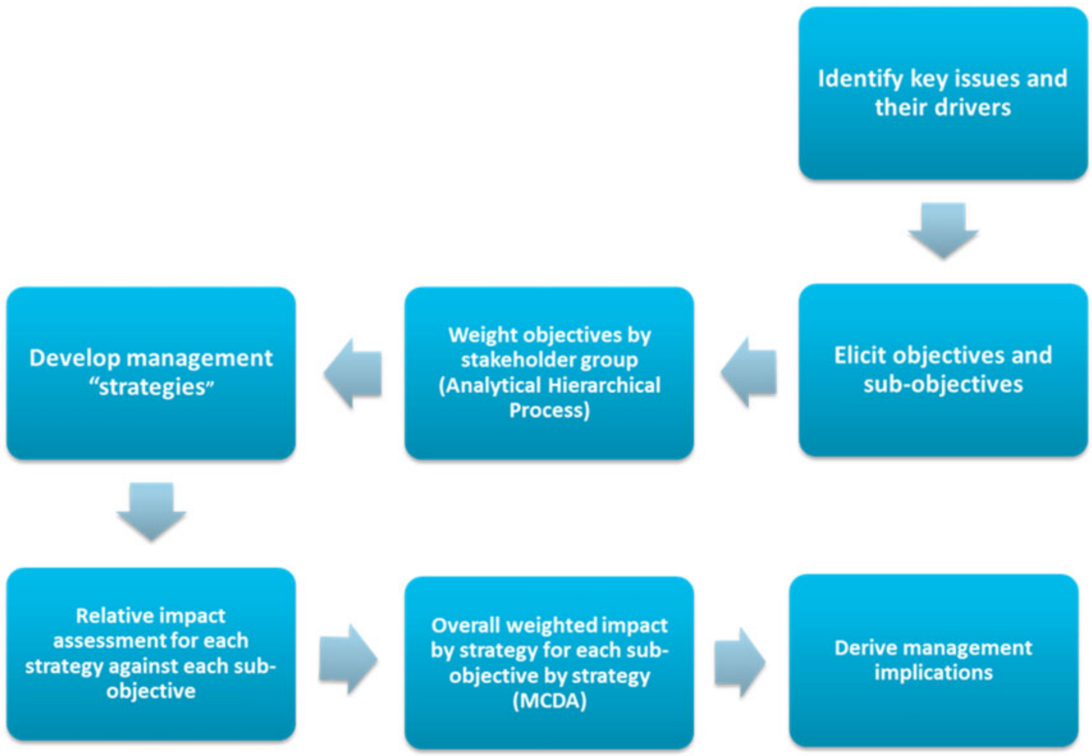

Figure 1. Steps to undertake a semi-quantitative approach to assessing difference harvest strategies.

Objective weighting

The method used here is fully described in Pascoe *et al.* (2019). In summary, a modified version of the analytic hierarchy process (AHP) (Saaty, 1980) was undertaken, through an online survey of 110 fishery stakeholders (including Working Group members) to elicit preference weights. The method used relative importance comparisons between each set of objectives at each level of the hierarchy and therefore ultimately produced relative weights by stakeholder at each level. These objectives and their relative weightings were used in both the MCDA and the objective-function approaches below.

Identification of potential HSs

A range of potential HSs were developed in collaboration with the Working Group through a further series of workshops. The "modified status quo" option included a target reference point for the stocks of 60% or more of the unexploited biomass, which is consistent with the Queensland Sustainable Fisheries Strategy (Department of Agriculture and Fisheries, 2017). This reference point was taken as a proxy for the biomass at maximum economic yield and to provide biological resilience to the stock. The "modified status quo" applied a commercial TAC to each of CT and red-throat emperor but managed the charter and recreational sectors through input controls such as season and bag limits, which are less flexible.

Alternative HSs comprised additional components to the "modified status quo", as suggested by Working Group members, to enhance at least one broader objective:

(i) a separate allocation of a TAC to the charter sector, aimed at enhancing the economic performance of this sector and social benefits accruing to the recreational fishers using these vessels (called "charter HS", important to charter industry stakeholders);

(ii) the use of environmental "overrides" where TACs are adjusted in response to a spatially or temporally isolated weather events (e.g. a tropical cyclone or coral bleaching) to enhance ecological outcomes and long-term sustainability (especially important to conservation-group stakeholders);

(iii) a combination of environmental overrides and spatially explicit management, where responses to the catastrophic event may vary in different areas of the fishery, and to ensure some form of spatial equity across regional communities (important to both conservation-group and fishing-industry stakeholders);

(iv) separate TACs for some key OS quota species currently not under TAC control (of some importance to all stakeholders, because of potential targeting of these species); and

(v) separate TACs for different CT species also presently not under individual TAC control (suggested by scientists to protect the less common CT species).

Performance of HSs against the objectives

MCDA approach

The assessment of the perceived performance of the above additional HS components was undertaken via a second survey to elicit "expert" opinion as to their effects on the objective outcomes. This survey mainly targeted those who had taken part in developing the set of management objectives and HSs, and most likely had expectations about how they would perform. This group consisted of scientists, conservation-group representatives, fishery managers, and commercial, recreational, and charter fishery operators. This group had already been exposed to the project's discussions around each of the HSs developed; hence, members were in a good position to assess the outcomes of the strategy against each of the identified objectives given their own knowledge and experience.

Respondents rated each potential HS option relative to the baseline of "modified status quo", against each of the 21 objectives on a scale ranging from of "Much worse than the baseline" to "Much better than the baseline". The full mathematical formulation is described in Pascoe *et al.* (2019): the outputs are relative frequencies of these scores that can be used to derive a subjective probability distribution of the expected benefits of each HS, which accounts for heterogeneity in the impact scores and the objective preference weightings. As they are based on distributions of opinion and expert judgement, as opposed to real-world outcomes, the probability distributions are subjective rather than objective.

Objective-function approach

To more quantitatively evaluate the TBL, a simple "proof of concept" simulation, approximating the three main species groups in the fishery (CT, RTE, and OS) was developed. The fishery and resource were split into two areas. No movement of fish between the areas, constant area-specific recruitment, and an equal distribution of recreational fishing mortality between areas were assumed. The method is described in detail in Dowling *et al.* (submitted) and summarized here. Catch time series for all three species groups and all three sectors from the past three decades of the fishery were used as a start-up period so that the relative biomasses of CT and RTE were similar to a stock assessment (Campbell et al., 2019) over the same period. Thereafter, the optimization process was used to determine the TAC for each species group for a subsequent 25 years, and therefore the final fishing mortality by species and region. In the case of the "modified status quo", the fishery is mainly managed through three (species-group-specific) commercial TACs, whereas in the "charter HS", additional sets of charter-specific TACs were applied, in that the optimized TAC from the model in any year was allocated across both the commercial and the charter sectors in fixed proportions (90% and 10%, respectively).

The model optimized a value function over a range of possible TAC levels for a given set of stakeholder group weightings. This approach could be used to test any HS decision rule, but here, it was limited to determining optimal species-specific TACs across TBL objectives for the "modified status quo" and the "charter HS". Given this is a test case, the simulation assumed perfect knowledge. Ultimately, it will be further developed with the different HS-specific models and tested within the ELFSIM MSE software (Little *et al.*, 2007, 2016) which was specially designed for the line fishery on the GBR. It could be run with a sampling model and an underlying assessment relevant to each HS via which to estimate biomass by species and area.

As described above, taking the same line as Richerson *et al.* (2010) and Munch *et al.* (2017), a quantitative performance indicator was defined for each of the 21 TBL operational objectives, each of which needed to be a function (directly or indirectly) of the management lever; in this case, the commercial or sector level TAC. A description of how these objective outcomes related to the model outcomes is given in the Supplementary Material S1. Each of the TBL objectives was denominated in different units (e.g. effort, catch, and value) and so was normalized from 0 to 1, to make the performance metrics commensurate (Richerson *et al.*, 2010).

The corresponding stakeholder preference weighting to each performance indicator was applied and summed to obtain an overall value function for each stakeholder group:

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

where $V_{g,y}$ is the value function for stakeholder group g in year y, $PI_{j,y}$ is the value of performance indicator j in year y, and $Wt_{j,g}$ is the preference weighting of performance indicator j by stakeholder group g. In each year of the simulation projections, an optimization was used to find the species-specific TACs that maximized the value function for the relevant stakeholder group, $V_{g,y}$ (Mangel and Dowling, 2016).

Given the optimum strategy (TACs) for each stakeholder group's weightings, the value functions for every other set of stakeholder group weightings were calculated. For each year, this gave a matrix of values according to each set of stakeholder group weightings, calculated using the performance indicators derived from the optimal TACs (the optimal strategy) for each stakeholder group. Each row represents one stakeholder group's optimal strategy, which was applied to each stakeholder group's preference weighting, by column:

$$\begin{bmatrix} V_{11} & \cdots & V_{1n} \\ \vdots & \ddots & \vdots \\ V_{n1} & \cdots & V_{nn} \end{bmatrix}.$$

The columns of this matrix are standardized relative to the value for the stakeholder group for which the strategy is optimal (such that the diagonals equate to 1).

Two alternative criteria are used to select the overall optimal TAC: (i) the highest average value across all stakeholder weightings (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 row of the matrix that has the highest minimum value across the row, indicating that this strategy results in the "minimum whinge" across all preference groups).

In principle, this level of output is available for all the HSs and the base case ("modified status quo"), which could then be converted into a result format similar to the MCDA approach if required.

Results

Objective preference weightings

The final set of objectives and objective preference weightings by stakeholder group are shown in Table 1 and Figure 2 respectively. Objective 4.2.2 from Table 1 was initially included but was later discarded as it was regarded as beyond the control of the HS. Of the 21 final TBL objectives, those pertaining to ecological sustainability consistently ranked the highest. Pascoe *et al.* (2019) provide a complete analysis, but, in general, commercially oriented groups (commercial fishers, quota owners, and buyers) rated economic

Table 1. Objective hierarchy identified with the Coral Reef Finfish Fishery Working Group.

Main objectives (level 1)	Subobjectives (level 2)	Operational objectives (level 3)
1. Ensure ecological sustainability	1.1 Ensure resource biomass sustainability	1.1.1 As per the Queensland Sustainable Fisheries Strategy, achieve B_{MEY} (biomass at maximum economic yield) (∼60% unfished biomass or defensible proxy), by 2027 for the main commercial, charter, and recreational species; if below biomass at maximum sustainable yield, B_{MSY}, aim to achieve B_{MSY} (∼40–50% B_0) by 2020
		1.1.2 Minimize risk to other species in the fishery, which are not included in 1.1.1
	1.2 Ensure ecosystem resilience	1.2.1 Minimize risk to bycatch species
		1.2.2 Minimize discard mortality of target species (e.g. high grading)
		1.2.3 Minimize broader ecological risks
		1.2.4 Minimize risk to protected species
	1.3 Minimize risk of localized depletion	1.3.1 Due to fishing
		1.3.2. In response to environmental events (e.g. cyclone)
2. Enhance fishery economic performance	2.1 Maximize 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 Maximize value of recreational fishers and charter experience (direct to participant)	
	2.3 Maximize flow-on economic benefits to local communities (from all sectors)	
	2.4 Minimize short-term (interannual) economic risk	
	2.5 Minimize costs of management associated with the harvest strategy: monitoring, undertaking assessments, and adjusting management controls	
3. Enhance management performance	3.1 Maximize willingness to comply with the harvest strategy	
4. Maximize social outcomes	4.1 Maximize 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) (recreational, commercial, charter, and indigenous)	4.2.1 Through sound fishing practices, minimize adverse public perception around discard mortality (compliance with size limits, environmental sustainability, and waste)
		4.2.2 Maximize utilization of the retained catch of target species
		4.2.3 Maximize the potential for fishing to be perceived as a positive activity with benefits to the community (commercial, rec, 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 Maximize spatial equity between regions or local communities

objectives similar to the ecological sustainability objectives, whereas other groups tended to rate economic objectives lower than ecological sustainability. The preference weightings of charter operators were closer to recreational fishers than commercial fishery weightings. Most groups tended to rate the social objectives the lowest, consistent with most other previous management objective studies in Australia (e.g. Pascoe and Dichmont, 2017).

MCDA approach

The MCDA approach yielded performance scores of the anticipated performance of alternative HS options against the 21 TBL objectives (Pascoe *et al.*, 2019). Only those for the "charter HS" are shown in this article (Figure 3a), as this was found to generally produce positive outcomes across the full range of objectives relative to the status quo (Pascoe *et al.*, 2019). The exceptions to this were for discard mortality (Ecol 1.2.2) (possibly because of a perception of some respondents that the quota will encourage high grading and discarding as in commercial fisheries, although the current recreational trip limits are also essentially a quota) and costs of management (Econ 2.5) (possibly because of a perception of higher monitoring, surveillance, and control costs). The scores originally involved discrete and categorical outcomes on a scale ranging between −3 and 3 but were rescaled in Figure 3a to between −1 and 1 for a better comparison with the model-based objective-function approach results.

In terms of the subjective probability distributions, no single alternative management option was considered an improvement in all cases, with both positive and negative outcomes expected given the variability in expected impact and objective weightings (Pascoe *et al.*, 2019).

Objective-function approach

The translation of operational objectives to quantitative performance indicators involved making many assumptions. This was exacerbated in the face of data paucity, particularly around the

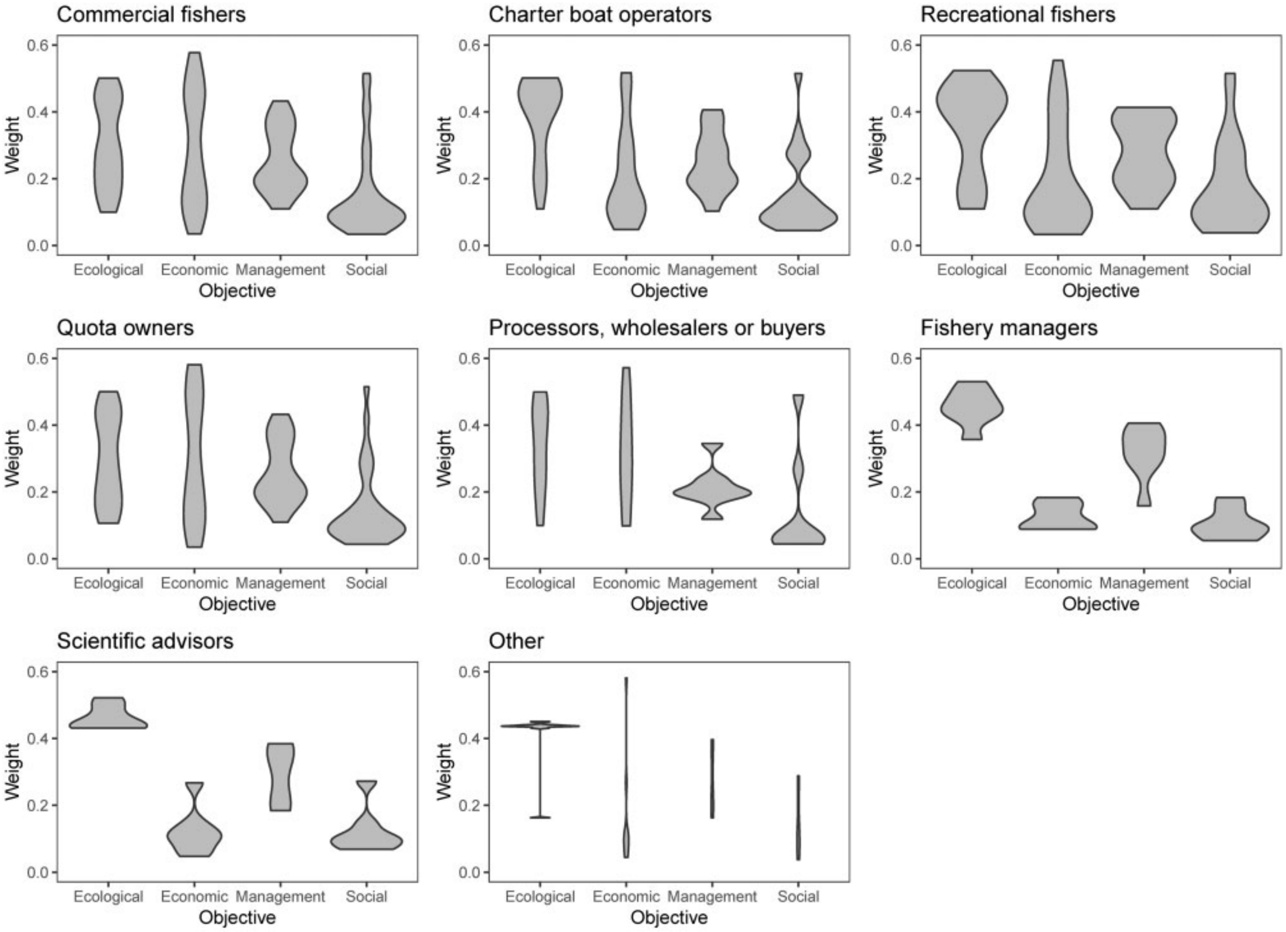

Figure 2. Objective weight distributions by stakeholder group.

non-target species ecological, some of the economic, and most of the social, objectives. The reason is that the performance indicator corresponding to each operational objective (such as social equity and access to local seafood) needed to be some mathematical function of catch or effort (direct or indirect—similar to the traditional "hockey stick" single-species rule). One alternative, simpler approach would have been to only have used the 12 higher-level objectives (Table 1), but these were not explicitly defined to readily equate to quantitative performance indicators, thus the lower level operational objectives and weights were used (as per the MCDA approach).

The relative performance scores for the "Charter HS" relative to the "modified status quo" are shown in Figure 3b. These are in a comparable format to those shown in Figure 3a for the MCDA approach, except that the scores range from between −1 and 1 (because quantitative scores were normalized from 0 to 1). In both approaches, 0 denotes no change between the charter HS and the status quo. Compared with the semi-quantitative approach, the model-based method generally showed minor change relative to the status quo. This stationarity is not that surprising in that the relative TAC allocation to the charter sector was based on historical performance, and therefore produced similar TACs to past unallocated charter catches.

The biggest positive (and overall) outcome was an increase in charter sector profits (Econ 2.1.2), presumably because of the increased catch afforded to the charter sector under a dynamic TAC. There was also a small increase in the value of recreational fishers and charter experience (direct to participant) (Econ 2.2), as a result of the increased charter sector catch. The strongest negative change was in the indicator minimizing short-term (interannual) economic risk (Econ 2.4), because introducing a charter TAC allowed for greater interannual flexibility (whereby interannual variability in TAC occurred, presumably, to attempt to satisfy other objectives). The HS also reduced equity between the sectors (Social 4.1), likely because "equity" was defined as a fixed proportion whereby the recreational and charter sector take are equal. Allowing a charter TAC introduced variability within this sector that was not tracked by the fixed recreational take. Other negative changes were against minimizing the risk to bycatch species, minimizing discard mortality, broader ecological risks, and risk to TEPS (Ecol 1.2.1–1.2.4). As a result, the public perception around discarding mortality worsened (Social 4.2.1). In the MCDA approach, the main negative impact was that management costs were seen as likely to increase (Econ 2.5). Although this was not seen for the simulation result, there was slightly less willingness to comply with the HS (Manage 3). There was an overall increase in variability around the level of access to local

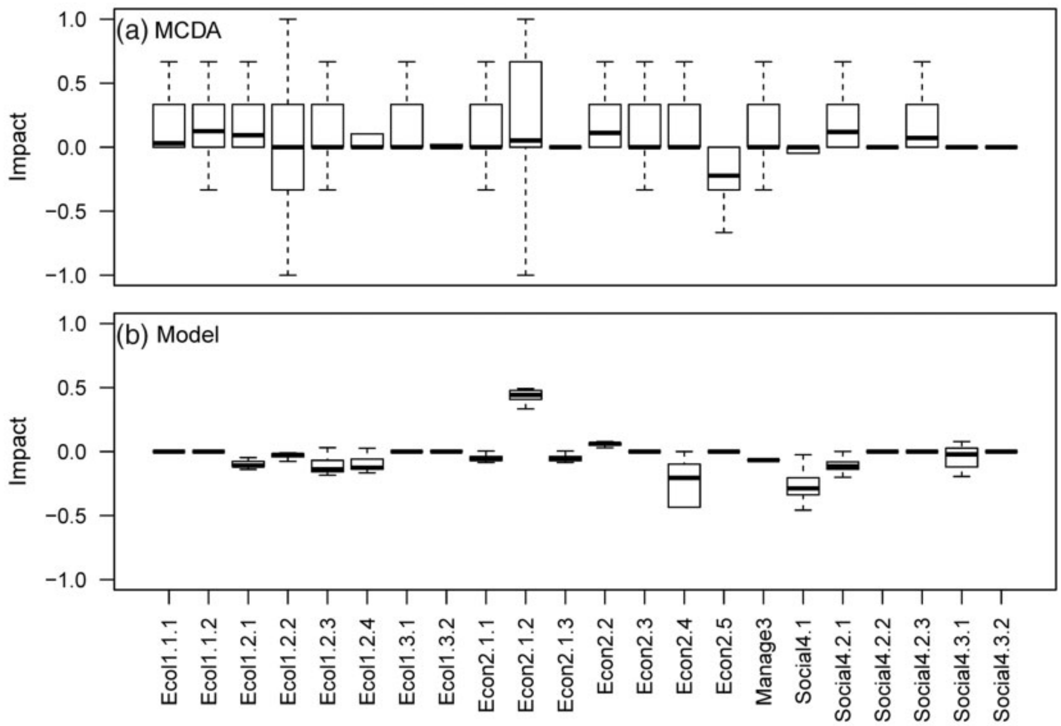

Figure 3. Distribution of the relative impacts of the separate charter harvest strategy for the MCDA (a) and the simulation model (b) approaches.

seafood (Social 4.3.1), presumably as a result of the increased variability in charter catch.

It is important to note the preliminary nature of these results; as for similar simulation approaches, the results are sensitive to the burn-in period catch values; that is the starting biomass and the parameters in the different objective-level control rules. The approach taken here was to run the simulation using historical catches in the earlier period of the fishery and place the biomass at a relative biomass close to that observed by a stock assessment.

Comparison of the outcomes from both approaches

A comparison of the results against each of the separate objectives from the MCDA and the objective-function approaches is presented in Table 2. The mean outcomes, although less meaningful for the MCDA outcomes, are presented for a simple comparison of the results. Given the different structures of the data, a non-parametric approach is required to compare the results. In this instance, a Wilcoxon rank sum test (Wilcoxon, 1945) is used to compare the distributions for outcomes against each objective from the two approaches.

The outcomes from the two approaches were significantly different for the majority of objectives. The MCDA expert-judgement-based outcomes were generally more optimistic against most objectives than the simulation model. The significantly different outcomes (13 of the 21 objectives) are identified in Figure 4. The lower right-hand quadrant of Figure 4 represents objectives that were estimated to improve in the MCDA approach but that declined or remained the same in the objective-function approach. Most of the significant objectives are found in this quadrant, suggesting that the different approaches do not favour one objective set over another, but that there is overall optimism associated with the MCDA approach.

Determining which outcomes are correct is not possible, whereas in principle "objective", simulation models are based on

Table 2. Comparison of the scores derived by MCDA and the objective-function model.

	Means		Wilcoxon rank sum test		
Objective	MCDA	Model	*W*	Pr(*W*)	Sig[a]
Ecol 1.1.1	0.063	0.002	288.000	0.094	–
Ecol 1.1.2	0.125	0.000	312.500	0.008	**
Ecol 1.2.1	0.188	−0.094	425.000	0.000	***
Ecol 1.2.2	0.000	−0.033	300.000	0.066	–
Ecol 1.2.3	0.083	−0.118	391.000	0.000	***
Ecol 1.2.4	0.104	−0.104	414.000	0.000	***
Ecol 1.3.1	0.021	0.001	280.500	0.111	–
Ecol 1.3.2	0.021	0.003	218.500	0.867	–
Econ 2.1.1	0.042	−0.055	367.000	0.000	***
Econ 2.1.2	0.104	0.433	151.000	0.070	–
Econ 2.1.3	−0.063	−0.055	358.000	0.001	***
Econ 2.2	0.111	0.058	250.000	0.546	–
Econ 2.3	0.042	0.000	287.500	0.046	*
Econ 2.4	0.044	−0.362	402.000	0.000	***
Econ 2.5	−0.222	0.000	100.000	0.000	***
Manage 3	0.000	−0.067	375.000	0.000	***
Social 4.1	−0.048	−0.277	354.000	0.002	**
Social 4.2.1	0.119	−0.110	396.000	0.000	***
Social 4.2.2[b]	0.000	0.000	237.500	0.504	–
Social 4.2.3	0.143	0.001	309.000	0.022	*
Social 4.3.1	0.000	−0.044	287.000	0.125	–

[a]"***": significant at 0.1% level; "**": significant at 1% level; "*": significant at 5% level.
[b]This objective outcome was not included in the simulation model.

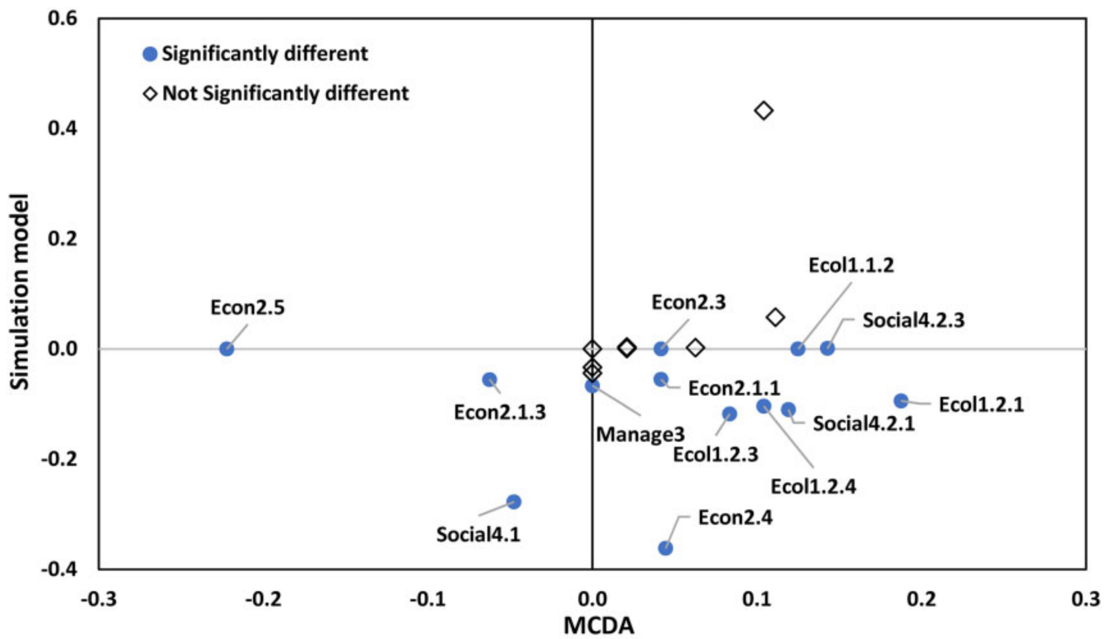

Figure 4. Comparison of the mean scores derived from the two approaches. The labelled (round) points represent objectives with significantly different outcomes using the different approaches. The unlabelled (diamond) points were not significantly different.

a number of assumptions, beliefs, and approximations that may influence the outcomes (Martin *et al.*, 2012). This is particularly the case for assessing some outcomes of management that are less directly quantifiable. For example, assessing social outcomes in the simulation model requires assumptions about the links between other modellable fishery outcomes (e.g. bycatch, income distributions, and fleet structures) and the level of social impact. In contrast, the subjective expert-knowledge-based outcomes may take into account a wider range of factors through use of cognitive models based on experience and a broader knowledge base, but being subjective, may also be influenced by personal bias. For example, unconscious biases may arise given the level of correspondence of the issue with the expertize of the expert, personal beliefs, and from the personal stake they might have in a particular outcome (Martin *et al.*, 2012). In contrast, conscious bias may also arise if the potential decision will have consequences for the expert, overstating the benefits if the outcome is seen as beneficial, or overstating the costs if the outcome is seen as negative (Murphy, 2001).

Discussion

Although TBL HSs have become mainstream in science and policy, they have yet to be routinely operationalized (Mangel and Dowling, 2016). More broadly, as EBFM becomes more common, decision makers and managers increasingly face TBL trade-offs. To date, this has been largely limited to a conceptual treatment (e.g. Stephenson *et al.* 2017) or semi-quantitative approaches (Dichmont *et al.*, 2012). This article has presented two extreme approaches to highlight the key points of an article on paradigms. The emphasis is not to undertake an in-depth analysis of each approach or the case study. What is more important are the similarities and differences between the two approaches and what is required to operationalize them. Both approaches are ultimately able to operationalize TBL HSs; however, the semi-quantitative MCDA approach already has a successful precedent in several cases. A fully conditioned and optimized TBL assessment simulation model for practical implementation is further away. However, several other approaches within these extreme ranges could have been applied.

Hjort-s (1914, 1926) publications were seminal texts that changed the biological paradigm of fish population dynamics towards the concept that variation in larval survival may strongly affect adult population dynamics. Hjort's hypotheses have been recognized as important concepts in marine science. However, the emphases from mainly the biological systems have changed to including human behaviour, which has led to the TBL becoming more central in fisheries. This work is not a replacement of Hjort's paradigms but provides key additions. Clearly fisheries science approaches together with the associated required data need to be broadened. When compared to Hjort's seminal articles, new fisheries science paradigms are required. These should emphasize (i) *representative* participatory management systems, (ii) TBL objectives, (iii) expansion/modification of the tools, and (especially) (iv) the data required to underpin these tools.

The MCDA approach allows managers to integrate all dimensions of sustainability and TBL objectives into the HS development process with transparency, and conceptual clarity. As described by Pascoe *et al.* (2019), this approach provides a formal and an explicit role for stakeholder engagement. Industry and other groups were directly involved in identifying the objectives, weighting the objectives, identifying the potential HSs, and providing input (based on expert knowledge) as to how these strategies were likely to perform. The process provides a formal framework in which this consultation can take place.

As shown in the case study, quantitative approaches can use similar products to semi-quantitative approaches (such as MCDA), but our objective-function approach needs to find a way to greatly simplify the resultant operational objective space or obtain a much broader range of information types, especially in the social objective space.

However, both the semi-quantitative and quantitative approaches share the need for stakeholder input to determine objectives, weights, and HSs, and, as such, the Working Groups should have greater diversity of representation for TBL approaches than would have been required for traditional management. The stakeholder group used in this case study did not include any social scientists or economists but did obtain insights in these areas from the project team (which in normal fishery management, as opposed to a research project, would not have been available). As such, stakeholder groups may need reformulating in the future to cope with the demands of TBL approaches.

The MCDA approach also identifies which areas of a HS option may require further consideration to be more acceptable to a greater proportion of stakeholders. For example, reducing the (perception of) costs associated with the "environmental overrides and spatial management" options would appear to result in universal benefits. However, the perceived relative benefits of alternative HS options are opinion based. The *post hoc* evaluation and revisiting for future revisions is an ongoing iterative process.

On the other hand, the objective-function approach provides a quantitative means to explicitly evaluate the TBL and its trade-offs in terms of clearly defined stakeholder objectives. It further allows the formal evaluation of performance of the TBL across stakeholder groups, providing an objective means to obtaining an overall optimum HS (here, a set of species-group-specific TACs).

However, the simulation part of the objective-function approach suffered from a lack of detailed data: it was not conditioned on historical data (as traditional MSEs would be), but tuned to replicate stock assessment models. Additionally, most of the performance indicators corresponding to each objective were based on very loose assumptions, not helped by the requirement that each must be some function of the management lever (i.e. the catch or effort). Although the simulation provides an impartial and replicable platform for operationalizing the TBL HSs, this comes at a high information premium. The objective-function approach also has less direct stakeholder involvement (it resembles a "black box") and is more technically challenging to interpret. Thus, there is a distinct need for a shift in paradigms in terms of the types of information that need to be collected. In reality, this data requirement is likely to be in addition to the existing data collection systems, so the challenge would be how to obtain this through existing or additional funding, or how to undertake the broader TBL work in a data-limited environment.

A pertinent issue when weighing alternative approaches is the nature of the risk that managers wanting to operationalize the TBL wish to take on. The MCDA approach carries the inherent risk associated with qualitative expert opinion. The objective-function model has inherent uncertainties associated with data gaps and assumptions. To some extent, these could be objectively addressed via sensitivity analyses. On the other hand, the level of stakeholder engagement and the sense of ownership and accountability conferred by the MCDA approach may, to some extent, avoid the liability around "getting it wrong".

More broadly, incorporating important external drivers like climate change and policy implementation will be difficult within both the MCDA and objective-function approaches. For both, these would have to be treated as alternative scenarios whose effects on the population, and on the fleet dynamics, are assumed known. Although the objective function would then analytically adjust the recommended TACs within the optimization process, expert judgement (as used in MCDA) would be challenging, as the experts would have to reconcile the assumed nature and impact of the driver, with its effect on HS performance.

Conclusion

In terms of paradigm shifts, the work presented here highlights the result of a much-expanded set of TBL objectives, elicited from stakeholder groups, that were either included in the conceptual trade-off analyses of different HSs or directly in a simulated optimization. This work has also shown the need for expanded teams from quite different backgrounds and viewpoints, from a representative group of stakeholders, through to social scientists and managers to help interpret, and quantitatively translate, the TBL objectives. Finally, the data required for a quantitative approach make extra demands on existing monitoring programmes, whereas the expert input required for the semi-quantitative approach requires the ongoing commitment of a dedicated identified expert panel.

Do we need a new paradigm to operationalize TBL HSs? Yes, our work here indicates that, indeed, a refinement of the existing set of tools, additional information, and team members are required, accompanied by policy changes and institutional readiness, to realize this goal. Although operationalizing the TBL in an HS is demonstrably possible, it is clearly early days and much work remains to be done to make this mainstream practice.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

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Supplementary material S1: Operationalising triple bottom line harvest strategies: model specifications

Dichmont, Catherine M.[1]; Dowling, Natalie A.[2]; Pascoe, Sean[3]; Cannard, Toni[3]; Pears, Rachel J.[4]; Breen, Sian[5]; Roberts, Tom[5]; Leigh, George M.[5], Mangel, M.[6,7]
[1]Cathy Dichmont Consulting, Bribie Island, 4507, Australia; cathydichmont@gmail.com
[2]CSIRO Oceans and Atmosphere, Hobart, Tasmania 7000, Australia
[3]CSIRO Oceans and Atmosphere, Queensland Biosciences Precinct, 306 Carmody Rd, St Lucia, Brisbane 4067, Australia
[4]Great Barrier Reef Marine Park Authority, PO Box 1379, Townsville, Queensland 4810, Australia
[5]Fisheries Queensland, Department of Agriculture and Fisheries, GPO Box 46, Brisbane 4001, Australia
[6]University of California, Santa Cruz, California, CA 95060, USA
[7]University of Bergen, Bergen, 9020, Norway

S1. Simulation model and performance indicator specifications

The model is a simulation approximating the 3 main species groups in the Coral Sea Finfish Fishery: coral trout (CT), red-throat emperor (RTE), and the "other species" collective (OS).

It is not conditioned on data and it assumes perfect knowledge of stock sizes, environmental parameters, and fishing mortality. That is, there is no stock assessment or sampling model to estimate underlying biomass. We also assume that the set of total allowable catches (TACs) are fully realised (i.e. no over-or under-catch).

We assume 2 latitudinal regions (noting that, longitudinally, all commercial fishers concentrate their effort on the mid-shelf).

Supplementary material S1: Operationalising triple bottom line harvest strategies: model specifications

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[1]Cathy Dichmont Consulting, Bribie Island, 4507, Australia; cathydichmont@gmail.com
[2]CSIRO Oceans and Atmosphere, Hobart, Tasmania 7000, Australia
[3]CSIRO Oceans and Atmosphere, Queensland Biosciences Precinct, 306 Carmody Rd, St Lucia, Brisbane 4067, Australia
[4]Great Barrier Reef Marine Park Authority, PO Box 1379, Townsville, Queensland 4810, Australia
[5]Fisheries Queensland, Department of Agriculture and Fisheries, GPO Box 46, Brisbane 4001, Australia
[6]University of California, Santa Cruz, California, CA 95060, USA
[7]University of Bergen, Bergen, 9020, Norway

S1.1 Historical: Setting up equilibrium structure and the population structure

The virgin age structure is determined assuming equilibrium dynamics, with natural mortality acting alone upon constant average unfished levels of recruitment. We assume that in year *y,* fish undergo half of natural mortality prior to being fished, and then the remaining natural mortality is applied. Mid-year abundance is calculated:

$$N_{a,s,A,y} = N_{a,s,A,y-1} \cdot e^{-M_{a,s}/2} \quad (1)$$

Over the historical years of catch data, fishing mortality by fleet *f*, species *s*, region *A*, and year, *y*, is calculated

$$F_{f,s,A,y} = \frac{C_{obs\ F,s,A,y}}{\sum_{a=1}^{a_{max,s}} S_{a,F,s} \cdot N_{a,s,A,y} \cdot m_{a,s}} \quad (2)$$

where

$C_{obs,f,s,A,y}$ is the observed catch (mass) of species *s* by fleet *f* from region *A* for year *y*

$S_{a,F,s}$ is the selectivity –at-age vector (where *a* is age) by fleet and species

$m_{a,s}$ is the mass-at-age of species *s*.

Note that the selectivity for RTE is age-based, but for CT it is length-based. This can be converted to selectivity-at-age using the length-age relationship.

Abundance is updated by applying the mortality due to catch, and finally the remainder of the natural mortality, to the interim (mid-year) numbers: to

$$N_{a,s,A,y} = N_{a,s,A,y} \cdot \left(1 - S_{a,f,s} \cdot \sum_f F_{f,s,A,y}\right) e^{-M_{a,s}/2} \quad (3)$$

We assume that no migration between regions occurs. CT show site-fidelity to the reefs on which they settle as larvae. We make the same assumption for the OS species.

The surviving cohort sizes are updated at the end of the year by incrementing the age classes:

$$N_{a+1,s,A,y} = N_{a,s,A,y}$$

$$N_{a_{max},s,A,y} = N_{a_{max},s,A,y} + N_{a_{max}-1,s,A,y} \quad (4)$$

The total spawner biomass by species, $B_{sp\ s,y}$, and total overall biomass by species, $B_{s,y}$, at the end of the year is calculated

$$B_{sp\ s,y} = \sum_{a=1}^{a_{max,s}} p_{a,s} \cdot m_{a,s} \cdot \sum_{i=1}^{Narea} N_{a,s,i,y}$$

$$B_{s,y} = \sum_{a=1}^{a_{max,s}} m_{a,s} \cdot \sum_{i=1}^{Narea} N_{a,s,i,y} \quad (5)$$

Annual recruitment, R_y is then determined using a Beverton-Holt stock-recruitment relationship with process uncertainty $E_{y,s}$

$$R_{y,s} = \frac{B_{sp\ s,y}}{\alpha_s + \beta_s \cdot B_{sp\ s,y}} \cdot e^{E_{y,s}} \quad (6)$$

where

$$\alpha = \frac{(1-h_s) \cdot SPR_s}{4h_s}$$

$$\beta = \frac{(5h_s - 1)}{4h_s \cdot R_{0,s}} \quad (7)$$

For the historical years of the model, we fitted $E_{y,s}$ to annual recruitment deviations.

The recruits are then distributed spatially according to:

$$N_{0,s,A,y} = Frac_{s,A} \cdot R_{y,s} \tag{8}$$

Supplementary material S1: Operationalising triple bottom line harvest strategies: model specifications

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[1]Cathy Dichmont Consulting, Bribie Island, 4507, Australia; cathydichmont@gmail.com

[2]CSIRO Oceans and Atmosphere, Hobart, Tasmania 7000, Australia

[3]CSIRO Oceans and Atmosphere, Queensland Biosciences Precinct, 306 Carmody Rd, St Lucia, Brisbane 4067, Australia

[4]Great Barrier Reef Marine Park Authority, PO Box 1379, Townsville, Queensland 4810, Australia

[5]Fisheries Queensland, Department of Agriculture and Fisheries, GPO Box 46, Brisbane 4001, Australia

[6]University of California, Santa Cruz, California, CA 95060, USA

[7]University of Bergen, Bergen, 9020, Norway

S1.2 Calculation of catchability

In the simulation, we assume a TAC for each species group and that the TAC is achieved for each species group each year, through a combination of targeted and incidental take. For the three species groups, we define the overall catchability on species *i*, q_i, using historical data of targeted catch and effort, and historically modelled biomass:

$$q_{i,f} = \exp\left(\frac{\sum_y \ln\left(C_{f,i,y}/B_{i,y}\right)}{\sum_y \ln\left(E_{f,i,y}\right)}\right) \tag{9}$$

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[6]University of California, Santa Cruz, California, CA 95060, USA

[7]University of Bergen, Bergen, 9020, Norway

S1.3 Projections: The harvest strategy

We model a harvest strategy equating to a total allowable catch (*TAC*), adjusted annually.

We assume size limits as an additional management measure but assume these are fixed over time.

For any given *TAC*, in scenarios when this was allocated across all sectors, we assume a fixed allocation matrix by sector and species of $\begin{pmatrix} 0.85 & 0.05 & 0.1 \\ 0.5 & 0.3 & 0.2 \\ 0.5 & 0.25 & 0.25 \end{pmatrix}$ where the columns are the commercial, charter and recreational sectors, and the rows represent each species group CT, RTE and OS, respectively. These proportions were based on historical averages. When the *TAC* was allocated only between the commercial and charter sectors, we assumed the charter sector allocation proportion was (0.15, 0.5, 0.5) for each of the three species groups. If a sector did not receive a dynamic *TAC* allocation, it was assumed they took a fixed amount for each species group, based on the averages over the final three historical years.

A bag limit applies to the recreational sector. As bag limits would not be adjusted by <=1 fish, nor do managers want to adjust them overly often, changes in (Total Allowable Recreational Catch (*TARC*) would in reality move infrequently and by sizable amounts. For this study, we have ignored the complexities of bag limits and assumed that a *TARC* can be set every year. It is still the case in our model that the *TARC* is adjusted only if the interannual change in the overall *TAC* is greater than 30%.

In each year, the TACs are determined as parameters that optimise the value function, described below as the sum of the relative performance indicators weighted by alternative stakeholder group preferences. An overall optimal (or “minimum whinge”) TAC is then obtained across the stakeholder groups.

Supplementary material S1: Operationalising triple bottom line harvest strategies: model specifications

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[6]University of California, Santa Cruz, California, CA 95060, USA
[7]University of Bergen, Bergen, 9020, Norway

S1.4 Projections: Fleet dynamics

If TAC is set by region, we assume perfect knowledge and no implementation error.

Otherwise (for the OS category, and where the *TAC* is not region-specific), we distribute the fishing mortality.

Commercial and charter (f <=2):

$$PropF_{f,s,A,y} = \frac{0.5\left(C_{f,s,A,y-1}+\frac{\sum_{y'=1}^{y-2} C_{f,s,A,y'}}{\left(yrsfished_{f,s,A}\right)-1}\right)}{\sum_A\left[0.5\cdot\left(C_{f,s,A,y-1}+\frac{\sum_{y'=1}^{y-2} C_{f,s,A,y'}}{\left(yrsfished_{f,s,A}\right)-1}\right)\right]}$$ if region A was fished (had non-zero catches of species s) in previous year by that fleet, where

$yrsfished_{f,s,A}$ is the number of years in which a non-zero catch of species s was reported by fleet f in region A.

$$PropF_{f,s,A,y} = \left[\frac{\sum_{y'=1}^{y-1} C_{f,s,A,y'}}{yrsfished_{f,s,A}}\right] \Big/ \sum_A \left[\frac{\sum_{y'=1}^{y-1} C_{f,s,A,y'}}{yrsfished_{f,s,A}}\right], \tag{10a}$$

otherwise we assume the recreational fishing effort is distributed equally between the two regions.

$$PropF_{rec,s,A,y} = \frac{1}{Narea} \tag{10b}$$

We apply these each $PropF_{f,s,A,y}$ proportions to distribute fishing mortality proportionately among regions when the *TAC* is not spatially explicit (i.e. is specified globally, $TACglob_{f,s,y}$), and hence calculate the region-specific catch by species.

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S1.5 Projections: Fishing mortality

We assume perfect knowledge and that the species-specific *TAC*s are achieved each year.

Species, and, in some versions, region-specific TACs will be achieved both via targeted and non-targeted fishing.

Overall, the fishing mortality by species, *s*, (and region, *A*) is determined by dividing the fleet-specific TAC by the biomass, as per equation (2). That is, when the TAC is specified globally, as $TACglob_{f,s,y}$, the fishing mortality is:

$$F_{f,s,A,y} = \frac{TACglob_{f,s,y}\cdot PropF_{f,s,A,y}}{\sum_{a=1}^{a_{max,s}} m_{a,s}\cdot S_{a,f,s}\cdot N_{a,s,A,y}} \tag{11a}$$

When the TAC is spatially explicit, the fishing mortality is

$$F_{f,s,A,y} = \frac{TAC_{f,s,A,y}}{\sum_{a=1}^{a_{max,s}} m_{a,s} \cdot S_{a,f,s} \cdot N_{a,s,A,y}} \tag{11b}$$

For the charter and recreational sectors (*f* = 2,3), the TAC will only be adjusted if it exceeds that of the previous year by more than a threshold, currently set at 30%.

To obtain the effort associated with the given *TAC* (and the catch by targeting practice), we use the catchabilities:

$$E_{f,s,A,y} = F_{f,s,A,y}/q_{s,f}$$

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S1.6 Projections: Population dynamics

As with the historical period, it is assumed that within any year, *y*, fish undergo half of natural mortality prior to being fished, and then the remaining natural mortality is applied. Mid-year abundance is calculated using equation (1), as for the historical period.

Catch (numbers) by species, fleet, region and year, $C_{f,s,A,y}$ is then:

$$C_{f,s,A,y} = F_{f,s,A,y} \cdot \sum_{a=a_{legal,s}}^{a_{max,s}} \cdot S_{a,f,s} \cdot N_{a,s,A,y} \tag{12}$$

where,

$S_{a,f,s}$ is the selectivity–at-age vector by fleet and species. For now, we assume the selectivity is the same across fleets (sectors), as they are all line fishing. However, the commercial fishers use larger hooks, so this may be re-evaluated.

$F_{f,s,A,y}$ is the fishing mortality from fleet *f* in region *A* and year *y* for species *s*

$a_{legal,s}$ is the average age at which the fish reaches legal size.

We assume all undersize catch, $UC_{f,s,A,y}$, below the minimum legal length (*MLL*) is discarded:

$$UC_{f,s,A,y} = F_{f,s,A,y} \cdot \sum_{a=1}^{a_{MLL,s}} m_{a,s} \cdot S_{a,f,s} \cdot N_{a,s,A,y} \tag{13}$$

Catch in mass is obtained by multiplying equation (13) by the species-specific mass-at-age, $m_{a,s}$.

Abundance is updated by applying the mortality due to catch, and finally the remainder of the natural mortality, to the interim (mid-year) numbers: to

$$N_{a,s,A,y} = N_{a,s,A,y} \cdot \left(1 - \sum_f PropF_{f,s,A,y} \cdot F_{f,s,y} \cdot S_{a,f,s}\right) e^{-M_{a,s}/2} \tag{14}$$

As per the historical period, the surviving cohort sizes are updated at the end of the year by incrementing the age classes, equating to growth:

$$N_{a_{max},s,A,y} = N_{a_{max},s,A,y} + N_{a_{max}-1,s,A,y}$$

$$N_{a+1,s,A,y} = N_{a,s,A,y} \tag{15}$$

The total spawner biomass by species, $B_{sp\ s,y}$ at the end of the year is calculated

$$B_{sp\ s,y} = \sum_{a=1}^{a_{max,s}} \sum_{i=1}^{Narea} p_{a,s} \cdot m_{a,s} \cdot N_{a,s,i,y} \tag{16}$$

Annual recruitment, R_y is then determined using a Beverton-Holt stock-recruitment relationship, and recruits are distributed among the regions, as per the historical period (equations (6)-(8)). An annual error, E_y can be applied, but it was here set to zero.

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S1.7 Projections: Performance indicators

In each projection year, we calculated the performance indicators (PIs). Each PI corresponds to a single TBL objective, as elicited from stakeholders -see Table S1.

We are trying to maximize the value for each PI.

Some PIs have alternative specifications, $PI_{i,j}$, where *j* is the alternative specification of PI *i*.

We calculated indicators assuming 100% (perfect) sampling of catch and biomass. Table S1 summarises the basics of the performance indicators.

Supplementary material S1: Operationalising triple bottom line harvest strategies: model specifications

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S1.8 The TBL value function

For each year *y*, we have a vector of 22 PIs: $PI_{1:22,y,1}$

We then calculate a TBL value function for any set of stakeholder group *g*'s weightings, by multiplying each PI by its weight, and summing:

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

In each year, we seek the harvest strategy (i.e. TAC) (assuming size limit is fixed) that maximises the value function for that group, $V_{g,y}$.

Alternative harvest strategy specifications are:

i) Species-specific TACs: this is a 3×1 array comprising TACs for coral trout, red throat emperor and SOCI.
ii) Region-specific, species-specific TACs: this is a 3×2 matrix, comprising TACs for each of the 3 species groups and regions.

The initial values for TACs are those from the previous year.

We use the R function *optim* to optimise the value function, with the TAC matrix as the model parameters.

In each year, we optimise the value function for each set of stakeholder group's weightings.

Once the optimal TACs are found, we call the CalcPerfInd function one more time to ensure that the corresponding values and PIs are obtained (for each preference/stakeholder group).

Given the optimum strategy (TACs) for the *g*th stakeholder group's weightings, we can calculate the value function for every other set of stakeholder group weightings, *k*:

$$V_{k,y} = \sum_{j=1}^{Nind} PI_{j,y,1} \cdot Wt_{j,k}$$

For each year, this gives a matrix of values according to each set of stakeholder group weightings, calculated using the PIs derived from the optimal TACs (the optimal strategy) for each stakeholder group. Each row represents one stakeholder group's optimal strategy, which is applied to each stakeholder group's preference weighting, by column:

$$\begin{bmatrix} V_{11} & \cdots & V_{1n} \\ \vdots & \ddots & \vdots \\ V_{n1} & \cdots & V_{nn} \end{bmatrix}$$

When each column is standardised relative to the value for the stakeholder group for which the strategy is optimal (i.e. each column's values are divided by the value in the row corresponding to that column), the result is a matrix of relative values whose diagonals equate to 1.

We can use two alternative criteria to select the overall optimal TACs (= harvest strategy) across all the stakeholder preference groups. We take either:

- The highest average value across all stakeholder weightings: that is, the row of the matrix that has the highest average, indicating that the strategy is overall optimal across all preference groups, or
- The highest minimum value across all stakeholder weightings: that is, the row of the matrix that has the highest minimum value across the row, indicating that this strategy results in the "minimum whinge" across all preference groups.

We then run the population dynamics and calculate the PIs, using the overall optimal TACs for that year.

Table S1. Descriptive summary of conceptual objectives, together with their translation into operational objectives, or performance indicators, that are direct or indirect functions of the catch

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
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)**	This performance indicator applies only to CT and RTE. We assume a "hockey stick" relationship: the performance indicator is 1 above a relative biomass of 0.6, 0 below a relative biomass of 0.2, and tracks linearly with relative biomass between these values.	The target reference point is assumed to be 60% of the unfished biomass, although this may be higher, from a conservation standpoint.
		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.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
	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 the maximum historical effort. For each target species, fleet and area, the effort is calculated relative to the historical high, and 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.	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 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 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).
		1.2.3 Minimise broader ecological risks	The broader ecological risk is assumed to be a weak linear function of effort but the performance indicator is set to 1 when effort is below a target, and 0 when it is above a 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)
		1.2.4 Minimise risk to TEPS	The TEP risk is a weak exponential function of effort, but the performance indicator is set to 1 when effort is below a target, and 0 when it is above a 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)

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
	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 mean across the species and areas.	Target and limit relative biomass reference points are set at 0.5 and 0.2.
		1.3.2. In response to environmental event (e.g. cyclone, climate change)	Handled as separate model scenarios	This performance indicator needs to reflect the need to be conservative and precautionary given these perturbations. As such, we will take indices of relative local depletion as per 1.3.1 above (but inclusive of OS species), and apply a (say, 20%) penalty, Pen = 0.8, to the estimates of biomass, to be precautionary given that availability is reduced as a result of ongoing perturbation effects. We have biomass by region, relative to that area's virgin biomass. 0 if penalised relative biomass below 20%, 1 if above 0.5 Bo, and linear in between.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
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 performance indicator is calculated by taking the mean across species of the ratio of profit to that at MEY.	Unit costs of fuel, gear and effort have all been assumed. Profit at MEY is currently a guesstimate based on the profit in the last year of the historical time series.
		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 same manner as for the commercial sector.	Unit costs of fuel, gear and effort have all been assumed. Profit at MEY is currently a guesstimate based on the profit in the last year of the historical time series.
		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.	We assume 5t (in addition to the commercial TAC) is set aside for each of the three species groups (for a total of 15t), for indigenous commercial development. We also assume that the existing indigenous fraction of the commercial TAC is 3%.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
	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 catch, catch-per-unit-effort (CPUE), and effort. The recreational utility is the weighted sums of recreational catch, CPUE, and effort, where each area's utility is, in turn, weighted according to the proportion of recreational effort in that area. The average is taken over all areas, and the performance indicator is calculated by standardizing this average by the maximum historical recreational catch, effort and CPUE.	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.
	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 a reference year.	The recreational dollar scalar, and the reference year, are both assumed.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
	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 variation of +/- 10% CV is optimal and equates to a performance indicator value of 1, and that +/- 25% is the limit below which the performance indicator 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.	The target and limit reference values are assumed.
	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 historical high, 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.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
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, 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 an inverse non-linear function of TAC and bag limit – that is, as these go down, the lack of compliance increases. The performance indicator is calculated by adding each of these two terms and taking the average.	The rate parameter in the "disagree fail" component of this PI is assumed. It is currently assumed that the "complexity fail" and the "disagree fail" terms are equally weighted.
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 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 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.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
	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)**	Using the indicators of discarding (1.2.2) and TEPS (1.2.4), we take a weighted average, with stronger weight on TEPS and low on discarding indicators. We then infer a hockey stick relationship between that weighted average and perception, reflecting a small tolerance, before people quickly get "anxious" and want to close the fishery. These performance indicators are subtracted from 1, so that the higher their value, the higher the risk. For the TEP risk, when the risk is less than 10%, the "perception score" is 0. When the risk is at 10%, the perception begins to linearly increase. 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 the 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 values, are assumed. The perceptions around discarding and TEPS are currently equally weighted.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
		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 fishing to be perceived as a positive activity with benefits to the community (commercial, rec, and charter)	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 fishing to be perceived as a positive activity scales directly with objectives 1.1.1, 1.1.2, and 2.3, and take an average across them.	Each of the three contributing performance indicators is currently equally weighted.
	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 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.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Supplementary Information)	Assumptions
		4.3.2 Maximise spatial equity between regions or local communities	We assume there was a period in time when catch by area was considered equitable. We compare relative regional catches to those in that were acceptable in the past, using a distance function. For now, assume the equitable proportions of catch (by weight) by area were equal. The deviation threshold, above which the area is “unhappy”, is set at 20% (score of 1 above this). The deviation tolerance, below which the area is “happy”, is set at 5% (score of 0 below this). 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 area gets no catch, then the performance indicator value is 0. Otherwise, the performance indicator is one minus the hockey stick scores, averaged over the areas.	The nature of the relationship, together with their threshold values, are assumed.