FINAL REPORT

Developing triple bottom line harvest strategies that include all environmental aspects for multi-sector fisheries

Natalie Dowling, Catherine Dichmont, Sean Pascoe, Toni Cannard, George Leigh, Rachel Pears, Tom Roberts, Sian Breen

16 October 2020

FRDC Project No **2015/013**

ISBN 978-1-925994-15-5

Developing triple bottom line harvest strategies that include all environmental aspects for multi-sector fisheries

FRDC 2015-013

2020

Ownership of Intellectual property rights

This publication (and any information sourced from it) should be attributed to **Dowling N.A.**, Dichmont, C., Pascoe, S., Cannard, T., Leigh, G., Pears, R., Roberts, T., & Breen, S. (2020). *Developing triple bottom line harvest strategies that include all environmental aspects for multi-sector fisheries.* Final Report, FRDC Project No 2015/013, Hobart, Australia, 301pp. CSIRO Oceans and Atmosphere, 16 October 2020.

Creative Commons licence

Disclaimer

The Fisheries Research and Development Corporation plans, invests in and manages fisheries research and development throughout Australia. It is a statutory authority within the portfolio of the federal Minister for Agriculture, Fisheries and Forestry, jointly funded by the Australian Government and the fishing industry.

Researcher Contact Details

Name:	CSIRO Oceans and Atmosphere
Address:	GPO Box 1538 Hobart TAS 7001
Phone:	02 6232 5148
Fax:	
Email:	Natalie.dowling@csiro.au

FRDC Contact Details

Address:	25 Geils Court Deakin ACT 2600
Phone:	02 6285 0400
Fax:	02 6285 0499
Email:	frdc@frdc.com.au
Web:	www.frdc.com.au

In submitting this report, the researcher has agreed to FRDC publishing this material in its edited form.

Contents

Tables

Figures

Acknowledgments

This research was funded by the Fisheries Research and Development Corporation (FRDC), grant number 2015-013. In-kind contributions were made by CSIRO, Queensland Department of Agriculture and Fisheries, and the Great Barrier Reef Marine Park Authority (GBRMPA). We are grateful for the support and engagement of the members of the Coral Reef Finfish Fishery Working Group, and to the associated stakeholders who partook in online surveys and provided feedback. Prof. Marc Mangel (University of California Santa Cruz) and Drs. Aaron Mamula and Cameron Speir (NOAA, Southwest Fisheries Science Center) are thanked for providing invaluable advice and input to the simulation model approach. Dr. Sue Pillans is thanked for her artwork summary of the General Methodology. Peter Campbell (CSIRO) provided invaluable scientific computing user support.

Abbreviations

BBN	Bayesian Belief Network
CRFFF	Coral Reef Finfish Fishery
CSIRO	The Commonwealth Scientific and Industrial Research Organisation
DAF	(Queensland) Department of Agriculture and Fisheries
GBR	Great Barrier Reef
GBRMPA	Great Barrier Reef Marine Park Authority
HS	Harvest Strategy
MCDA	Multi-Criteria Decision Analysis
MEY	Maximum Economic Yield
TBL	Triple Bottom Line
WG	Working Group

Executive Summary

What the report is about

- Multi-sector fisheries are commonly faced with competing objectives when attempting to obtain optimal outcomes using formal harvest strategies. Harvest strategies are the monitoring, assessment and decision rules used to achieve the fishery's objectives. Traditionally, formal harvest strategy evaluation has focused on target species sustainability and some economic performance indicators. However, multi-sector (i.e. commercial, recreational, indigenous) fisheries, particularly in multi-species fisheries that occur in environmentally sensitive environments such as the Great Barrier Reef (GBR), are directly confronted with the "triple bottom line" (TBL) of a range of environmental, economic and social objectives, as well as a fourth pillar relating to institutional objectives. Given that Queensland has instituted management reforms requiring the development of harvest strategies for all of its fisheries, there is a legislative need to operationalise triple bottom line harvest strategies for these fisheries. However, these have yet to be operationalised. This project represents the first effort to do so.

- In what was primarily a methodology project, the expertise of scientists (from CSIRO, Queensland DAF and GBRMPA), DAF managers, and fishery stakeholders was combined to explore approaches to operationalise TBL harvest strategies for a Queensland case study fishery, namely, the Coral Reef Finfish Fishery (CRFFF). We engaged directly with the CRFFF Working Group and fishery stakeholders to elicit and weight TBL objectives and develop harvest strategy options, with particular attention to the unique environmental aspects impacting the fishery (including the acute effects of cyclones, and the chronic effects of climate change). We then explored two different approaches to evaluate the harvest strategies against the TBL objectives. The first was a semi-quantitative expert judgement process that applied multi-criteria decision analysis. The second was a data-hungry, novel simulation approach that optimised a total allowable catch across the suite of TBL objectives, as well as over the range of stakeholder group preferences (weightings). Both approaches were able to operationalise TBL harvest strategies, though the former is not defensibly quantitative and the latter makes many assumptions and is not implementation ready. A key deliverable was a General Methodology summarising our key learnings in a practical stepwise process, to assist managers with future approaches to TBL harvest strategy development.

Background

The triple bottom line (TBL) of environmental/ecological, economic and social management objectives (and in some cases the extended fourth pillar relating to governance) is important for stock sustainability, supporting environment health, resource access, certainty, export approvals and public confidence. However, it has yet to be formally operationalised within a harvest strategy context. A harvest strategy (HS) framework specifies pre-determined management actions in a fishery to achieve management objectives via monitoring, assessment and decision rules. As opposed to a broader management strategy or procedure, HSs focus on controlling exploitation rates for relevant species.

The National HS Guidelines (Sloan et al. 2014) recommended the development of case studies to test their practical application, with priority assigned to multi-sector (e.g. commercial and recreational) fisheries that are directly confronted with TBL objectives (noting their mixed data quality and quantity). Queensland multi-sector GBR fisheries provided an ideal starting point to develop and recommend TBL HSs, and one that was timely given the recent reviews and reforms of Queensland fisheries management.

Aims/objectives

The aim of this project was to outline a process to develop TBL harvest strategy approaches for complex multi-sector fishery to achieve ecological, economic and social sustainability objectives. The Coral Reef Fin Fish Fishery (CRFFF) was selected as a case study fishery.

The main objectives were:

1. To undertake a review of (existing work around) multi-objective management systems and associated assessment approaches.

    Outcome: This review is embedded within the Method section of the report ("Identifying Triple Bottom Line Objectives – Literature Review: identified pitfalls"; "Process of trading off and reconciling objectives – Literature Review: conceptual overview"; "Determining a methodology for TBL evaluation: a review and summary of the process". Background material is presented in Appendices A ("Achieving the triple bottom line in fishery harvest strategies: challenges identified in the literature") and B ("The process of objective elicitation: a brief literature review"). Table 14 summarises the available approaches for evaluating TBL harvest strategies, and gives an overview of the advantages and disadvantages of each. This may assist practitioners to determine which approach may best be suited to their circumstances.

2. To compile an inventory of current environmental, economic and social objectives and consider how to translate such conceptual management objectives to operational objectives for multi-sector fisheries.

    Outcome: The TBL objectives inventory prepared as an objective of this project is provided in Table 6, within the Results section of the report.

3. To develop a theoretical framework (incorporating existing) methods and approaches with which to evaluate trade-offs between, and/or priorities for, environmental, economic and social objectives

    Outcome: We took two approaches to evaluating trade-offs between TBL objectives: a semi-quantitative expert judgement process that applied multi-criteria decision analysis, and a novel simulation approach that optimised a total allowable catch across the suite of TBL objectives, as well as over the range of stakeholder group preferences (weightings). Both approaches were able to operationalise TBL harvest strategies, though the former is not defensibly quantitative and the latter makes many assumptions and is not implementation ready. Each has been published in the primary literature. The same method for objective elicitation and preference weighting underpinned both approaches. These collectively form the bulk of the Method, Results and Discussion section of the report.

4. To develop a General Methodology for harvest strategy development against the triple bottom line for multi-sector fisheries

Outcome: The General Methodology is presented as the final sub-section of the Results section of the report, and offers succinct, user friendly guidance to practitioners. It is also available as a standalone document. The General Methodology distils a stepwise, more generalised summary of the process, to assist practitioners to apply the approaches to their own fisheries. This General Methodology was presented as part of a final online workshop to managers, stakeholders and scientists in September 2020.

5. To finalise the choice of Queensland state-based multi-sector case study fishery

Outcome: A formal approach was taken to finalise the choice of Queensland state-based multi-sector case study fishery. This is presented as the first sub-section in the Method section of the report, which includes Table 1.

6. To develop and recommend a triple bottom line harvest strategy framework for the case study fishery

Outcome: We assisted Fisheries Queensland, within the Coral Reef Finfish Fishery Working Group, by leading a process of identification of alternative TBL harvest strategy options. This process is defined in the Method subsection, “Process of engagement to identify alternative harvest strategy options”. The resulting alternate HS options were subsequently evaluated using both the semi-quantitative MCDA and quantitative simulation approaches. However, there was a divergence of Queensland fishery management reforms and this project, as discussed in the section “Modification of objectives”. Ultimately, the implemented harvest strategy objectives were driven by i) the timeframe for delivering a harvest strategy (versus the longer timeline of the project); ii) the need for consistency across all Queensland fisheries, and the legislated target reference point of 60% of the unfished stock biomass; iii) the fact that it was more practical to adopt a hierarchical set of objectives, where the stock status was paramount; and iv) the limited ability to pull management levers relating to social/economic measures. This is discussed in the “Extension and Adoption” section.

Methodology

A literature review of (existing work around) multi-objective management systems and associated assessment approaches underpinned the choice of approaches taken for this project. We formally appraised a short-list of fishery options to select the Queensland Coral Reef Finfish Fishery as the choice of Queensland state-based multi-sector case study fishery, and applied two approaches (a semi-quantitative approach, and a quantitative simulation) to develop triple bottom line harvest strategy frameworks. As part of our literature review, we compiled an inventory of current environmental, economic and social objectives, which formed the basis of our approach to elicit TBL objectives for the CRFFF.

To identify priorities for environmental, economic and social objectives, we undertook a modified Analytic Hierarchy Process (AHP) based on a series of pair-wise comparisons. This enabled individual preference weightings to be obtained from over 100 stakeholders.

In direct consultation with the CRFFF WG (a DAF WG that provides advice on operational aspects of the management of the fishery), we identified 6 alterative "harvest strategies" that built on the current management arrangements to explicitly address key environmental, economic and social concerns.

Based on the results of the literature review and the collective expertise of the project team, we developed two alternative approaches with which to evaluate trade-offs between triple bottom line objectives and stakeholder preferences: a semi-quantitative multi-criteria decision analysis (MCDA) framework, and a quantitative simulation model approach. The MCDA semi-quantitative approach carries the inherent risk associated with qualitative expert opinion. The quantitative simulation model has inherent uncertainties associated with data gaps and assumptions. In particular, the simulated performance indicators corresponding to many of the objectives, and especially the social objectives, were crude and highly simplified guesses that assumed that these metrics were directly related to catch, effort or CPUE.

Results/key findings

The Working Group agreed on a set of 21 TBL objectives. The preference weighting survey indicated that all groups tended to rank the ecological sustainability objectives the highest. However, the weightings across the 21 lower level TBL objectives were fairly similar across the different stakeholder groups. This was an artefact to some extent of the "dilution" effect of distributing the higher level objective weights over many sub-objectives.

Both the semi-quantitative MCDA and simulation approaches demonstrated how triple bottom line outcomes can be affected by relatively small changes in harvest strategy, and through combining HS approaches targeted at particular aspects. In the case of the simulation approach, we also illustrated the sensitivity to the specification of performance indicators.

A comparison of the results against each of the separate objectives from the MCDA and simulation model was undertaken for a harvest strategy option where TAC was allocated separately to both the commercial and charter sectors. For the majority of objectives, the outcomes from the two approaches were not significantly different, although the magnitudes of the mean outcomes varied.

The MCDA expert-judgement based outcomes were generally more optimistic against most objectives than the simulation model, although more pessimistic in some social and economic outcomes given the management scenario examined. Determining which outcomes are correct is not possible. While in principle "objective", simulation models are based on a number of assumptions, beliefs and approximations that may influence the outcomes. This is particularly the case for assessing some outcomes of management that are less direct functions of the harvest strategy – for example, linking social outcomes to changes in catch and effort.

The quantitative simulation provides a means to consider a richer range of trade-offs than possible with bio-economic models only. While novel in its ability to consider multiple objectives in terms of their (often, assumed) relationship to harvest controls (and hence, catch effort, or CPUE), and its ability to optimise across the range of difference stakeholder group weightings, it is highly data-hungry and requires the ability to quantitatively define each performance indicator. While uncertainties and assumptions can be objectively explored via sensitivity analyses, right now, our quantitative approach is not ready to be used for management advice. To do so requires a clear review and evaluation of the kinds of information required and how stakeholders wish to quantitatively translate their TBL objectives into operational objectives and performance indicators.

On the other hand, qualitative/semi-quantitative approaches have demonstrated that managers can have a hands-on approach to operationalising TBL harvest strategies. While they are based on expert opinion, the level of stakeholder engagement and the sense of ownership and accountability conferred by such approaches means that all parties share responsibility for the success of the harvest strategy. A fully conditioned and optimised TBL management strategy evaluation (MSE) model for practical implementation is further away. That stated, semi-quantitative approaches cannot explicitly specify management lever values and inter-annual adjustments: these are agreed upon through a process of expert judgement.

Implications for relevant stakeholders

Policy and legislation demand that we need to continue to move to a quantitative approach for reconciling TBL objectives and operationalising these defensibly within harvest strategies. This project demonstrated a clear process to identifying TBL objectives, from a comprehensive inventory and weighting these across stakeholders. It has embraced two alternate approaches to operationalising the triple bottom line within fishery harvest strategies, using the Coral Reef Finfish Fishery as a case study.

Stakeholders are key to the process of TBL HS development, and are critical for the process of eliciting and weighting objectives, and for identifying alternative possible HS options. However, they require careful direction (not manipulation) for these processes to be successful. A Working Group must commit to ongoing engagement toward the development, evaluation and implementation of the harvest strategy.

Overall, operationalising the TBL in a harvest strategy is demonstrably possible, but it is clearly early days and much work remains to be done. Although the recognition of the importance of consideration of TBL outcomes in fisheries management has occurred concurrently with the recognised benefits of the use of harvest strategies to aid management decision making, the approaches that have been applied here represent, to the best of our knowledge, the first dedicated efforts to operationalise the TBL within harvest strategies.

A General Methodology provides accessible guidance to managers faced with developing TBL harvest strategies.

Recommendations

This work has 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. The work also points to the need for a coordinated approach to monitoring, centred on data that serves the TBL and ultimately enables the calculation of the performance indicators corresponding to all TBL objectives.

The number of TBL objectives should be reviewed with the aim of capping these at a lower number than the 21 considered here (say, 10). This is both to reduce both the "dilution effect" of preference weightings not showing strong difference across objectives, and to make the harvest strategy practical and workable in terms of its monitoring requirements and evaluation. The maximum number of objectives is currently arbitrary, but should balance the observed need for more than one environment, economic and social objective, against the observed "dilution effect" that resulted from having 21 objectives. While an optimal number of objectives could theoretically be formally tested, this would require multiple rounds of stakeholder questionnaires and was beyond the scope of this project.

More generally, there must be a balance between developing TBL-defensible harvest strategies and what is practical, cost-effective and achievable. A key priority should be to identify which of the TBL objectives (which can directly include institutional objectives such as the cost of management) and which external environmental factors are most critical, not only in terms of stakeholder weightings, but in terms of their perceived sensitivity (for the former) and impact (for the latter), and to focus efforts around these.

If a quantitative TBL evaluation approach is to be pursued, then this needs to be evaluated in the context of a full MSE that integrates the existing available stock assessments for Coral Trout and Red Throat Emperor, and likely a more comprehensive operating model that can better account for spatial, fleet and population dynamics. The assumptions underpinning the performance indicators will also need to be reviewed. The use of Bayesian Belief Networks (BBNs) to capture non-quantitative objectives may be a sensible compromise that avoids the need to define explicit relationships between performance indicators and management controls whilst still evaluating objectives in a semi-quantitative probability-based manner (i.e. the outputs of a quantitative model would feed into a BBN model). This approach warrants further investigation.

The Queensland CRFFF fishery manager's recommendations included that, for optimal consideration of a TBL HS, the existing management framework should, ideally, be mature (as the existing management framework influences the available harvest control rule levers). It was therefore also recommended that a fully explicit TBL harvest strategy should not be the first harvest strategy a fishery should ever have, but rather, a second or third generation of harvest strategy where the stakeholders and the management framework are very settled. In this context, it is also helpful if the fishery is in a healthy, non-contentious space, with the stocks to be managed being already at or near their sustainable targets. Fisheries Queensland also agreed that a high number of objectives is fraught, because it becomes difficult to keep stakeholder expectations in check. The need for quality data, including economic, social and recreational data, was also acknowledged.

Due to their costs and data requirements, TBL harvest strategies are unlikely to be able to be developed for small-scale, low-value fisheries, many of which lack adequate data to undertake a stock assessment on the main target species. Cost-benefit trade-offs must be considered in the context of developing a TBL harvest strategy.

Finally, this work needs to be seen to be used: the approaches taken in this project to directly incorporate the triple bottom line in harvest strategies should form the basis for further, coordinated approaches moving forward.

Keywords

Triple bottom line, multi-sector fishery, objective weighting, multi-criteria decision analysis, simulation model, harvest strategy.

Introduction

What is the Triple Bottom Line?

The triple bottom line (TBL) is a term attributed to John Elkington (1997) and adopted as an accounting concept wherein performance is reported explicitly against economic, ecological and social criteria (Suggett and Goodsir, 2002). Similar concepts with three perspectives to balance include the People Planet Profit, 3BL and three pillars. Goldberg (2001) describe the 'triple bottom line of sustainability' in similar terms to Head et al. (2004) as:

- "the environmental imperative of living within ecological means
- the economic imperative of meeting basic material needs, and
- the social imperative of meeting basic social needs and cultural sustainability".

Stephenson et al. (2017a) propose a quadruple bottom line for fisheries, or four "pillars of sustainability" that includes institutional aspects in addition to economical, ecological and social "pillars. Institutional or managerial objectives of "simplifying and improving management structures" were also considered by Pascoe et al. (2013d).

Considerable progress has been made against the application of TBL in international fisheries due to the Food and Agriculture Organization of the United Nations (FAO) development of a Code of Conduct for Responsible Fishing. This code addresses issues of bycatch and responsible fisheries management (FAO, 1995). Various international conventions including the Convention on Conservation of Migratory Species (CMS) and the Convention on Biological Diversity (CBD) were preludes to the establishment of marine protected areas (MPAs) that typically manage fisheries and biodiversity considerations. These conventions encompass the social part of the triple bottom line in terms of the need for intergenerational equity and the ecological sustainability of the resource itself, and since fisheries have always had an economic component it is therefore accepted that TBL has been applied implicitly in fisheries management. In more recent years, explicit application of TBL to fisheries has been conducted; for example, Anderson et al. (2015) recently published their Fishery Performance Indicators (FPI) tool for assessing performance in individual fisheries, and for identifying the links between enabling conditions, fisheries management strategies and triple bottom line outcomes.

Defining the TBL and its means of application

The Triple Bottom Line attempts to formally acknowledge the ecosystem/environmental, social and economic objectives that underpin fishery operations and management. Often these are competing; at times, they are complementary. Further, the nature of the trade-offs is definition-dependent. (Halpern et al., 2013) showed a range of trade-off typologies between equity and conservation, depending on how social equity was defined and measured.

In categorising fishery objectives, the question is whether the paradigm should be a Triple Bottom Line, or whether it is better considered as a Quadruple Bottom Line, or simply a

Multiple Bottom Line. Alternative structures have been proposed. Firestone et al. (2009) suggest that “cultural aspects” should be the fourth bottom line. Others suggest that the additional pillar should be ‘progress’ (Levin et al., 2010). Others still suggest that “ease of management” or “governance” as additional components (Pascoe et al., 2013d; Stephenson et al., 2017b).

Even within the traditional three pillars (environmental, economic and social), objectives can embrace a range of considerations. For example, environmental or ecosystem objectives can be from many contexts, including:

- Target-species
- Byproduct species
- Bycatch species
- Broader ecosystem
- Habitat impact
- Interaction with threatened, endangered and protected (TEP) species, protected areas
- Pollution, damage due to loss of gear, carbon footprint
- Ecosystem services and non-market valuation challenges.

The social bottom line implies goals for improving total quality of life (wellbeing); and widening the ‘social’ beyond socioeconomic improvements to include topics such as social capital, social cohesion, social equity (Klein et al., 2015). However, there is currently no agreed paradigm for the social bottom line (Rindorf et al., 2017; Stephenson et al., 2017a). Moreover, as with environmental objectives, social objectives can have many aspects according to user group:

- Fishers (and within this, commercial, recreational, charter, indigenous)
- Users of marine environment for other reasons (e.g. tourism diving)
- Broader community
- State/national/international perception.

Social objectives are also notoriously difficult to quantify (Symes and Phillipson, 2009; Vieira et al., 2009; Triantafillos et al., 2014; Brooks et al., 2015; Pascoe et al., 2017). In attempting to derive quantitative performance indicators, they often reduce to environmental or economic proxies. While such proxies can simply be interpreted as having weight from both environmental/economic and social perspectives, independent social metrics are desirable.

The triple bottom line can be applied in four different concepts along a gradient of increasing complexity. This complexity can span from the level of broad values to detail and comparative national reporting achievement towards the internationally accepted sustainability goals. Complexity of this type engages indicators commonly used in corporate sustainability reports, they entail monetary values, such as sales volume ($), and physical values, such as waste (kg), fresh water (m^3), working accidents (cases). By the nature of the indicator it is usually possible to distinguish between the three sustainability pillars (Global Reporting Initiative, 2004).

More broadly, environmental, economic and social indicators are used in many countries and international initiatives (e.g. The Global Reporting Initiative™, 2000[1]) for sustainability and TBL performance. Indicators are used to signpost values. However, such values vary widely across cultures. The World Values Survey provides insights to such variance with an interactive interface for the plotting and visualisation of various countries and their respondents' responses to various value statements including fisheries (Graham et al., 2014; Turner et al., 2015; World Values Survey, 2015).

TBL has special relevance given Queensland fishery interaction with the GBR, and the multi-sector nature of many of its fisheries (large recreational and charter, also indigenous). Sloan et al. (2014) recommend that further investigations as to application of harvest strategies were need in both cases of a large recreational take and/or significant indigenous harvests. The social part of TBL is to date the least developed except in very broad terms e.g. global reporting initiative. Broad social indicators have been criticised as being too fragmented and lacking in inexplicit evaluation (Tullberg, 2012). Yet there is increasing acknowledgement of the importance of social objectives – even objectives those that defy quantitative measurement.

The ultimate aim of good fisheries management should be to acknowledge all relevant objectives, where the categorisation thereof is secondary to their quantitative definition, relative stakeholder group weighting, and evaluating outcomes against each.

To that end, there is a difference between acknowledging the TBL in the conceptual space and operationalising this in a harvest strategy context. Little has been done to date against the latter. If so, it has generally occurred more as a post-hoc discussion against MSE trade-offs, rather than as an inherent aspect of the harvest strategy (Fletcher et al., 2016). It is the aim of the current project to explore ways to directly incorporate TBL objectives within the harvest strategy.

Why is the Triple Bottom Line important?

The triple bottom line of environmental/ecological, economic and social management objectives is important for stock sustainability, supporting environment health, resource access, certainty, export approvals and public confidence.

Some environmental and many economic objectives are already in place through relevant Australian fisheries legislation, industry codes and standards. The triple bottom line concept goes beyond biologically sustainability concepts to more fully incorporate issues around social licence to operate, and the integrative potential of considering more than just stocks of resources. In fact the idea is that communities linked with resource in focus creates TBL sustainability through creation and maintenance of healthy ecosystems and the vibrant industries that benefit from resource extraction and this is reflected in the term 'well-being' (Brooks et al., 2015).

[1] https://www.globalreporting.org/STANDARDS/G4/Pages/default.aspx

Similarly the concept of triple bottom line requires more than a corporate social responsibility (CSR) approach (Elkington, 2006b; Kleine and von Hauff, 2009). Group of 100 Incorporated (2003) includes an Australian business perspective on TBL reporting and provides examples of best practice highlights the following considerations:

- Indicators should address requirements and concerns of stakeholders
- Indicators should align with objectives and policy
- Indicators should provide information to guide decision-making
- There should be systems to generate accurate, reproducible performance data
- Identify risks of publicising specific measures of performance
- Indicators should facilitate comparison with competitors.

A variety of well-being indexes have been developed – for example, the Measure of Economic Welfare (MEW) the Genuine Progress Indicator (GPI); the Index of Economic Well-Being (IEWB); the Human Development Index (HDI); the Index of Social Health (ISH); the Quality of Life Index (QOL); and the Index of Social Progress (ISP) (Sharpe, 1999). The construction of economic and social well-being indexes highlights technical issues of constructing indicator indices alongside challenges of using multiple versus single indicator approaches other than monetary indicators.

However, the DEWHA (2003) identified the following key benefits of triple bottom line reporting:

- Embeds sound corporate governance, values-driven culture and ethics systems through all organisational levels
- Improves management of risk with improved management systems and regular performance monitoring and transparent resource allocation decisions
- Formalises and enhances communication with key stakeholders and community
- Ability to benchmark performance.

Project need

As stated above, the triple bottom line (TBL) of environmental/ecological, economic and social management objectives is important for stock sustainability, supporting environment health, resource access, certainty, export approvals and public confidence. However, it has yet to be operationalised within a harvest strategy context.

A harvest strategy (HS) framework specifies pre-determined management actions in a fishery to achieve management objectives via monitoring, assessment and decision rules. As opposed to a broader management strategy or procedure, HS focus on controlling exploitation rates for relevant species.

To address the TBL requires:

a) understanding the impact of environmental (bycatch, habitat, broader), economic and social aspects on a fishery,

b) elicitation of objectives and an understanding of the trade-offs between these,

c) assessment methods that may be applied within a HS.

The National HS Guidelines (FRDC 2010/061) stated the importance of establishing operational TBL objectives (Sloan et al., 2014). The National Strategy for Ecologically Sustainable Development stipulates that these objectives must be considered simultaneously, with none predominating. Queensland fishery management in the Great Barrier Reef Region aims to simultaneously achieve the objectives of the Fisheries Act (1994) (Qld) and the GBRMP Act (1975) (that permits ecologically sustainable use provided it is consistent with the main object of long-term environmental protection). Thus, addressing the TBL in a HS context is paramount for Queensland GBR fisheries.

The National HS Guidelines recommended the development of case studies to test their practical application, with priority assigned to multi-sector (e.g. commercial and recreational) fisheries that are directly confronted with TBL objectives (noting their mixed data quality and quantity). FRDC 2010/040 developed and tested social objectives for fisheries management but emphasised the outstanding need to integrate social objectives/indicators within HS frameworks. Queensland multi-sector GBR fisheries provide an ideal starting point to develop and recommend TBL HSs, and one that is timely given the recent reviews and reforms of Queensland fisheries management.

Policy and legislative context

Under the Fisheries Management Act 1991 (FMA), decision-making processes should integrate short- and long-term dimensions of economic, environmental, social and equity considerations (Wilson et al., 2010). There is an implicit TBL intent within the primary legal instrument, the *Environment Protection and Biodiversity Conservation Act (EPBC) Act* (1999), in which the definition of the environment is detailed as Section 528 (Commonwealth of Australia, 2007):

(a) ecosystems and their constituent parts, including people and communities; and

(b) natural and physical resources; and

(c) the qualities and characteristics of locations, places and areas; and

(d) the social, economic and cultural aspects of a thing mentioned in paragraph (a), (b) or (c).

The EPBC Act incorporates a triple bottom line and specifically requires people and communities to be included as part of ecosystems, in other words the social aspect is explicit in Section 528 (a). Section 528 (d) further requires the triple bottom line to be considered as well – although in this case the triple consideration is social, economic and cultural. However, beyond this definition the *EPBC Act* provides little guidance as to how to achieve a triple bottom line.

TBL could be considered as a problem of co-viability. In this sense the space of co-viability would include the ecologically based Population Viability Analyses (PVA) as per Burgman and Possingham (2000) combined with Socio-Economic Viability Analyses (EVA) for the social pillar and economic yield for the economic pillar (Gourguet et al., 2013; Thébaud et al., 2014). However there are limits as to how many dimensions that can be analysed for co-viability. In the material presented by Thébaud et al. (2014), the social viability relates only to the size of the fleet and how much the fleet size can alter in any one given year.

Any application of triple bottom line to Australian fisheries much also take into account the diversity of policies and harvest control rules across fisheries, differences in state legislation and the issues related to the devolution of federal government powers to states and territories. Fisheries management in most countries, including Australia, have multiple objectives that are implicit in policy statements and there is little consistency in explicit requirements for triple bottom line approaches (Pascoe et al., 2013d; Brooks et al., 2015).

One well-known example in which federal powers are devolved to the Queensland State government is the Great Barrier Reef (GBR). Management of fishery resources in the Great Barrier Reef region aims to simultaneously achieve the objectives of both the *Queensland Fisheries Act* (1994) and the *Great Barrier Reef Marine Park Act* (1975) *(GBRMP Act)* (and be consistent with other relevant legislation) (COMLAW, 2008; State of Queensland, 2017a). The main objective of the *GBRMP Act* is to provide for the long-term protection and conservation of the environment, biodiversity and heritage values of the Great Barrier Reef Region. This Act allows ecologically sustainable use (including fishing) of the GBR Region, so far as it is consistent with the main object of long-term protection and conservation.

Recent proposed fisheries management reforms in Queensland provide a prime opportunity for harvest strategies to be developed that directly address the TBL.

Changes made in the *Queensland Fishery Act (1994)* in order to “allow for more responsive decision-making through the use of harvest strategies” took effect on 28 May 2019. While the TBL is known and cited in various Australian policies and legislation, the specifics of how to apply the TBL in a harvest strategy sense, let alone in the actual harvest strategies, have been lacking. Previously, under predominantly cost benefit analyses, economic objectives predominated, such that those goods and services lacking market values have been broadly ignored (Hanley and Spash, 1993; Hanley and Barbier, 2009; Marre et al., 2016b). Such ignorance of ‘externalities’, as they are termed in economics, may have further enabled delayed uptake of prescriptive means with which to create the values needed under the social side. There are also burgeoning needs to include and specifically measure, monitor and report regularly under indigenous and recreational legislation that are linked with co-management.

The major policy drivers that effectively impose restrictions related to TBL sustainability and more importantly processes that accredit sustainability are linked to four main processes: (i) access to export markets (under provisions of the EPBC Act and international agreements), (ii) eco-certification e.g. Marine Stewardship Council, (iii) licences and entitlements to fishery resource extraction, and (iv) the intersections between fisheries and conservation marine parks or high-level conventions or treaties (e.g. GBRMPA as a World Heritage Status).

Finally, while the Commonwealth Fisheries Act mandates the maximising net economic returns as the main objective by using MEY targets, this focuses on resource sustainability, and on having regard to (accounting for) the interests of all stakeholders, without explicit social objectives. Australian state fisheries policies mostly include economic and social considerations with resource sustainability as key focus.

Broader environmental values, condition and trends

Ultimately, fisheries utilise a natural resource and, as such, the consideration of environmental values, conditions and trends is an undeniable aspect of good fisheries management and stewardship. Fisheries management depends on good management today for ongoing resource sustainability in the future. Clark and Munro (2017) state that "The theory of capital is about stocks, addressing the question of what is the optimal stock of a particular type of capital. The theory of investment is about flows, addressing the question of what the optimal rate is at which a stock of capital should be increased, or depleted, if current stock of the capital is below or above the optimal stock level."

In most developed nations fisheries need to be sustainable (and accountable) in an ecological context broader than just fisheries stocks targeted by commercial fishers (e.g. EPBC Act, U.S.A. Magnuson–Stevens Fishery Conservation and Management Act). Additionally, since the environment in which many Australian fisheries operate is subject to multiple use management, other extractive industries occur concurrently with fishing. Additionally, fisheries are subject to environmental impacts and ecosystem effects, including pollution, run-off, changes in predator and prey abundance, and seasonal and long-term oceanographic changes affecting temperature, salinity and primary productivity over both short and long timescales.

Monitoring of environmental values, condition and trends is necessary to provide early identification of change drivers which can guide the tracking and analysis of both short- and long-term trends. Given this, management may be able to take steps to minimise cumulative threats and related pressures. This is particularly pertinent in the context of the (likely) chronic effects of climate change, and, for Queensland fisheries, acute events such as tropical cyclones.

The TBL concept has been put forward as an accepted international standard for robust full cost accounting of market and non-market natural resource values in a way such that both static information and changing trends can be embraced (Bebbington et al., 2007; Carpenter et al., 2009). Halpern et al. (2013) suggest that maximising conservation goals and achieving equity in social outcomes, while minimising overall costs, is the ideal TBL outcome.

TBL objectives

TBL objectives (and associated indicators) seek to include these wider aspects of sustainability. In Australia a Ministerial Direction 2005[2] had a prominent focus on integrated management (although the focus here was on stocks and economics, without explicit consideration of social objectives) and over the last decade objective-based management has become an expected and accepted approach (Fulton et al., 2014). TBL was developed to increase the likelihood of successful integrated management and is suited to adaptive management. The TBL approach also encourages the development of hierarchical objectives as long as the higher-level objectives are underpinned by indicators of performance measures and metrics. In the fisheries context, these performance measures (the value of

[2] available at http://www.daff.gov.au/fisheries/domestic/fishingfuture

performance indicators relative to some (typically, target or limit) reference level) are, in effect, monitoring the outcomes of the harvest control rule processes that are prescribed within a harvest strategy. It is also important when defining sustainability indicators for TBL to avoid “tokenism”, “greenwashing” and “box-ticking”, while appreciating both the utilitarian and values based arguments that currently drive conservation (Auster et al., 2009).

In the next subsection we consider social, environmental and economic objectives in more detail, based on a review of the literature. We emphasise that no matter what TBL objectives, indicators and performance measures are selected, the reliance on primary input and output measures will endure and take priority. If the price of fuel skyrockets or the market price of the catch is suddenly halved, then fishers and fishery managers may entertain options previously not considered viable. We suggest that during these times the consideration of alternative management strategies that widen the objectives of the fishery might enable adaption and resilience to outside market conditions.

Social objectives

Social objectives are arguably the least developed to date, except in very broad terms. There is increasing acknowledgement of the importance of including social objectives – beyond establishing and maintaining a social licence to operate and meeting corporate social responsibilities (CSR) (Elkington, 2006a; Kleine and von Hauff, 2009), even if those objectives defy quantitative measurement. However, broad social indicators have been criticised as being too fragmented and as such are lacking in explicit evaluation (Tullberg, 2012).

Anderson et al. (2015) acknowledge that assessing progress on social-ecological outcomes presents an urgent need for new frameworks to evaluate how management approaches interact with resource, community and market conditions to not only assure stock health, but also create economic and community benefits. It is also important to identify indicators around societal responses to policy, given that policy itself can trigger a social tipping point (Lenton, 2013).

The lack of clear, simply and widely adopted definitions of the social objectives is a key challenge of applying TBL to fisheries harvest strategies. The social bottom line implies goals for improving total quality of life and wellbeing by widening the ‘social’ beyond socioeconomic improvements to include topics such as social capital, social cohesion, social equity (Klein et al., 2015) and intergenerational equity. However, there is currently no agreed paradigm for a satisfactory quantitative social bottom line (Rindorf et al., 2017; Stephenson et al., 2017a).

Initially it was accepted that expanding the economic bottom line to include socioeconomic considerations (employment, training, and indirect benefits) was sufficient to assess the social bottom line. However, as Kennelly (2014) point out, despite challenges of social aspects often having no direct monetary value (therefore not easily measured), these social aspects must be specifically included in their own objectives. Similarly, it is insufficient to consider including social objectives by including only a sectoral approach within multi-sector fisheries.

It makes sense to limit social objectives to those strongly linked to the fishery in question. Social objectives can often be broader than socioeconomic objectives even though those objectives are much more likely to be within the area of fisheries management influence and control, per Figure 1 and Figure 2 (Triantafillos et al., 2014; Brooks et al., 2015).

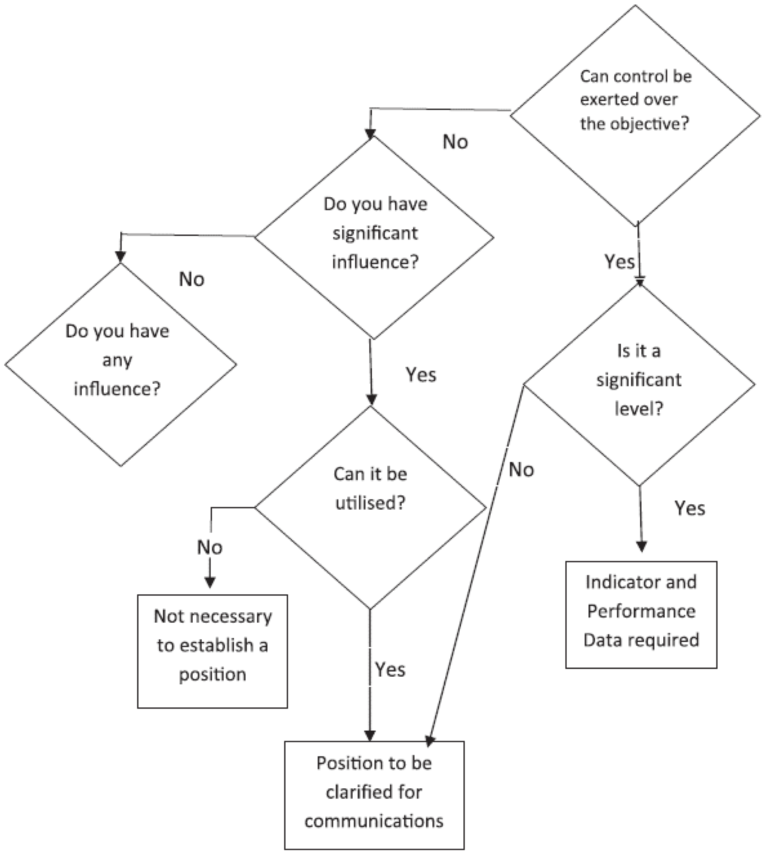


Figure 1: Decision-making flowchart guiding development of potential social objectives (Brooks et al., 2015)

Social objectives can be broken down into multiple sub-categories:

- Social for main users (commercial – e.g. supporting local community employment and economy, numbers of days off, safety, enjoyment of fishing, recreational – supporting local community and economy, quality of experience, whether by metrics of strike rate or otherwise (e.g. quality of the environment), indigenous – e.g. able to provide subsistence)
- Social for broader community locally (e.g. fishing activities support employment, attractiveness and lifestyle of area given fishing activities)
- Social for users other than fishers (e.g. quality of environment for tourism divers)
- From an economic theory, whereby non-use values and option values (today's willingness to pay by individuals who wish to conserve the asset for future use) fall into the social pillar.

Bebbington et al. (2007) state that social impacts capture "both positive and negative aspects of: indirect employment associated with a project, offset by deaths and accidents arising during employment above the entity paid costs; contributions to creating a socially sustainable society; and perceived benefits of products or other outputs of the project”.

Currently integration of social objectives in fisheries management does not extend beyond routine requirements (Rindorf et al., 2017). Symes and Phillipson (2009) say that, in an Australian fisheries context, social issues should include “including access to fishing rights, renewal of the industry’s social capital and the sustainability of fishing communities”.

Defining social objectives quantitatively is a challenge (Symes and Phillipson, 2009; Vieira et al., 2009; Triantafillos et al., 2014; Brooks et al., 2015; Pascoe et al., 2017). Often innovative proxies need to be used (Mangel and Levin, 2005; Mangel and Dowling, 2016). While such proxies can simply be interpreted as having weight from both environmental/economic and social perspectives, independent social objectives and associated metrics are desirable (Anderson et al., 2015). Also, social metrics are not always independent, and even if they are, they are often strongly correlated with environmental and/or economic objectives.

FAO have progressed to using indicators to measure performance against specific social objectives of fisheries management, rather than measuring only the more generic social characteristics of fishing communities (Brooks et al., 2015). A relatively recent comprehensive study developed a hierarchical structure for social objectives in Australian fisheries (Figure 2).

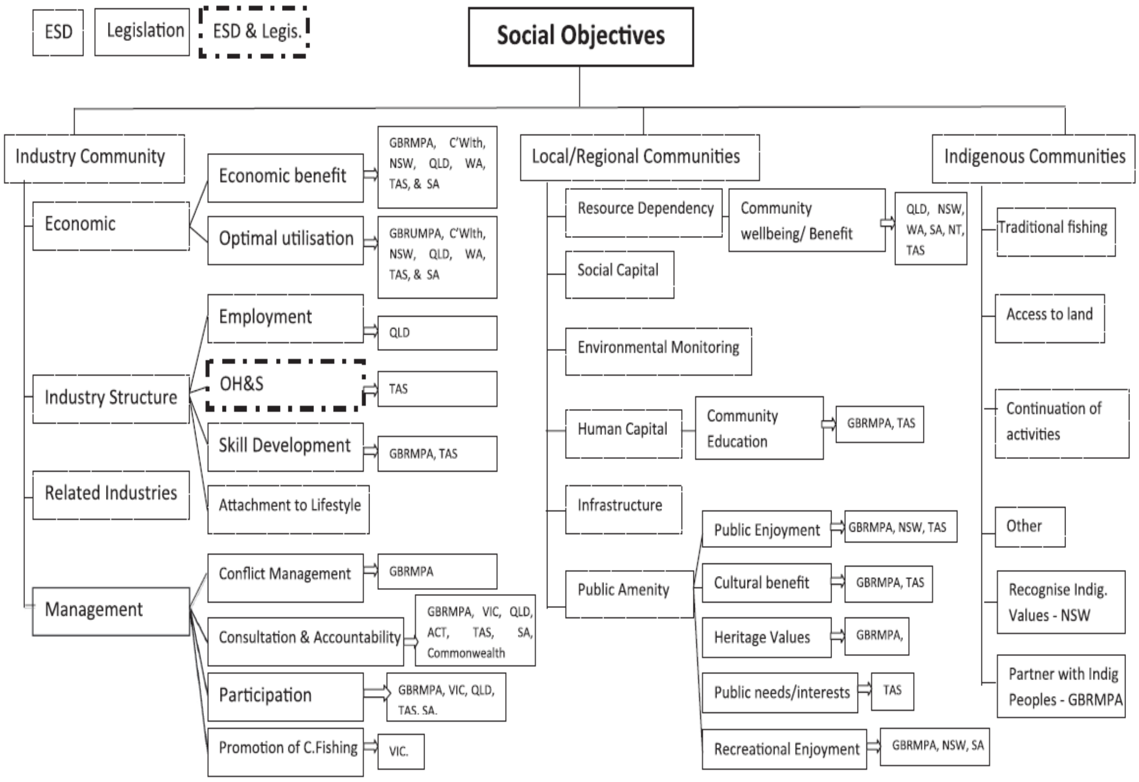


Figure 2: Detailed substructure proposed for social objectives. Source: (Triantafillos et al., 2014; Brooks et al., 2015)

Taking ecosystem services and representing them as social values is a relatively new approach, and there have been recent statistical advances in mapping presence data of social values (Sherrouse and Semmens, 2013). These enable prediction of the likely social values of an area even if social values data for that particular community have not been surveyed (or included in some other manner). These models have interlinking components and are able to calculate indicator values from survey data and generate value indices, and maps of those values and environmental metrics.

Social objectives and indicators to measure progress towards these objectives should be reflections of underlying social values. For example, Pascoe et al. (2015) showed that fisher satisfaction was built upon a sense in pride in their profession and the importance of maintaining family tradition. Other social values are the indirect, non-use or future option values; these are sometimes referred to as stewardship values (Wills, 2006).

Kenter et al. (2015) identify five dimensions across which shared (social) values need further appreciation and understanding: scale, intention, value concept, provider and elicitation processes. They highlight the relationship between social values and total economic value and review non-monetary and deliberative methods to establish shared values in a decision-making setting.

Finally, Bond and Morrison-Saunders (2011) urge the consideration of appropriate timescales for social objectives, since these usually apply over longer time periods than, say, updated fishery stock assessments may be updated. They provided evidence from both England and Western Australia showing that intergenerational equity, a social sustainability principle enshrined in policy, does not form a significant part of many sustainability assessments and they suggest that is due to the assessment being driven through a target-species-based decision-making context.

Environmental objectives

Environmental objectives are important from both the exploited and non-exploited stock sustainability contexts plus they can capture the wider importance from an intergenerational equity perspective as healthy, functional ecosystems ought to be passed onto future generations (Millennium Ecosystem Assessment, 2005; TEEB, 2010). In addition to inducing pressure on fish stocks, certain types of fishing can pose other environmental threats such as illegal fishing, bottom trawling, “ghost” fishing due to loss of gear, pollution, and the use of explosive devices for fishing. Environmental objectives, then, need to go beyond sustainability of just the targeted stocks to encompass the sustainability of the ecosystems in which the fishing stocks are found in order to limit environmental impacts of fishing activities. As stated earlier, environmental (or ecological) objectives can be from many contexts beyond the target species.

Environmental objectives also need to consider the impact of non-fishing activities occurring in the region of interest, such as shipping, oil and gas mining, as well as the chronic effects of regime shifts, most notably due to climate change, and from acute events such as tropical cyclones.

Environmental objectives frequently interact with economic objectives. For instance, the recent environmental damage caused by Severe Tropical Cyclone Debbie to the GBR coastal and islands infrastructure caused damage across the triple bottom line. It is also important to consider timeframe with all of the environmental sustainability and associated objectives: short-term objectives can differ, and in fact be contradictory to, long-term objectives.

DEWHA (2003) provide detailed information on the benefits of TBL as well as methods for identifying and selecting environmental indicators. However, there can be substantial regional differences between what are considered appropriate environmental objectives (Rindorf et al., 2017). It is a legislated requirement that assessments be conducted of fisheries environmental performance (Part 10 of the *EPBC Act)*, impacts on protected marine species (Part 13) and species/fisheries product that require export approval (Part 13A). In these considerations, Chesson and Whitworth (2004) recommend general ecosystem impacts to be further subdivided into:

1. Impacts on ecological communities
    - Benthic communities
    - Ecologically related, associated or dependent species
    - Water column communities
2. Impacts on food chains
    - Structure
    - Productivity/flows
3. Impacts on the physical environment
    - Physical habitat
    - Water quality

As such, scientific advice to fisheries managers regarding stock and conservation limits and targets involves complex trade-offs, having moved beyond the original static and deterministic maximum sustainable yield (MSY) target, the latter of which led to leading to economic and social problems in fishing communities (Rindorf et al., 2017). Governance objectives are specified in policy documents, including the EU Common Fisheries Policy: “principles (of good governance) include decision making based on best available scientific advice, broad stakeholder involvement, and a long-term perspective”.

When developing environmental objectives in a triple bottom line context there may be less need to cover all possibilities (areas of direct impact and indirect impact) since some environmentally-linked issues will be covered in the social and the economic objectives. This can be different to when an environmental management plan is created in the absence of social and economic objectives.

Economic objectives

At the Commonwealth fisheries level, a key management objective is that of maximising sustainable economic returns (DAFF, 2007), defined primarily in terms of sustainable fishery profits over time (Pascoe et al., 2017). Management strategy evaluations have long focused on the trade-offs between ecological/sustainability objectives, and economic objectives. As with ecological/environmental objectives, economic objectives can be short- or long-term, with the latter more closely aligning with sustainability objectives. Short-term volatility is typically undesirable, with preferences for certainty in returns and economic stability. However, profiles of economic objectives naturally vary, both across and within stakeholder groups. For some sets of economic objectives, multi-sector balancing calls for objectives that span the trade-off spaces. For instance, in the case of an industrial harbour, one of the economic objectives explicitly recognised the trade-offs with the social and environmental aspects via the objective “enhance values of recreational and environment assets” (Pascoe et al., 2016a).

Commercial fisheries throughout the world commonly use economic objectives that span economic prosperity and viability, access and distribution of benefits, as well as contributions to the regional economy (Stephenson et al., 2017b). More recently, the focus in economic objectives has broadened further based on ecosystem services to the inclusion of livelihoods derived from ecosystems that support fisheries, e.g. coral reefs (Cinner, 2014).

For the most part economic objectives are generally easily measured, quantitative in nature and well accepted conceptually by the public and stakeholders alike. Fisheries landings in Australia are recorded as part of the gross value of production (GVP) which amounted to $313.8 million in 2008-09 financial year (Wilson et al., 2010; Pascoe et al., 2016a). Fisheries provide employment and training opportunities and economic objectives for these metrics are recorded.

Australian fisheries were early adopters of maximising net economic returns as a primary economic objective of fisheries management. The more recently prioritised social objectives (needed to balance the triple bottom line) which are usually more difficult to define and quantitatively measure, therefore present further challenges and the need to link economic and environmental objectives with social objectives in fisheries models (Pascoe et al., 2017).

In some situations, institutional and governance objectives are part of the economic objectives. Some authors suggest governance as the fourth part of a quadruple bottom line (Garcia et al., 2003; Rindorf et al., 2017), the argument being that without reliable, appropriate governance structure that considers both the broader institutional objectives, ensures effective decision-making processes and also fulfils legal obligations including to indigenous peoples, the economic returns could be compromised.

Emery et al. (2017) discussed challenges to the implementation and continued use of economic analyses and instruments. These included: (i) short-term transition costs and associated trade-offs between ecological, economic, social and political objectives; (ii) scarce logistical and financial capacity to collect and analyse economic data; (iii) a lack of desire among industry to change and transition to economic targets such as maximum economic yield (MEY), particularly when it is associated with lower catches; and (iv) a lack of economic literacy among fisheries managers and industry. It is contended that many of these challenges may apply to all of the TBL objectives, and initially arise from an absence of clearly identified and prioritised objectives within overarching legislation and management plans. Once objectives are prioritised, limited resources can be allocated more efficiently to improve data collection, economic analysis and increase awareness as well as education of managers and industry.

Is the Triple Bottom Line Achievable in an operational, harvest strategy context?

In the Queensland context, many Great Barrier Reef fisheries, and, more generally, many fisheries globally, are multi-sector, with commercial, recreational and sometimes charter and/or indigenous stakeholders. The challenge, therefore, is achieving the TBL of environmental/ecological, economic and social management objectives, in accordance with legislative requirements, and simultaneously across different user groups. In practice, multi-sector fisheries are most directly confronted with the TBL, yet their data quantity and quality are often mixed, reference points and performance indicators vary between them, and environmental, economic and social information for all sectors is often limited. Effectively implementing a TBL management framework still requires much work, particularly for mixed fisheries.

In fisheries, several jurisdictions have legislated the use of TBL objectives. For example, the United States Magnuson-Stevens Fishery Conservation and Management Act (1996) mandates their use in National Standard 8. In Australia, the Queensland Government's Sustainable Fisheries Strategy 2017–2027 states that TBL objectives should be used in conjunction with harvest strategies for all major fisheries that fall within their jurisdiction (State of Queensland, 2017b). Harvest strategies comprise pre-agreed monitoring, 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 (Sainsbury et al., 2000; Punt et al., 2002; Butterworth and Punt, 2003).

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). (Note that, although social objectives within a TBL harvest strategy may including maintaining equity between fishing sectors (and such objectives may be heavily weighted by certain stakeholders), the formal allocation of TAC between sectors is not the mandate of a harvest strategy). The implementation of TBL, however, remains problematic and it has not been operationalised with fishery harvest strategies (Mangel and Dowling, 2016). Indeed, Elkington (2018) sought to recall and rethink the concept, stating that it has "failed to bury the single bottom line [economic] paradigm". To date, consideration of the TBL has been largely limited to conceptual treatment (Stephenson et al., 2017a) or intuitive forecasting methods using expert opinion (Bernstein and Cetron, 1969; Dichmont et al., 2012b; Dichmont et al., 2014; Pascoe et al., 2019). A further review of pertinent literature around operationalising TBL HSs is provided in Appendix A.

Beyond the explicit incorporation of all TBL objectives, formal methods that have attempted acknowledge the TBL result in discrete strategies do not consider stakeholder's preferences (weightings) across the range of objectives, and provide no formal means of determining the optimal solution given these weightings. Pascoe et al. (2013d) 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 Eastern 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.

Thus, operationalising the triple bottom line, beyond a simple conceptualisation is complex. To embed the TBL in formal management, 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.

Overview of the project

The recent Queensland Sustainable Fisheries Strategy (Department of Agriculture and Fisheries, 2017) aims to move the State's fisheries towards a more ecologically, economically and socially sustainable future. As part of this process, a number of Fisheries Working Groups have been established to identify fishery objectives and harvest strategies aimed at moving the fisheries forward. These Working Groups include a wide range of stakeholders, including the commercial, charter and recreational sectors, with explicit consideration given to the indigenous sector; processors and buyers; scientists, fishery managers, marine park managers and conservation groups.

In 2017, Queensland was essentially at the starting gate and was in prime position to begin to engage with stakeholders in the context of harvest strategy development. While triple bottom line objectives should be elicited and weighted, the technical challenge of evaluating these in a harvest strategy context for all its fisheries perhaps should be a longer-term goal for Queensland fisheries. The aim of this project is to outline a process to develop TBL harvest strategies for a complex multi-sector fishery to achieve ecological, economic and social sustainability objectives. The Coral Reef Fin Fish Fishery (CRFFF) is one of Queensland's most valuable in terms of export earnings, and has important recreational, commercial and charter sectors.

The process developed in the project involved first identifying and prioritising the objectives for the fishery with the key stakeholders. Given these objectives, modifications to the current harvest strategy were developed that potentially considered these broader objectives that stakeholders believed may enhance the fishery's performance.

The effectiveness of these additions at improving performance was examined using a Multi Criteria Decision Analysis, through a qualitative impact assessment against the objectives. The overall probabilities that the options would improve the fishery's outcomes were determined. The process provides a roadmap for future harvest strategy development to achieve ecological, economic and social objectives.

Given the legislative mandate to set total allowable catches (TACs) based on TBL objectives and their associated performance indicators, the challenges around TBL management 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, across 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 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.

As such, we take a second, quantitative, non-commensurable unit approach, via a multi-indicator objection function (a function comprising all of the relevant TBL indicators, normalised to range between 0 and 1, with each weighted according to stakeholder preference, and summed to form an overall value function whose maximum value is sought for any given stakeholder group) within a simulation, with explicit objective preference weights to set total allowable catches for three main species groups in a reef line fishery. The emphasis is on the methodology, and on offering a means to explicitly incorporate all TBL objectives as quantifiable and comparable.

This approach is consistent with the "efficiency frontier" presented by Halpern et al. (2013), whereon optimal solutions lie, and represent different importance (weight) given to conservation versus equity goals. As opposed to the approach of Rindorf et al. (2017) that takes a suite of fishing mortalities corresponding to sustainable yield, and progressively refines this, we consider the TBL objective weighting profile for given stakeholder groups as an integrated value function that is optimised across a suite of catch levels.

Based on our learnings from "living" the process of stakeholder elicitation and preference weighting of TBL objectives, guiding the development of TBL harvest strategy options, and evaluating these against the objectives using the above-described two approaches, we developed a General Methodology that is intended to serve as a user-friendly process that a manager can practically apply from this project.

Indigenous fishers and Queensland's Aboriginal and Torres Strait Islander commercial fishing development policy

In general, engagement with Aboriginal and Torres Strait Islander communities and stakeholders across Queensland is challenging, particularly on broad policies such as harvest strategies. Fisheries Queensland has found that if an action is not directly impacting a community it is difficult to obtain input, which is not surprising given the amount of feedback being sought from Aboriginal communities by various agencies.

However, the indigenous sector was consulted on the Aboriginal and Torres Strait Islander commercial fishing development policy[3]. The Aboriginal and Torres Strait Islander commercial fishing development policy specifies that an Indigenous commercial allocation be set aside in a harvest strategy where appropriate to make sure there is an amount of harvest available for use under an Indigenous fishing permit (for commercial use).For the CRFFF, this policy allocates 5 tonnes to indigenous fishers to use under indigenous fishing permits, which is their allocation in the harvest strategy.

---

[3] https://www.daf.qld.gov.au/?a=109113:policy_registry/fish-comm-atsi-dvlp-policy.pdf

All Queensland harvest strategies recognise traditional fishing rights are not impacted by the harvest strategy, and all include the wording, “ *The traditional fishing rights of Aboriginal peoples and Torres Strait Islanders are protected under native title legislation and relate to harvest for domestic, communal and non-commercial purposes. Accordingly, traditional and customary fishing is not a defined allocation”.*

Objectives

Original objectives

1 To undertake a review of (existing work around) multi-objective management systems and associated assessment approaches (Phase 1)

2 To compile an inventory (incorporating existing work, and particularly that of 2013/204) of current environmental, economic and social objectives and consider how to translate such conceptual management objectives to operational objectives for multi-sector fisheries (Phase 1)

3 To develop a theoretical framework (incorporating existing) methods and approaches with which to evaluate trade-offs between, and/or priorities for, environmental, economic and social objectives (Phase 1)

4 Key Objective: to develop a General Methodology for harvest strategy development against the triple bottom line for multi-sector fisheries (Phase 1)

5 To finalise the choice of Queensland state-based multi-sector case study fishery (Phase 1)

6 Key Objective: to develop and recommend (but <u>not</u> to formally implement and/or operationalise) a triple bottom line harvest strategy framework for a Queensland multi-sector case study fishery (Phase 2)

Modification of scope

Broader engagement with Fisheries Queensland: benefits provided by the project team to the management reform process

The project team devoted considerable time early in the project to assisting with the Queensland fishery management reforms in a broader sense. In July 2016, a project workshop in Brisbane was largely devoted to reviewing the DAF Green Paper on proposed Fishery Management Reforms. The project team provided detailed but informal advice and feedback.

In June 2017, the project team held a day-long workshop with eight staff from DAF, including the Directors of Management and Reform, and of Monitoring and Research. The goals of the workshop were to provide an update of project progress to DAF, for DAF to update the project team with regard to the current status with regard to their proposed management reform process, to share a range of tools, products and pieces of work that may be of relevance to DAF in their current management reform/harvest strategy development process, and to offer the support of the project team against the same, and to finalise the choice of case study fishery for the project.

Following this workshop, at the request of DAF, Natalie Dowling travelled to Brisbane in July 2017 to hold a dedicated, two-day workshop that provided training on the FishPath data-limited harvest strategy decision support tool.

Within the CRFFF Working group, the project team generally helped to guide the harvest strategy development process, and to educate stakeholders about harvest strategies and their role in the context of their development and implementation.

These engagements provided insight to the project team of the needs and challenges within the Queensland fishery management reform process, which helped to obtain a broader appreciation of how to approach our project.

Divergence of Queensland fishery management reforms and this project

As the project progressed, there was a divergence of the Queensland fishery management reforms, and the project. The original intention had been to develop and recommend a triple bottom line harvest strategy framework for a Queensland multi-sector case study fishery, such that this could inform the choice of harvest strategy in alignment with the timeframes of the reform process.

However, due to a combination of:

i) The need to have placed emphasis on the education of the process of harvest strategy development
ii) The delays caused by the 2017 Queensland State Election, and the timing of, and attendance at, the CRFFF WG
iii) The technical demands of developing the simulation model
iv) The practical constraints for DAF to implement radically alternative harvest strategies to the status quo, at least in the first instance.

The project was limited in its ability to provide a recommended, operational TBL harvest strategy in time for the DAF deadline at the end of 2018.

The project team did lead the process of eliciting, identifying, and fleshing the details of, alternative harvest strategy options, and the MCDA approach was completed in time to assist the selection of the initial harvest strategy of choice.

Ideally, the outcomes and recommendations from this project will be able to more comprehensively assist DAF, and the CRFFF in particular, at the point of the first formal review of the harvest strategies.

Method

Choice of case study fishery: a formal approach

Finalising the choice of Queensland state-based multi-sector case study fishery was a key goal of the June 6, 2017 workshop held with the project team and 8 members of DAF.

We undertook a transparent process of listing potential case study fisheries, scoring these according to available expert opinion against a list of relevant caveats, and thus determining the most suitable case study fishery for the purpose of the project. This process is detailed in Table 1.

Whilst acknowledging the urgency from a DAF viewpoint to formulate harvest strategies for their three major fisheries, the group determined that the Coral Reef Finfish Fishery, with an initial focus on Coral Trout, best served the purposes of the project.

The Queensland Coral Reef Finfish Fishery (CRFFF) operates mostly within the Great Barrier Reef Marine Park, spanning a broad latitudinal range from Cape York (101˚41'S) in the north, to Bundaberg (24˚ 30'S) in the south. In order of decreasing value, 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 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 hour) 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, 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 total allowable commercial catch (TACC), allocated via individual transferable quota units (ITQs), trade-ability of input and output entitlements, and seasonal spawning closures. A fishery specific Working Group (WG), 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, as well as managers, and scientists.

While there is a small amount of indigenous harvest, and this was considered explicitly in the formulation of TBL objectives, the WG was the body with whom the project engaged, and its membership composition was determined independent of this project. Currently there is no indigenous representative on the Reef Line Working Group. However, as stated above, the Aboriginal and Torres Strait Islander commercial fishing development policy specifies that an Indigenous commercial allocation be set aside in a harvest strategy where appropriate to make sure there is an amount of harvest available for use under an Indigenous fishing permit (for commercial use). For the CRFFF, this policy allocates 5 tonnes to indigenous fishers to use under indigenous fishing permits, which is their allocation in the harvest strategy.

Significant management challenges within the fishery included:

- that the Coral Trout quota group is a complex of seven Coral Trout species - increasingly, there is a need to ensure management of each species separately

- that the fishery can be significantly impacted by cyclones. For example, in 2009 cyclone Hamish travelled the length of the Queensland coast, resulting in in depressed catch rates, and fleet displacement
- the complexity of implementing regional management on a fishery with allocated TACC's already in place
- that there are limited economic/social data that are able to be used with confidence.
- the changing of working group membership over the course of the project
- legislated timeframes to have a HS completed.

Table 1. Process for “traffic light” scoring of potential Queensland case study fisheries, against relevant caveats. A red “score” indicates the fishery fails against the caveat, yellow indicates the caveat is partially addressed, while green indicates that the fishery adequately addresses the caveat.

Fishery	Multi-sector	**Scope for consideration of multi-faceted social objectives**	**Adequate breadth of environmental objectives**	Established WG/ industry group (touchdown group)	Minimally politically contentious	Aligns with Qld DAF priorities	Effective existing management measures	Simple, but not trivial in ability to demonstrate proof of concept	Existing models/ analysis	Transferability	Confronted by over-arching environmental changes	Willingness of stakeholders	Number of stakeholders (red if high; green if low)
EC Trawl (Eastern King Prawn)				Has had									
Coral Reef Finfish Fishery			Severe cyclone damage to reef and significant risk to this fishery					Can be staged; Complexity comes in with OS	Not for OS				
Ocean Beach	Yes Taylor, no mullet							Too simple			Mullet rely on rainfall and freshwater flow		
Spanish Mackerel								Too simple					
Crab											Rely on rainfall	Want to engage on reform but potential to be volatile	
Tropical Rock Lobster	Spatial delineation			Industry group				Too simple			Affected by currents		
East Coast Net								Too complex			Rely on rainfall		
Moreton Bay regional								Too complex	Multispecies prawn - but large non-trawl sectors				

Identifying Triple Bottom Line objectives

Here we review the process of eliciting objectives, considering also appropriate performance indicators and measures that correspond to each. We then review principles of what has worked in identifying, trading off and reconciling those objectives, as well as likely pitfalls. Underlying these is a tacit agreement that stakeholder and community participation is both desirable (Rice et al., 2012), necessary, and required in Australia (under various legislative instruments e.g. EPBC Act (1999), Queensland Fisheries Act (1994)).

Two decades ago de la Mare (1998) criticised management objectives as being vague and unclear as to what they actually mean in practical terms and called for measurable operational objectives that are clear and able to address both long and short time scales. Considerable research efforts have been devoted to progressing best practice in this regard. Most recently, the chapter of Schwermer et al. (2020), entitled "A literature review on stakeholder participation in coastal and marine fisheries", lists types of intentions for participation and under the definitions provide the sub-types of objectives, criteria for their evaluation and management implications.

The process of eliciting objectives is multi-stage. In some cases, higher-level objectives are imposed by the relevant government policy and fishery management plans, but these may need to be locally specified in greater detail, and augmented by other objectives. A brief literature review of the process of eliciting objectives is provided in Appendix B.

Importantly in the context of this project, Pascoe et al. (2013d) developed an objectives hierarchy. A preliminary hierarchy was drawn from a comprehensive review of natural resource management objectives, and was cross-referenced to policy documents related to the fishery and the GBR marine park, and to key legislation by interdisciplinary researchers. Subsequently, the Scientific Advisory Group (scientists, fisheries managers and industry members from catching and processing sectors) agreed on the final hierarchy by consensus and this was adjusted slightly with minor additions by the government department responsible for the management of the fishery (the Department of Employment, Economic Development and Innovation[4]); see Figure 3.

[4] Now split into several Departments, including DAF and the Department of Environment and Science

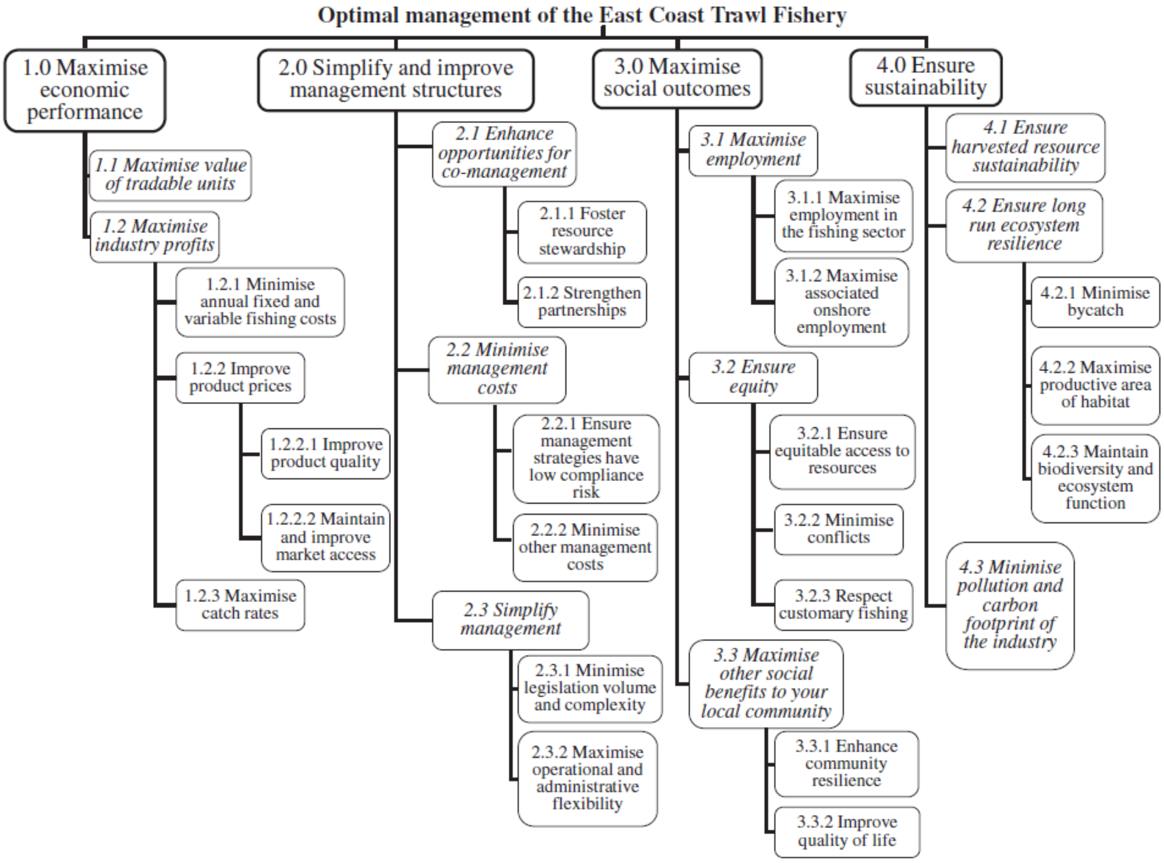


Figure 3. Objectives hierarchy developed for the Queensland East Coast Trawl Fishery (Source: Pascoe et al. (2013d))

Literature Review: identified pitfalls

One current pitfall is the challenge of accounting for the breadth of environmental objectives, and absence of explicit social objectives for fisheries, though the introduction of mandatory impact assessments are expected to ensure closer attention to social issues at an earlier stage in the policy development (and review) process (Symes and Phillipson, 2009). The process of eliciting of objectives should not be done in isolation as a ‘feel good’ exercise. Instead eliciting objectives needs to be an integral part of the whole management system (indicators, performance measures, triggers, control rules, and thresholds). But what might work will be highly dependent on the context and the priorities at that time. That is, the triple bottom line is dynamic in the sense that different objectives may have different weightings at different times. The management system must therefore be subject to regular review and be adaptive to change.

Social objectives need to consider social issues and how those interact with projects, policies, infrastructure programs and other planning activities. While considerable guidance for eliciting fisheries social objectives is provided in Pascoe et al. (2014a) and in more detail can be found in Brooks et al. (2015) there are still considerable practical challenges for the inclusion of a wide suite of social objectives. Typically, the process of eliciting social objectives has focused on prioritisation rather than identifying risks and determining objectives to minimise and manage those risks. Therefore ideally setting overall policy objectives prior to the determination of a harvest strategy or management plan can ensure that the social objectives are elicited for the industry community, the indigenous community and the local/regional community (Triantafillos et al., 2014).

From an ecosystem perspective, Long et al. (2017) found that the most important principles of EBM, from the fishers' perspective, can differ greatly from those in the EBM literature. The practical implications of this is the need for better recognition by management players of fishers' priorities embedded within EBM principles. In turn this may well generate greater on-the-ground support and thereby aid EBM implementation. Moreover, fishers need and want to be included in the process from the beginning, as bringing their expertise and perspectives, in addition to academic and institutional analyses, alters the selection of appropriate EBM principles (Long et al., 2017).

Another pitfall is the lack of integration across TBL objectives. Stephenson et al. (2017a) give three key impediments: a relative lack of explicit social, economic and institutional (management) objectives; a general lack of process (frameworks, governance) for routine integration of all four pillars of sustainability; and a bias towards biological considerations. They provide five practical steps to overcome this issue and move toward integrating ecological, economic, social and governance aspects are advocated, as follows:

(1) adopt the perspective of the fishery as a 'system' with interacting natural, human and management elements;
(2) be aware of both strategic and operational aspects of fisheries assessment and management;
(3) articulate overarching objectives that incorporate all four pillars of sustainability;
(4) encourage appropriate (and diverse) disciplinary participation in all aspects of research, evaluation and management; and
(5) encourage development of (or emulate) participatory governance (management) processes.

Chesson and Whitworth (2004) reviewed 27 Australian regional marine planning assessment systems (analysing objectives, indicators, and performance measures linked to decision rules, data collection, monitoring and reporting) and found that none of the cases "provide a comprehensive, ongoing assessment of all the types of objectives that will be required for regional marine planning purposes." This is a clear reminder that even if considerable effort is put into eliciting TBL objectives and measures of achieving these, the objectives and metrics must be fit for the overall purpose of fisheries management.

Rules of thumb (heuristics) for objectives, and associated reference points, triggers and thresholds need further development to meet multi-sectoral sustainability criteria, though some metrics are available for single sectors or at the individual project level (Senner, 2011). Lack of metrics has implications for the trading-off process (see next section). However, risk analysis approaches do allow analysis for a range of potential scenarios and this in itself can be a form of trade-off exploration; trade-offs in supported decision-making and management actions selection can occur across sectors, or just within a single sector (Dickey-Collas, 2014). Similarly, constraints mapping is another stakeholder engagement tool that can be used to determine areas of non-negotiability and areas that may provide trade-offs in the exploration phase (Bond and Morrison-Saunders, 2011). By understanding early in the process where the ‘no go’ areas are, the options left for possible selection become more visible and, indeed, obvious to all stakeholders.

Lack of appropriate metrics (especially in multi-sector fisheries) is further complicated in Australia by State, Territory and Commonwealth agencies collecting, analysing and reporting different fisheries performance measures against inconsistent objectives (van Putten et al., 2015). Conversely, the European Commission (EU) guidelines provide clear, consistent standards and require measurement and reporting of social impacts in terms of employment, social inclusion, non-discrimination, privacy, health and safety. Consistency across Australian fisheries agencies would enhance standards of practice by allowing comparison between fisheries in terms of their TBL sustainability performance. Hobday et al. (2016) draft Healthcheck has begun to address this by providing an approach to summarise available information to document the sustainability of Australian fisheries. The draft Healthcheck categories include social, economic and governance factors that have not consistently been included in fishery assessments to-date, alongside common biological considerations, such as stock status. A pertinent reason for the lack of objectives in multiple use cases is simply that, in general, the objectives and performance measures are put in place and monitored independently by each agency responsible for managing the particular resource, industry, or sector (e.g. fisheries, forests or national parks) (van Putten et al., 2015).

A final pitfall is selecting only easy-to-measure objectives. Bond and Morrison-Saunders (2011) found that sustainability assessments in Western Australia based on the existing environmental impact assessment (EIA) process to more explicitly accommodate socio-economic factors risk being reductionist by breaking each proposal down into discrete parts and then assigning environmental objectives which may not actually adequately represent environmental functions.

Process of TBL objective elicitation for this project

Previous studies of fisheries management objectives (and natural resource management objectives in general) identify that generally a hierarchy of objectives is developed, with higher level objectives being the typical triple bottom line categories of economic, social and environmental objectives, and lower level objectives being more detailed or specific objectives for the fishery in question (Leung et al., 1998; Mardle et al., 2002; Soma, 2003; Wattage and Mardle, 2005; Pascoe et al., 2009c; Jennings et al., 2016b). A similar approach was adopted for this study, although a fourth higher level objective – simplifying management – was also included as these had been previously considered important to both managers and fishers (Pascoe et al., 2013d). Consideration of institution objectives is also considered best practice in a developing good management (Stephenson et al. 2017).

Our broader review of triple bottom line objectives (Table 11), included the review of objectives previously applied in Australian fisheries (Pascoe et al., 2013b; Pascoe et al., 2014a; Brooks et al., 2015; Jennings et al., 2016b; Farmery et al., 2019). We identified a subset 75 different potential objectives that were relevant or applicable to the Coral Reef

Finfish Fishery. Each of the objectives fell in one of four triple bottom line categories: ecological/environmental, economic, social and management (institutional).

A series of workshops were held with members of the Coral Reef Finfish Fishery Working Group to identify the triple bottom line objectives that were of relevance to the fishery. The list of identified objectives as well as some example objectives identified in an earlier Queensland study (Pascoe et al., 2013b) and the concepts around the development of an objective hierarchy were presented to the working group in the first of these meetings (November 2017). Working group members broke into smaller groups to identify which of these earlier objectives may be applicable to their fishery, which needed modification and which new objectives specific to their fishery were required. A revised potential set of objectives was then compiled based on the outcomes from the group discussions.

Between meetings, the project team translated each of these potential conceptual objectives into operational objectives. To be considered operational, they needed to be: i) realistic, ii) simulation-achievable, and iii) have performance indicators against which each objective could be assessed.

A revised set of potential objectives was presented at the subsequent working group meeting (March 2018), and the set of objectives for the fishery was finalised through further discussion with the working group.

A total of 22 operational objectives were agreed by the working group (Table 2, from Pascoe et al. (2019)). These were arranged into a three-level hierarchy, with the top level consisting of sustainability, economic, governance and social objectives.

A number of other objectives (mostly governance and social objectives) were also considered important by the working group, but it was recognised that these could not be influenced by a harvest strategy, so were not subsequently considered in the further analysis. It was noted that these additional objectives need to be considered when developing broader management structures. These overarching but non-operational objectives, are presented in Table 3.

Table 2. Objective hierarchy identified with the working group

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, 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 Minimise risk to Other Species in the fishery which are not included in 1.1.1.
	1.2 Ensure ecosystem resilience	1.2.1 Minimise risk to bycatch species
		1.2.2 Minimise discard mortality of target species (e.g. high grading)
		1.2.3 Minimise broader ecological risks
		1.2.4 Minimise risk to TEPS
	1.3. Minimise risk of localised 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 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)	

Overarching objective	Sub-objectives	Specific objectives
	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 (rec), 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 Maximise 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 Maximise spatial equity between regions or local communities

Table 3. Objectives considered important by the CRFFF WG, all of which are largely, or wholly, beyond the direct control of a harvest strategy. They remain important objectives within the broader fishery management regime but cannot to be directly addressed within a TBL harvest strategy. The “Description” column is informal and captures the notes from a follow-up project team discussion. Different overarching objectives are assigned different colour shadings.

Overarching objective	Specific objectives	Description
3 Enhance management performance	3.3 Ensure management and compliance is appropriately resourced Managers may want to discuss whether there is need for a more explicit objective re: reducing illegal fishing (including catching legally but selling illegally)	Management should also be prioritised to focus on high-risk compliance activities. Conduct sufficient on-water surveillance to minimise IUU fishing. If this cannot be appropriately resourced due to top-down ministerial decisions, we have no control over how we can ensure appropriate resourcing. DAF has little control over the level of resourcing provided. Does this speak to instead taking a flexible and pragmatic approach in scaling and prioritising management and compliance given the available resources?

Overarching objective	Specific objectives		Description
	3.4 Ensure management acceptability: transparency and inclusiveness	3.4.1 By recreational, charter, commercial and indigenous fishers	This is about appropriate bottom-up engagement and consultation from point of inception, as well as undertaking the process in a defensible, standardised and transparent manner.
		3.4.2. By the Community (Social licence to operate)	https://socialicense.com/definition.html: Social License has been defined as existing when a project has the ongoing approval within the local community and other stakeholders, ongoing approval or broad social acceptance and, most frequently, as ongoing acceptance. At the level of an individual project the Social License is rooted in the beliefs, perceptions and opinions held by the local population and other stakeholders about the project. While there are dedicated social objectives defined elsewhere, this objective is about management being accepted and approved by the community beyond those directly involved in the fishery. The level of responsibility of an agency to ensure this is questionable. However, it is to the fishery's advantage if the general community perception that it is well managed. Fish are a community resource, but the community often is indifferent.
	3.5 Ensure appropriate data are collected (economic, environmental, etc.) and research		"Supporting effective management" equates to tailoring the monitoring program and stock assessment (calculation of performance indicators) appropriate to the fishery objectives. This also embraces research needs identified (e.g.) out of the Resource Advisory Committees, that will lead to an improved understanding and management of the fishery.

Overarching objective	Specific objectives	Description
	undertaken to support effective management	
	3.6 Management evaluation/ review undertaken	This is simply ticking off against pre-specified evaluations and review. These should be set in timeframes appropriate to the fishery's context (e.g. in terms of life history – e.g. within-season management for squid vs. updating a TAC every 2-3 years for a longer-lived species, and in terms of perceived stock status and risk).
	3.7 Ensure management enhances stewardship (recreational, charter, commercial)	This is desirable, but requires a balance of "stick and carrot" incentives given the current context. Who are the stewards? GBRMPA? Or is this a more social question – i.e. from the perspective of outsiders looking in? And how do we measure this? Does this boil down to "has the reason for the journey been clarified, and is this compelling?" and adjusting the position on the stick/carrot spectrum given the strength of the response?
	3.8 Develop management strategies that move the fishery towards third party	We feel that this is subsumed by Qld Policy and legislative requirements that by design should ensure that harvest strategies are developed consistent with the principles underpinning third party accreditation. (Ultimately we would hope to remove this objective – certification is not the responsibility of managers, and the criteria used to determine whether a fishery is certifiable are metrics that we already should be embracing with existing objectives.

Overarching objective	Specific objectives		Description
	certification/ accreditation		Managers have responsibility to the resource, not to develop 3rd party accreditation. This is more a by-product/consequence/fringe benefit of good management). Also, this objective indirectly speaks to maximising commercial economic benefits, because accreditation should result in a better price. It is also embraced by social perceptions (i.e. it is subsumed by other primary objectives).
4. Maximise social outcomes	4.1 Ensure equity between recreational, charter and commercial fishing	4.1.2 Minimise conflicts between sectors	Is this more that management should be acting to minimise conflict? While this speaks to 4.1.1. also (because a sense of equity reduces conflict), it could be a management performance to reduce levels of conflict, through transparency, standardised processes, open communication channels, and bottom up engagement.
		4.1.3 Increase the level of respect for customary fishing (heritage values)	This is about (e.g.) “I grew up in a fishing family, and I would like to continue to fish in my local region” or “commercial fishing is inherent in the local community – this is a fishing town.” There are also lifestyle considerations “I just want to go fishing, and this is where I go fishing”. This is about (e.g.) “I’ve always gone fishing here and these are my favourite spots, and it’s not just about the fishing, it’s about the trip and the location”.
		4.1.3.1 Commercial	

Overarching objective	Specific objectives		Description
		4.1.3.3. Indigenous (cultural)	This is about respecting indigenous rights, customary practices and determinations (e.g. under Native Title). The declaration of areas as having indigenous rights goes beyond a HS, with overarching obligations per Policy and legislation.
	4.3 Enhance the net social value to the local community from use of the resource	4.3.2. Increase sense of place (community value of having a fishery):	This is about the community acknowledging the fishery's value and contribution to the community, such that participants feel valued and recognised contributors to the local and broader community. For example, Cairns is associated with Coral Trout due to local promotion. A local hotelier finding a draw in local seafood equates to acknowledgement of the fishery's importance.
		4.3.2.1 Large-scale commercial	Is this really more about spatial distribution than "sense of place"? This is also very regionally focused.
		4.3.2.2 Small-scale commercial 4.3.2.3 Recreational 4.3.2.4 Charter 4.3.2.5 Indigenous	This is very complicated, and so many variables affect it, and it's very difficult to know what's good or bad given the complexity. Additionally, only a subset of this is under the direct control of a HS.

Process of trading-off and reconciling objectives

Literature Review: Conceptual overview

We here focus on the literature and practice of trading-off and reconciling objectives in a TBL context. Pascoe et al. (2017) list the four key challenges for modelling multiple objectives in fisheries as “(i) the importance of relative weights of objectives; (ii) pros and cons in the tools available for conveying multiple objectives; (iii) challenges in the definition of the objectives, especially those of a social nature; and (iv) the need for stakeholder buy-in in the process”.

Long et al. (2017) suggest that success in implementation of ecosystem-based management may depend on reconciling differing priorities among its underlying principles as well as combining knowledge and expertise from fishermen with research and institutional sources. They use a comparative methodology to achieve this. The same philosophy applies when attempting to assimilate and reconcile a suite of TBL fishery objectives elicited across all stakeholders.

Trade-offs between conflicting objectives can be achieved more readily if the objective preferences are explicit in a participatory process to “develop consensus, as different stakeholders can evaluate their own proposals from the other’s perspective” (Pascoe et al., 2013d). For each of the stakeholder groups there are considerations of overall net benefit (or cost).

From the National Harvest Strategy Guidelines (Sloan et al. 2016):

- When developing the conceptual management objectives, the trade-offs between the ecological, economic and social outcomes being sought must be surfaced and agreed upon (preferably in consultation with all key stakeholders) and any contradictions resolved so that they are simultaneously achievable, i.e. there should be no unreconciled conflicts between them (Cochrane, 2000; Cochrane, 2002).
- Where there are multiple user groups, the impacts these objectives will have on the outcomes that each user group aspires to achieve should be considered at the beginning of the harvest strategy design process.

In its most general sense, reconciling objectives is not about resource sharing, or allocation, or inter-sectoral conflict *per se*. Rather, it is about acknowledging that, even given good relations between sectors and an equitable division of fisher rights, objective weightings (priorities) will naturally differ by user and stakeholder groups. It also needs be acknowledged that certain objectives (in our case, that Fisheries Queensland mandates a target reference point of 60% of the unexploited biomass (Department of Agriculture and Fisheries 2017) may be hardwired via legislation or policy, which constrains the process of objective reconciliation in the long run. The path to achieve this, however, is variable, and trade-offs between economic and social objectives in the short term may still influence how the target is to be achieved over time. Ideally, the aim is to achieve the optimal compromise (equitable distribution of fisher rights) among user groups given their preferences, within mandated constraints.

To enable objectives to be traded off and reconciled, common currencies are required. As such, a common approach is to give individual weights to the objectives so as to be able to model optimal trade-offs by prioritising some objectives over others. In this context, Jennings et al. (2016) considered climate change adaptation strategies and suggested these can help elucidate the "possibility of conflicts between groups when determining appropriate adaptation strategies, as alternative strategies will have different economic and social outcomes even if achieving comparable environmental outcomes".

To assist the trade-off and reconciliation process, fishery objectives should be classified as either “conceptual” (strategic) or “operational” (tactical) (Punt, 2015). In this context, conceptual objectives are generic, high-level policy goals. They are frequently expressed in broad terms and are typically too vague to be particularly useful as actual targets for a harvest strategy (Cochrane, 2002; Fletcher et al., 2002). However, without the conceptual objectives, there is no clarity on how the fishery should operate in terms of addressing ecological, economic and social performance outcomes, which can result in *ad-hoc* decisions and sub-optimal use of resources, which increases the probability of serious conflicts as different interest groups jostle for greater shares of the benefits (Cochrane, 2002).

Conversely, operational management objectives are very precise and are formulated in such a way that the extent to which they have been achieved during a specified period should be easily measured (Cochrane, 2002; Fletcher et al., 2002). An operational objective has a direct and practical interpretation in the context of a fishery and against which performance can be evaluated (Fletcher et al., 2002). Operational objectives should be easily measured and linked to the performance indicators, reference points and decision rules of a harvest strategy.

As such, to be included in harvest strategies and management strategy evaluation conceptual objectives need to be converted or translated into one or more operational objectives, expressed in terms of related, quantitative, performance measures.

The following example is taken from the South Australian Pipi Fishery to demonstrate the linkages between the three tiers of management objectives: “(1) overarching legislative objectives; (2) conceptual management objectives established for an individual fishery; and (3) ‘operational’ management objectives established for defined species (see Box 1)”.

Box 1: Example of the linkage between 'high-level' legislative objectives, 'conceptual' fishery management objectives and 'operational' management objectives for the South Australian Pipi Fishery

TIER 1-High level legislative objective (South Australian *Fisheries Management Act 2007*)

-To protect, manage, use and develop the aquatic resources of the State in a manner that is consistent with ecologically sustainable development

TIER 2-Conceptual fishery management objective (Lakes and Coorong Fishery Management Plan)

-Ensure the Lakes and Coorong Fishery resources are harvested within ecologically sustainable limits

TIER 3-Operational management objective for Pipi Fishery (Lakes Coorong Fishery Management Plan)

-Maintain a target Pipi relative biomass above 10 kg/ 4.5 m^2 and not less than 8 kg/ 4.5 m^2

-Ensure the Pipi relative biomass does not drop below 4 kg/ 4.5 m^2.

-Maximise Fishery Gross Margin

For countries or jurisdictions that have quite specific legislative objectives, such as in New Zealand where the legislative objective for all fisheries, as stated in the *Fisheries Act 1996*, is to manage fisheries in a way that will lead to production of the Maximum Sustainable Yield (MSY), the need to translate the legislation into operational objectives is more a technical exercise and may not require the step of developing 'conceptual' management objectives. For Australian Commonwealth fisheries, higher-level operational objectives (pertaining to target species sustainability and economics) are defined by adoption of the MEY target and the limit reference point whose default value is half the biomass corresponding to MSY. That stated, social and broader environmental and economic objectives should still be elicited and reconciled within the bounds of the legislated objectives.

Halpern et al. (2013) agree that target reference points for each objective should be articulated to the extent possible. However, if this is not possible *a priori*, it should not be cause for concern: this may instead be undertaken in a post-hoc manner, after stakeholders can see the output of management strategy evaluation (MSE) analysis, and adjust their weightings and targets in response to these. Usually, stakeholders want to see what they are trading off before they are able to weight (prioritise) the objectives.

Weighing TBL objectives for this project: preferences survey

Different harvest strategies are likely to have different impacts against the different objectives. To assess the overall suitability of the harvest strategy, the objectives need to be weighted so that the different strategies can be compared on an effective performance basis.

It is reiterated and acknowledged that some objectives are constrained by legislation: the harvest strategy policy for Queensland mandates a target reference point of 60% of target stock unexploited biomass (Department of Agriculture and Fisheries 2017). That this is a specified operational objective that cannot be breached will ultimately constrain other objectives, such that commercial and/or recreational benefits may be compromised. While we placed no restrictions on the extent to which stakeholders could weight each objective (e.g. that corresponding to the legislated target biomass could have been assigned a minimal preference), the legislated bounds immediately rule out certain harvest levels that may optimise short-term commercial fishing catch and revenue. In this context, stakeholders will consciously and unconsciously be protecting their interests, and this can be reflected as biases in their weighting of objectives.

A range of methods have been applied in the literature to assess objective weights, each with advantages and disadvantages (Doyle et al., 1997; Bottomley et al., 2000; Hayashi, 2000; Bottomley and Doyle, 2001; Roberts and Goodwin, 2002; Wang et al., 2009). Comparative studies of these methods suggest in some cases that the weights may vary considerably (Pöyhönen and Hämäläinen, 2001), although others have found higher correlations between the methods (Van Ittersum et al., 2007).

To this end, we apply modified versions of two commonly used approaches to determine potential weights of the general community and key stakeholder and management groups. These include simple scoring based approaches (Bottomley and Doyle, 2001) and the Analytic Hierarchy Process (AHP) (Saaty, 1980a) based on a series of pair-wise comparisons. Each method relies on a selected group of individuals (e.g. key stakeholders) to indicate a preference for each objective within a set of objectives. They differ in how these preferences are captured and analysed, both between and within the different approaches.

An online survey of fishery stakeholders was developed and implemented to elicit weights using the two approaches, and to assess how the methods used affected the overall objective weights.

Scoring approaches

Scoring based methods, or direct rating methods, generally involve allocating a score, for example 100, to the most preferred (first ranked) sub-component, then allocating a lower score somewhere between 1 and 100 for subsequent sub-components based on their relative importance. Direct rating methods have been applied in a number of coastal and resource management studies (Yang et al., 2011; Koschke et al., 2012; Liu et al., 2013; Pascoe et al., 2016b).

There have been several different approaches proposed for weight derivation from scoring based systems. These include direct rating approaches such as the max100 approach, where the highest ranked sub-component is allocated 100 points and subsequent (lower) sub-components allocated less than 100 points; and the min10 approach where the least preferred sub-component is allocated 10 points and the higher ranked sub-components allocated higher points relative to these (Bottomley and Doyle, 2001). Alternative approaches include direct point allocation where the set of all sub-components are allocated 100 points, and individuals share these 100 points out across all sub-components (so that they sum to 100) (Bottomley and Doyle, 2001).

Direct scoring out of 10 is a common approach applied in social science research (De Vaus, 2013). Unlike a Likert-type scale that is generally linked to a verbal statement (e.g. strongly disagree, …, strongly agree), the 10-point format places greater reliance on the respondent using a numerical response for which the precise meaning has not been defined. As a result, differences in interpretation make interpersonal comparisons difficult (e.g. does my "8" mean the same as your “8”). However, this disadvantage is balanced against the fact that many people are familiar with the notion of rating 'out of 10' (Dawes, 2008).

In all cases, the final weight is determined by:

$$w_{i,j} = S_{i,j} / \sum_{i} S_{i,j} \qquad (1)$$

where $S_{i,j}$ is the initial score given to each sub-component *i* (i.e. between 1 and 100 or 1 and 10) and $w_{i,j}$ is the final weight used in the analysis.

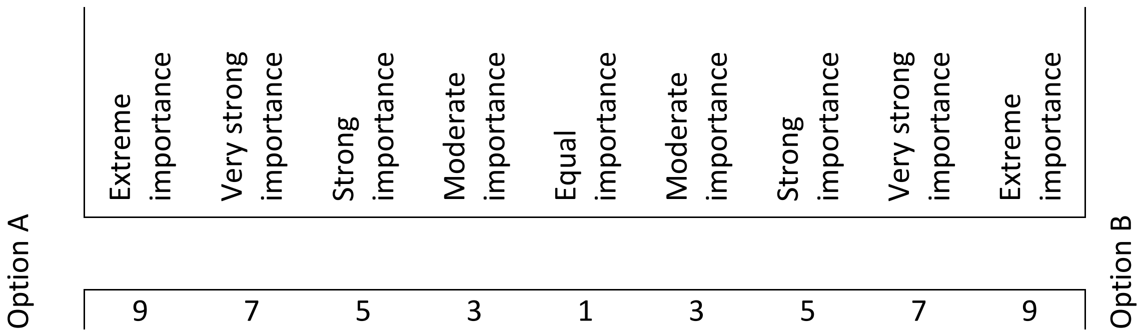
Several studies have suggested that the max100 approach is the most reliable in test-retest studies (Doyle et al., 1997; Bottomley et al., 2000; Bottomley and Doyle, 2001). However, respondents of a previous online survey indicated that they had little confidence in their response using this approach (Pascoe et al., 2014b). The normalisation process in the final weight calculation removes some of the problems associated with potential differences in perception by individuals in the case of the “out of 10” scaling.

Analytic Hierarchy Process (AHP)

AHP has been used in a number of marine and coastal applications to determine management sub-component importance and assist in decision making (Leung et al., 1998; Soma, 2003; Mardle et al., 2004; Wattage and Mardle, 2005; Nielsen and Mathiesen, 2006; Himes, 2007; Pascoe et al., 2009b; Pascoe et al., 2009c; Baby, 2013; Pascoe et al., 2013a), and is the most common approach used for preference elicitation in a wide range of applied natural resource case studies.

Traditional AHP is based upon the construction of a series of pair-wise comparison matrices which compare sub-components to one another (Figure 4), and a hierarchical structure that groups similar sub-components into subgroups, and builds the hierarchy with progressive layers of groupings. The pair-wise comparison method makes the process of assigning weights much easier for participants because only two sub-components are being compared at any one time rather than all sub-components having to be compared with each other simultaneously.

Figure 4. Example 9 point scale used in traditional AHP studies



Preferences are expressed on a nine point scale, with a 1 indicating equal preference, and a 9 indicating an extreme preference for one of the sub-components. Preferences are assumed symmetrical, such that if A against B has a preference of $a_{12} = 9$, then $a_{21} = 1/a_{12} = 1/9$. For each set of comparisons, a matrix of scores can be developed, given by:

$$A = \begin{bmatrix} 1 & a_{12} & \dots & a_{1n} \\ a_{21} & 1 & \dots & a_{2n} \\ \dots & \dots & \dots & \dots \\ a_{n1} & a_{n2} & \dots & 1 \end{bmatrix} \quad (2)$$

There are two general approaches used for determining the weights; the original eigenvalue method (EM) developed by Saaty (1980b)[5] and the Geometric Mean Method (GMM) developed by Crawford and Williams (1985). While the former approach has been employed in a wider range of coastal and resource management studies, the latter approach has been found to be less susceptible to influence from extreme preferences, as well as having better performance around other aspects of theoretical consistency (e.g. less susceptible to rank reversibility if the preference set changes, and greater transitivity properties) (Aguarón and Moreno-Jiménez, 2000).

The objectives weight (ω_i) using the GMM are determined by

$$\omega_i = \frac{\left(\prod_{k=1}^{n} a_{i,k}\right)^{1/n}}{\sum_k \left(\prod_{k=1}^{n} a_{i,k}\right)^{1/n}} \quad (3)$$

The analysis is undertaken within each level of aggregation in the hierarchy. The weights of the individual sub-components are determined by the product of their initial weight estimate (i.e. when compared with the other sub-components that they are grouped with) multiplied by the weight of the higher order aggregation (i.e. which is compared with other higher order aggregations) under the principle of hierarchic composition (Saaty, 1986). This reduces the number of direct comparisons that need to be made, as only sub-components at the same level and within the same broader sub-component need to be compared.

A challenge facing the use of traditional AHP is the propensity for respondents to be inconsistent in their responses. Preference weightings are highly subjective, and inconsistency is a common problem facing AHP, particularly when decision makers are confronted with many sets of comparisons (Bodin and Gass, 2003). Respondents do not necessarily cross check their responses, and even if they do, ensuring a perfectly consistent set of responses when many sub-components are compared is difficult. The discrete nature of the 1-9 scale can also contribute to inconsistency, as a perfectly consistent response may require a fractional preference score. Baby (2013) also suggests that inconsistency can arise through errors in entering judgments, lack of concentration and inappropriate use of extremes.

[5] Alternative approaches have also been used to derive the weights, with the row geometric method Crawford, G., and Williams, C. 1985. A note on the analysis of subjective judgment matrices. Journal of Mathematical Psychology, 29: 387-405. gaining increasing interest Aguarón, J., and Moreno-Jiménez, J. M. a. 2000. Local stability intervals in the analytic hierarchy process. European Journal of Operational Research, 125: 113-132..

In the case of perfect consistency, we would expect the ratio of the weights $A/B*B/C = A/C$. Such a condition may not hold due to several reasons, mostly relating to human error when undertaking multiple different bivariate comparisons. At an extreme level, this may result in a circular triad, where preferences for A>B>C>A.

The level of inconsistency in the survey responses is measured by the Geometric Consistency Index (GCI):

$$GCI = \frac{2}{(n-1)(n-2)} \sum_{i<j} \log^2 \left(a_{i,j} w_j / w_i \right) \tag{4}$$

where *n* is the number of attributes being compared within a level of the hierarchy (Aguarón and Moreno-Jiménez, 2003). This is compared to a randomly generated value for an n x n matrix (Random Indicator or RI) to derive a consistency ratio, CR, where CR=GCI/RI. Values of CR≤0.1 are generally considered acceptable (Aguarón and Moreno-Jiménez, 2003).

Development of the online survey and modified scoring approaches

In natural resource management in particular, management agencies are turning to online surveys to understand the priorities of a wide range of stakeholders to better support policy development and management decision making, with many of these surveys using AHP approaches (e.g. Whitmarsh and Wattage, 2006; Dichmont et al., 2012a; Dichmont et al., 2013b; Marre et al., 2016a; Pascoe et al., 2016b; Pascoe and Doshi, 2018). The use of online surveys to elicit preferences is not unique to natural resource management, with a range of other AHP studies implemented through online surveys (e.g. Benlian, 2011; Samvedi et al., 2013). A key advantage of the use of online surveys is that allows access to relevant stakeholders who may be geographically dispersed, even if not large in absolute numbers. For example, Thadsin et al. (2012) employed an online AHP survey to assess satisfaction with the working environment within a large real estate firm with offices spread across the UK.

The lack of direct interaction with the respondents creates additional challenges for deriving priorities through approaches such as AHP. Direct interactions with the individual respondents is not generally feasible, and in many cases responses are anonymous. At the same time, high levels of inconsistency are relatively common in online surveys. For example, Hummel et al. (2013) found only 26% of respondents satisfied a relaxed threshold consistency ratio of 0.3 (compared to the standard threshold of 0.1) in their online survey; Sara et al. (2015) found 67% of respondents satisfied a relaxed threshold consistency ratio of 0.2; Marre et al. (2016a) found 64% of the general public and 72% of resource managers provided consistent responses, while Tozer and Stokes (2002) found only 25% or respondents satisfied the standard threshold consistency ratio. Most previous online-based AHP studies have tended to exclude responses that have a high level of inconsistency, resulting in a substantially reduced, and potentially unrepresentative, sample (Tozer and Stokes, 2002; Hummel et al., 2013; Sara et al., 2015; Marre et al., 2016a).

In this study, we avoid some of these pitfalls by modifying the way in which the data are collected and analysed, taking into account the symmetry assumption underlying AHP.

In the survey, respondents are presented with a nine-point importance scale against which they can assess the importance of each objective. A nine-point scale was selected (rather than an "out of 10") as it allows five categories to be defined with mid-points between them. An example of one of the questions is presented in Figure 5. As with the traditional scoring approach, respondents can indicate which response best approximates their belief around the importance of each objective. These can be converted to a score between 1 and 9, and the individual objective weight derived as in Equation 1.

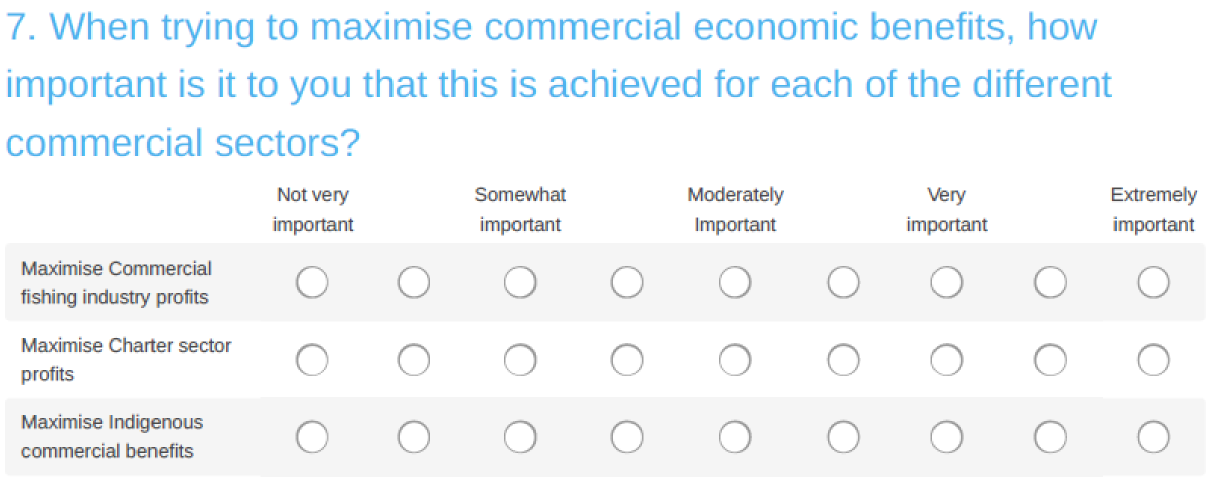


Figure 5. Example of question in the objective importance survey

The results can also be analysed using a modified version of the AHP GMM. An equivalent comparison matrix (Equation 2) can be derived from the objectives in Figure 5 by deriving each element based on the differences between the stated objective importance (expressed on a 1-9 scale):

$$a_{i,j} = \begin{cases} a_i - a_j + 1 & if\ a_i - a_j > 0 \\ \dfrac{-1}{a_i - a_j - 1} & if\ a_i - a_j \le 0 \end{cases} \quad \text{and} \quad a_{j,i} = \frac{1}{a_{i,j}} \qquad (5)$$

Given these, the GMM weights can be derived using Equation 3.

An advantage of this approach over the traditional pairwise comparison is that the respondent is able to compare all objectives at the same time, avoiding issues around inconsistency (which does not need to be calculated as a result). That is, the respondent will immediately be able to see how each of the options compares to the full set when making their response.

A further key advantage of the approach is that the issues around interpersonal comparisons are reduced compared with the scoring method. As the differences between the preference scores are used rather than their absolute levels, a score of [5,6,7] from one respondent will produce the same weighting of each objective as a score of [7,8,9] using the modified AHP method, but different weights using the scoring method.

The online survey

The online survey was developed in *SurveyMonkey* (www.surveymonkey.com) and consisted of three sections. The first section asked about the level of experience in the fishery and which stakeholder groups the respondent identifies with. The second section of the survey required respondents to indicate the level of importance they attached to each of the different alternative objectives presented. Figure 5 above provides an example of one such question. Further details on each of the objectives was also provided before the response was required. The final section asked respondents to leave any additional comments if they wished, and also asked respondents if they would be prepared to participate in a follow-on survey (in which case they were asked to provide a contact email address). A copy of the full objectives survey is available through the Supplementary Material associated with Pascoe et al. (2019).

The survey was distributed to stakeholders in the fishery with the assistance of Queensland Fisheries and the members of the Working Group. As with any online survey, the potential for non-response bias exists. Further, as the survey was distributed also by members of the Working Group (e.g. to processors and wholesalers), there is the additional risk that that the selected recipients may not be representative of the group as a whole, or that the recipients may have colluded in their response with the Working Group members. As the survey was focused on the objective weightings (not the potential management options themselves), and substantial variation in these between individuals in all groups was observed, the potential for such bias was not considered problematic.

The survey was approved by CSIRO's Social Science Human Research Ethics Committee (Project 113/17) in accordance with the Australian National Statement on Ethical Conduct in Human Research.

Process of engagement to identify alternative harvest strategy options

A range of potential harvest strategies were developed in collaboration with the CRFFF Working Group through a series of workshops. Initially, information was fed back to the Working Group about the outcomes of the objectives surveys to help identify key areas that required further consideration. At the July 2018 CRFFF WG meeting, the project team used a "strawman" technique to develop harvest strategy options for the CRFFF. These were subsequently discussed by the project team and DAF who finalised a shortlist of options, based on what was practically feasible and had a scientifically defensible rationale.

The Working Group first established a "modified status quo" option (the "baseline") based around Coral Trout, which was considered the minimum amount of change required to meet the ecological sustainability objectives of the new Queensland Sustainable Fisheries Strategy. Other "Modified status quo" options were considered for the Red Throat Emperor and "other species" quota categories. These allowed the group to critically confront the positive and negative attributes of the current management arrangements.

Working Group members were then encouraged to consider alternative "blue sky" options by "thinking outside the box" of the current management arrangements to raise or suggest alternative harvest strategy options. This was done to provoke thought, allow issues to be identified, and bring new ideas that may be worthy of consideration. These other alternatives were postulated as having the potential to enhance at least one or more of the broader objectives of the fishery.

The key proposed modification underlying the “modified status quo” harvest strategy was the adoption of a target reference point for the stocks of 60% of the unexploited biomass, which is consistent with the Queensland Sustainable Fisheries Strategy (Department of Agriculture and Fisheries, 2017).

The target of 60% unexploited biomass recognises TBL considerations (and aligns with some maximum economic yield estimates to maximise economic benefits, as this applies to the total catch across all sectors, so likely not economically optimal for all). Queensland’s Strategy approach to setting sustainable catch limits is not just about economics or the minimum necessary for sustainability. It recognises that a higher stock level is more resilient, and more likely to recover from adverse environmental conditions. Many of Queensland’s fisheries are multi-sector and multi-species fisheries and so in addition to commercial profitability and stock resilience, this is also about the quality of fishing and managing the risks around environmental limitations. Broader environmental influences are already affecting some stocks, and managers are trying to also prepare for likely further changes into the future. This is particularly important given the cumulative pressures on marine ecosystems (including the very poor long-term outlook for the Great Barrier Reef ecosystem).

Harvest control rules were developed to adjust the fishery-level TAC each year, where these changes are then applied proportionally to both the commercial TAC (TACC) and the recreational trip limits to maintain relative equity between the sectors. For Coral Trout species, the TAC is based on a stock assessment every 5 years and a suite of other indicators in the intermediate years. For Red Throat Emperor, the changes are proposed to be based on a risk assessment undertaken at least every 5 years, with empirical indicators used to adjust the quota and bag limits in the intermediate years. Similarly, for the other species (OS) component, the combined TAC would be retained, with both commercial and recreational catches adjusted proportionally in response to changes in the level of catch and catch composition. Within this cap, species considered “at risk” would potentially be subject to separate commercial caps and recreational trip limits.

Alternative harvest strategies identified by the Working Group were postulated, subject to subsequent formal evaluation, to further enhance one or more ecological, economic or social objectives.

Determining a methodology for TBL evaluation: a review and summary of the process

Literature Review: Defining and addressing technical issues and conceptual platforms for TBL harvest strategies

Overview

The Australian experience has identified four key challenges for modelling multiple objectives in fisheries: (i) the importance of relative weights of objectives; (ii) pros and cons in the tools available for conveying multiple objectives; (iii) challenges in the definition of the objectives, especially those of a social nature; and (iv) the need for stakeholder buy-in in the process (Pascoe et al. 2017). Against (ii), there are a number of different ways that multiple objectives can be modelled and the results presented, each with different advantages and disadvantages.

Incorporating social and economic relationships, together with ecological sustainability objectives into models to provide management advice creates a number of challenges, particularly when this advice requires complex trade-offs between objectives. This is further complicated by differences in quality and quantity of data across fisheries, and difficulties in quantifying some measures, particularly around social objectives and outcomes (Pascoe et al. (2017). The approach chosen is often based on data availability and the complexity of the issues, including an evaluation of how best to present the results to the different stakeholder groups.

The two main qualitative approaches are:

- Multi-criteria decision analysis techniques
- Qualitative models.

The three main quantitative approaches (from less complex to more complex) are:

- Commensurable units e.g. socio-bio-economic optimisation model
- Non-commensurable units with explicit objective weights e.g. goal programming bioeconomic model
- Non-commensurable unit without explicit objective weights which provides separate outcomes under each objective (e.g. hybrid models, simulation approaches) and viability analysis approaches.

These are reviewed in detail in the following sections.

Review of theoretical frameworks of methods/approaches to evaluate TBL objectives

Various tools exist, bracketing approaches based in reality and those considering optimal states. Table 14 provides a summary of the available approaches for evaluating TBL harvest strategies, together with an overview of the advantages and disadvantages of each. This may assist practitioners to determine which approach may best be suited to their circumstances.

We structure the following as per the qualitative to quantitative categories outlined by Pascoe et al. (2017).

A basic harvest strategy embedded in TBL considerations

The types of assessment that fit into this category are the more traditional stock assessments and associated management levers (such as an adjustment to a total allowable catch). They can also include data-limited stock assessments, such as those reviewed in

Dowling et al. (2016)). In this approach, assessments are embedded within a simulation-based MSE that is tuned to achieve optimal TBL performance. Indeed, most of Australian Commonwealth harvest strategies have been developed in this way. This approach acknowledges that, particularly against social and economic objectives, there is likely to be data limitation. It also acknowledges the need for pragmatism in terms of the available capacity, and nature of the fisheries.

There is a limited extent to which all objectives would be explicitly acknowledged and an overall optimum achieved. The development of such a harvest strategy may be easier and less confronting as the long time scales are generally ignored with the logic that if each year sustainability is achieved then the long-term social and economic objectives would likely be met in any case. However, we believe that most social scientists would see this as a failure to meet TBL theoretical requirements of explicitly reporting against all parts of the TBL. Conceptually this is analogous to the social objectives being the "outer onion layers".

In the absence of sufficient resources to complete a full TBL approach this "shallower" approach to TBL could still be useful in data-poor fisheries, and in those which there are other reasons (such as political sensitivities, or a desire to allow market forces to operate) that a more "comfortable" arm's-length approach may be preferred. If this is considered as an easier starting point that will be further developed in the future it may make even more sense. In this approach the social objectives would be process-driven rather than outcome-driven, so that explicit indicators would be needed only for the environmental and economic objectives. The process of stakeholder engagement and objective elicitation to some extent acknowledges social aspects and at least should seek to do so in a way that achieves some level of social equity.

The clear advantage of this approach is the ability to select only objectives and associated indicators that allow a commensurate approach. Another advantage is that complexity is minimised.

Using this approach, Dichmont et al. (2014) took an iterative process of expert and community consultation (a "Delphic approach") to develop and MSE tested a "strawman" harvest strategy that acknowledged the TBL, while achieving, for the first time, unanimous agreement across all stakeholder groups. However, a clear risk associated with the simplified approach is that only some non-market values are appropriate for social benefits and the absence of others could mean the optimisation is skewed.

Qualitative and semi-qualitative approaches

Multi-criteria decision analysis techniques

Multi-criteria decision analysis (MCDA) has been widely applied to support environmental and natural resource decision making. For example, Huang et al. (2011) identified over 300 papers published between 2000 and 2009 that used MCDA approaches to assess environmental management alternatives, while Cegan et al. (2017) identified over 3000 MCDA papers published between 2000 and 2015 relating to environmental and terrestrially based natural resource management (i.e. not including fisheries studies).

MCDA has also been widely applied in fisheries cases, with several review articles identifying a wide range of applications in fisheries (e.g. Mardle and Pascoe, 1999; Leung, 2006; Kjaersgaard, 2007; Andalecio, 2011) and aquaculture (Vergara-Solana et al., 2019) over recent decades. More recent applications (not captured in the above reviews) include assessments of constraints to small-scale fisheries development (Kimani et al., 2020) and fisheries management in Italy (De Boni et al., 2018) and Scotland (Nielsen et al., 2019).

A wide range of multi-criteria methods exist. However, all involve a common approach, namely the identification of objectives (the criteria), the weighting of the importance of these objectives, and a measure of the impact. The first two stages require stakeholder involvement to identify and weight the objectives. The latter stage can involve either expert judgement or more quantitative approaches to assess impacts. A study comparing expert opinion in MCDA studies to more empirically (and objectively) derived impacts found that expert opinion provided robust assessment of impacts, provided the “experts” had a high level of knowledge about the system being assessed and the related management issues (Pashaei Kamali et al., 2017). .

In Australia, MCDA approaches have been used to assess and weigh fisheries management objectives at the Commonwealth (Pascoe et al., 2009c; Jennings et al., 2016a) and State level (Pascoe et al., 2013b; Jennings et al., 2016a). Similarly, all three stages of the MCDA approach have been applied in Australia to assess fisheries management options. For example, Pascoe et al. (2009a) used expert opinion to assess spatial management options against a range of management objectives, which were combined with objective weights to determine the most appropriate options. Similarly, in Queensland, a stakeholder elicitation process was used to develop social, governance, economic and ecological objectives in the Queensland Trawl Fishery, and then weight the relative importance of these objectives (Dichmont et al., 2012b). An expert group was used to develop different governance strawmen (or management strategies) and 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, but was not optimal in terms of every objective.

Other Qualitative approaches

Qualitative approaches can also take the form of qualitative risk assessments and other types of qualitative models, such as Bayesian Belief Networks (BBN). Ecosystem risk assessments (ERA) may be used to determine whether proposed management tools, such as marine parks, may achieve the desired objective. For example, Read and West (2010) assessed the effectiveness of managed-use zones in six multiple-use marine parks located within NSW using qualitative ERA.

BBN models have also been used to generate the social and economic parts of report cards in a multi-sector coastal management setting, e.g. Gladstone Healthy Harbours (Pascoe et al., 2016a). In Torres Strait Rock Lobster Fishery, van Putten et al. (2013) used a Bayesian Network model to assess how the islander sector might respond to different management strategies and allocations. Pascoe et al. (2020) developed a BBN to assess social, economic and environmental outcomes under individual transferable quota management.

The overall triple bottom line outcomes can also be presented as a series of multiple indicators that are evaluated using such techniques as "traffic light" approaches (Caddy, 2004; Caddy et al., 2005; Caddy, 2009), cumulative sum multiple indicator systems (Scandol, 2005), and multidimensional scaling analysis (RAPFISH) (Pitcher and Preikshot, 2001; Pitcher and Cheung, 2013).

Quantitative approaches

Quantitative approaches that may be taken to evaluate TBL HSs include approaches that scale to commensurable units, such as dollar terms in a cost benefit analysis (e.g. Freese et al., 1995), or utility terms in multi-attribute utility analysis (e.g. Healey, 1984)); scale to non-commensurable units but with explicit objective weights as in a goal programming bio-economic model (e.g. Charles, 1989; Pascoe and Mardle, 2001); scale to non-commensurable units without explicit objective weights, thereby providing separate outcomes under each objective as in hybrid models (e.g. Mapstone et al., 2008; Little et al., 2015); and, finally, co-viability analysis (Gourguet et al., 2013; Gourguet et al., 2016).

The three main quantitative approaches (from less complex to more complex) are reviewed in greater detail below.

4) Commensurable units (i.e. can be combined in single unit – e.g. biomass terms, dollar terms) e.g. socio-bio-economic optimisation model

<u>Simulations quantifying trade-offs between objectives (reality-based)</u>

Gaichas et al. (2017) used a length-structured multispecies, multi-fleet simulation model to explore alternative status determination criteria and reference points that could simplify fisheries management, and to illustrate trade-offs between objectives pertaining to yield, biomass, community composition and revenue.

Per Pascoe and Dichmont (2017), for the Torres Strait Rock Lobster Fishery, together with the Bayesian Network Model of van Putten (2013), a separate model described the fleet adjustment of the non-islander fleet under different quota allocations (Pascoe et al., 2013b). These in turn determine the level of available effort in the fishery in each fleet. The implications of these effort levels were assessed using a bioeconomic model, building on existing stock assessment models in the fishery, which provided information on the economic outcomes to both islander and non-islander fleets (Plaganyi et al., 2012).

McDonald et al. (2008) calculated multiple performance indicators (including water quality, high-value stock size, icon species, overall profit, and habitat cover) using a multiple-use MSE agent-based model.

Zimmerman and Yamazaki (2017) use a stylised bioeconomic model of a multi-stock fishery to study how different management objectives are affected by the nature of stock interactions and to identify potential trade-offs between multiple objectives in stock rebuilding. The type and strength of stock interactions were shown to directly determine the trade-offs between the biological and economic objectives of the fishery as well as the short-term and long-term objectives in stock rebuilding. Despite the multi-stock novelty of this model, in terms of the TBL it was limited to a biological-economic trade-off considerations.

Voss et al. (2014) introduce such a triple-bottom line approach to the management of multi-species fisheries using the Baltic Sea as a case study. A coupled ecological-economic optimization model was applied to address the actual fisheries management challenge of trading-off the recovery of collapsed cod stocks versus the health of ecologically important forage fish populations.

Modelling approaches calculating various reference points (MSY, MEY, MSocY, MSEY), and trying to optimise over each.

The Commonwealth Fisheries Management Act mandates the maximising net economic returns as the main objective by using MEY target and focussing on resource sustainability without explicit social objectives. An alternative could instead be aiming for "maximum social yield" defined by Tony Charles: an effort level that maximises the full set of objectives (social, economic and environmental) taking into account their relative weights (Charles, 1989).

The Northern Prawn Fishery bio-economic model, focusing on MEY-based management (Buckworth et al., 2015), included the tiger prawn fishery bioeconomic model that estimated the level of effort that should be applied to each of two fishing strategies that maximises the net present value of fishery profits over a 50 years period with a discount rate of 5%, with the condition that stocks of the key target species are at their equilibrium level (Smey) within 7 years.

That being the case, it is understandable that Rindorf et al. (2016) emphasised that management with MSY on a single-species basis does not ensure that TBL objectives are addressed. They expanded the concept of a "pretty good yield" range of fishing mortalities (assumed to provide 95% of the average yield for a single stock) to a pretty good multispecies yield (PGMY) space. In this way, "pretty good" multidimensional yield can accommodate situations where the yield from a fishery stock affects the ecosystem, economic and social benefits, or more broadly TBL sustainability. As PGMY provides a safe operating space for management that adheres to the principles of MSY, it allows the consideration of other aspects to be included in operational management advice.

For fishery managers and others, MSY-based PGMY ranges may provide a way to account for mixed fisheries, ecosystem issues and possibly economic considerations to allow policy makers to address 'choke' species issues, while providing scientific limits to policy choices (Rindorf et al., 2017). This approach can also provide a formal way to integrate annual fluctuations of all stocks and fleets in mixed fisheries. However, there are situations where simultaneous good yields of different stocks cannot be achieved or where ecological, economic, and social objectives are conflicting (Rindorf et al., 2017). Further, social objectives may not be directly related to fishing pressure and, therefore, a 'Pretty Good Social Yield' may not be ensured by defining specific combinations of fishing mortalities. Guillen et al. (2013) investigated the MSY and MEY estimation in multi-species and multi-fleet fisheries in comparison to single species assessments. Analyses were applied to the Bay of Biscay demersal fishery using a bio-economic model. The impact of exploiting at MSY and

MEY on the optimal effort allocation between fleets with different exploitation patterns and economic structures was analysed.

Using the risk-cost-catch approach to quantitatively evaluate trade-offs

Fulton et al. (2016) explain risk equivalency in the context of the Australian Commonwealth Harvest Policy (HSP)'s requirement of "ensur(ing) that the stock stays above the limit biomass level at least 90% of the time"[6]. This concept is at the core of trading off based on the risk-cost-catch approach. Where risk is higher due to increased uncertainty associated with the available data or stock assessment, managers use precautionary meta-rules and buffers to ensure that the undesired state of the fishery stock is prevented and/or actively averted.

In fisheries, these risk management approaches may be accompanied by spatial management regimes that include spatial and temporal closures and or restriction of practices. Dichmont et al. (2013a) analysed a series of spatial closures to investigate trade-offs between biodiversity, ecosystem function, benthic impacts, at-risk species plus economic and sustainability objectives. They found that in actively managed fisheries MSE could string together the required and useful management tools to satisfy the TBL objectives.

However, Dichmont et al. (2015) in their comparison of four case studies warn that, due to the substantial differences in data requirements within tiered harvest strategies, risk needs to be assessed within each tier without creating a composite risk that would alter the relevant management action set. MSE is valuable in this case and can be used to select which values are suitable and can also guide buffer setting and/or periodic review.

5) Non-commensurable units with explicit objective weights e.g. goal programming bioeconomic model

These include multi-objective modelling approaches that places explicit weightings on objectives, and consider trade-offs for various objectives each expressed in different units (profits in dollars, social in terms of numbers of jobs, fish stocks in kg of biomass), but that are all standardised to a common unit (e.g. from 0 and 1).

Goal programming bioeconomic models compare model outputs to pre-defined goals that policy makers wish to achieve (e.g. the maximum potential profits; the maximum observed level of employment; half the level of discards etc.). Deviations from the goals – either positive or negative – are expressed as a percentage of the goal, converting the non-commensurable units into commensurable measures (i.e. percentage positive of negative deviation). Solved as an optimisation model, goal programming models estimate the combination of catch and effort that minimises the sum of the undesirable deviations, weighted by their importance. In doing so, the model estimates the pareto optimal set of outcomes that best satisfices the set of management objectives given the prevailing economic, social and environmental conditions. At this point, no single outcome can be

---

[6] DAFF, 2007. Commonwealth Fisheries Harvest Strategy Policy Guidelines. Australian Government Department of Agriculture, Fisheries and Forestry, Canberra, Australia, pp. 55 http://www.agriculture.gov.au/fisheries/domestic/harvest strategy policy.

improved without making another outcome worse off, resulting an overall reduction in total social welfare.

Examples of applications of goal programming approaches applied to multi-objective fisheries management are limited, although several studies have been undertaken for European fisheries (e.g. Mardle et al., 2000; Pascoe and Mardle, 2001; Kjærsgaard et al., 2007). The approach is more broadly used in other natural resource management decision making (e.g. Ren et al., 2016; Colapinto et al., 2017; Gupta et al., 2018; Gosling et al., 2020).

6) Non-commensurable unit without explicit objective weights

Viability Analysis

Viability analysis identifies objectives and goals and seeks solutions within feasible bounds. That is, it avoids explicit trade-offs between objectives, but rather shows, given constraints, the likelihood of staying within these.

The viability analysis approach recently developed in the Northern Prawn Fishery does not aim to identify an "optimal" outcome and hence does not require objective weightings, but instead aims to ensure at least a minimal acceptable levels for each of the objectives (Gourguet et al., 2016; Pascoe et al., 2017). This itself raises additional issues: for example, while any stock levels above a limit reference point may be considered "acceptable" to some degree, it is far from desirable (and is counter to the current Commonwealth Harvest Strategy Policy). Quantifying acceptable levels of social and economic objectives is also highly subjective. Studies based around the limits of acceptable change framework have found that perceptions of these limits varies substantially between individuals and stakeholder groups and references therein (Ahn et al., 2002; Roman et al., 2007; Pascoe et al., 2017), resulting in similar issues as those with determining appropriate objective weights (e.g. which set of minimal acceptable levels to use). Further, once a set of viable options have been identified, identifying which option to implement still requires some implicit weight for each of the objectives.

Péreau et al. (2012) added the social objective which seeks to achieve the maximum number of active fishers in their bioeconomic model for an ITQ agent-based model. They used the viability kernel concept to characterise the sustainability of the system. In this case, the kernel is the feasibility set of initial stock sizes for which an acceptable regime of quotas exists and satisfies in time the ecological sustainability constraints while also achieving cost efficiency with economic constraints such that it "emphasizes that the viability of TAC management strategies in an ITQ system where the social constraint applies depends on the current status of the stock as compared to the minimum stock threshold x_{lim} and on the effects of fishing on the stock as compared to the limit mortality rate F_{lim}" (Péreau et al., 2012). The resulting stock extraction was considerably lower than under MSY or MEY for all scenarios tested.

Frontier analysis (outcomes when behaviour is optimal relative to different objectives/targets)

Frontier analysis shows where behaviour is optimal relative to difference objectives or targets. Per Halpern et al. (2013), the solutions that lie along the "frontier" are triple-bottom-line solutions, where one can optimise conservation goals and equity while minimizing costs. Solutions interior to these frontier solutions are all possible. As in other trade-off assessments, finding the frontier does not then prescribe a single correct solution but instead presents the range of options, all optimal, that represent the trade-off between stated goals.

Griffin and Woodward (2011) simulated a wide range of recreational management strategies for their impacts on red snapper yield, economic surplus and the fish stock. Data Envelopment Analysis (DEA) inspired policy efficiency frontiers that lead to finding those strategies that offered the greatest level of economic surplus for any biological target.

Weninger (2001) describe fishery management implications of an efficient production frontier (EPF), where a directional technology distance function model of the harvesting technology is used to measure changes in the EPF over time. Frontier shifts are summarised with input- and output-based frontier indicators that are interpreted as measures of bioeconomic productivity change. A value function approach is outlined by Mangel and Dowling (2016) takes the stock status estimate from an assessment, and optimises, over the range of possible catch levels, a value function for a given set of stakeholder group weightings.

Constraints and criteria mapping

Constraints mapping is a stakeholder engagement tool that can be used to determine areas of non-negotiability and areas that may provide trade-offs (Bond et al., 2011). By understanding early in the process where the ‘no go’ areas are, the options left for possible selection become more visible and, indeed, obvious to all stakeholders. Criteria mapping is a resource-intensive process that uses actual spatial maps with layers of uses and users, to evaluate how well a proposed strategy meets objectives. Constraints map displays a set of feasible alternatives that fits some kind of objective profile (Malczewski and Rinner, 2015).

Operational approaches for this project

As reviewed above, there are many tools applied within the environmental and other research domains that can be applied in fisheries. In an operational context, Benson and Stephenson (2018) 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 management strategy evaluation (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 stakeholders’ weightings (preferences) and provide no formal means of determining the optimal solution given these weightings. Even MSEs that do aim to include more than just sustainability objectives (e.g. Plagányi et al., 2012a), have no means to formally make recommendations that reconcile different stakeholder groups.

The importance of stakeholder preferences was illustrated by Pascoe et al. (2013) assessment of relative importance of the different objectives to various stakeholder groups in the Queensland East Coast Otter Trawl Fishery, Australia, using the Analytic Hierarchy Process (AHP). Across stakeholder interest groups, preference weightings showed a 4-fold difference in economic outcomes, 2-fold in social outcomes, and almost 2-fold in environmental outcomes. This motivates the need to reconcile weightings (priorities), and therefore, TBL HSs, across interest groups.

Despite the extensive availability of tools, the process of operationalising TBL HSs, beyond a simple conceptualisation, 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, e.g., catch or effort.

As outlined in the Introduction, for this project, we take two alternative approaches:

- a semi-quantitative multi-criteria decision analysis (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 TACs for the three main species groups.

As such, TBL objectives are included either in the MCDA trade-off analyses of the different HSs using stakeholder input, or directly in an optimisation model.

The simulation is a more defensible approach from a research perspective, but it is resource-intensive and data-hungry. The MCDA approach, on the other hand, acknowledges pragmatic constraints, both financial- and capacity-related, and with respect the Queensland's current state of progress with respect to harvest strategy development. The MCDA approach is also appealing given that the vast bulk of Qld stocks do not have model-based assessments.

It must be noted that there are boundaries around developing a TBL HS placed upon the project by the jurisdiction, fishery and scope chosen. Legislated & policy boundaries around TBL HSs – some things are hard wired and have to be accounted for when setting out on determining a TBL HS – so people's expectations must be tempered.
Queensland's B60 target is such a boundary (optimisation of the objectives may enabled other choices if it was set at say B40) as was the project's choice of which stakeholders to engage with (mainly harvesters, managers, scientists & eNGOs, but no wholesalers, retailers or consumers). Involvement of these latter groups may have changed objective weightings). Acknowledgement of these issues makes clearer why perhaps weighting and optimisation ended up where it did.

Queensland setting B60 as a target is such an example as it immediately rules out harvest levels that may optimise commercial fishing catch and revenue (profit is another matter). All stakeholders will consciously and unconsciously be keeping an eye on their interests which, as the authors state, can be reflected as biases in their weighting of objectives. For commercial fishers this is usually catch, for charter and recreational strike rate and/or fish size, for indigenous ease of access for community use and for eNGOs minimising TEP mortality and broader environmental impacts. The MCDA approach will be particularly prone to such biases and can lead to perverse outcomes overall as a result, e.g. far from optimal for the Queensland community at large. In this sense independent leadership of the MCDA process is critical to both point out and reduce bias.

A MCDA approach to determining and evaluating TBL HS

Performance of each harvest strategy against the objectives

The assessment of the performance against each of the objectives followed a similar approach as that outlined in Pascoe et al. (2009a) and Dichmont et al. (2013b). An online survey was developed in *SurveyMonkey* and administered to those individuals from the first survey who agreed to participate in a follow-on survey.

The respondents were asked to rate each potential harvest strategy *relative to the baseline* against each objective (Figure 6) on a scale ranging from "Much worse than the baseline" to "Much better than the baseline". An example of one such comparison applied in the survey is given in Figure 6.

The resultant choices were converted to a 7-point scale, ranging from –3 ("Much worse than the baseline") to +3 ("Much better than the baseline"), with "About the same as the baseline" having a value of 1. The final output of this process is an impact matrix $I^{s}_{i,j}$ where *s* is strategy, *i* is the number of objectives and *j* is the total number of respondents of the second survey.

The relative weights for each respondent for each objective derived from the objective survey were combined into a single relative weight matrix, $W^{t}_{i,r}$ by stakeholder group, *t*, where *r* is the number of respondents to the objective survey and *i* is the number of objectives considered. The overall results were derived by the product of these two matrices, $W'I$ for each stakeholder group and harvest strategy, producing (*r*j*i*) observations. Summing this over all objectives (*i*) provides a score representing how well the harvest strategy performs against the baseline given the objective preference of each respondent *r* and the expectations about how the strategy performs against these objectives from each respondent *j*. A positive score indicates an overall positive contribution relative to the present system and a negative score indicates an overall negative result.

From this, we can derive a probability distribution of the expected benefits of each harvest strategy that takes into account heterogeneity in both the impact scores and also the objective preference weightings.

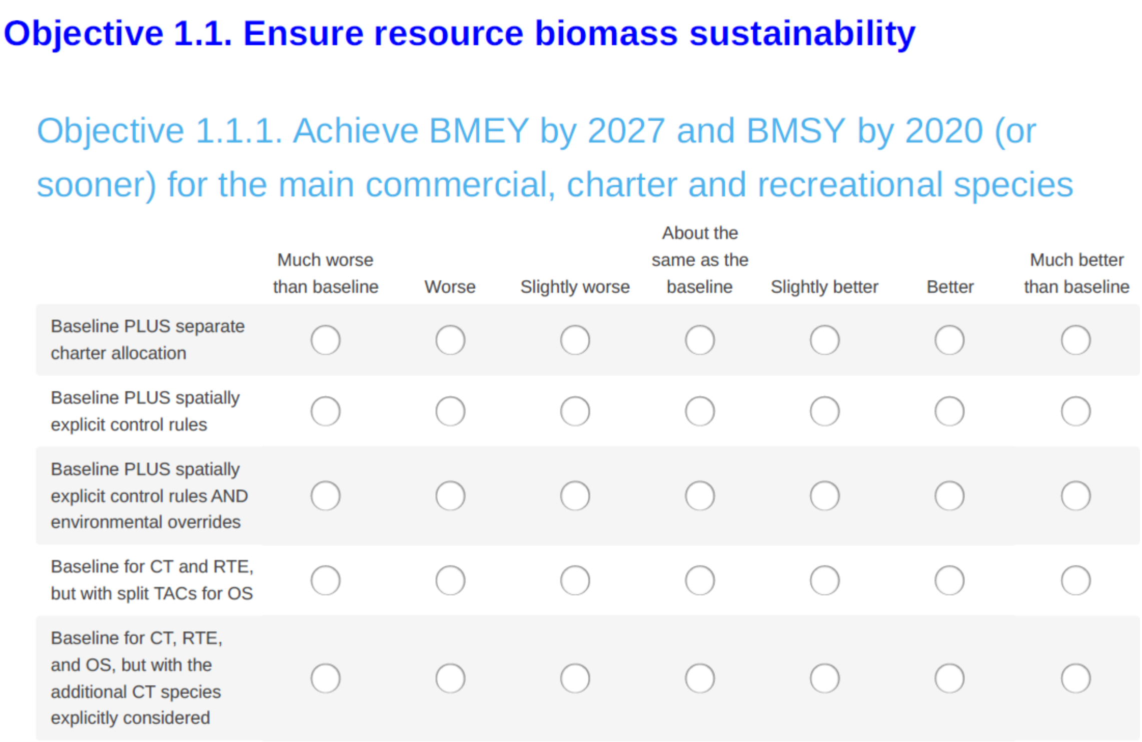


Figure 6. Example of question in the harvest strategy performance survey

A quantitative simulation model approach to evaluating TBL harvest strategies

Background and pilot study

In operationalising TBL harvest strategies, the question remains as to how to optimise a TBL value function, given a set of weightings, across a range of scenarios, across 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 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. In Appendix C, we generate a Pareto frontier over which a given strategy can be optimised for any combination of weightings, against which trade-offs can be assessed.

Our approach is consistent with the "efficiency frontier" presented by Halpern et al. (2013) whereon optimal solutions lie, and represent different importance (weight) given to conservation versus equity goals. As opposed to the approach of Rindorf et al. (2017) that takes a suite of fishing mortalities corresponding to sustainable yield, and progressively refines this, we consider the TBL objective weighting profile for given stakeholder groups as an integrated value function that is optimised across a suite of catch levels. While Pareto frontiers have seen extensive applications in other contexts, for details see Enríquez-Andrade and Vaca-Rodríguez (2004), these have not been leveraged to the maximum extent in a fisheries management context.

While our Pareto frontier can find an optimal strategy for a given set of preferences, it cannot, however, directly reconcile among different stakeholders with different sets of weightings. The question then becomes how to make sense of, and seek, an overall optimum solution among differing sets of stakeholder preferences.

This approach provides a rational formal means to reconcile the stakeholder preferences. That is, we illustrate a formal way in which to trade off the values across the various sets of weightings, where these show a lack of agreement among stakeholders. This can alternatively be seen as a demonstration of a rational approach to "mutually disagreeing".

The initial background theoretical study to test the idea of applying a TBL-type value function to stakeholder groups using weightings loosely based on the East Coast Trawl Fishery in Queensland, is presented in Appendix C. We used empirically estimated value functions for each of the three TBL components and weightings from each from 5 interest groups (including fishery managers). We chose a simple model with a limited number of components to each of the TBL values and based our value function around a simple biological model, so that the concepts are explicit and to facilitate the clarity of the ideas.

Even if everyone agrees on weights (preferences), our Pareto frontier provides a more elegant way of optimising over multiple strategies, and multiple indicators (e.g. Mapstone et al. (2008)) comprising a value function. But where there are a range of different preferences (e.g. Pascoe et al. (2014a)), we apply a single value function and determine the cost to each group of what is being lost, if their optimal strategy is not adopted. This can be considered analogous to game theory, or Nash equilibrium – the best overall compromise will see everyone sacrificing a little bit, but by moving too far away from this optimal point of compromise, someone will do worse.

Simulation model approach

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). This is a quantitative, non-commensurable-unit approach, via a multi-indicator objective function. 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 Appendix D (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 Appendix D, 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 (Appendix Table E1). 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 Appendix D, Figures D1.8.1-15.

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

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

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).

We assumed that the optimised TACs were fully realised, with no over- or under-catch.

The full model and performance indicator specifications may be found in Appendix D.

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 CT, 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 Table 4, 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 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.

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.

Table 4. Summary of model and biological input parameters

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
Selectivity-at-age (Cont'd)	SelAge	Age			

Input parameter	Abbreviation		Value		
			CT	RTE	OS
		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

The number of age classes (20) was sufficient to cover 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 as they grow relatively quickly and the final age class is a "plus group" containing all fish aged 19 years or more.

Alternative TAC specifications

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 7). 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.

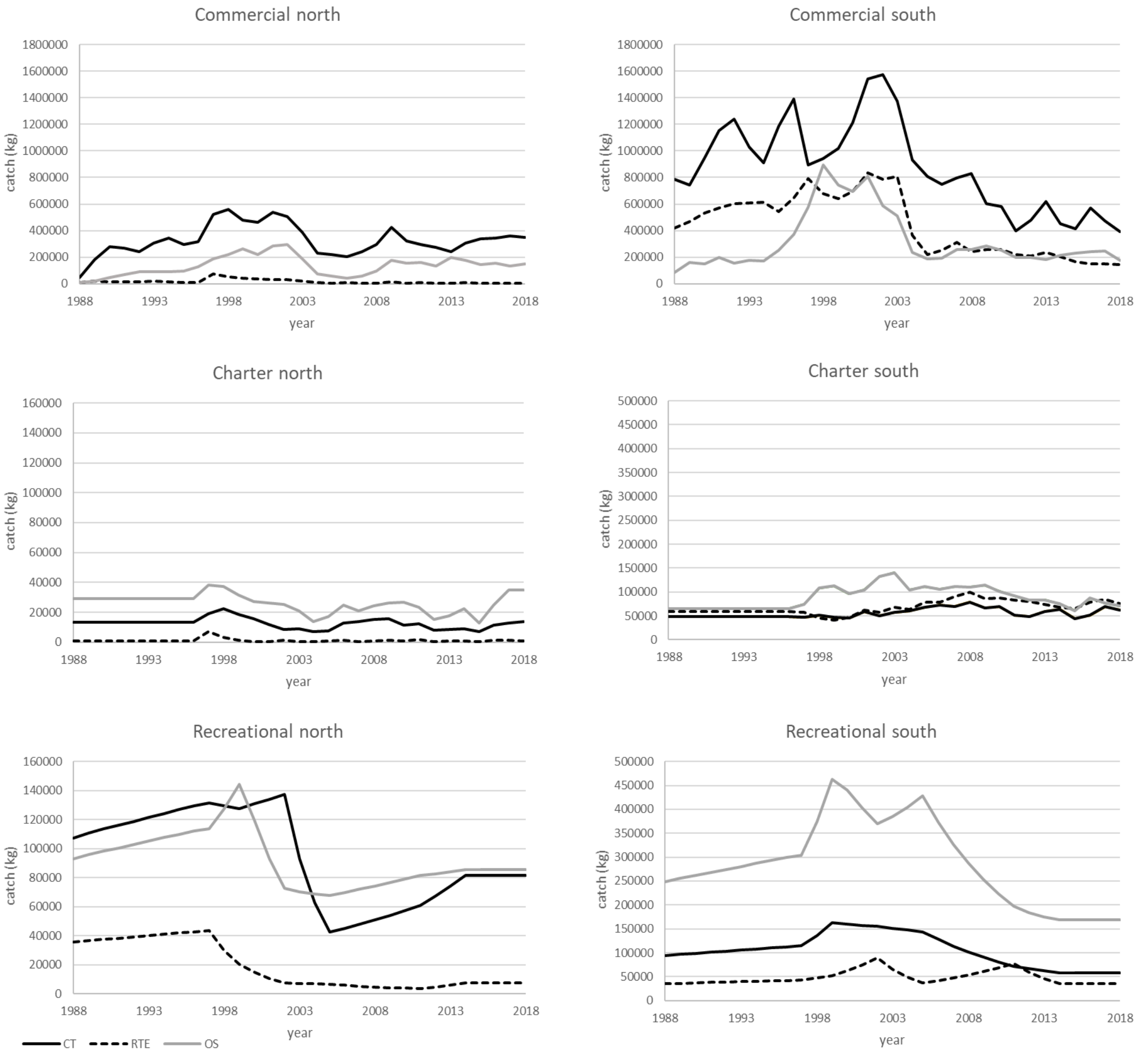


Figure 7. Reconstituted or actual historical time series of commercial, charter and recreational catch in the Coral Reef Finfish Fishery, by species group and region.

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.

Commercial and charter TAC

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.

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 (Courtney et al., 2015; Bureau of Meteorology, 2019; Queensland Government, 2019). We simulated a single cyclone event in the 5th 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.

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.

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.

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.

Model uncertainties and sensitivity analysis

Because the emphasis here 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 5). 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 E1 and Appendix Table E2, and a detailed consideration of each sensitivity test is provided in Appendix E. Our use of only one alternative parameter value for each performance indicator provides broad insight as to which indicators are most sensitive, and the extent to which changes to the parameterisation of one performance indicator have follow-on effects. Scenario-specific results are reported in detail in Appendix E.

Appendix E also summaries the outcomes of additional scenarios undertaken around external environmental perturbations (simplified “climate change” and “cyclone” scenarios), region-specific TACs, and the use of a metarule that constrains the permitted interannual change in magnitude of the TAC. We also consider the effect of an alternate optimisation algorithm.

Within the sensitivity analyses, 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.

Only two of the 17 sensitivity scenarios resulted in no change to modelled performance indicator values, due to these being well above or below specified thresholds. 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 (objective 1.2.1) and discarding (objective 1.2.2), and the related social perception of the fishery (objective 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.

Given the general sensitivity of any one performance indicator when its parameter values are changed, we emphasise the importance of agreeing on the form in which each conceptual TBL objective is operationalised as a performance indicator, and an awareness of the data limitations that may be associated with the definition of any performance indicator.

Table 5. Descriptive summary of conceptual objectives together with their translation into operational objectives, or performance indicators.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	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 (CT, RTE and key other species yet to be identified)**	We use a dome-shaped specification (Figure D1.8.1). 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).
		1.1.2 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).
	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, normalised 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 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	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	The worst possible discard percentage is assumed to be 0.5. We assume zero high grading for this fishery

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions
		from high-grading of target species)	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 normalised according to the worst possible expected discard percentage.	(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 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)
		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)
	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 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)	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 penalised 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.
2. Enhance fishery economic performance	**2.1 Maximise commercial economic benefits, as combined**	**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	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,

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions
	totals for each of the following sectors		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.	noting that these corresponded approximately to 0.6B0 for CT.
		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 was approximated by taking the historical high profits for each fishing sector, noting that 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.
	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.
	2.3 Maximise flow-on economic benefits to local		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	The recreational dollar scalar, and the historical maximum as the reference, are both assumed.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions
	communities (from all sectors)		performance indicator is obtained by normalising relative to the historical maximum.	
	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 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.	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 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.
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	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.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions
			(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.	
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.
	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.
		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	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 (CT	Each of the three contributing performance indicators is currently equally weighted.

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions
		with benefits to the community (commercial, rec, and charter)	and RTE sustainability), 1.1.2 (OS sustainability), and 2.3 (flow-on economic benefits), and take an average across them.	
	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.
		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.

Results

Literature review: Inventory of TBL objectives

The TBL objectives inventory prepared as an objective of this project is provided in Table 6.

Table 6. Triple bottom line objectives inventory

Domain	Objectives	Dimension (subject)	Reference
Management	Strengthen partnerships between and within industry and government	Simplify/improve management structures	(Pascoe et al., 2013b)
Management	Ensure management strategies have low compliance risk	Simplify/improve management	(Pascoe et al., 2013b)
Management	Minimise other management costs e.g. industry compliance costs[7]	Simplify/improve management	(Pascoe et al., 2013b; van Putten et al., 2015; Dutra et al., 2016; Ogier et al., 2020)
Management	Minimise legislation volume and complexity and Remove regulatory barriers to flexibility (alternative harvesting techniques, zoning, diversification in the economy)	Simplify/improve management	(Pascoe et al., 2013b; van Putten et al., 2015; Dutra et al., 2016)
Management	Maximise operational and administrative flexibility	Simplify/improve management	(Andalecio, 2011; Pascoe et al., 2013b; van Putten et al., 2015; Dutra et al., 2016; Jennings et al., 2016a)
Management	Foster resource stewardship through fisher understanding of rules and regulations	Simplify/improve management	(Pascoe et al., 2013b; Brooks et al., 2015; van Putten et al., 2015; Dutra et al., 2016; Jennings et al., 2016a)
Management	Appropriate access - Appropriately allocate shares of access to stocks	Administration of managed fisheries	(Ogier et al., 2020)
Management	Recognise Aboriginal traditional fishing access	Administration of managed fisheries	(Ogier et al., 2020)
Management	Encourage participation in management including ensuring appropriate mechanisms exist for fisher involvement in development of fisheries management advice.	Administration of managed fisheries	(Brooks et al., 2015; van Putten et al., 2015; Dutra et al., 2016; Ogier et al., 2020)

[7] See Pascoe et al 2009

Domain	Objectives	Dimension (subject)	Reference
Management	Maintain or Increase compliance with fishing regulations and compliance with environmental and resource use regulations	Administration of managed fisheries	(Andalecio, 2011; van Putten et al., 2015; Dutra et al., 2016; Ogier et al., 2020)
Management	Ensure sufficient information exists to inform management decisions	Administration of managed fisheries	(Ogier et al., 2020)
Management	Achieve government targets for the recovery of costs of management	Administration of managed fisheries	(Ogier et al., 2020)
Management	Ensure obligations under international agreements	Administration of managed fisheries	(Ogier et al., 2020)
Management	Minimise [stakeholder] conflict	Administration of managed fisheries	(Ogier et al., 2020)
Management	Quantify the economic rents being obtained from the Commonwealth fisheries	Fisheries rent	(Chesson and Whitworth, 2004)
Management	Increase management acceptability	Increase management support	(van Putten et al., 2015; Dutra et al., 2016)
Management	Increase policy, regulatory and implementation integration	Increase management integration	(van Putten et al., 2015; Dutra et al., 2016)
Management	To ensure that ESD principles are underpinned by legal, institutional, economic and policy frameworks capable of responding and taking appropriate pre-emptive and remedial actions.	Governance	(Chesson and Whitworth, 2004)
Management	Within management responses recognise the impacts of the environment on fisheries from both natural and non-fishery human induced sources and incorporate these	Impacts of the Environment	(Chesson and Whitworth, 2004)
Management	The fishery is conducted, in a manner that minimises the impact of fishing operations on the ecosystem generally	General ecosystem	(Chesson and Whitworth, 2004)

Domain	Objectives	Dimension (subject)	Reference
Social or Management	Industry stakeholders have a high level of trust in the management of fisheries	Industry Community wellbeing	(Brooks et al., 2015)
Social or Management	Maximise stewardship of fisheries resources	Industry Community wellbeing	(Brooks et al., 2015)
Social or Management	Perceived transparency in fisheries management and decision-making processes	Industry Community wellbeing	(Brooks et al., 2015)
Social or Management	Aboriginal and Torres Strait Islander communities associated with 'Sea Country' resources have a high level of trust in the management of fisheries	Indigenous Community wellbeing	(Cowx and Van Anrooy, 2010; Brooks et al., 2015)
Social	To satisfy traditional (customary) fishing needs, cultural/economic development and ESD of indigenous communities	Socio-cultural	(Leung et al., 1998; Chesson et al., 1999; Chesson and Whitworth, 2004; Davis and Wagner, 2006; Tobin et al., 2010; Urquhart et al., 2011; Brooks et al., 2015)
Social	Equity [in employment] [maximise flow-on economic and employment benefits - Fair access for all local people with both full-time and part-time employment	Socio-cultural	(Plagányi et al., 2013; Brooks et al., 2015; Jennings et al., 2016a)
Social	New entrants - Potential for new and young Islanders to enter the fisher	Socio-cultural	(Plagányi et al., 2013)
Social	Social benefits for recreational users, including recreational fishers	Social benefits	(Leung et al., 1998; Mapstone et al., 2008; Ogier et al., 2020)
Social	Safety of all types of users from shark attack [by fishing for shark species]	Social benefits	(Ogier et al., 2020)
Social	Conserve traditional activities and culture {indigenous and non-indigenous] to facilitate and support the cohesion and connectedness	Increase social cohesion	(Plagányi et al., 2013; Brooks et al., 2015; van Putten et al., 2015; Dutra et al., 2016)

Domain	Objectives	Dimension (subject)	Reference
Social	Improve workplace and family health, sanitation and safety in the region	Increase social capacity	(Mardle et al., 2002; Soma, 2003; Coulthard, 2012; Nunan, 2014; Anderson et al., 2015; van Putten et al., 2015; Dutra et al., 2016)
Social	Improve education, training, social infrastructure and networks for captains, crew, processing workers	Increase social capacity	(Anderson et al., 2015; van Putten et al., 2015; Dutra et al., 2016)
Social	Provide flexible opportunities to ensure fishers can maintain or enhance their livelihood, within the constraints of ecological sustainability	Industry Community wellbeing	(Symes and Phillipson, 2009; Brooks et al., 2015)
Social	Improve or maintain ability of fishers to participate effectively in fisheries management advisory processes	Industry Community wellbeing	(Brooks et al., 2015)
Social	Ensure equitable treatment and access for fishers	Industry Community wellbeing	(Brooks et al., 2015)
Social	Ensure adequate access to infrastructure and fisheries information needed for successful operation of fishing activities	Industry Community wellbeing	(Brooks et al., 2015)
Social	Provide opportunities for Aboriginal and Torres Strait Islander communities to participate in fisheries management decision making processes	Indigenous Community wellbeing	(Brooks et al., 2015)
Social	Make fisheries collected data available in a timely and publicly accessible manner to Aboriginal and Torres Strait Islander communities	Indigenous Community wellbeing	(Brooks et al., 2015)
Social	Ensure collaborative inputs by Aboriginal and Torres Strait Islander communities, regional and industry sectors on the benefits each sector offers to fisheries management	Indigenous Community wellbeing	(Brooks et al., 2015)
Social	To contribute to community, regional and national well-being, lifestyle and cultural needs	Regional and National Wellbeing	(Chesson and Whitworth, 2004; Symes and Phillipson, 2009; Brooks et al., 2015)

Domain	Objectives	Dimension (subject)	Reference
Social	Positively influence fisheries related socioeconomic benefits for regional communities, within the constraints of ecological sustainability	Local and Regional wellbeing	(Brooks et al., 2015)
Social	Facilitate capacity building (through skills and knowledge development) for community members to enhance stewardship of fisheries resources [improve opportunities for co-management and stakeholder participation]	Local and Regional wellbeing	(Brooks et al., 2015; Jennings et al., 2016a)
Social	Ensure fisheries information is available in a timely and publicly accessible manner [enhance accountability and transparency]	Local and Regional wellbeing	(Brooks et al., 2015; Jennings et al., 2016a)
Social	Minimise conflicts with recreational and sports fisheries	Minimise externalities	(Pascoe et al., 2009a)
Social[8]	Harvest [food] safety	Harvest performance	(Anderson et al., 2015)
Social	Processing workers remuneration - earnings, social standing	Labour returns	(Anderson et al., 2015)
Social	Non-resident employment for captains, crew, processing workers	Local ownership	(Anderson et al., 2015)
Ecological	Sufficient spawning biomass	Biological	(Plagányi et al., 2013)
Ecological	Sustainable target species stocks by maintaining biomass of mature and legal sized target species, or increase to, within established reference levels	Sustainable resource use	(Plagányi et al., 2013; Ogier et al., 2020)
Ecological	Minimise percentage of overfished stocks, degree of overfishing	Fish stock health & environmental	(Anderson et al., 2015)

[8] Nominated as Community rather than Social in the paper.

Domain	Objectives	Dimension (subject)	Reference
Ecological	Stock trends improved	Fish stock health & environmental	(Anderson et al., 2015; Ogier et al., 2020)
Ecological	Minimise and monitor IUU[9] landings	Fish stock health & environmental	(Anderson et al., 2015)
Ecological	Proportion of harvest with 3rd party certification	Fish stock health & environmental	(Anderson et al., 2015)
Ecological	Ensure harvested resource sustainability with ecologically viable stock levels, reduction of target species discards and maintain positive trends	Ensure harvest resource sustainability	(Chesson and Whitworth, 2004; Pascoe et al., 2013b; Anderson et al., 2015; Jennings et al., 2016a; Ogier et al., 2020)
Ecological	Minimise bycatch and discards to maintain or improve ecological viability of non-target species	Bycatch species	(Chesson and Whitworth, 2004; Pascoe et al., 2013b; Jennings et al., 2016a)
Ecological	Reduce impacts on threatened, endangered, protected (TEP) species	Conserve inshore living resources	(van Putten et al., 2015; Dutra et al., 2016)
Ecological	Maximise productive area of habitat	Ensure long run ecosystem resilience	(Pascoe et al., 2013b; Jennings et al., 2016a)
Ecological	Minimise impacts of fishing on biodiversity and ecosystem function and avoid mortality of, or injuries to, endangered, threatened or protected species	Ensure long run ecosystem resilience	(Chesson and Whitworth, 2004; Pascoe et al., 2013b; Jennings et al., 2016a; Ogier et al., 2020)
Ecological	Minimise pollution and carbon footprint of the industry	Ensure sustainability and improve water quality	(Pascoe et al., 2013b; van Putten et al., 2015; Dutra et al., 2016; Jennings et al., 2016a)
Ecological	Where the fished stock(s) are below a defined reference point, the fishery will be managed to promote recovery to ecologically viable stock levels within nominated timeframes.	Primary stocks and by-product	(Chesson and Whitworth, 2004)

[9] Illegal, Unregulated or Unreported landings

Domain	Objectives	Dimension (subject)	Reference
Ecological	The fishery is conducted in a manner that and avoids or minimises impacts on threatened ecological communities	Threatened species and Threatened communities	(Chesson and Whitworth, 2004)
Ecological	Fishing is conducted in a manner that does not compromise the integrity of the broader marine ecosystem	Broader marine ecosystem	(Chesson and Whitworth, 2004)
Ecological	Prevent and manage marine pest incursions	Marine pests	(Chesson and Whitworth, 2004)
Ecological	Reduce direct impacts of infrastructure and development	Improve ecosystem connectivity	(van Putten et al., 2015; Dutra et al., 2016)[10]
Ecological	Sustainable human use of marine resources	Conserve inshore living resources	(van Putten et al., 2015; Dutra et al., 2016)
Ecological or Economic	To maintain/increase fisheries production and value for Commonwealth and state fisheries	Fisheries production	(Chesson and Whitworth, 2004)
Economic	Maximise value of tradable units (through ratio asset value to gross earnings)	Maximise economic performance	(Dichmont et al., 2013a; Pascoe et al., 2013b; Anderson et al., 2015; Jennings et al., 2016a)
Economic	Minimise annual fixed and variable fishing costs and maintain/increase economic efficiency	Maximise economic performance	(Pascoe et al., 2013b; Plagányi et al., 2013; Jennings et al., 2016a; Ogier et al., 2020)
Economic	Improve product quality to improve product price	Maximise economic performance	(Pascoe et al., 2013b; Anderson et al., 2015; Jennings et al., 2016a)
Economic	Maintain and improve market access to improve price	Maximise economic performance	(Pascoe et al., 2013b; Ogier et al., 2020)

[10] These objectives include fisheries as well as coastal management

Domain	Objectives	Dimension (subject)	Reference
Economic	Maximise catch rates	Maximise economic performance	(Pascoe et al., 2013b; Jennings et al., 2016a)
Economic	Maintain/increase productivity and increase fisheries trade volume and value, economic performance of the fishing industry	Economic benefits	(Chesson and Whitworth, 2004; Ogier et al., 2020)
Economic	Improve economic efficiencies	Economic benefits	(Chesson and Whitworth, 2004; Anderson et al., 2015; Ogier et al., 2020)
Economic	Ensure fishers can maintain or enhance their livelihood	Social benefits	(Lane, 1989; Leung et al., 1998; Chesson et al., 1999; Mascia, 2003; Soma, 2003; Chesson and Whitworth, 2004; Glaser and Diele, 2004; Davis and Wagner, 2006; Hilborn, 2007; Marshall and Marshall, 2007; Symes and Phillipson, 2009; Tobin et al., 2010; Stouten et al., 2011; Urquhart et al., 2011; Coulthard, 2012; Ogier et al., 2020)
Economic	Minimise spillover effects to other fisheries	Minimise externalities	(Pascoe et al., 2009a)
Economic	Excess capacity and capacity utilization rate	Harvest performance	(Chesson and Whitworth, 2004; Anderson et al., 2015)
Economic	Capacity to export to US & EU, and other international trade	Owners, Permit holders, rights and access	(Anderson et al., 2015)
Economic	Maximise processing yield	Owners, Permit holders, rights and access	(Anderson et al., 2015)
Economic	Product improvement	Owners, Permit holders, rights and access	(Anderson et al., 2015)
Economic	To maintain/increase fisheries trade volume and value	Fisheries trade	(Chesson and Whitworth, 2004)

Domain	Objectives	Dimension (subject)	Reference
Economic	Improve regional economic development and industry diversity	Increase economic growth	(van Putten et al., 2015; Dutra et al., 2016)
Economic	Improve family livelihoods in the region	Increase economic growth	(Symes and Phillipson, 2009; van Putten et al., 2015; Dutra et al., 2016)
Economic	Ensure that natural resource based industries are profitable and sustainable	Increase economic growth	(van Putten et al., 2015; Dutra et al., 2016)
Economic	Employment (Active part-time, full-time, casual fishers, transferrable vessel holders)	Economic contribution	(Plagányi et al., 2013; Jennings et al., 2016a)
Economic	Maximise value added opportunities	Economic contribution	(Plagányi et al., 2013)
Economic	Minimise [negative] impacts on indigenous communities and fisheries	Minimise externalities	(Pascoe et al., 2009a)

The concept of explicit integration of objectives has been developed further for fisheries management and we include Table 7 below to illustrate how this has been achieved (Anderson et al., 2015). Table 8 is an example of various inventory lists used when compiling our objectives inventory.

Table 7. Fishery Performance Indicators: a management tool for TBL outcomes (Source: (Anderson et al., 2015)). Note the authors suggest ecology, economics and community as TBL indicators and showing these alongside sector indicators (stock performance, harvest sector performance and post-harvest sector performance); they provide 61 case studies.

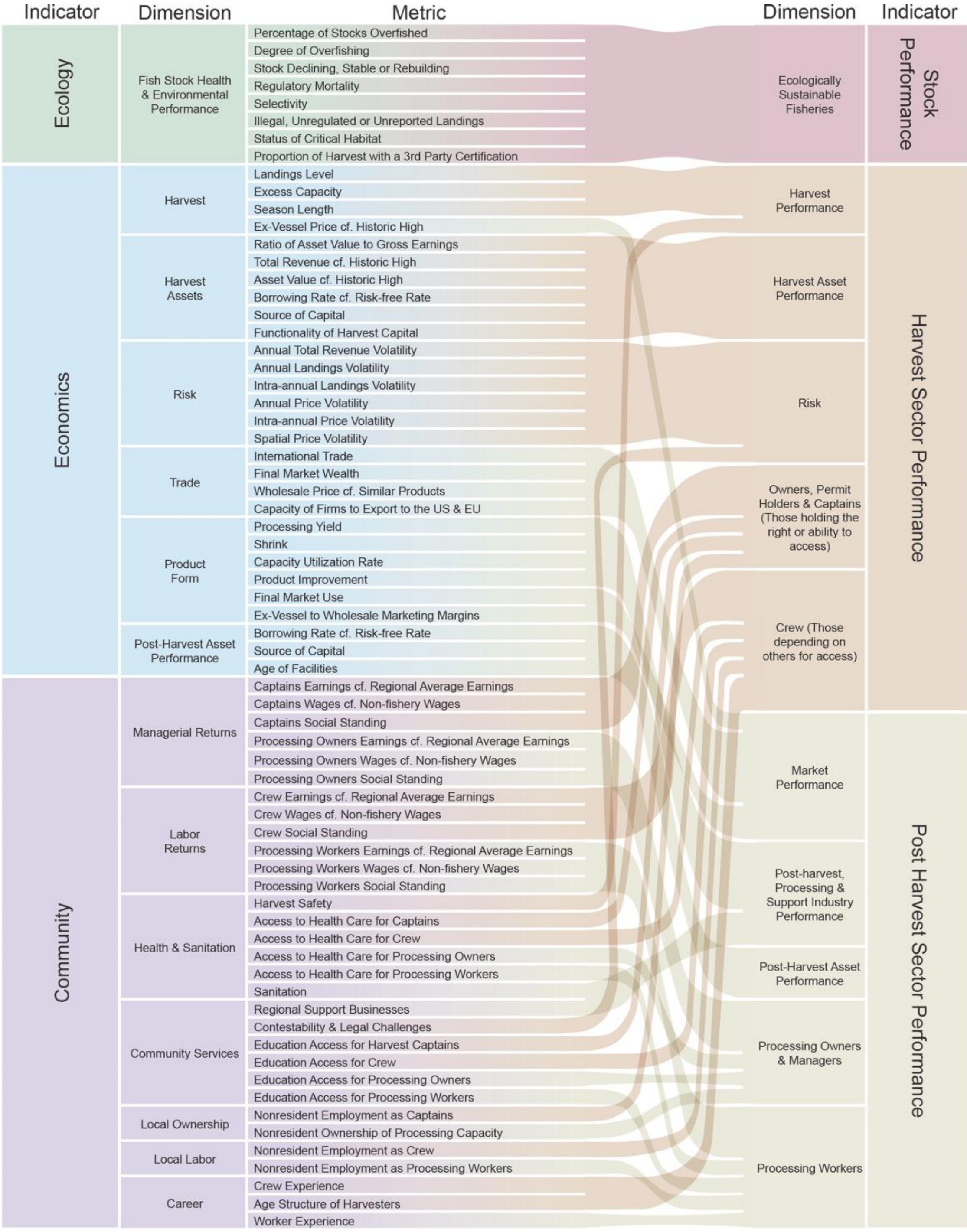


Table 8. An example of the inventories and lists used in the compilation of the objectives inventory. Source: Pascoe et al. (2014a)

Social objectives relating to fisheries management.

Objective	Commercial fisheries	Recreational fisheries	Indigenous fisheries
Maintain or enhance family incomes and livelihoods	Chesson et al. (1999); Coulthard (2012); Davis and Wagner (2006); Glaser and Diele (2004); Hilborn (2007); Lane (1989); Leung et al. (1998); Marshall (2007); Mascia (2003); Soma (2003); Stouten et al. (2011); Symes and Phillipson (2009); Tobin et al. (2009); Urquhart et al. (2011)		Andalecio (2011); Plagányi et al. (2013)
Maintain or maximise employment	Chesson et al. (1999); Cheung and Sumaila (2008); Fulton et al. (2007); Hilborn (2007); Mardle et al. (2002); Mardle et al. (2004); Nunan (2013); Pascoe et al. (2013a); Stouten et al. (2011); Symes and Phillipson (2009); Urquhart et al. (2011)		Plagányi et al. (2013)
Maintain communities	Fulton et al. (2007); Hilborn (2007); Mardle et al. (2002); Mardle et al. (2004); Marshall (2007); Pascoe et al. (2009b); Symes and Phillipson (2009); Tobin et al. (2009); Urquhart et al. (2011)	Cowx and Van Anrooy (2010)	Plagányi et al. (2013)
Equity	Andalecio (2011); Davis and Wagner (2006); Fulton et al. (2007); Glaser and Diele (2004); Mardle et al. (2004); Marshall (2007); Nunan (2013); Pascoe et al. (2013a); Tobin et al. (2009)		Plagányi et al. (2013)
Maintain social capital	Brooks (2010); Davis and Wagner (2006); Marshall (2007); Soma (2003); Urquhart et al. (2011)		
Ensure health and safety	Coulthard (2012); Mardle et al. (2002); Nunan (2013); Soma (2003)		
Conserve traditional activities, culture and products	Chesson et al. (1999); Davis and Wagner (2006); Leung et al. (1998); Tobin et al. (2009); Urquhart et al. (2011)	Cowx and Van Anrooy (2010)	Plagányi et al. (2013)
Maintain or improve recreational access	Leung et al. (1998); Mapstone et al. (2008)	Leung et al. (1998); Mapstone et al. (2008)	
Maintain or enhance resilience	Brooks (2010); Marshall (2007, 2010); Marshall and Marshall (2007); Tobin et al. (2009); Urquhart et al. (2011)		
Enhance quality of life	Coulthard (2012); Lane (1989); Leung et al. (1998); Schirmer and Casey (2005); Tobin et al. (2009)		
Avoid social exclusion (improve public perception)	Fulton et al. (2007); Symes and Phillipson (2009)		
Minimise conflicts between alternative users	Andalecio (2011); Davis and Wagner (2006); Fulton et al. (2007); Leung et al. (1998); Mardle et al. (2002); Mardle et al. (2004); Pascoe et al. (2009b)	Cowx and Van Anrooy (2010)	
Ensure food supply	Chesson et al. (1999)		
Ensure management stability	Fulton et al. (2007)		
Ensure management acceptability	Andalecio (2011)		

Objective weighting survey responses and levels of fishery experience

The objective weighting survey received a total of 110 responses, of which around half were from commercial fishers (Figure 8). Most respondents indicated that they fell into two different stakeholder groups, with the biggest overlap being between the commercial fishers and quota holders. Only three responses were received from conservation groups, so these were incorporated into the “other” category, which also contains commercial and recreational fisheries group representatives (but not actively fishing).

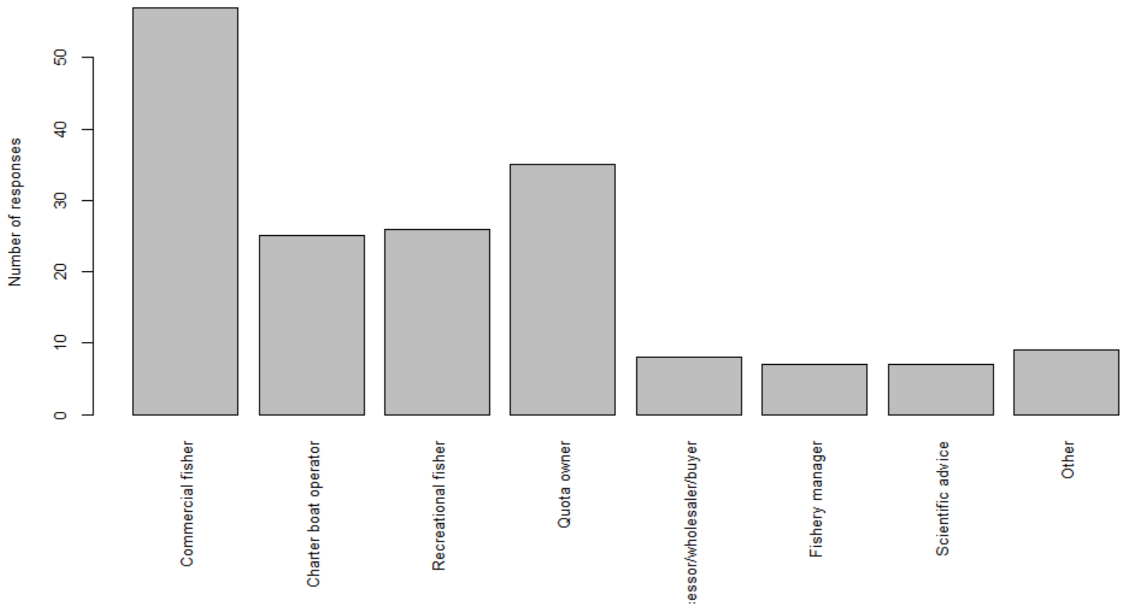


Figure 8. Distribution of survey responses by stakeholder group

All groups had considerable experience on average with the fishery and with fisheries in general (Figure 9). Fisheries managers had the least experience on average with the fishery, but even this was considerable, averaging around 10 years’ experience directly with the fishery and over 15 years’ experience in fisheries management in general. Given this, the view of the respondents can be considered to be based on considerable experience.

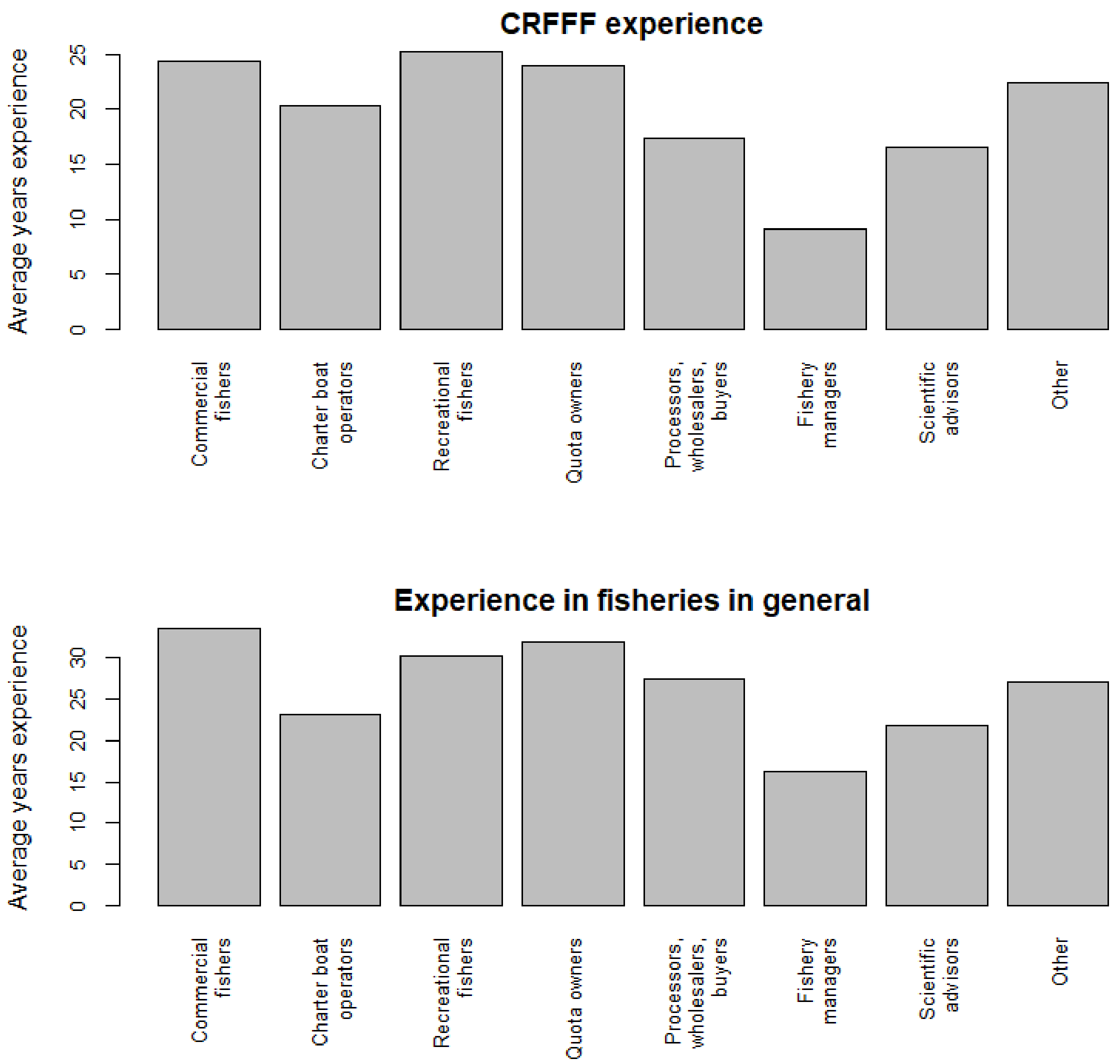


Figure 9. Average experience by stakeholder group

Two approaches were used to derive the weights from the survey responses. A ratings approach, which assessed the relative contribution of the objective to the overall sum of preference scores, and a variant of the Analytic Hierarchy Process (AHP), which is based on the differences between the individual preference scores. The following results are based on the ratings approach. Both approaches yield similar average weightings.

Objective importance

As noted in the methods section, two alternative approaches were applied to derive the objective weights.

Scoring approach

The distributions of the derived higher level objective groups (Economic, Social, Ecological Sustainability and Governance) by stakeholder group are shown in Figure 10 and relative comparisons of the average weights shown in Figure 11. All groups tended to rank the sustainability objectives the highest although there was not a substantial difference between sustainability and other objectives for most commercial and industry related groups. Managers, scientists and the “other” groups all had a higher weighting on ecological sustainability. The commercially oriented groups (fishers, charter boat operators, quota owners and buyers) all rated economic objectives above social objectives.

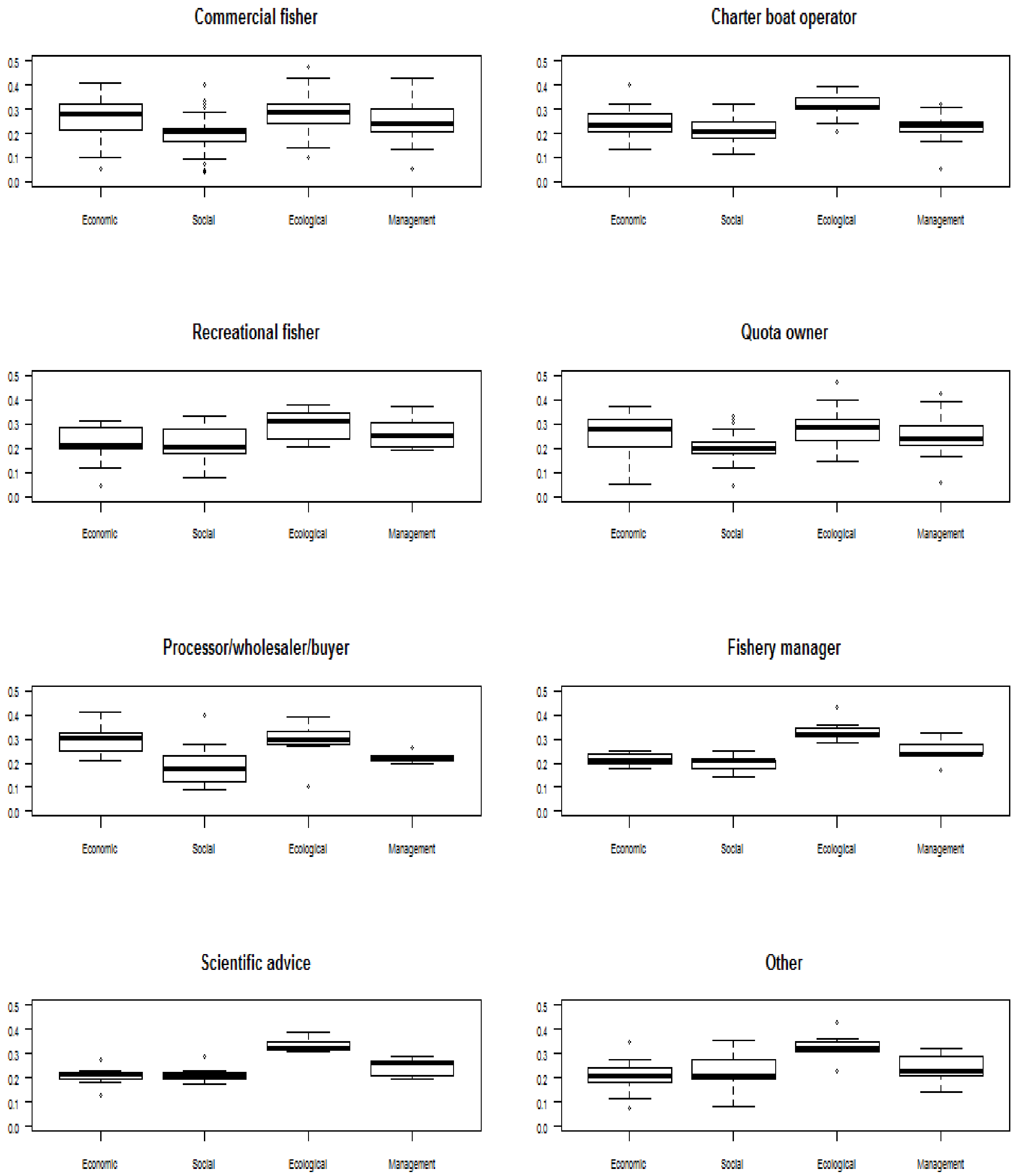


Figure 10. Objective weight distributions by group: scoring method

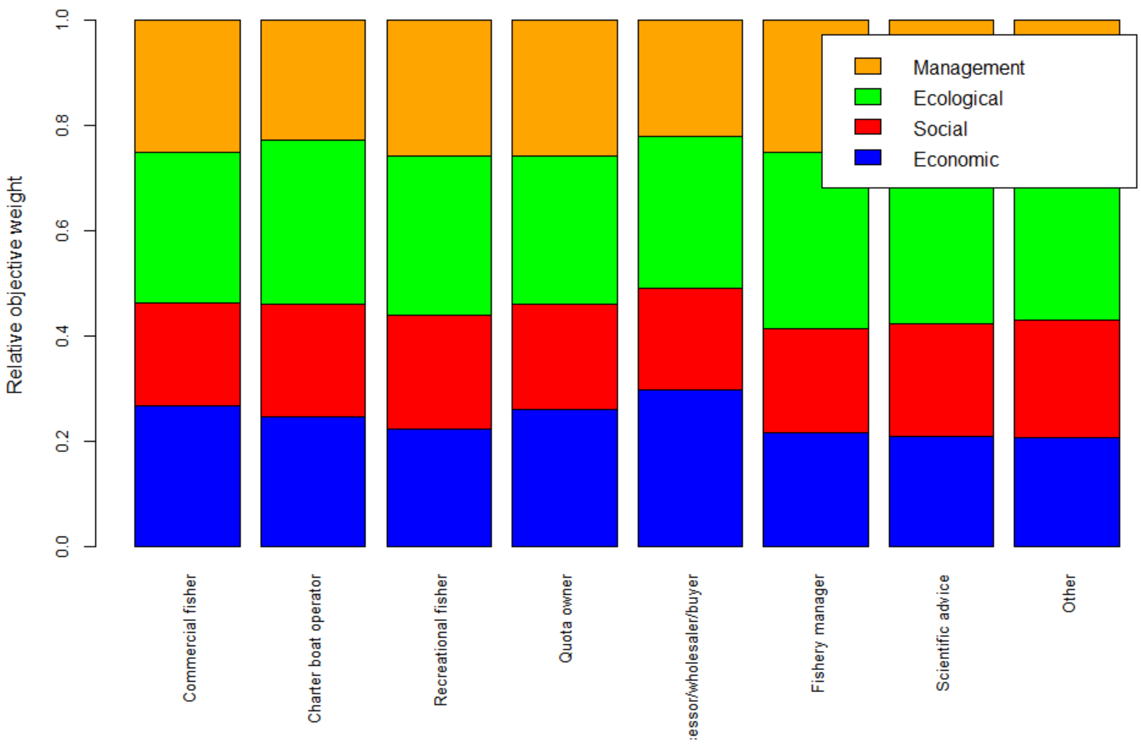


Figure 11. Average objective weight by group: scoring method

The full set of average objective weightings are given in Table 9. The relative standard errors (RSEs) in Table 9 are the relative standard errors expressed as a percentage, and provide an indication of the confidence in the mean value. As the data are derived from a sample, the RSEs provide a measure as to where we expect the population average to be found. As an approximation, we would expect the true average value to be within two RSEs of the sample mean. For example, we would expect the true population average weight for the economics group of objectives to be within 6% (i.e. 2 × 3%) of 0.29 for commercial fishers (i.e. between 0.27 and 0.31). As RSEs are influenced by the sample size, the stakeholder groups with the larger sample sizes tended to also have the smallest RSEs (and vice versa).

The weightings on the lower level objectives appear fairly similar across the different stakeholder groups. This is an artefact to some extent of the "dilution" effect of distributing the higher level objective weights over many sub-objectives. The cumulative effect of these small differences at the lower level, however, may result in overall different preferences for different harvest strategies at the stakeholder level.

Modified AHP method

The same analyses was repeated using the modified AHP method to derive the weights. The distributions of the derived higher level objective groups (Economic, Social, Ecological Sustainability and Governance) by stakeholder group are shown in Figure 12 and relative comparisons of the average weights shown in Figure 13. As with the scoring approach, all groups tended to rank the sustainability objectives the highest although there was greater difference between sustainability and other objectives for most commercial and industry related groups compared with the scoring approach. As before, managers, scientists and the “other” groups all had a higher weighting on ecological sustainability. The commercially oriented groups (fishers, charter boat operators, quota owners and buyers) generally rated economic objectives above social objectives.

The full set of average objective weightings are given in Table 10. As with the scoring approach, the weightings on the lower level objectives appear fairly similar across the different stakeholder groups. This is an artefact to some extent of the “dilution” effect of distributing the higher level objective weights over many sub-objectives. Again, the cumulative effect of these small differences at the lower level, however, may result in overall different preferences for different harvest strategies at the stakeholder level.

Table 9. Average weights by stakeholder group: ratings method

Group	Commercial fisher		Charter boat operator		Recreational fisher		Quota owner		Processor/ wholesaler/ buyer		Fishery manager		Scientific advice		Other	
Responses	57		25		26		35		8		7		7		9	
	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE
Ecol1	0.290	3%	0.310	3%	0.294	4%	0.281	4%	0.326	5%	0.335	5%	0.336	3%	0.329	6%
Ecol1.1	0.097	4%	0.103	4%	0.103	4%	0.097	5%	0.134	13%	0.133	6%	0.121	10%	0.110	6%
• Ecol1.1.1	0.052	6%	0.049	4%	0.055	5%	0.052	7%	0.071	17%	0.076	10%	0.059	10%	0.052	13%
• Ecol1.1.2	0.045	6%	0.054	4%	0.048	5%	0.044	9%	0.063	10%	0.056	9%	0.063	16%	0.058	11%
Ecol1.2	0.097	5%	0.103	6%	0.098	5%	0.094	6%	0.098	6%	0.107	4%	0.125	5%	0.108	10%
• Ecol1.2.1	0.023	7%	0.024	6%	0.023	7%	0.021	9%	0.022	6%	0.022	3%	0.029	7%	0.026	8%
• Ecol1.2.2	0.027	5%	0.028	6%	0.026	7%	0.028	6%	0.03	12%	0.029	7%	0.030	9%	0.032	17%
• Ecol1.2.3	0.025	6%	0.025	8%	0.025	6%	0.024	8%	0.025	16%	0.028	9%	0.031	12%	0.027	19%
• Ecol1.2.4	0.023	6%	0.026	6%	0.024	7%	0.021	8%	0.021	7%	0.028	8%	0.034	4%	0.023	14%
Ecol1.3	0.096	5%	0.103	4%	0.094	5%	0.090	6%	0.094	8%	0.096	10%	0.089	12%	0.110	7%
• Ecol1.3.1	0.056	6%	0.062	7%	0.053	9%	0.055	8%	0.045	14%	0.05	12%	0.049	16%	0.055	9%
• Ecol1.3.2	0.040	7%	0.041	7%	0.040	8%	0.035	9%	0.049	13%	0.046	13%	0.041	10%	0.055	10%
Econ2	0.261	4%	0.254	5%	0.221	7%	0.253	5%	0.291	5%	0.216	5%	0.208	8%	0.206	14%

Group	Commercial fisher		Charter boat operator		Recreational fisher		Quota owner		Processor/ wholesaler/ buyer		Fishery manager		Scientific advice		Other	
Econ2.1	0.063	5%	0.051	8%	0.043	11%	0.061	6%	0.072	5%	0.045	15%	0.045	17%	0.044	19%
• Econ2.1.1	0.032	7%	0.018	14%	0.015	14%	0.032	8%	0.033	10%	0.018	17%	0.016	21%	0.022	32%
• Econ2.1.2	0.016	7%	0.021	9%	0.017	12%	0.015	9%	0.023	5%	0.013	17%	0.014	19%	0.013	23%
• Econ2.1.3	0.014	8%	0.012	8%	0.011	11%	0.013	12%	0.016	20%	0.014	13%	0.014	19%	0.009	22%
Econ2.2	0.029	8%	0.048	8%	0.051	8%	0.027	11%	0.049	13%	0.049	5%	0.037	4%	0.033	18%
Econ2.3	0.055	6%	0.056	7%	0.051	8%	0.051	7%	0.059	17%	0.041	11%	0.045	13%	0.052	19%
Econ2.4	0.047	6%	0.045	8%	0.034	10%	0.048	8%	0.045	9%	0.040	9%	0.037	11%	0.033	19%
Econ2.5	0.067	5%	0.055	8%	0.043	9%	0.066	6%	0.067	11%	0.041	18%	0.044	12%	0.044	14%
Manage3	0.255	4%	0.219	5%	0.264	4%	0.265	4%	0.227	3%	0.251	7%	0.242	6%	0.239	8%
Social4	0.195	4%	0.217	4%	0.221	6%	0.202	4%	0.156	7%	0.197	7%	0.214	7%	0.226	12%
Social4.1	0.053	7%	0.064	9%	0.063	9%	0.059	8%	0.042	21%	0.068	8%	0.068	14%	0.062	16%
Social4.2	0.068	5%	0.078	5%	0.078	7%	0.072	5%	0.052	15%	0.059	12%	0.070	10%	0.092	17%
• Social4.2.1	0.019	6%	0.023	8%	0.024	7%	0.021	7%	0.016	22%	0.019	14%	0.017	20%	0.028	18%
• Social4.2.2	0.024	6%	0.027	7%	0.025	9%	0.025	6%	0.017	10%	0.021	14%	0.028	18%	0.030	20%
• Social4.2.3	0.025	6%	0.028	6%	0.029	8%	0.027	6%	0.019	20%	0.020	13%	0.024	14%	0.034	19%
Social4.3	0.073	6%	0.075	6%	0.079	8%	0.071	7%	0.062	11%	0.070	15%	0.076	11%	0.072	19%

Group	Commercial fisher		Charter boat operator		Recreational fisher		Quota owner		Processor/ wholesaler/ buyer		Fishery manager		Scientific advice		Other	
• Social4.3.1	0.040	7%	0.037	8%	0.038	8%	0.040	9%	0.030	13%	0.036	13%	0.043	17%	0.045	27%
• Social4.3.2	0.032	6%	0.038	7%	0.041	9%	0.032	6%	0.032	14%	0.034	23%	0.034	9%	0.027	15%

Table 10. Average weights by stakeholder group: modified AHP method

Group	Commercial fisher		Charter boat operator		Recreational fisher		Quota owner		Processor/ wholesaler/ buyer		Fishery manager		Scientific advice		Other	
Responses		57		25		26		35		8		7		7		9
	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE	Mean	RSE
Ecol1	0.308	3%	0.384	3%	0.335	3%	0.297	3%	0.387	3%	0.454	3%	0.461	3%	0.405	3%
Ecol1.1	0.100	4%	0.130	4%	0.124	4%	0.110	4%	0.207	4%	0.254	4%	0.193	4%	0.138	4%
• Ecol1.1.1	0.058	6%	0.055	6%	0.070	6%	0.064	6%	0.133	6%	0.170	6%	0.093	6%	0.062	6%
• Ecol1.1.2	0.042	6%	0.075	6%	0.054	6%	0.046	6%	0.074	6%	0.083	6%	0.100	6%	0.076	6%
Ecol1.2	0.109	5%	0.132	5%	0.115	5%	0.101	5%	0.099	5%	0.120	5%	0.191	5%	0.133	5%
• Ecol1.2.1	0.022	7%	0.023	7%	0.026	7%	0.018	7%	0.018	7%	0.016	7%	0.040	7%	0.027	7%
• Ecol1.2.2	0.037	5%	0.045	5%	0.038	5%	0.039	5%	0.039	5%	0.038	5%	0.044	5%	0.051	5%
• Ecol1.2.3	0.028	6%	0.033	6%	0.027	6%	0.023	6%	0.027	6%	0.033	6%	0.054	6%	0.029	6%
• Ecol1.2.4	0.023	6%	0.031	6%	0.024	6%	0.020	6%	0.016	6%	0.033	6%	0.053	6%	0.026	6%
Ecol1.3	0.099	5%	0.122	5%	0.096	5%	0.087	5%	0.081	5%	0.080	5%	0.077	5%	0.134	5%
• Ecol1.3.1	0.066	6%	0.084	6%	0.061	6%	0.061	6%	0.038	6%	0.044	6%	0.051	6%	0.073	6%
• Ecol1.3.2	0.033	7%	0.038	7%	0.035	7%	0.025	7%	0.042	7%	0.036	7%	0.027	7%	0.062	7%

Group	Commercial fisher		Charter boat operator		Recreational fisher		Quota owner		Processor/ wholesaler/ buyer		Fishery manager		Scientific advice		Other	
Econ2	0.286	4%	0.239	4%	0.214	4%	0.289	4%	0.311	4%	0.125	4%	0.123	4%	0.169	4%
Econ2.1	0.099	5%	0.062	5%	0.053	5%	0.098	5%	0.094	5%	0.033	5%	0.035	5%	0.036	5%
• Econ2.1.1	0.064	7%	0.023	7%	0.025	7%	0.064	7%	0.057	7%	0.017	7%	0.016	7%	0.023	7%
• Econ2.1.2	0.018	7%	0.031	7%	0.020	7%	0.019	7%	0.024	7%	0.008	7%	0.010	7%	0.008	7%
• Econ2.1.3	0.017	8%	0.008	8%	0.008	8%	0.015	8%	0.013	8%	0.008	8%	0.009	8%	0.006	8%
Econ2.2	0.023	8%	0.043	8%	0.053	8%	0.025	8%	0.036	8%	0.032	8%	0.014	8%	0.018	8%
Econ2.3	0.061	6%	0.057	6%	0.048	6%	0.060	6%	0.086	6%	0.022	6%	0.032	6%	0.059	6%
Econ2.4	0.050	6%	0.036	6%	0.028	6%	0.052	6%	0.043	6%	0.018	6%	0.019	6%	0.026	6%
Econ2.5	0.053	5%	0.041	5%	0.032	5%	0.055	5%	0.053	5%	0.020	5%	0.022	5%	0.030	5%
Manage3	0.264	4%	0.232	4%	0.277	4%	0.271	4%	0.241	4%	0.317	4%	0.293	4%	0.270	4%
Social4	0.142	4%	0.145	4%	0.174	4%	0.143	4%	0.061	4%	0.104	4%	0.123	4%	0.156	4%
Social4.1	0.033	7%	0.047	7%	0.048	7%	0.038	7%	0.016	7%	0.038	7%	0.045	7%	0.039	7%
Social4.2	0.048	5%	0.051	5%	0.056	5%	0.051	5%	0.021	5%	0.029	5%	0.037	5%	0.067	5%
• Social4.2.1	0.010	6%	0.011	6%	0.013	6%	0.011	6%	0.006	6%	0.008	6%	0.006	6%	0.017	6%
• Social4.2.2	0.017	6%	0.018	6%	0.018	6%	0.018	6%	0.005	6%	0.012	6%	0.016	6%	0.021	6%
• Social4.2.3	0.021	6%	0.022	6%	0.026	6%	0.022	6%	0.010	6%	0.009	6%	0.015	6%	0.029	6%

Group	Commercial fisher		Charter boat operator		Recreational fisher		Quota owner		Processor/ wholesaler/ buyer		Fishery manager		Scientific advice		Other	
Social4.3	0.061	6%	0.047	6%	0.070	6%	0.054	6%	0.024	6%	0.037	6%	0.041	6%	0.050	6%
• Social4.3.1	0.039	7%	0.023	7%	0.032	7%	0.034	7%	0.013	7%	0.017	7%	0.026	7%	0.035	7%
• Social4.3.2	0.022	6%	0.024	6%	0.038	6%	0.02	6%	0.012	6%	0.021	6%	0.015	6%	0.016	6%

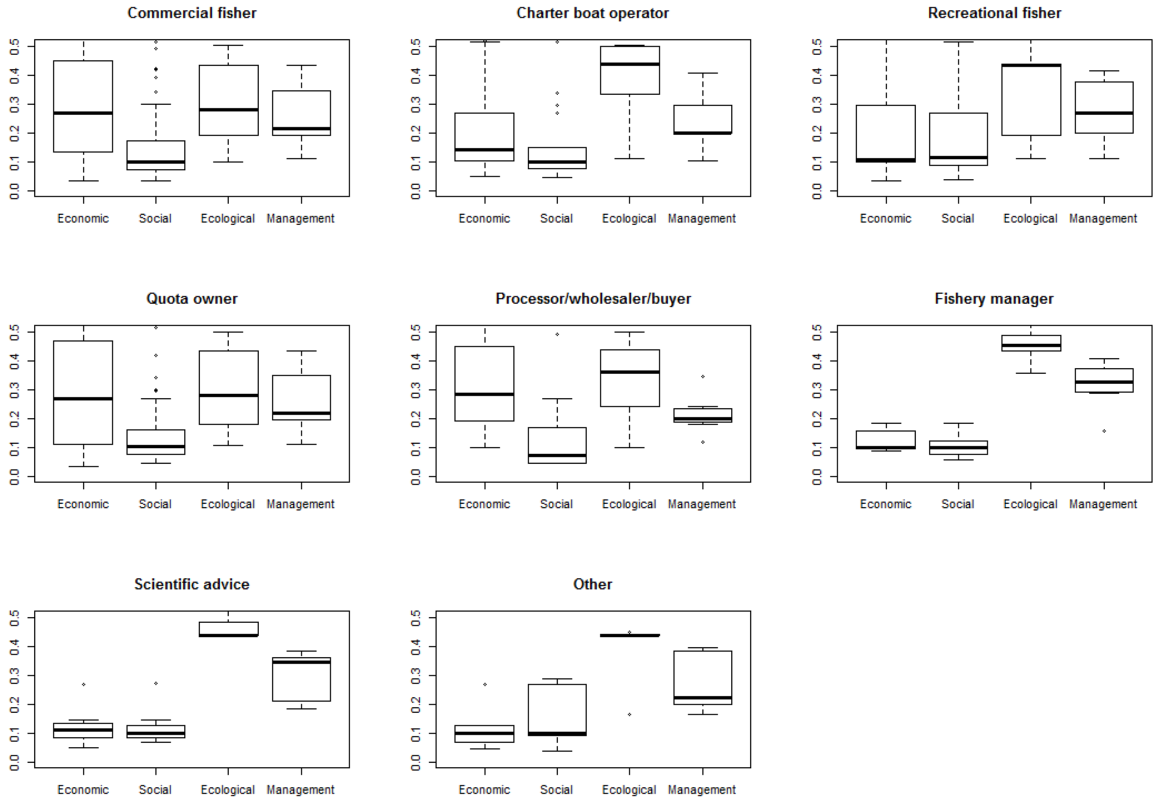


Figure 12. Objective weight distributions by group: modified AHP method

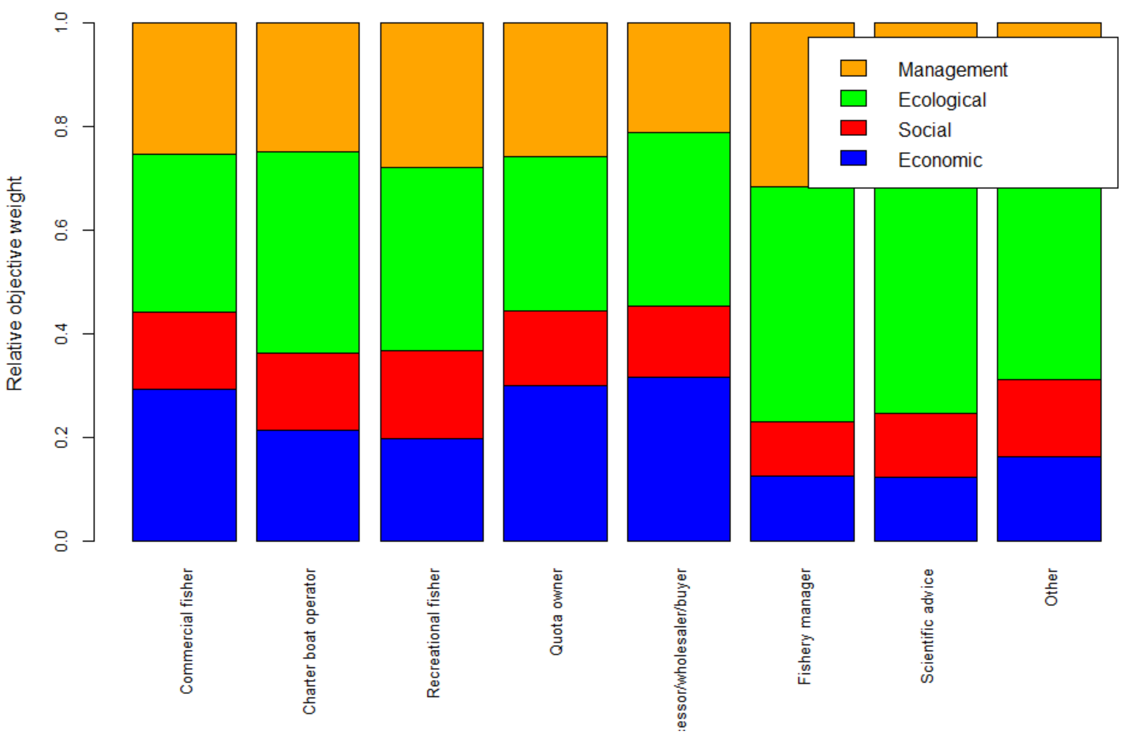


Figure 13. Average objective weight by group: modified AHP method

How do the approaches compare?

Comparing the relative distribution of objective weights in Figure 11 and Figure 13, the modified AHP method resulted in greater differences between groups than the scoring method, in which the relative weights were fairly homogenous across groups. From Table 9 and Table 10, the relative standard errors were generally lower using the modified AHP than the ratings approach. That is, the derived scores within a group were more consistent between group members when using the modified AHP than the scoring approach. This potentially reflects issues around interpersonal comparisons using the scoring approach. For example, one person may score an objective set (6,7,8), while another may score it (4,5,6), but the relative importance of the objectives is the same in both cases. The modified AHP approach would produce the same relative weight in both instances, while the scoring approach would produce different weights (hence greater variability between individuals).

The scores using the different methods are directly compared in Figure 14. The red lines in Figure 14 represent the 45 degree line (i.e. the line of equality between the two measures), while the black lines are linear regression lines between the observations. From this, the scoring approach tends to produce higher weights are the lower end, and lower weights at the upper end compared with the modified AHP approach. The range of estimated weights is also larger using the AHP approach (e.g., upper weights exceed 0.5) that the scoring approach, that tends to be limited to scores of 0.4 or less.

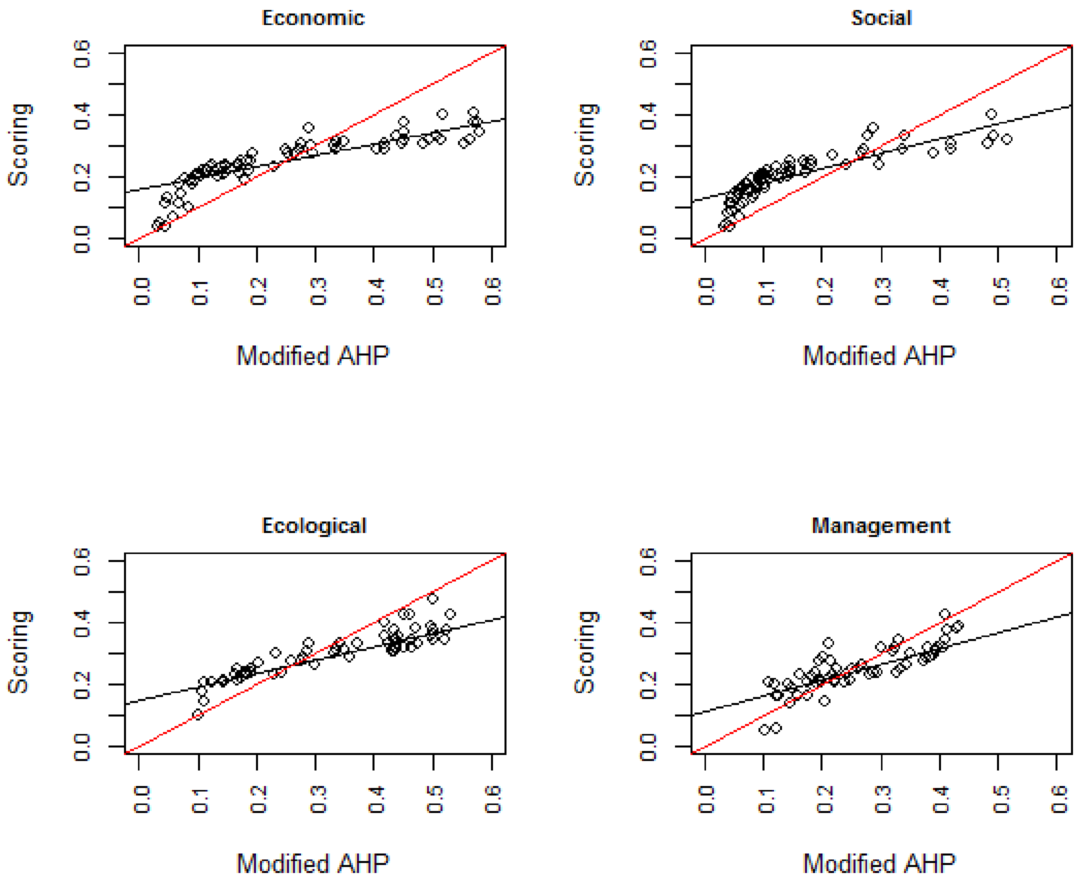


Figure 14. Comparison of weights from scoring approach and modified AHP method

These differences will potentially affect the estimates of the overall harvest strategy performance. Given that the modified AHP approach results in greater variability between the objectives (which will allow greater differentiation between the harvest strategy performance measures) but lower variability between individual respondents within the stakeholder groups (suggesting it is a more consistent measure), only this will be used for the final analysis.

Alternative harvest strategy options for the CRFFF

The key proposed modification underlying the “modified status quo” harvest strategy was the adoption of a target reference point for the stocks of 60% 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. Harvest control rules were developed to adjust the fishery-level TAC each year, where these changes are then applied proportionally to both the commercial TAC (TACC) and the recreational trip limits to maintain relative equity between the sectors. For Coral Trout species, the TAC is based on a stock assessment every 5 years and a suite of other indicators in the intermediate years. For Red Throat Emperor, the changes are proposed to be based on a risk assessment undertaken at least every 5 years, with empirical indicators used to adjust the quota and bag limits in the intermediate years. Similarly, for the other species (OS) component, the combined TAC would be retained, with both commercial and recreational catches adjusted proportionally in response to changes in the level of catch and catch composition. Within this cap, species considered “at risk” would potentially be subject to separate commercial caps and recreational trip limits.

Alternative harvest strategies identified by the Working Group were believed to further enhance one or more ecological, economic or social objectives. These are detailed below, but included: i) a separate allocation of the TAC to the charter sector (managed through vessel level possession limits, as opposed to being based on the number of recreational fishers they carried), aimed at enhancing the economic performance of this sector and social benefits accruing to the recreational fishers using these vessels; ii) the use of environmental “overrides” where TACs are adjusted in response to a spatially or temporally isolated weather event (e.g. a tropical cyclone) to enhance ecological outcomes and long term fishery performance; iii) the combination of environmental overrides and spatially explicit management, where responses to the catastrophic event may vary in different areas of the fishery and also to ensure some form of spatial equity across regional communities; iv) formally identifying separate TACs for a number of key OS quota species with ITQs and bag limits also allocated for these species; and, v) formally identifying separate TACs for a number of key Coral Trout species, again with separate ITQs and bag limits for each of these species for largely ecological benefits.

It was recognised that these alternative options would have greater benefits than the modified status quo against some objectives but potentially reduce benefits against others.

Modified Status Quo (Baseline)

The modified status quo harvest strategy option has three components:

i) the Coral Trout Modified Status Quo
ii) the Red Throat Emperor Modified Status Quo
iii) the "Other Species" Modified Status Quo.

These each attempt to retain the valuable parts of the current management approach, while identifying and attempting to fill any gaps.

i) Coral Trout (CT) Modified Status Quo

Description

For Coral Trout (Box 1) (Figure 15), the current formal stock assessment is retained, but the resulting global TAC is to be allocated between the commercial and recreational sectors, with the latter's share translated to a bag limit. In interim years, a suite of indicators will be evaluated to collectively infer stock status, and possibly make adjustments to the TAC. This replaces the current use of nominal CPUE time series with standardised CPUE, plus a range of additional "lead" and "lag" indicators. Finally, environmental events can trigger overrides such as spatial management or spawning closures.

Box 1: Coral Trout baseline harvest strategy

Every 5 years:

- Undertake a formal, model-based stock assessment (as per those currently undertaken). This will yield a global total allowable catch (TAC). This is then split into a commercial TACC, and recreational bag limits (or an alternative recreational control).

- This requires that we review the respective recreational and commercial catches, so that we can meaningfully allocate the TAC between the sectors.

In interim years:

- Use a suite of empirical indicators in a multiple-indicator framework (for example, in a decision tree) that collectively infer stock status. The empirical indicators may include:
    - Standardised commercial catch-per-unit effort (CPUE)
    - Recreational CPUE
    - Fishery-independent abundance
    - Environmental health indicators
    - "Lead" indicators (frequency of small fish; charter discards).
- The inferred stock status may results in proportional adjustments (with buffers) to TACC and recreational bag limits (or an alternative recreational control)

- In addition, and independent to the above, separate, environmental indicators may be monitored, and override triggers may be invoked that correspond to extreme environmental events, such as cyclones. Invoking these triggers may result in such measures as:
    - Spatial management
    - Spawning closures.

As part of this harvest strategy, other monitoring that may occur in addition to that currently undertaken may include:

- Boat ramp surveys (but note that it is not trivial to meaningfully compare these data with commercial catch data)
- Quota price
- Beach price
- Lease price.

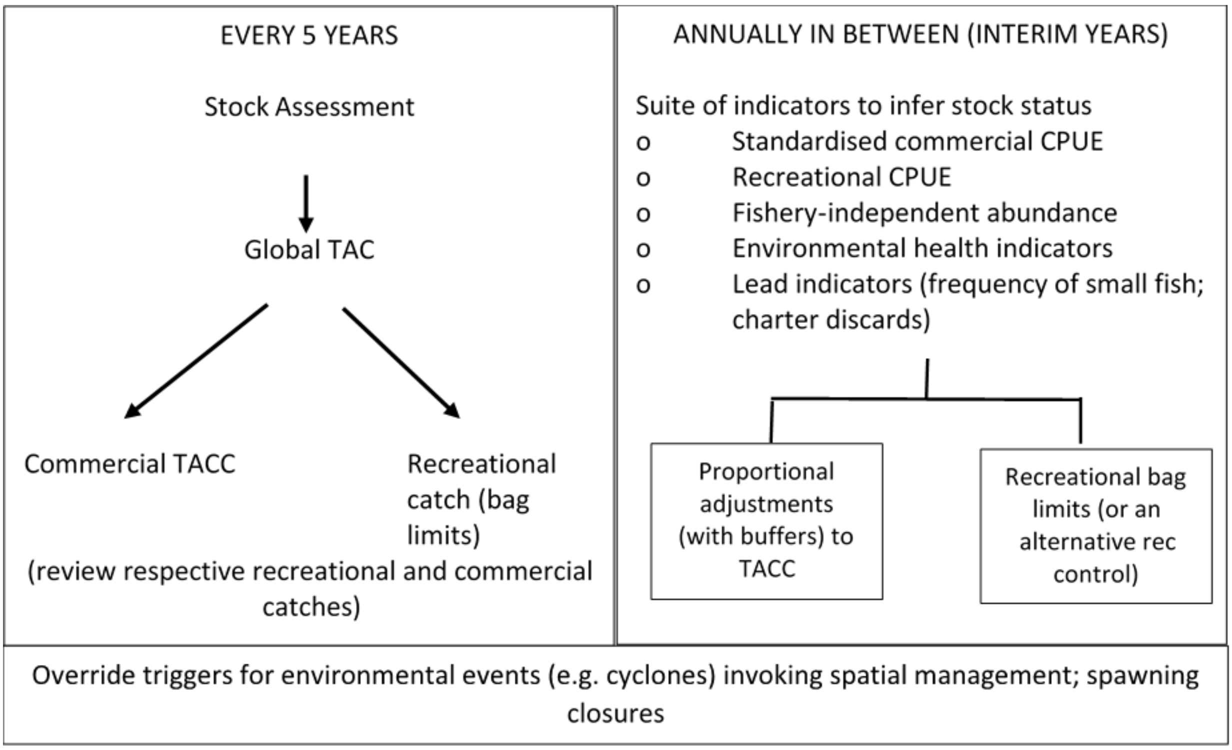


Figure 15. Summary of the modified status quo harvest strategy for Coral Trout

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Determine the relative allocation between the commercial and recreational sectors
- Translate the recreational allocation into a meaningful bag limit
- Determine the suite of interim empirical indicators (noting that we need both “lead” and “lag” indicators)
- Determine what magnitude of change in each indicator should equate to a trigger level
- Develop a multi-indicator framework, such as a decision tree, to determine what combination of indicator triggers should invoke an interim management response
- Determine the strength of the adjustment to the global TAC in response to the formal assessment and the interim decision tree
- Determine the environmental override triggers
- Determine the nature of response to the environmental override triggers
- Determine the strength of the response to the environmental override triggers
- Determine the criteria for ‘switching off’ the environmental override response.

ii) Red Throat Emperor (RTE) Modified Status Quo

Red Throat Emperor species are of most value to the recreational and charter sectors, and so management measures should be directed primarily at these sectors (Box 2).

While there is a lack of understanding of stock status for Red Throat Emperor due to limited data, its low-risk life-history justifies the modified status quo approach here described.

This strategy is underpinned by regular risk assessments, augmented by sets of triggers against a suite of empirical indicators. If the risk assessment returns a “harm” outcome, or one or more triggers are invoked, then an appropriate management response is triggered. Otherwise, status quo arrangements will apply.

If we obtain an updated formal Red Throat Emperor stock assessment, then the use and choice of empirical harvest strategy indicators would be reviewed.

To address the current lack of data, this harvest strategy also includes a commitment to ongoing “banking” of biological samples or data, to establish a time series against a possible future analysis.

Box 2: Red Throat Emperor baseline harvest strategy

Annual, or 5-yearly risk assessments
The risk assessments are augmented by triggers against empirical indicators:

- if the risk assessment outcome = "harm", or one or more triggers exceeded, management response triggered. Otherwise, status quo
- note that the empirical indicator triggers are needed less if the risk assessment is undertaken more frequently.

The empirical indicators may include, or be derived from:

- a catch-based assessment (focusing on dedicated OS fishers only, to eliminate non-targeted confounding)
- identifying decreasing standardised CPUE (focusing on dedicated OS fishers only, to eliminate non-targeted confounding)
- identifying changes in discards
- identifying changes in age structure (though this data is currently unavailable)
- identifying big jumps in catch per year (e.g. if markets opened up)
- identifying repeated years of poor catch in the charter sector
- tracking changes in habitat/reef health/oceanographic effects.

If an updated formal Red Throat Emperor stock assessment is undertaken at any stage, then the suite of empirical indicators would be reviewed.
There is an additional ongoing commitment to "banking" biological samples or data. These could include otoliths, age/length samples (e.g. obtained from cameras mounted over the filleting table). Sampling methods would have to be practical and affordable, ideally enabling biological data to be obtained for all species.

Sampling methods would have to be practical and affordable, ideally enabling biological data to be obtained for all species.

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Determine which risk assessment is to be used (e.g. productivity-susceptibility analysis)
- Determine the suite of empirical indicators
- Determine what magnitude of change in each indicator should equate to a trigger level
- Develop a decision tree to determine what combination of indicator triggers should invoke a response, and how or whether the strength of the response should scale with the number of triggers reached.
- Determine the strength of the adjustment to the global TAC in response to risk assessment "harm" response, and the empirical indicator trigger system

- Determine the relative allocation between the commercial and recreational sectors, noting that the commercial sector is secondary to the recreational sector when considering Red Throat Emperor
- Translate the recreational allocation into a meaningful bag limit.

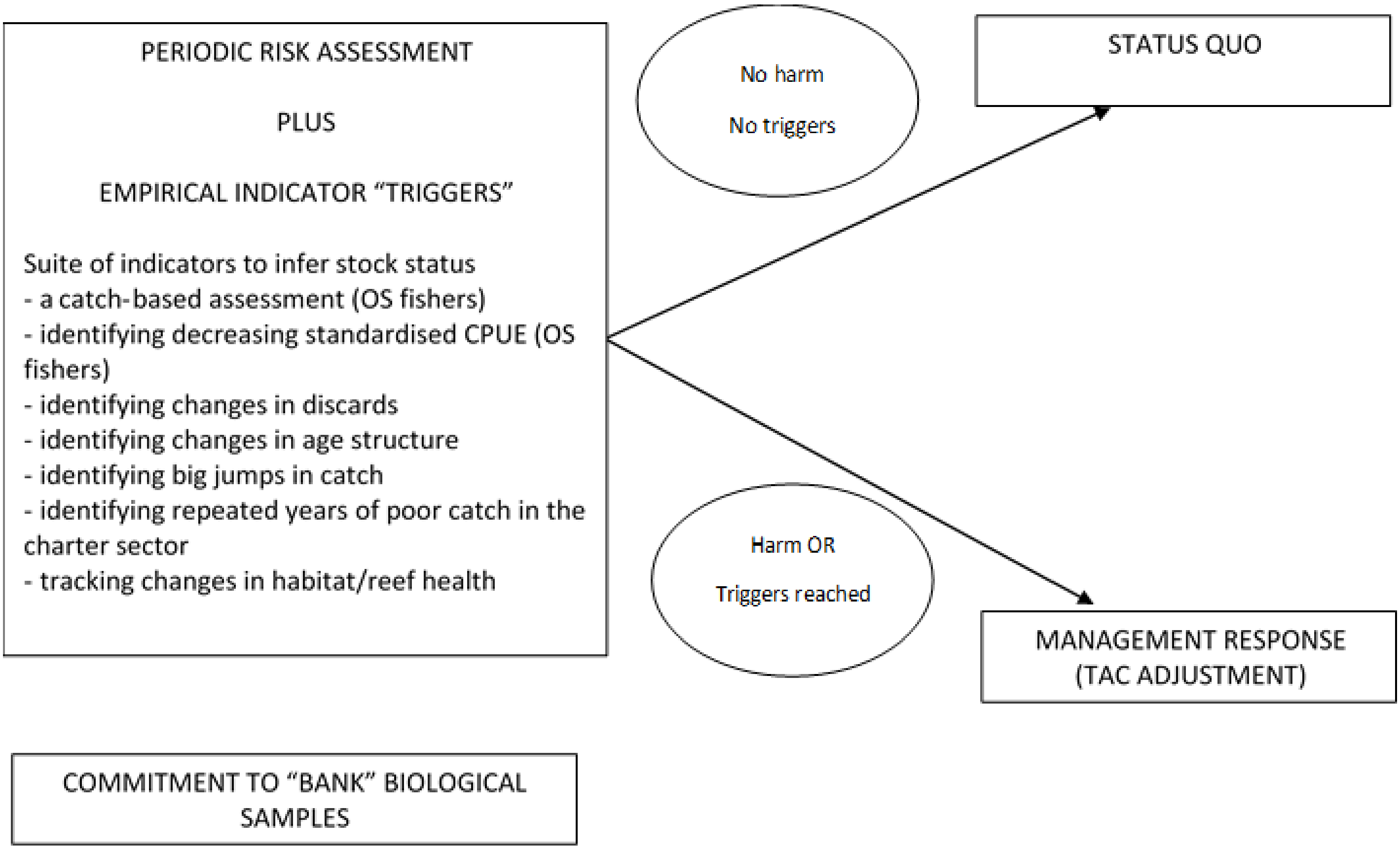


Figure 16. Summary of the modified status quo harvest strategy for Red Throat Emperor

iii) "Other Species" (OS) Modified Status Quo

There are 13 other species, including 5 cod species, which were identified as "high risk" by an Ecological Risk Assessment, and as such, must be appropriately and explicitly managed within the harvest strategy (Box 3).

This component is underpinned by a pragmatic, tiered approach, whereby the combined OS TACC is retained, unless a species is deemed "at risk", whether as one of the 13 currently identified "high risk" species, or because its higher relative level of catch warrants species-specific management. Species-specific ITQs and triggers can then be invoked to manage the species accordingly.

It should be noted that the existing Green Zones and the current, conservative fishery management centred around Coral Trout, already provides a buffer in affording indirect protection to these species.

It is recommended that a cluster analysis should be undertaken to better understand species profiles within the OS category, to make monitoring and analysis more efficient. For example, it is useful to understand whether the species is caught with Coral Trout or not, whether it is a predominantly commercial or recreational species, or spatial catch patterns.

Box 3: Other Species (OS) baseline harvest strategy:

The OS modified status quo component comprises a tiered approach:

1. At the bottom tier, the current, combined OS TACC would be retained, but recreational catch would also be considered. The combined TACC may be adjusted if the total OS catch approaches or exceeds it, or if there are major species compositional changes in the catch. If the recreational catch cap is exceeded, bag limits and/or seasonal closures should be invoked.

2. The next-level tier corresponds to “at risk” species. This involves applying commercial caps and recreational bag limits to those 13 “high risk” species.

   If these caps or bag limits are exceeded for any of those 13 species, OR if there any species whose catch exceeds a (pre-defined) proportion of the TACC, then a separate commercial ITQ will be applied to those species, and bag limits will be adjusted.

3. A system of commercial triggers, based on empirical indicators, will also apply to species considered to be “at risk” (noting that CPUE may not be a useful indicator). These should be structured such that they would be invoked at catches below their corresponding ITQ value. If these triggers are reached, then management responses such as move-on provisions, decrementation, or commercial trip limits may be invoked.

It is noted that TACCs or caps or catch may incite a “race to fish”. The 3rd tier commercial triggers described above should help mitigate against this, provided that appropriate indicators are monitored.
Recreational bag limits should possibly be set by weight instead of numbers to discourage high-grading.

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Undertake a cluster analysis of OS species-specific catch compositions
- Determine the relative allocation between the commercial and recreational sectors
- Translate the recreational allocation into a meaningful bag limit
- Determine whether bag limits should be by number or weight
- Determine indicators for changing the OS TACC
- Determine what magnitude of change in each indicator should equate to a trigger level

- Develop a decision tree to determine what combination of indicator triggers should invoke i) a change in the OS TACC, ii) determine whether a species should be moved to its own commercial ITQ, and iii) should invoke a management measure against an "at risk" species
- Determine which current at-risk species should have their own catch caps
- Determine the values for the commercial caps and recreational bag limits for these at-risk species
- Determine the type of management response to triggers and caps being reached
- Determine the strength of the management response to triggers and caps being reached.

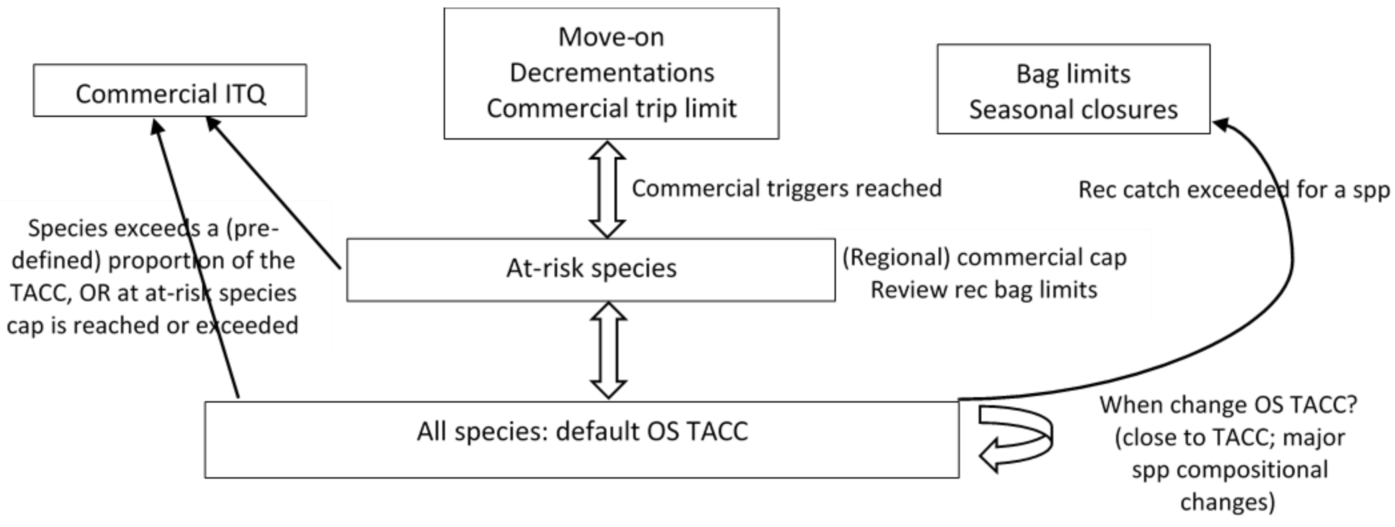


Figure 17. Summary of modified status quo harvest strategy for the "other species" group

Modified Status Quo PLUS a separate charter allocation

The harvest strategy is as per the Modified Status Quo, but with additional elements pertaining to the charter sector. These are not specific to any species, and pertain only to the charter operators. They acknowledge the charter sector's objectives for how they would prefer to be managed.

These elements include treating the charter operators as a separate industry or sector, independent of the remaining recreational sector.

The most important requirements for charter operators are i) the availability of good quality fish, and ii) big enough bag limit for trips to be attractive to clients. A concern of charter operators would be if bag limits were to drop so low that their business was threatened. (Although it is likely that a stock status corresponding to such a reduction in recreational bag limits would also significantly impact the commercial sector, commercial operators have the option to consolidate, whereas charter operators would be put out of business).

Additionally, bag limits do not protect charter operators from an influx of new operators; however, having their own TAC and allocation would protect them.

Box 4: Modified status quo harvest strategy with a separate charter sector allocation:

This comprises:

- A possession limit but no individual bag limit for charter operators.
- Controlling individual catch by charter clients through within-season changing of size limits.

Possession limits arrangements explicitly for the charter operators would achieve an implied TAC.
An additional rule would be that bait must be sustainably sourced.
To acknowledge the crossover with the recreational sector (given that charter clients are classed as recreational fishers), dockets or tags would be issued to recreational fishers on board.

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Treat charter operators as a third fleet, distinct from the recreational sector
- Determine charter operator possession limits (no TAC)
- Determine the control rules for changing charter size limits
- Consider how to roll this into the modified status quo options above – i.e. does this harvest strategy option occur as a subset of having an overall TAC? How do we account for charter catch when determining a sustainable overall TAC for the commercial and recreational sectors?

Modified Status Quo PLUS environmental overrides

The harvest strategy is as per the Modified Status Quo, but with the additional option to uses "override" triggers to detect and respond to significant environmental change, be these long-term, such as climate change, or catastrophic, such as cyclones (Box 5). The term "override" is used because the management response to such environmental changes can override other potential responses based on estimates of stock status. This directly addresses sustainability objectives by detecting and responding to changes or events that are not explicitly incorporated in most of the species-centric harvest strategy options.

When considering long-term effects, such as climate change, the key point is that the fishery may no longer be resilient to factors that have hitherto not been problematic. There is therefore a need to build in adaptivity to long-term changes.

Also, this is about being pre-emptive and responsive rather than reactive. Also, as our understanding of relationships (between, for example, climate and fisheries) improves, this will hopefully be more directly addressed within stock assessments. But in meantime, this is about being pre-emptive in avoiding decline or promoting a more rapid recovery.

This should be flexible enough to be adaptive to incorporate new knowledge as this becomes available. i.e. build in appropriate review timeframes with this as an explicit purpose.

Box 5: Modified status quo harvest strategy with environmental overrides

A suite of indicators would be used to identify both isolated, catastrophic events, and to detect long-term "drift". Against each of these two categories, trigger values would be used to invoke management responses.
In establishing indicators and triggers, the following points should be noted:

- The strength and extent of cyclonic impacts are specific to each cyclone
- Analysis of fleet dynamics responses to cyclones should be undertaken to inform appropriate management responses
- Long-term bleaching/reef health/flooding needs to be tracked, and this is regionally specific
- GBRMPA and science partners are happy to provide real-time habitat impact data.

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Determine what indicators will be used to identify
  - Catastrophic impact (cyclone watch, other?)
  - Long-term bleaching /reef health
- Determine how suites of indicators will be used in a "trigger" based assessment
- Specify the type of management responses (by appropriate spatial units?) for short- and long-term events
- Specify the strength of management responses for short- and long-term events
- Consider how resilience should be built into all other decision rules – i.e. how do we build in adaptivity to long-term change, across the board of harvest strategies?
- Specify how to "turn off" short-term override conditions.

Modified Status Quo PLUS environmental overrides AND spatially explicit management

This harvest strategy builds on that proposed as option 3 above, in that, in addition to the option for environmental overrides, there is the option to enable management measures to be spatially-specific, in accordance with the event or indicator that has triggered that response. Alternatively, this could involve undertaking assessments on a region-specific basis, but management responses occurring at a fishery-wide level.

This option is not necessarily specific to any species and may be considered as a common approach across species, or as something to be invoked on an as needed basis. It enables management to be responsive to local events or to localised depletion.

Note that this option is distinct from (does NOT include) having regional ITQs and TACs.

Box 6: Modified status quo harvest strategy with environmental overrides and spatially explicit management

This option could involve:

- Invoking management responses at spatially-temporally appropriate scales, given the event or indicator that has triggered the response. Forms of management measures could include regional temporary closures, or effort caps. At the same time, flexibility needs to be granted to fishers to enable them to accommodate local responses to events.

- Identifying spatially-explicit assessment regions, with region-specific indicators. If indicator trigger levels are reached, the overall TACC and bag limits could be reduced to reduce pressure on the stock.

Spatial management measures may have to consider what levels of flexibility may need to be afforded to industry, should measures excluded them from an area in which they frequently fish.

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Determine and specify region-specific indicators and responses
- Alternatively, determine and specific spatially explicit assessment regions, whose outcomes invoke whole-of-fishery responses.
- Consider how this form of management could be incorporated into the modified status quo harvest strategy options.

Modified Status Quo for Coral Trout and Red Throat Emperor, but with split TACs for “Other Species”

This option retains the modified status quo options, inclusive of most of the species within the “Other Species” category. However, under this option, 6 species currently within the “OS” basket would have their own individual ITQs. As stated, the remainder would be managed as per modified status quo approach for “Other Species” as described above. The six species are among those 13 identified as “high risk” from the Ecological Risk Assessment.

Box 7: Modified status quo harvest strategy for Coral Trout and Red Throat Emperor, but with split TACs for "Other Species"

Individual ITQs would apply to the following species: - 3 reds - Stripeys - Spangled - Bar cod. The remainder of the OS species would be managed via the modified status quo approach described in option 1 above. A key issue with increasing the number of species managed by ITQs is that the risk of discarding significantly increases, due to fishers attempting to reach their quota for certain species inadvertently catching other species whose quota they have already achieved. Species whose quota is reached such that this invokes changes in fisher behaviour (in attempting to actively avoid them while still seeking quota for other species), are termed "choke species".

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Effectively address the outstanding issues per the modified status quo options above, PLUS
- Determine the ITQ values for 6 species
- Consider how discarding and fleet dynamics (in terms of "choke" species effects) may play out.

We should also retain as a broader management regime objective the key aspect of an eliminated option raised in the CRFFF WG meeting, which was to educate fishers to change their behaviour.

Modified Status Quo, but with the additional CT species explicitly considered

This option retains the details of the modified status quo options. However, it would consider the various species of Coral Trout (CT) individually, as opposed to the current practice of lumping them together in a broad "Coral Trout" category. Under this option, we would consider the 5 Coral Trout + 2 other (different genus) species individually. The approach described for the Coral Trout Modified Status Quo would apply to data-rich Coral Trout species, but others would be managed under separate ITQs, while acknowledging these would not necessarily be underpinned by data-rich assessments.

This approach would use an ecological risk assessment as a first step, supplemented by empirical indicators, to identify individual species requiring explicit management.

Box 8: Modified status quo harvest strategy, but with the additional CT species explicitly considered

As a first step, an Ecological Risk Assessment (ERA) will be used to identify species as susceptible to harm. Meaningful empirical indicators will be used to supplement the ERA, as per the **Red Throat Emperor Modified Status Quo** approach.
If a species is deemed to be susceptible to harm per the ERA, or at risk according to the empirical indicators, that species will be pulled out for management on an individual basis, as per the **OS Modified Status Quo** approach.
The **main Coral Trout species** captured would be (individually) subject to management described under the Coral Trout modified status quo component.
When considering the choice of empirical indicators, the following points are relevant:

- Monitoring dead vs alive Coral Trout as a time series should be undertaken no to invoke a decision rule, but to provide context in allowing managers to anticipate changes to the nature of the fishery.
- Commercial catch will need to be reported by species (as non-retained fish are not reported, logbooks may be changed to include discard reporting as a requirement).
- As the various Coral Trout species have different relative spatial distributions, catch compositions will vary spatially.

This approach could be rolled in with the Red Throat Emperor (RTE) Modified Status Quo approach, noting, though, that RTE availability increases after cyclones, so we would need to be mindful of this when increasing or reducing quota in any combined approach.
In order to implement a species-specific Coral Trout harvest strategy, there would be increased monitoring requirements around species-specific identification and reporting. Currently the only available data is from boat ramp surveys. Possible monitoring approaches include

- Voluntary recreational reporting
- Crowd fundraising? Frames? Via Tackle Shops?
- Data-banking – "Train the Trainers".

Outstanding issues

In order to effectively design and implement this harvest strategy option, we would need to:

- Articulate ERA and indicators, as per the RTE Modified Status Quo
- Determine indicator thresholds for pulling out individual CT species, as per (3) OS Status Quo
- Determine how to set individual CT ITQs and management measures in response to empirical indicator triggers
- Explicitly articulate monitoring requirements
    - Commercial catch by species
    - Discard reporting
    - Dead vs alive status

- Recreational reporting (voluntary, crowd-funded, tackle shops etc.).

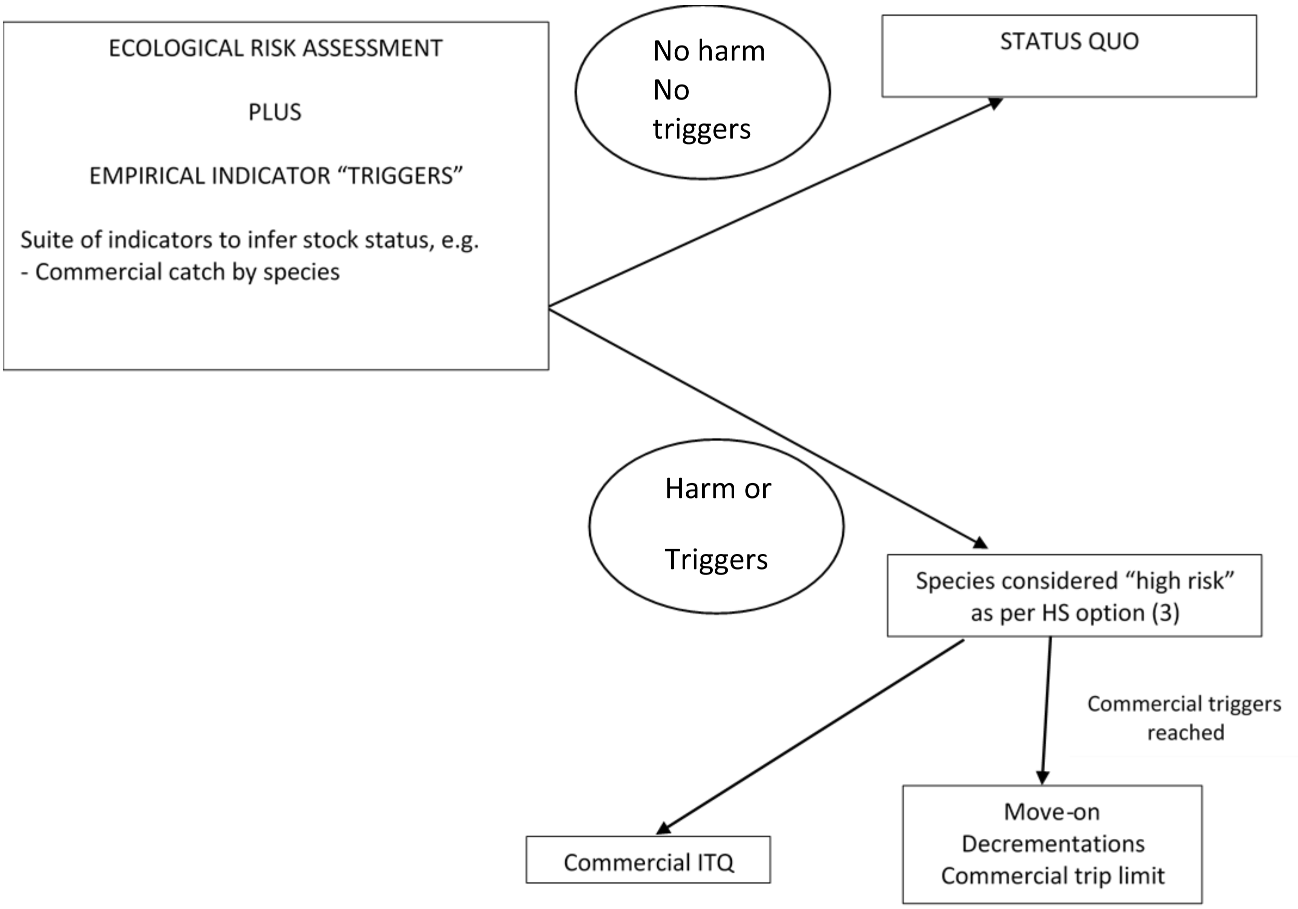


Figure 18. Summary of harvest strategy option where the additional Coral Trout species are explicitly considered.

A Multi-Criteria Decision Analysis approach to determining and evaluating TBL harvest strategies

Performance of harvest strategies against the objectives

From the first survey of management objective importance, 48 respondents expressed a willingness to participate in a follow up survey. These were sent a second questionnaire asking them to assess each of the potential harvest strategies against each of the objectives. The survey was only open for a week, during which 28 responses were received. From these, a total of 18 useable responses were received, with the other respondents not providing more than just details of their involvement with the fishery. The distribution of the usable responses by stakeholder group is shown in Figure 19. Two thirds (12) of the usable responses were from working group members, the remainder from individuals not involved in determining the harvest strategies.

The expected impact of each harvest strategy option against the objectives is shown in Figure 20. The charter sector allocation is generally expected to provide benefits against nearly all objectives, with the exception of the discard mortality (Ecol1.2.2) and indigenous economic benefits (Econ2.5) objectives. The additional of environmental overrides is generally considered to have ecological sustainability benefits, but potentially negative consequences for the economic objectives. Social outcomes are expected to be mixed. The addition of spatially explicit control rules and environmental overrides is expected to produce similar pattern of impacts with respect to ecological and economic outcomes as the previous scenario, although the magnitude of some of these impacts is greater.

Separating out some of the OS species and applying TACs to these is expected to potentially have more substantial ecological benefits, with a mixed result in terms of economic objectives. In contrast, separating out the Coral Trout species into separate TACs is expected to have little or no ecological benefit, but economic costs.

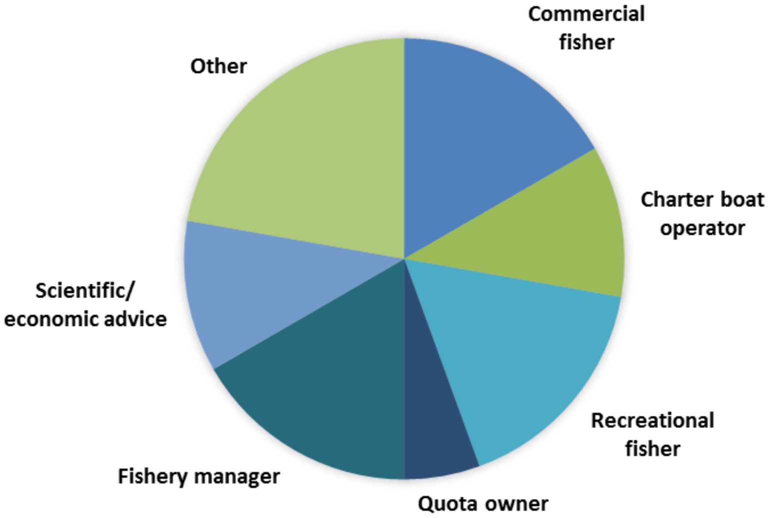


Figure 19. Composition of usable responses from the impact survey

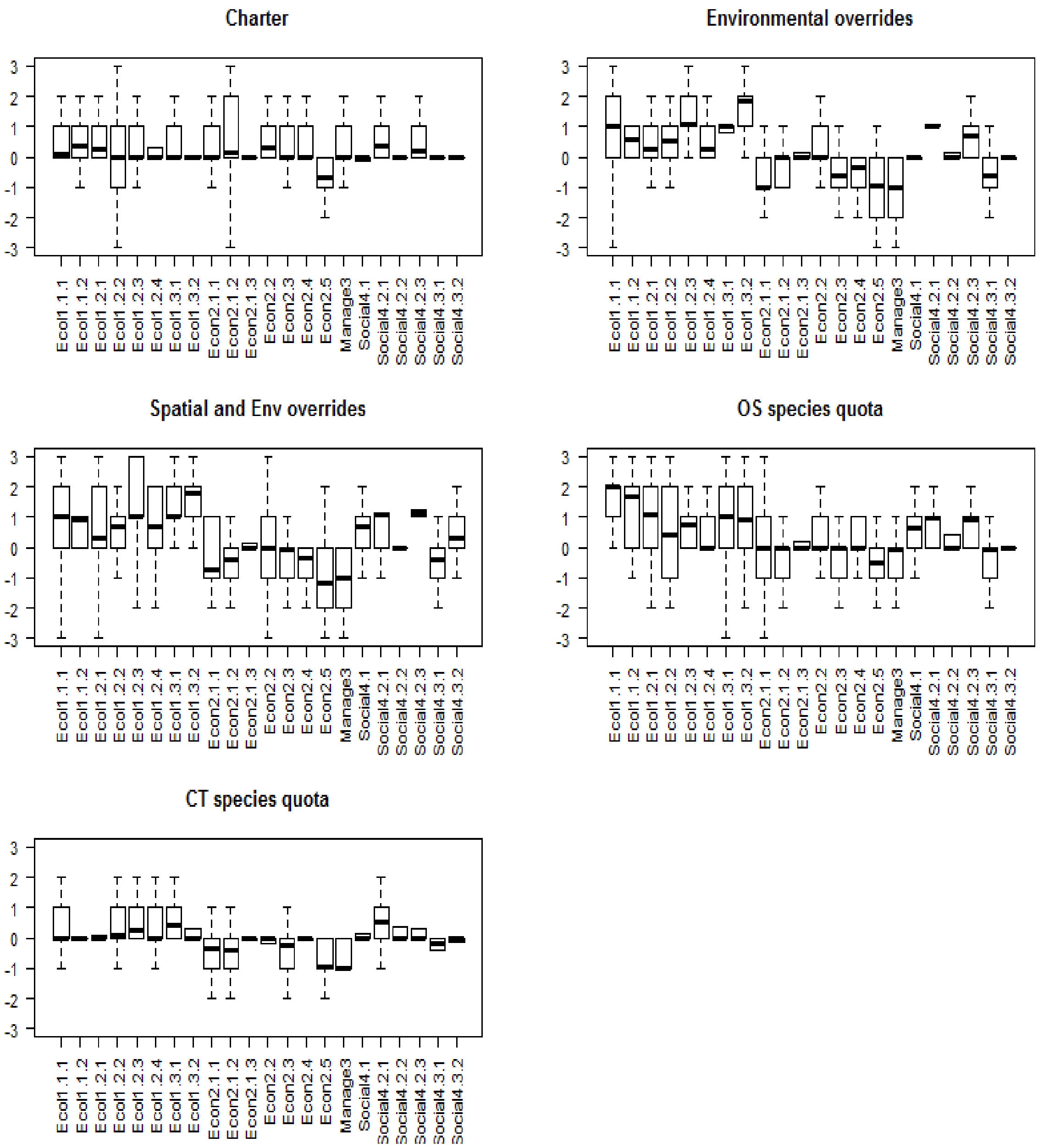


Figure 20. Distribution of the impacts against the management objectives

Overall performance measures of each of the harvest strategies

The derived impact matrix was multiplied by the matrix of objective weights to produce a matrix of potential weighted outcomes from each of the management options. The distributions of these outcomes are illustrated in Figure 20. The cumulative probabilities in Figure 21 represent the likelihood that a harvest strategy alternative would perform better than the baseline harvest strategy alone.

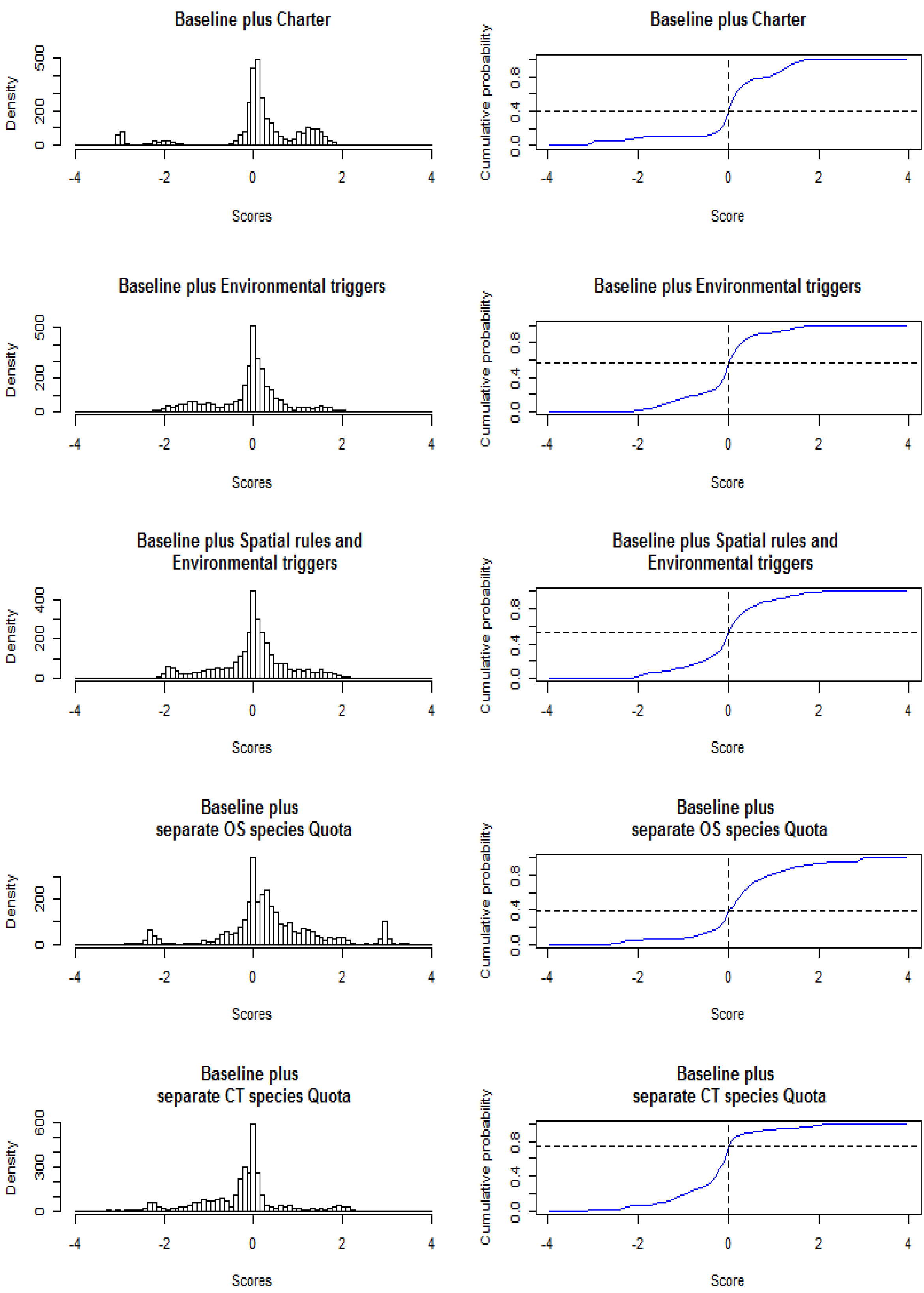


Figure 21. Weighted impact distributions

The results were also considered by Stakeholder group (totalling the 28 survey respondents) to determine if there were substantial differences of opinion in terms of the relative merit of particular harvest strategy additions. The average score for the charter allocation option was small but positive for all stakeholder groups (Table 11). Similarly, the average score for the split OS quota was also positive across all stakeholder groups. This suggests that these options would be beneficial from the perspective of all groups (on average). In contrast, the split CT quota was negative for all groups, suggesting it is not a desirable option.

Environmental overrides were generally negative on average for the commercial and recreational sectors, but positive for the manager, scientist and other groups with a stake in the fishery (Table 11). Adding spatially explicit control rules improved the average score for all stakeholder groups, although the overall average was still negative for quota holders and commercial fishers.

Table 11. Average net "score" by stakeholder group

Stakeholder group	Charter	Environmental overrides	Spatially explicit control rules and environmental overrides	Split OS quota	Split CT quota
Commercial fisher	0.06	-0.13	-0.06	0.28	-0.28
Charter boat operator	0.09	-0.02	0.07	0.37	-0.23
Recreational fisher	0.09	-0.07	0.02	0.34	-0.26
Quota owner	0.06	-0.15	-0.07	0.28	-0.29
Processor/wholesaler/buyer	0.09	-0.02	0.04	0.38	-0.24
Fishery manager	0.10	0.03	0.08	0.49	-0.23
Scientific/economic advice	0.11	0.02	0.09	0.46	-0.22
Other	0.10	0.00	0.08	0.40	-0.24

An alternative to considering the average score is to consider the proportion of outcomes with a negative score (Table 12). Generally, the charter allocation and split OS quota produced negative outcomes on fewer than 35% of cases, whereas the split CT option resulted in negative scores more than 57% of cases.

Table 12. Proportion of estimates with a negative value by stakeholder group

Stakeholder group	Charter	Environmental overrides	Spatially explicit control rules and environmental overrides	Split OS quota	Split CT quota
Commercial fisher	0.30	0.51	0.48	0.31	0.64
Charter boat operator	0.29	0.37	0.35	0.29	0.59
Recreational fisher	0.29	0.45	0.39	0.32	0.61
Quota owner	0.29	0.53	0.49	0.31	0.64
Processor/wholesaler/buyer	0.33	0.34	0.38	0.25	0.58
Fishery manager	0.35	0.33	0.32	0.27	0.59
Scientific/economic advice	0.31	0.36	0.34	0.25	0.57
Other	0.30	0.36	0.34	0.29	0.58

The results in Table 12 can also be presented visually. In Figure 22, the darker the colour the greater the proportion of expected negative outcomes. In contrast, lighter colours indicate fewer expected negative outcomes.

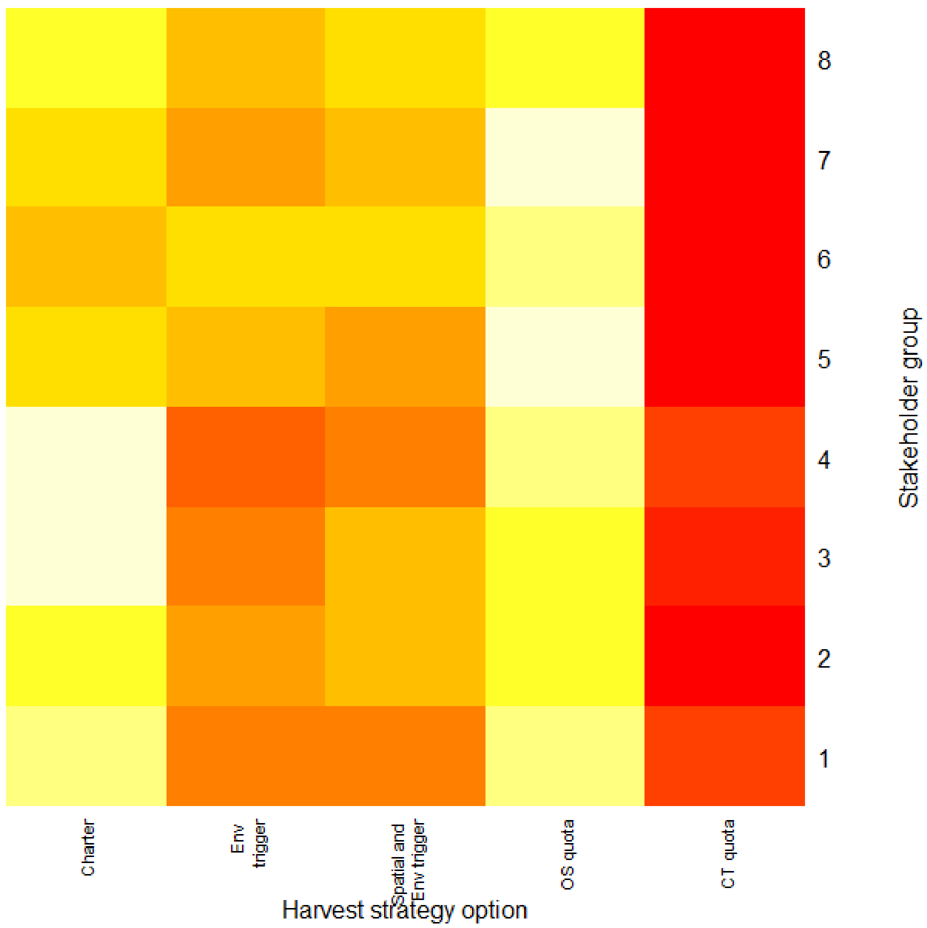


Figure 22. Heat map of the proportion of estimates with a negative value by stakeholder group

A quantitative simulation model approach to evaluating TBL harvest strategies

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 there was an increase in no-take areas (Figure 22). 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.

Key scenarios

For each scenario, we present time series of total catch (Figure 23) (species-specific catch time series are also provided in Figure 26), total final biomass (Figure 24) (biomass time series are also provided in Figure 27) for each species group, as well as the mean of each of the 21 performance indicators, taken across the 25 projection years (Figure 25) (means with standard deviations are also provided in Figure 28).

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 23). 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 24). 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 25). 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 23) such that RTE biomass achieved its target (Figure 24). 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 23). 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 24). 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 25). 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 25). 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 23), 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 24). 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 25). 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 23). 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 24). 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 25). 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 23, Figure 24). Recall that we simulated a cyclone in the 5th 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 23).

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 24). There was no effect on overall catch or biomass, nor most of the performance indicators (Figure 25). 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 23) (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 25) 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.

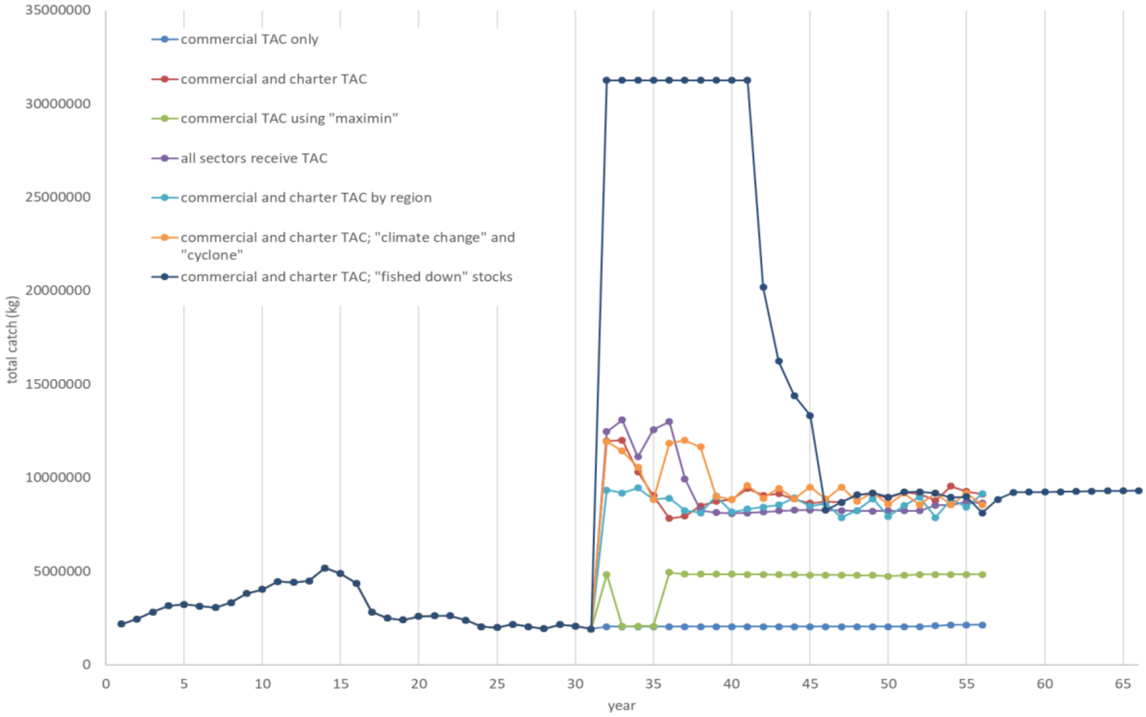


Figure 23. Time series of total catch (kg) summed across each species group, for each scenario considered.

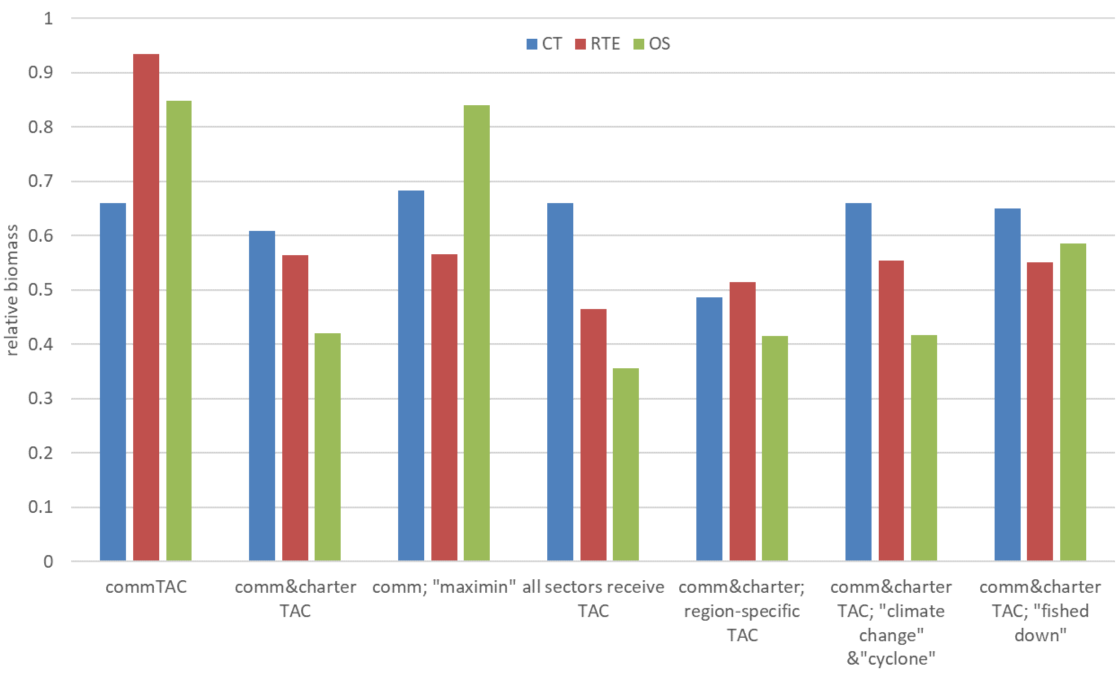


Figure 24: Barplot of final year biomass, relative to the initial year, for each species group and scenario considered.

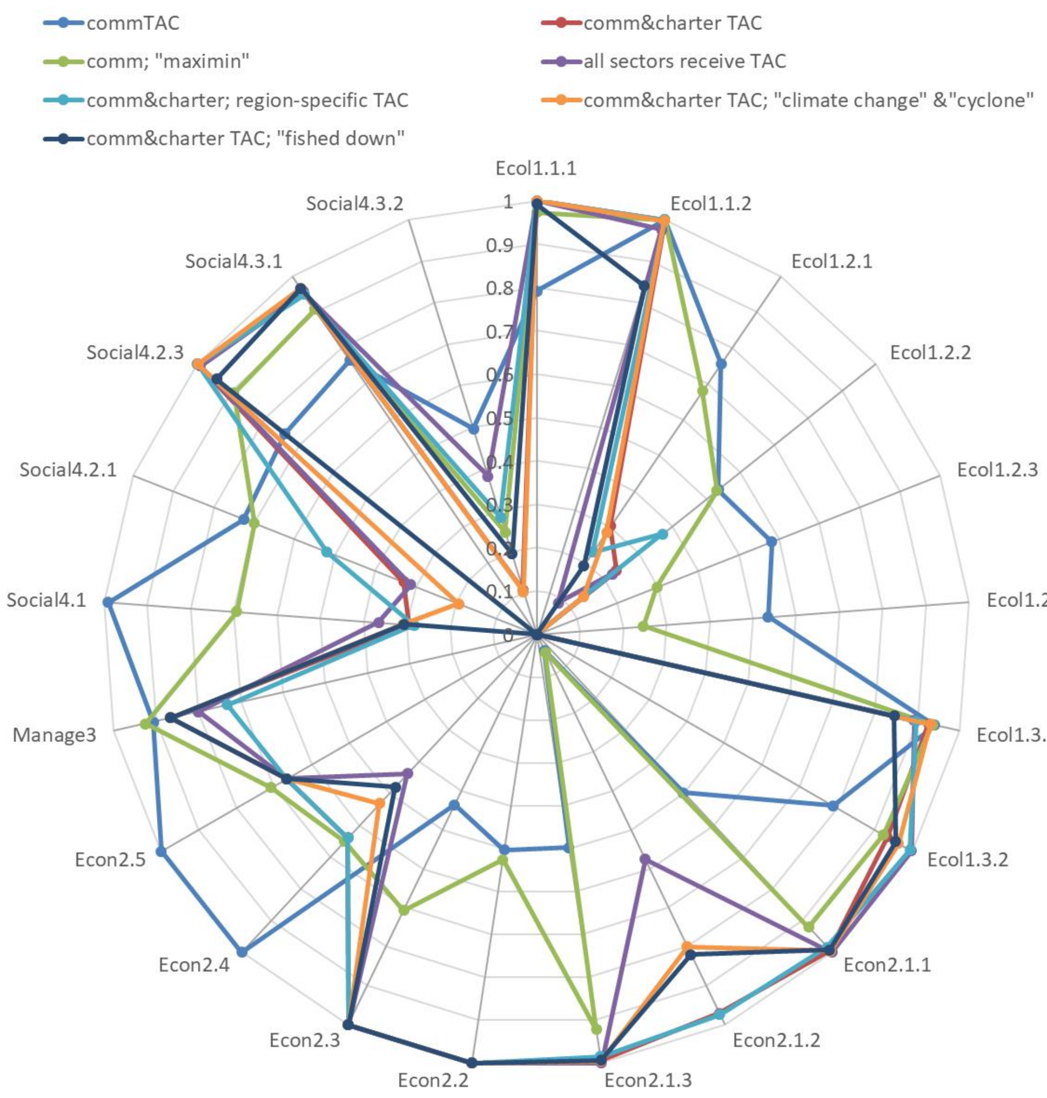


Figure 25: Radar plot of mean value across the projection years, for each of the 21 performance indicators, for each scenario examined.

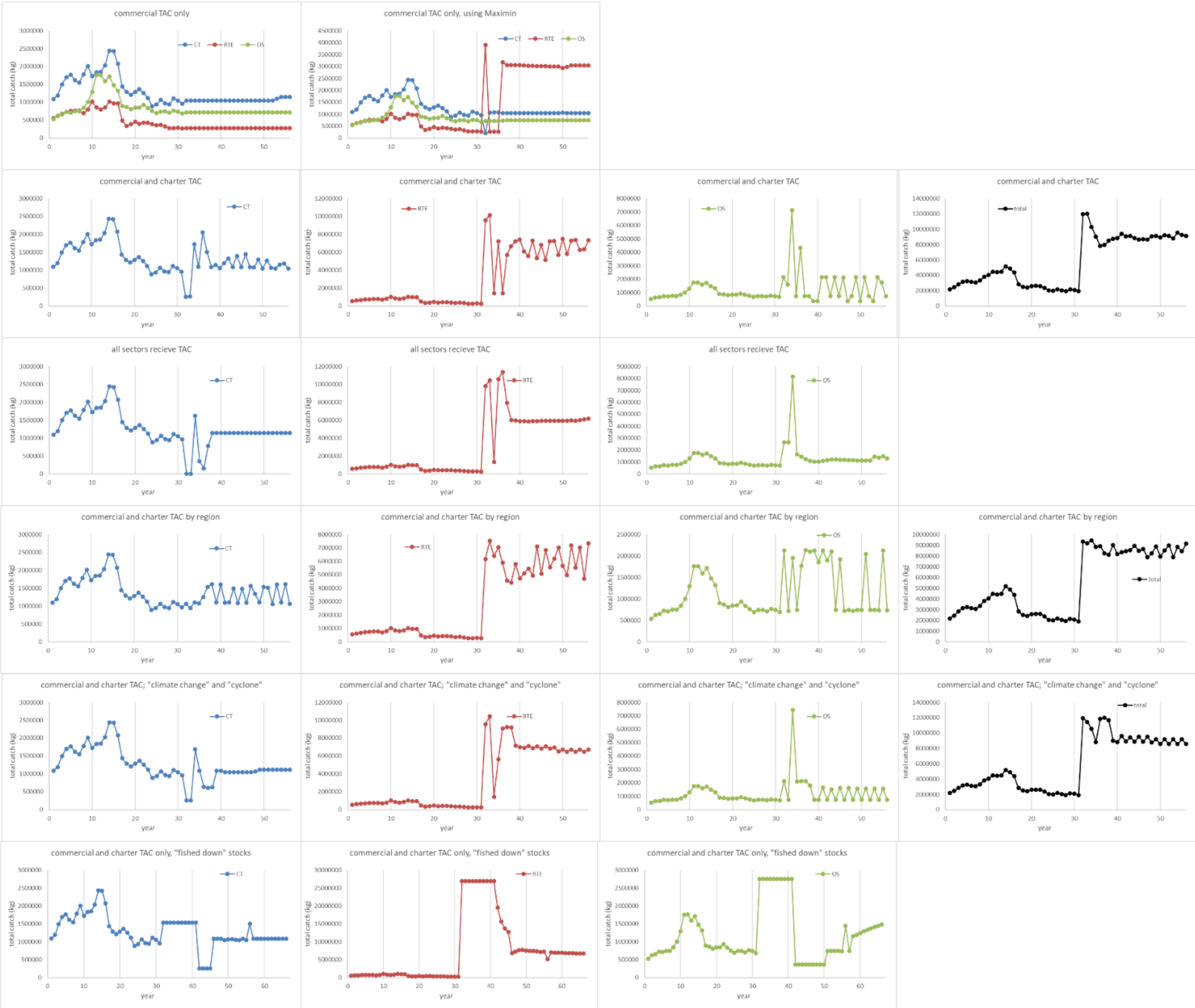


Figure 26: Time series of total catch (kg) for each species group (columns) and scenario (two scenarios in first row; one scenario per row thereafter) considered. For some scenarios, the time series are presented in individual panels for each species, due to differences in magnitude precluding ease of reading if these were overlaid.

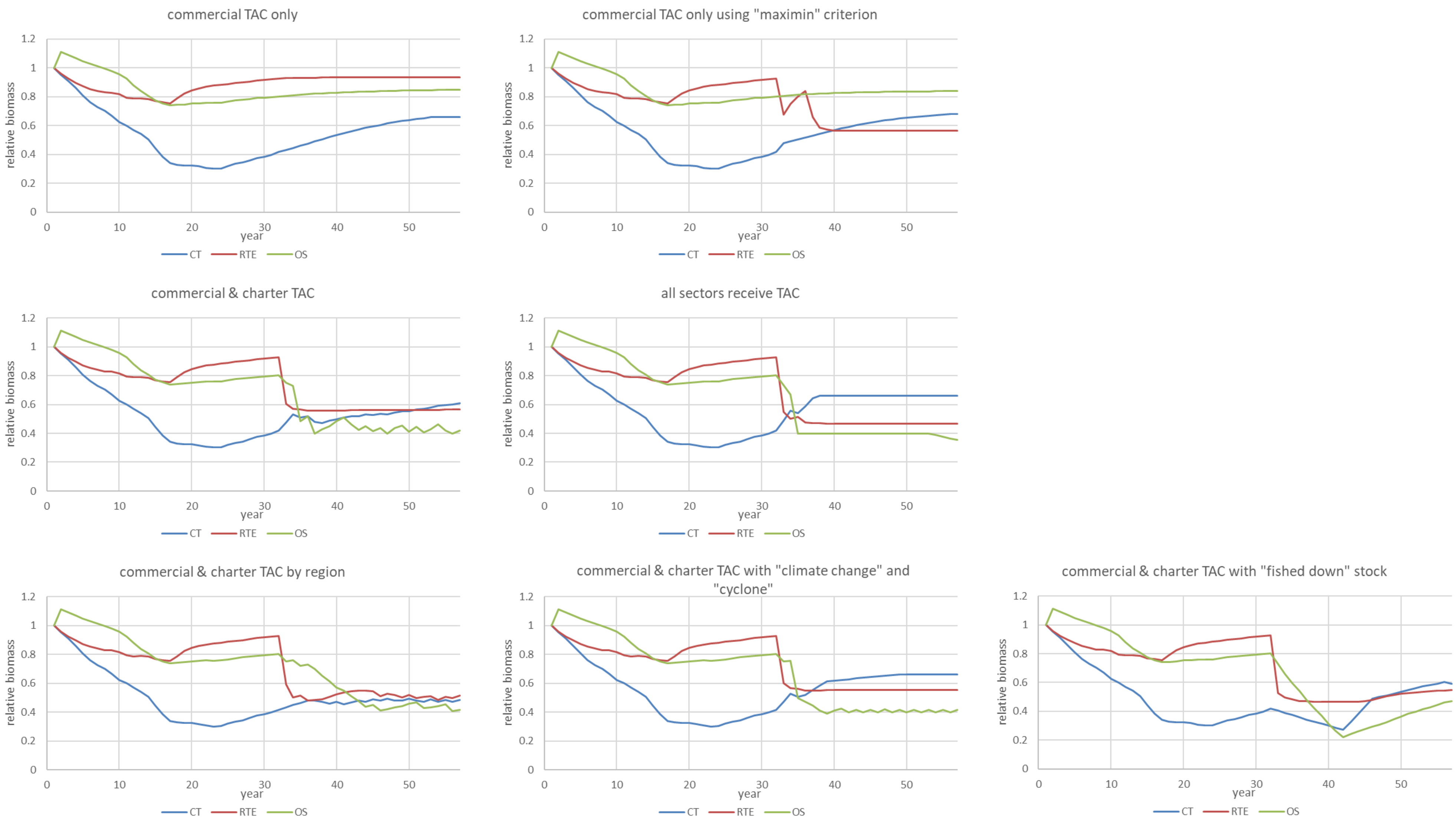


Figure 27: Time series of biomass, relative to the initial year, for each species group and scenario considered.

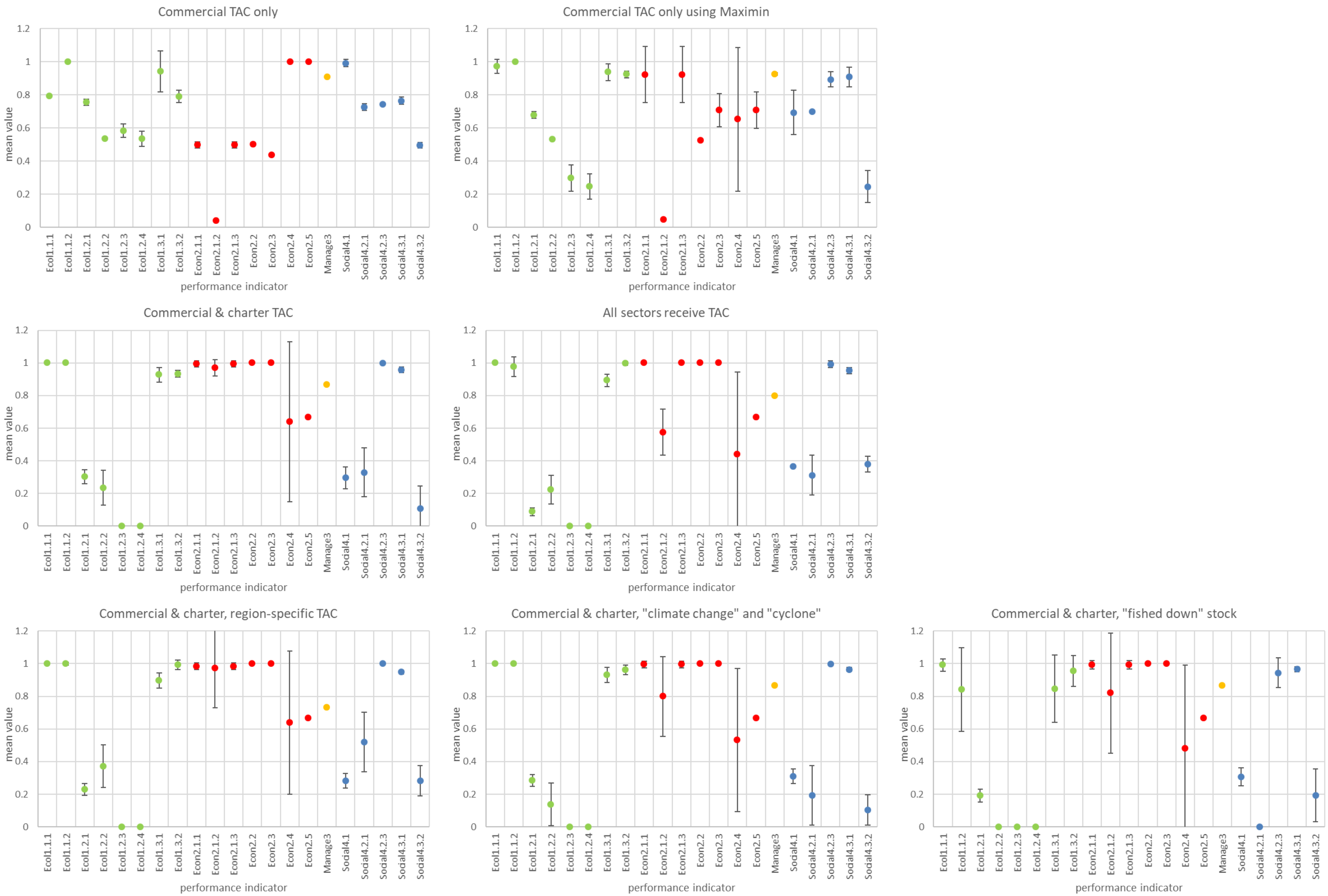


Figure 28: Mean, plus and minus one standard deviation, of each of the 21 performance indicators, for each scenario examined.

Comparing the MCDA and simulation approaches

A comparison of the results against each of the separate objectives from the MCDA and simulation model was undertaken using the scenario where TAC was allocated separately to both the commercial and charter sectors. This is the "commercial and charter" harvest strategy, or Scenario 2a (Appendix Table E2) for the quantitative model. The results are presented in Table 13 and Figure 29. The simulation model produces continuous outcomes against each objective (ranging between -1 and 1). The MCDA involves discrete and categorical outcomes ranging between -3 and 3, although the scores were normalised (divided by 3) to produce a similar range of outcomes as the model scores for comparison. The mean outcomes, although less meaningful for the MCDA outcomes, are presented in Table 13 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 each objective from the two approaches.

The outcomes from the two approaches were significantly different for the majority of objectives (Table 13), although in most cases moved in the same direction (i.e. improved or made worse). The significantly different outcomes (18 of the 21 objectives compared[11]) are identified in Figure 30. The upper right-hand quadrant of Figure 30 represents objectives that were estimated to improve or remained the same in both approaches (i.e. not decrease in both approaches). Ten of the 21 objectives fell into this category. Similarly, the bottom left quadrant of Figure 30 represents objectives that were estimated to decrease or remained the same in both approaches (i.e. not increase in both approaches). Five of the objectives fell into this category. Five of the remaining six objectives were found in the bottom right quadrant, with the MCDA approach suggesting an improvement in outcomes and the model suggesting a decline relative to the status quo.

The distributions of the outcomes were more dispersed in the MCDA results that the model results (Figure 29), although the medians (Figure 29) and mean values (Table 13) were small, suggesting only small improvements or decreases on average. The MCDA expert-judgement based outcomes were generally more optimistic against most objectives than the simulation model. Outcomes of only three objectives were expected to decrease in the MCDA (i.e. top and bottom left hand quadrants in Figure 30). In contrast, 11 of the 21 objectives were estimated to decrease in performance from the model (Table 13).

Determining which outcomes are correct is not possible. There are uncertainties in each approach that are inherent to their methodology, and many unquantified biases which makes an unbiased comparison almost impossible. While in principle "objective", simulation models are based on 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 requires assumptions about the links between other modellable fishery outcomes (e.g. bycatch, income distributions, fleet structures, etc.) 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

[11] The social objective 4.2.2 was not included in the model analysis.

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 expertise 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).

Table 13. Comparison of the outcome scores derived by MCDA and the objective-function model.

	Means		Wilcoxon rank sum test		
Objective	MCDA	Model	W	Pr(W)	Sig[a]
Ecol1.1.1	0.063	0.208	175	0.222	
Ecol1.1.2	0.125	0.000	317.5	0.008	**
Ecol1.2.1	0.188	-0.453	425	0.000	***
Ecol1.2.2	0.000	-0.299	344	0.003	**
Ecol1.2.3	0.083	-0.582	425	0.000	***
Ecol1.2.4	0.104	-0.535	425	0.000	***
Ecol1.3.1	0.021	-0.014	330	0.010	**
Ecol1.3.2	0.021	0.144	75	0.000	***
Econ2.1.1	0.042	0.496	100	0.002	**
Econ2.1.2	0.104	0.929	25	0.000	***
Econ2.1.3	-0.063	0.496	25	0.000	***
Econ2.2	0.111	0.498	50	0.000	***
Econ2.3	0.042	0.563	50	0.000	***
Econ2.4	0.044	-0.360	345	0.001	**
Econ2.5	-0.222	-0.333	250	0.500	
Manage3	0.000	-0.041	375	0.000	***
Social4.1	-0.048	-0.695	387	0.000	***
Social4.2.1	0.119	-0.396	400	0.000	***
Social4.2.2[b]	0.000	-	-	-	
Social4.2.3	0.143	0.257	125	0.014	*
Social4.3.1	0.000	0.192	100	0.002	**
Social4.3.2	0.000	-0.389	405	0.000	

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

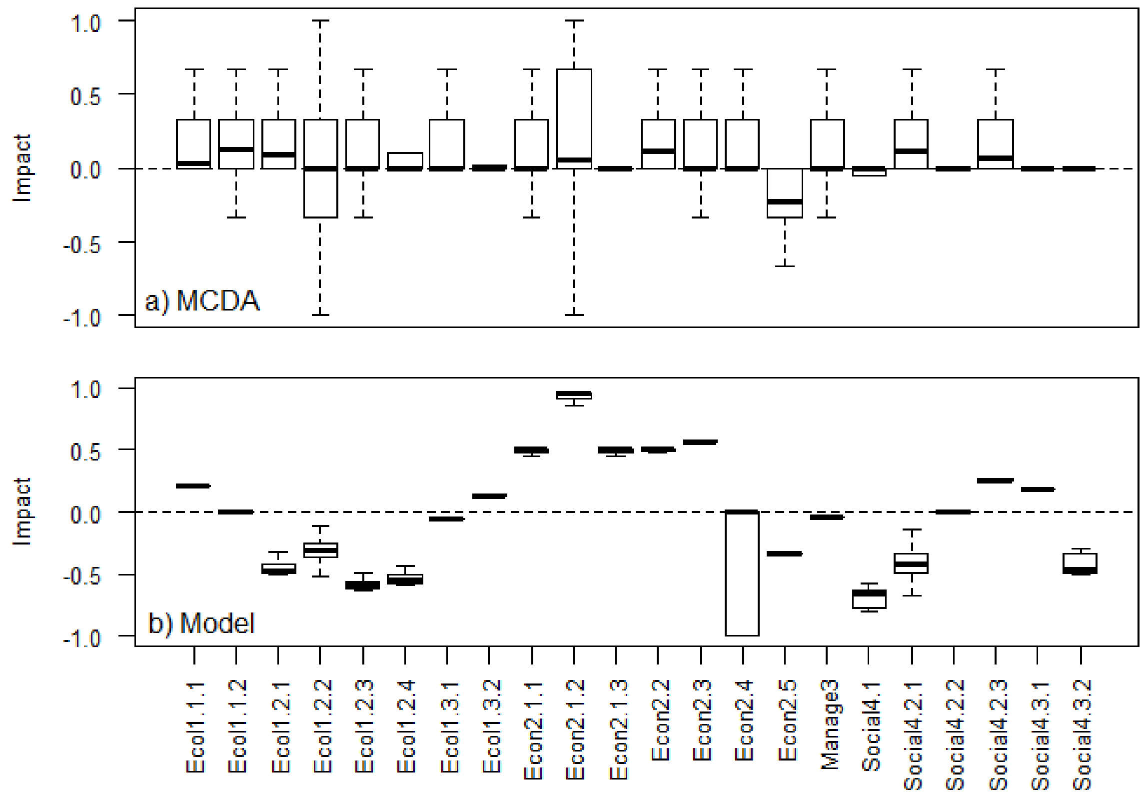


Figure 29. Distribution of the relative impacts of the separate charter harvest strategy for the semi-quantitative approach (top) and the quantitative approach (bottom). Note that objective 4.2.2 ("Maximise utilisation of the retained catch of target species") was not used in the simulation as it was beyond the direct control of harvest strategy levers, so an impact of zero was used for this objective in panel b).

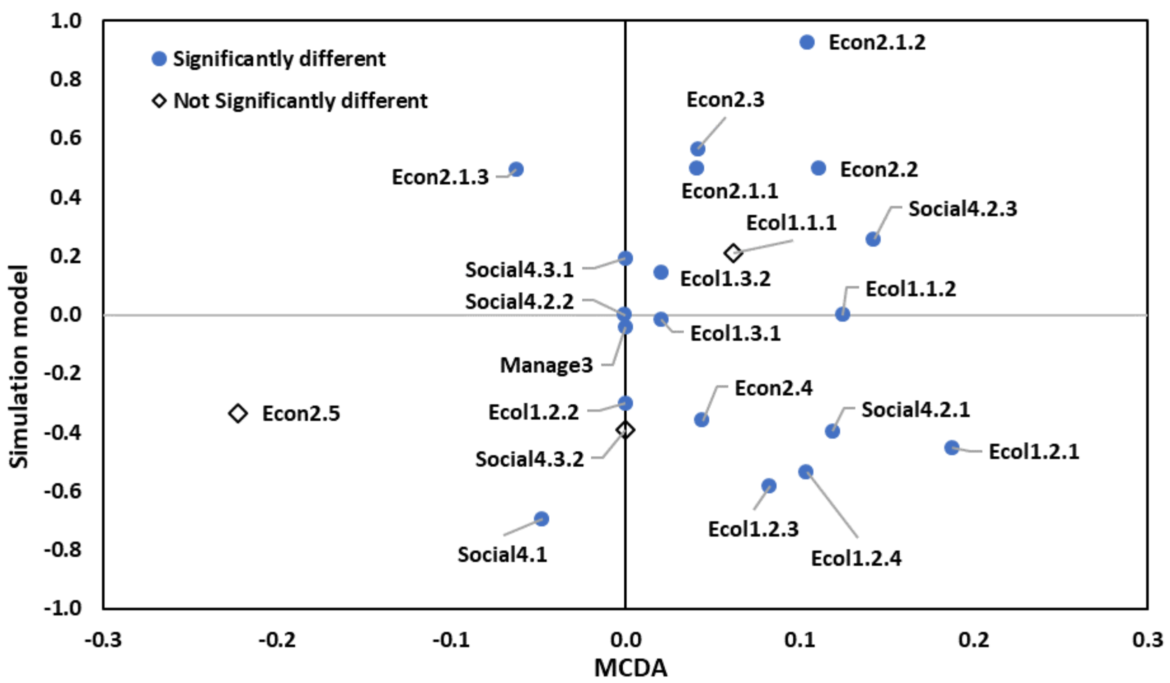


Figure 30. Comparison of mean MCDA and model scores.

A General Methodology for harvest strategy development against the triple bottom line for multi-sector fisheries

This project has illustrated, using the Queensland Coral Reef Finfish Fishery as a case study fishery (and, as such, our experiences "living the process"), a methodology to develop and operationalise triple bottom line (TBL) harvest strategies for multi-sector fisheries.

This section distils a stepwise, more generalised summary of the process, to assist practitioners to apply the approaches to their own fisheries. This Methodology may be used as standalone, succinct, user-friendly guidance. Figure 31, developed by Dr. Sue Pillans (www.drsuepillans.com) depicts the General Methodology as a cartoon flowchart.

Our experiences with the Queensland Coral Reef Finfish Fishery (CRFFF) are summarised in italics at the end of the each of the below 12 steps.

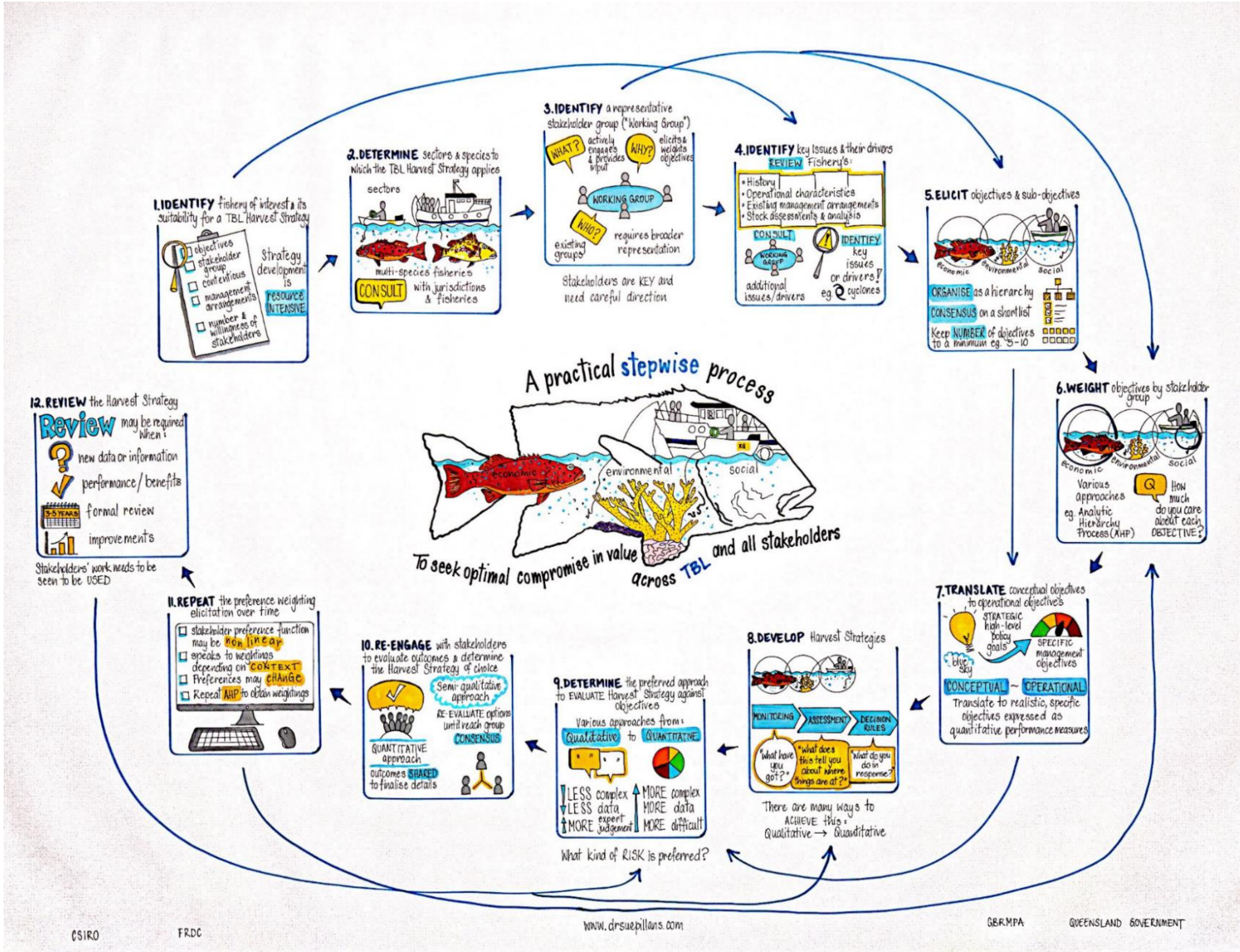


Figure 31: The 12 steps of the General Methodology for TBL HS development, represented as a pictorial cartoon. Art courtesy of Dr. Sue Pillans (www.drsuepillans.com)

1. Identify fishery of interest and its suitability for a TBL harvest strategy

Developing a TBL harvest strategy is a resource-intensive undertaking. In considering whether a TBL harvest strategy should be a priority for a multi-sector fishery, the following points should be considered:

- The scope for consideration of multi-faceted social objectives: are a range of social objectives readily able to be identified?
- The breadth of environmental objectives: do environmental objectives pertain to bycatch and the ecosystem more broadly?
- The existence of an established stakeholder group (e.g. a formal Working Group, or an industry group or association)
- The level to which the fishery is politically contentious: might such tensions challenge the efforts to develop a TBL harvest strategy?
- The extent of the fishery's alignment with governance agency priorities
- The extent of effective existing management arrangements: will a TBL harvest strategy equate to a major management change, or to a modification of existing arrangements?
- Whether previous stock assessments, management strategy evaluation, or other data analyses have been undertaken: if so, these could be useful to incorporate in the TBL harvest strategy
- The presence of over-arching environmental changes: are there environmental drivers or perturbations that affect the fishery?
- The number and willingness of stakeholders: it may be more difficult to obtain adequate representation if the stakeholder group is large, and the chances of success of the TBL harvest strategy's implementation is optimal if stakeholders as a whole are willing to buy-in to the development process.

These are the same points that were considered when formally selecting the CRFFF as the case study fishery.

In the case of the CRFFF, we had to finalise the choice of fishery at a meeting of managers. We listed the above considerations, and scored each possible fishery against them.

2. Determine sectors and species to which the harvest strategy applies

A clear definition of the sectors (and gears, or fishery operation categories [e.g. small vs. large vessels]) and species to which the TBL harvest strategy will apply is the next step.

Ideally, a harvest strategy should be applicable to commercial, recreational and indigenous sectors, but there may be circumstances where existing arrangements for certain sectors are felt to be appropriate.

For multi-species fisheries, managers may choose to focus only on the main target species, key species according to life history/vulnerability, high value species, or species groups.

If stocks straddle or cross jurisdiction boundaries or fisheries, the relevant jurisdictions and fisheries must be consulted to determine the manner in which the shared stocks are to be managed under harvest strategies.

In the case of the CRFFF, we had Coral Trout ("CT") as a "basket" of various species, Red Throat Emperor ("RTE"), and a basket of other species ("OS"). We explicitly considered the commercial (of which there were two kinds), charter, and recreational sectors, all of which employ different targeting practices. We also acknowledged the indigenous commercial catch. Catch proportions also varied by region.

3. Identify a representative stakeholder group (a "Working Group")

The next step is to identify a core group of representative stakeholders, who will actively engage and provide input to the harvest strategy, and who will be the main points of contact.

Active engagement and buy-in to the process of the development of any harvest strategy is critical to its success, but the active input of stakeholders is imperative to developing a TBL harvest strategy, in order to elicit and weight objectives and to identify alternative possible harvest strategy options.

Existing Working Groups or other industry groups are an obvious starting point of contact, but care will have to be taken to ensure adequate representation across all interest groups. Typically, these include the commercial, recreational and indigenous fishing sectors (across all gear types/operating practices), conservation groups, community representatives, other fisheries targeting the same stocks, buyers/processors, scientists, and managers. It is important that members of the group are respected among their peers and that their views are felt to reflect those whom they represent.

Both of the TBL approaches taken in this project share the need for stakeholder input to determine objectives, weights and harvest strategies. As such the Working Groups should have greater diversity of representation for TBL approaches than traditional management. The stakeholder group used in this case study did not include any social scientists or economists. As such, stakeholder groups may need reformulating in the future to cope with the demands of TBL approaches.

The identified group should commit to ongoing engagement toward the development, evaluation and implementation of the harvest strategy. This is likely to equate to at least two to three, one to (ideally) two day workshops per year, and inter-sessional reviews or questionnaires. Managers will also have to be prepared to canvass all relevant stakeholders to complete the latter.

In the case of the CRFFF, we did not have the opportunity to custom-design out Working Group, but rather engaged with the existing Working Group, which experienced changes to its membership midway through the project.

4. Identify key issues and their drivers

A review of the fishery's history, operational characteristics, existing management arrangements, stock assessments and analyses should be undertaken. Subsequently, there should be consultation with the Working Group to elicit any additional issues or driving forces in the fishery. These should be along the TBL pillars: ecological, economic, social, and institutional. The aim here is to set the foundation for determining the key TBL objectives for the fishery.

It may help to ask stakeholders to write a “vision statement” paragraph summarising their aim for the fishery.

At this stage, it is helpful to begin to identify which key issues or drivers that may equate to ‘exceptional circumstances’ that may trigger departure from or even suspension of the harvest strategy. This is one way to allow flexibility in a structured way, but not so much flexibility that it undermines the intent of having a harvest strategy.

In the case of the CRFFF, we had the advantage of a well established Working Group, and a history of quantitative assessments and sustainable management. We found that asking members for “vision statements” helped set the foundation for the elicitation of objectives.

5. Elicit objectives and sub-objectives

It is sensible to organise objectives as a hierarchy, with higher level objectives being the triple (quadruple) bottom line categories of economic, social, environmental/ecological, and institutional/management objectives, and lower level objectives being more detailed or specific objectives for the fishery in question (Leung et al., 1998; Soma, 2003; Wattage and Mardle, 2005; Pascoe et al., 2009c; Jennings et al., 2016).

The Working Group should achieve a consensus on a shortlist of fishery objectives. At this stage, individual opinions of the relative importance of each objective are irrelevant – this comes into play at the “preference weighting” stage.

While it is important to capture the key objectives for the fishery across the triple bottom line, efforts should be made to keep the number of objectives to a minimum. As the number of objectives increases, preference weightings are likely to be diluted across them, the trade-offs will be more difficult to evaluate, and gathering the required information around their corresponding performance indicators becomes impractical. Between 5 and 10 objectives is ideal, but more may be warranted depending on the fishery’s circumstances.

To begin the process of identifying fishery-specific TBL objectives, it may be useful to share the inventory of TBL objectives prepared as part of this project. This inventory should be confronted with the identified key issues and drivers in order to begin to flag fishery-specific objectives of interest. It may also be helpful for an outside expert or facilitator to share the concepts around the development of an objective hierarchy.

Once an initial shortlist has been created, Working Group members may break into smaller groups to identify which objectives may be most applicable to their fishery, which may need modification, and whether any new objectives specific to their fishery may be required.

The process is often iterative, and can occur over more than one workshop.

Part of this process will include delineating between the objectives that are within the control and mandate of a harvest strategy, and those that are broader. If objectives are unable to be influenced by a harvest strategy’s harvest control rules, they should be retained but not be included within the TBL harvest strategy. However, it is likely that these objectives, which can tend to be more over-arching in nature, will need to be considered when developing broader management structures.

In the case of the CRFFF, objective elicitation was an iterative process that occurred over 2 sets of meetings. We found it was easier to have an inventory of objectives as a basis for the conversation, and that break-out groups were helpful. Many of the proposed objectives were outside of the mandate of a harvest strategy. In hindsight, 21 objectives was too many.

6. Weight objectives by stakeholder group

Different harvest strategies are likely to have different impacts against the different objectives. To assess the overall suitability of the harvest strategy, the objectives need to be weighted so that the different strategies can be compared on an effective performance basis.

There are various approaches that may be used to obtain objective stakeholder weightings. In this project, however, we have applied modified versions of simple scoring-based approaches and the Analytic Hierarchy Process (AHP) based on a series of pair-wise comparisons. Each method relies on a selected group of individuals (e.g. key stakeholders) to indicate a preference for each objective within a set of objectives. They differ in how these preferences are captured and analysed, both between and within the different approaches.

An online survey of fishery stakeholders is recommended to elicit weights using the two approaches, and to assess how the methods used affected the overall objective weights. A key advantage of the use of online surveys is that allows access to relevant stakeholders who may be geographically dispersed, even if not large in absolute numbers. Ideally, as many stakeholders as possible need to participate to give representative samples of individual weights.

The lack of direct interaction with the respondents creates additional challenges for deriving priorities through approaches such as AHP. Direct interactions with the individual respondents is not generally feasible, and in many cases responses are anonymous. Most previous online-based AHP studies have tended to exclude responses that have a high level of inconsistency, resulting in a substantially reduced, and potentially unrepresentative, sample.

Our suggested approach avoids some of these pitfalls by modifying the way in which the data are collected and analysed, accounting for the symmetry assumption underlying AHP.

In the survey, respondents are presented with a nine-point importance scale against which they can assess the importance of each objective. A nine-point scale was selected (rather than an “out of 10”) as it allows five categories to be defined with a neutral mid-point between them. Respondents indicate which response best approximates their belief around the importance of each objective.

The results were analysed using a modified version of the AHP Geometric Mean Method (GMM).

In the traditional AHP approach, preferences are expressed on a nine-point scale, with 1 indicating equal preference, and 9 indicating an extreme preference for one of the sub-components. Preferences are assumed symmetrical, such that if A against B has a preference of $a_{12} = 9$, then $a_{21} = 1/a_{12} = 1/9$. For each set of comparisons, a matrix of scores can be developed, given by:

$$A = \begin{bmatrix} 1 & a_{12} & \dots & a_{1n} \\ a_{21} & 1 & \dots & a_{2n} \\ \dots & \dots & \dots & \dots \\ a_{n1} & a_{n2} & \dots & 1 \end{bmatrix}$$

In the approach used in this project, a separate value is derived for each a_i, a_j, and the relative score $a_{i,j}$ is derived from the difference between them. From this, an equivalent comparison matrix to the traditional AHP approach can be derived, with the value for each element based on the differences between the stated objective importance levels (expressed on a 1-9 scale):

$$a_{i,j} = \begin{cases} a_i - a_j + 1 \ \ if \ \ a_i - a_j > 0 \\ \dfrac{-1}{a_i - a_j - 1} \ \ if \ \ a_i - a_j \le 0 \end{cases} \quad \text{and} \quad a_{j,i} = \frac{1}{a_{i,j}}$$

The analysis is undertaken within each level of aggregation in the hierarchy. The weight of an individual sub-component is determined by the product of its initial weight estimate (i.e. when compared with other sub-components in that same group) multiplied by the weight of the higher order aggregation (i.e. compared with other higher order aggregations) under the principle of hierarchic composition. This reduces the number of necessary direct comparisons, as only sub-components at the same level and within the same broader sub-component need to be compared. Given these, the Geometric Mean Method (GMM) weights (ω_i) can be derived:

$$\omega_i = \frac{\left(\prod_{k=1}^{n} a_{i,k} \right)^{1/n}}{\sum_k \left(\prod_{k=1}^{n} a_{i,k} \right)^{1/n}}$$

An advantage of this approach over the traditional pairwise comparison is that the respondent is able to compare all objectives at the same time, avoiding issues around inconsistency (which does not need to be calculated as a result). That is, the respondent will immediately be able to see how each of the options compares to the full set when making their response.

A further advantage of the approach is that the issues around interpersonal comparisons are reduced compared with the scoring method. As the differences between the preference scores are used rather than their absolute levels, a score of [5,6,7] from one respondent will produce the same weighting of each objective as a score of [7,8,9] using the modified AHP method, but different weights using the scoring method.

It is important to acknowledge that the weightings on some objectives may be fixed according to legislative defaults that apply regardless of individual preferences. For example, the Queensland Fishery Management Act mandates that fisheries resources are utilised in an ecologically sustainable way. This automatically confers a high weighting on associated objectives.

Once individual weightings have been obtained, a decision has to be made on how these should be grouped. In this project, we simply aggregated weightings according to which of the following groups each respondent most strongly identified:

- Commercial fisher
- Charter boat operator
- Recreational fisher
- Quota owner
- Processor/wholesaler/buyer
- Fishery manager
- Scientific advice
- Other.

Note that additional groups could include “Indigenous fisher” and “Conservation group”.

An alternative approach to simply classifying individual responses by the group the individual best represents, could be to undertake a cluster analysis of the results and delineate the resultant clusters accordingly.

Practitioners may wish to consider more closely the role of managers in the context of objective preferences. Here, we assign fishery managers to be their own stakeholder group. However, their role may more accurately be described as neutral coordinators, who assimilate the preferences of the other stakeholders in order to make an overall recommendation. As such they could be considered to be weighting across stakeholder groups, rather than objectives. However, it is far more politically problematic to have managers formally assign weights to stakeholder groups, so treating them as a separate group that assigns weights to objectives may be preferable.

In the case of the CRFFF, an online survey worked well, but we had difficulty in getting people to participate and complete it. The importance of a fishery manager meeting face-to-face and explaining the value of the survey to fishers was instrumental in raising the response rate.

7. Translate conceptual objectives to operational objectives

Objectives for fisheries management can be categorised as either “conceptual” (strategic) or “operational” (tactical). Conceptual objectives are generic, high-level policy goals. To be included in a management strategy evaluation, conceptual objectives need to be converted into operational objectives (expressed in terms of the values for performance measures). This involves translating each conceptual objective into operational objectives and performance measures.

Operational management objectives are very precise and are formulated in such a way that the extent to which they have been achieved during a specified period should be easily measured (Fletcher 2002; Cochrane 2002).

An operational objective is defined as "An objective that has a direct and practical interpretation in the context of a fishery and against which performance can be evaluated". Operational objectives should be easily measured and linked to the performance indicators, reference points and decision rules of a harvest strategy. To be considered operational, objectives have to be: i) realistic, ii) have performance indicators against which each objective could be assessed, and, ideally, iii) simulation-achievable.

In the case of the CRFFF, we found this very tough and challenging. In attempting to translate each objective, we needed to ensure that we were correctly interpreting its intent. We also had to work in the absence of data to directly inform many of the objectives. Furthermore, defining the relationship between the performance indicator and the management lever (the TAC) was difficult for many performance indicators. The process involved much guesswork and we made many assumptions.

8. Develop harvest strategies

Given the key issues and objectives in the fishery, the task now is to determine alternative:

i) Monitoring
ii) Assessment
iii) Harvest control rules/decision rules

that may be applied to achieve the TBL objectives.

In this project, we took the approach of workshopping "straw man" options within the Working Group. For each of the three main species groups, modified versions of the current management arrangements were proposed, in an attempt to address key TBL issues more directly, or to include TBL aspects currently not incorporated. We also considered "blue sky" ideas that were not rooted in the current management arrangements but rather addressed other key aspects of the fishery, or that took a novel approach. In this, it was important not to be judgemental, but rather to encourage creative thinking and "out of the box" suggestions.

The broad approach was to:

- Identify a set of principles (e.g. buffers, targets, spatial management….)
- Consider modifications to the existing management arrangements (can we enhance or improve these?)
- Re-evaluate existing decision rules
- Consider "blue sky" ideas
- Consider the extent to which each proposed harvest strategy enhances one or more ecological, economic or social objectives.

In workshopping alternative harvest strategy options, a decision should be made about the extent to which the overall scope of the strategy, and the decision or harvest control rules (i.e., the management levers) are intended to directly address TBL issues and objectives, or whether these will focus primarily on target species fishing mortality. For the latter, the intention would be that the TBL performance indicators (associated with each objective) would detect and correct for any broader problems. An example of a TBL-explicit harvest strategy would be one that, for example, assigns separate TAC to different sectors, that has spatially explicit control rules, or that imposes "exceptional circumstance" override rules in the face of an acute environmental or anthropological event.

Generally, the harvest strategy components are developed in the same manner as described in the National Harvest Strategy Guidelines (Sloan et al. 2016). At the same time, careful attention needs to be paid to the TBL in the following ways:

Monitoring:

- How can data collection protocols most cost-effectively provide the required inputs for each of the performance indicators? Data have to usefully serve the harvest strategy in terms of informing the identified performance indicators.
- Consider the extent to which available data can link harvest strategy management levers to performance indicators. When collecting environmental, economic and social data, consider how this best serves the performance indicators associated with each TBL objective. Be mindful that such data must meaningfully relate to the harvest strategy. That is, if a decision rule is adjusted, can the relationship between this adjustment, and the data, be articulated?
- If current data collection protocols do not adequately serve the harvest strategy, a short-to-medium term plan may need to be put in place to modify such protocols.
- Determine who is responsible for collecting data, and ensure that it is understood how this data serves the harvest strategy development and implementation.

Assessment:

- Assessments for TBL harvest strategies require that all performance indicators are calculated or empirically derived. For example, additional calculations are likely to be required to obtain certain TBL performance indicators that are not directly output from a formal stock assessment.
- Determining certain TBL performance indicators can be difficult, particularly when practitioners are more familiar with traditional indicators (e.g. around target species sustainability and economic profitability).
- The sources of data used to inform the assessment will need to be agreed upon. Typically, the most reliable and representative data are used. This may mean that available information (e.g. from a more marginal sector) may not be incorporated in the assessment.

Harvest control/decision rules:

- Consider the manner in which each decision rule relates to each performance indicator. This can be challenging in a TBL HS context given that there are relatively few levers that are in the control of fishery managers. Harvest strategies focus on controlling the level of fishing mortality and the nature of fishing activities. As such, control rules are limited to various input controls around effort, various output controls around catch, various spatial, gear, and temporal controls, limits on size and sex of captured individuals, applying precautionary buffers, invoking overrides in case of exceptional circumstances, and use of incentives. How these controls directly, or indirectly, affect each objective (i.e. each performance indicator) needs to be clearly understood. This should at least be a conceptual understanding, but for quantitative evaluation approaches, a formal relationship will need to be specified.

- Consider how acute events (environmental and anthropological) are to be acknowledged, including by the invoking of "exceptional circumstances".
- Consider how chronic events (such as climate change) are to be acknowledged, including the need to be precautionary in this context.
- Consider how decision rules are to be applied across fishery sectors and species.

More generally, obtaining the broad form of a harvest strategy should be relatively straightforward. Fleshing this out and fully articulating the details, however, is a non-trivial exercise.

In the case of the CRFFF, we found that careful chairing and guidance were required, and that it was helpful to have some a priori "straw men" suggestions to encourage discussion and creative thinking around "blue sky" HS options. It was also useful to directly ask people to consider what was not being addressed by current management arrangements.

There were various management challenges in establishing a TBL HS for the CRFFF. These included the fact that the Coral Trout quota group is a complex of seven Coral Trout species. Increasingly there is a need to ensure management of each species separately. Additionally, the fishery can be significantly impacted by cyclones. For instance, 2009's Cyclone Hamish travelled the length of the Queensland coast, resulting in in depressed catch rates, and fleet displacement. There is also the complexity of implementing any proposed regional management on a fishery with allocated TACC's already in place. Other challenges included the fact that there were limited economic/social data that were able to be used with confidence, the changing of working group members throughout the course of the project, and the legislated timeframes to have the HS completed.

9. Determine the preferred approach to confront harvest strategy against objectives

Having identified alternative potential harvest strategies, the next step is to agree upon how to evaluate the harvest strategy according to the TBL performance indicators.

TBL harvest strategy evaluation may be either qualitative or quantitative.

Qualitative risk assessments can take the form of multi-criteria decision analysis, 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 (ERAs) 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 management strategies), which were assessed by a group of industry stakeholders and experts using multi-criteria decision analysis 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 include "data-limited" assessment approaches embedded within a simulation-based MSE framework, tuned to achieve optimal TBL performance; approaches that scale to commensurable units (i.e. can be combined into a single unit – e.g. all outputs expressed in dollar terms such as a cost benefit analysis (e.g. Freese et al., 1995), or utility terms such as multi-attribute utility analysis (e.g. Healey, 1984)); scale to non-commensurable units but with explicit objective weights, for example, using a goal programming bio-economic model (e.g. Charles, 1989; Pascoe and Mardle, 2001); scale to non-commensurable units without explicit objective weights, which provide separate outcomes under each objective (e.g. hybrid models, simulation approaches (e.g. Mapstone et al., 2008; Little et al., 2015)); and, finally, there are co- viability analysis approaches (Gourguet et al., 2013; Gourguet et al., 2016).

Table 14 provides a summary of the available approaches for evaluating TBL harvest strategies, together with an overview of the advantages and disadvantages of each. This may assist practitioners to determine which approach may best be suited to their circumstances.

Table 14. Overview of alternative approaches for TBL harvest strategy evaluation, with a summary of the advantages and disadvantages of each. The two approaches taken in this project are highlighted.

Category	Types of approach	Description/Examples	Advantages	Disadvantages
Basic Harvest Strategy, by embedded in TBL consideration.		Social objectives are not directly incorporated, but are rather an "outer onion layer". More traditional assessments and levers, but acknowledging TBL ito the extent to which all objectives are met. Can get around social aspect in that social objectives often process-driven rather than outcome-driven – so indicators env and economic, but process of engagement and objective elicitation acknowledges social aspects and is done in a socially equitable manner.	Less confronting. "Comfortable" arm's length approach. Might be an easier starting point More traditional approach	Does not directly incorporate TBL objectives. Social scientists would see this as a fail
Qualitative approaches	Multi-criteria decision analysis (MCDA) techniques	"Traffic light" approaches (Caddy, 2004, 2009; Caddy et al., 2005; Halliday et al., 2001)	User friendly, transparent, allow direct stakeholder input	Will not tell you how hard to pull a management lever. Will only give preferred strategies.
		CUSUM multiple indicator systems (Scandol 2003, 2005)		
		Multidimensional scaling analysis (RAPFISH) (Pitcher et al. 2013; Pitcher and Preikshot 2001)		
		Analytic hierarchy process; the weighted sum model; the ordination technique; concordance analysis; the regime method; Evamix		
		Ecosystem risk assessments (ERAs)		
	Qualitative models	BBN models/report cards, e.g. Gladstone Healthy Harbours see (Pascoe et al. 2016)		
Quantitative approaches	"Data-limited" assessment approaches embedded within an simulation-based MSE, that is tuned to achieve optimal TBL performance	Includes Delphic approach – "suck it and see" - how broad is value function wrt ecosystem, social (what's in objectives, what's realistically in decision rule and how estimable are indicators, evaluating HS performance)	Acknowledge, particularly against social and economic objectives, that there is likely to be data limitation. Also acknowledge need for pragmatism ito available capacity, nature of fisheries etc.	
	Commensurable units (i.e. can be combined in single unit – e.g. biomass terms, dollar terms) (e.g. socio-bio-economic optimisation models)	Simulations quantifying trade-offs between objectives (reality-based) (e.g. revenue vs biomass vs strike rate etc.); cost-benefit analyses; multi-attribute utility analyses		
		Modelling approaches calculating various reference points (MSY, MEY, MSocY, MSEY), and trying to optimise over each		
		Using the risk-cost-catch approach to quantitatively evaluate trade-offs		
	Non-commensurable units with explicit objective weights	Multi-objective modelling (places explicit weightings on objectives and considers trade-offs). Includes "Pretty Good Yield", "Pretty Good Sustainable Yield". Goal programming bioeconomic models	This gives you where you want to end up.	Leads in to Value functions that combine different objectives. This is really to identify target reference points. Doesn't tell you how to get there.
	Non-commensurable unit without explicit objective weights which provides separate outcomes under each objective	Hybrid models; simulation approaches; viability analysis and co-viability analysis. These involve identifying objectives and goals and seeking solutions within feasible bounds (avoids explicit trade-offs between objectives). Given constraints, what is the likelihood of staying within these? Not hard constraints; soft constraints whereby you want to stay within that region. This is a bit like MSE in that you are testing a harvest strategy, but you are trying to avoid a minimum rather than achieving a target. Gives probability of achieving > minimum.	Good if want to be no worse off than where you were; don't care about being optimal/best. Viability analysis gives minimum acceptable space, per Pope's "minimum whinge" principle – everyone unhappy, but nobody extremely unhappy. Does not aim to identify an "optimal" outcome and hence does not require objective weightings, but instead aims to ensure at least a minimal acceptable levels for each of the objectives.	While any stock levels above a limit reference point may be considered "acceptable" to some degree, it is far from desirable (and is counter to the current Commonwealth Harvest Strategy Policy). Quantifying acceptable levels of social and economic objectives is also highly subjective. Studies based around the limits of acceptable change framework have found that perceptions of these limits varies substantially between individuals and stakeholder groups (e.g. Ahn et al., 2002; Roman et al., 2007), resulting in similar issues as those with determining appropriate objective weights (e.g. which set of minimal acceptable levels to use). Further, once a set of viable options have been identified, identifying which option to implement still requires some implicit weight for each of the objectives.
		Frontier analysis (outcomes when behaviour is optimal relative to different objectives/targets)	The solutions that lie along the "frontier" in the case studies are triple–bottom-line solutions, where one can optimize conservation goals and equity while minimizing costs. Solutions interior to these frontier solutions (most of which are not plotted) are all possible and represent the many ways decision making can miss the mark on the triple bottom line.	As in other trade-off assessments, finding the frontier does not then prescribe a single correct solution but instead presents the range of options, all optimal, that represent the trade-off between stated goals
		Multi-objective optimisation of a value functions: take the stock status estimate from an assessment, and optimise, over the range of possible catch levels, a value function for a given set of stakeholder group weightings.	Can objectively provide the optimal TAC (or other management currency unit) across both the range of objectives and stakeholder weightings.	This approach is data-demanding, and requires explicit quantitative relationships to be defined relating each performance indicator to the management lever.
		Constraints mapping = actual spatial mapping. Criteria map looks at how well fits criteria; Constaints map displays a set of feasible alternatives that fits some kind of objective profile.		Very resource intensive. This is really to ID target reference points. Doesn't tell you how to get there.

This project took two alternative approaches:

i) a semi-quantitative multi-criteria decision analysis (MCDA) approach to compare broad management options or reforms, and
ii) a quantitative, non-commensurable-units simulation model, via a multi-indicator objection function, with explicit objective weights to set TACs for the three main species groups.

The former (MCDA) approach uses expert opinion to derive ordinal rankings of relative impacts against each objective. In our case, these were obtained from members of the CRFFF WG who were familiar with (and developed) the potential harvest strategies. By applying the weights for the individual stakeholders, we derived a subjective probability distribution of the net benefits of each option, which gave the mean outcome by stakeholder group.

The Working Group then conceptually considered the TBL trade-offs they perceived to be associated with their alternative identified harvest strategies.

Figure 32 illustrates the problem of TBL optimisation within and between harvest strategies, as addressed by the quantitative simulation model approach. Consider a situation with only one objective, and hence, one corresponding performance indicator (top left panel of Figure 32). Applying an individual's weighting to that performance indicator gives the value. Here we see that this performance indicator increases with increasing TAC (so it may be an economic one), and that stakeholder 1 cares more about this performance indicator than stakeholder 2.

As the number of objectives, and hence, performance indicators increases, an individual's value profile changes from a line to a 3D surface (top right panel of Figure 32). The shape of this varies according to the preferences assigned to each objective by each individual.

When there are many objectives, this value profile, or solution surface, becomes highly complex, so in the bottom left panel of Figure 32 we have stylised it as two sets of "tropical islands". Each set of islands represents one form of harvest strategy (e.g. non-regional-specific TACs, vs. region-specific TACS). The blue and purple alternate renderings of the islands represent the different values, according to stakeholder group preferences, for any given TAC. So on the upper left set of islands, we can see that stakeholder group 1's (blue) value is optimal at TAC1, as indicated by the blue peak, while stakeholder group 2's (purple) value is optimal for a different TAC value, TAC2. To get the "overall optimal" TAC across the different stakeholder groups, one can use either: i) the best compromise, i.e. the highest intersection point (TAC3), or ii) consider the relative loss in value to each group at any other group's optimal TAC. Here, we see that stakeholder group 1 (blue) has more to lose at stakeholder group 2's optimal TAC (TAC2), than stakeholder group 2 does at stakeholder group 1's optimal TAC (TAC1).

So, our simulation applies each stakeholder group's preference weightings to each performance indicator, and sums to get the overall value for that stakeholder group. As the performance indicators are all some function of the management lever (here, the TAC), we can then maximise the value across all possible TACs, thus obtaining the optimal TAC for that group. We then reconcile the optimal TACs across all stakeholder groups as stylised with the "island" diagrams. A pertinent issue when weighing alternative approaches is the nature of the risk on which managers wanting to operationalise the TBL wish to impale themselves. Qualitative or semi-quantitative approaches carry the inherent risk associated with qualitative expert opinion. Quantitative approaches such as simulation models have 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 qualitative or semi-quantitative approaches may, to some extent, avoid liability around "getting it wrong".

More broadly, incorporating important external drivers like climate change and policy implementation is difficult regardless of the approach selected. These would have to be treated as alternative scenarios whose effects on the population, and on the fleet dynamics, are assumed known. While quantitative approaches would then analytically adjust the recommended TACs within the optimisation process, expert judgement, as used in the MCDA approach, would be challenging, as the experts would have to reconcile the assumed nature and impact of the driver, with its effect on harvest strategy performance.

Regardless of the approach chosen, it should undertake the following, whether qualitatively or quantitatively:

i) Undertake a relative impact assessment for each "strategy" against each sub-objective
ii) Undertake an overall weighted impact by strategy, given the weighted sub-objectives for any one stakeholder group
iii) Derive the management implications.

Model-based approaches should inherently combine these three steps (e.g., within an MSE); qualitative approaches will undertake them as separate, explicit steps.

In the case of the CRFFF, the semi-quantitative MCDA approach had the benefit of an established precedent, but required close contact with the Working Group. The quantitative simulation was technically highly challenging, data hungry, and required performance indicators to be carefully refined against their operational objectives.

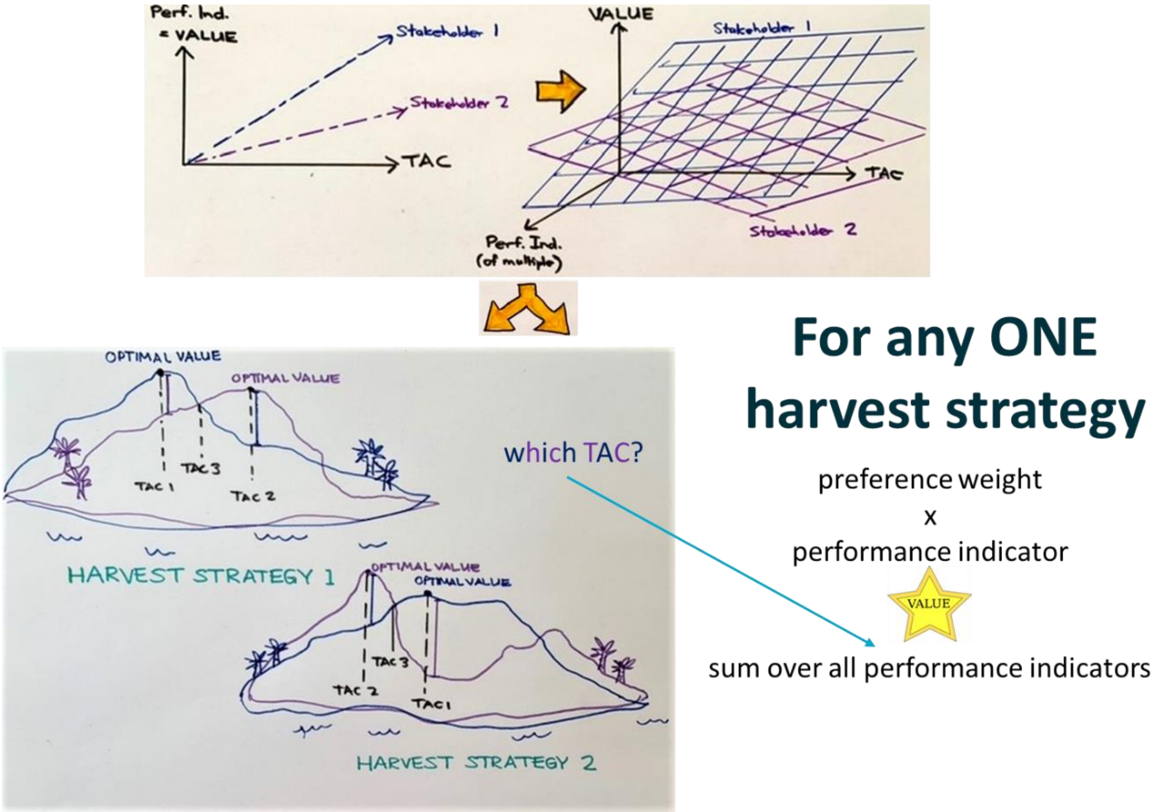


Figure 32: Simplified illustrative description laying out the problem of TBL optimisation within and between harvest strategies, for the quantitative simulation approach taken in this project.

10. Re-engage with stakeholders to evaluate outcomes and determine the harvest strategy of choice

The nature of this engagement will vary according to the type of TBL analysis undertaken. If a semi-quantitative or Delphic approach has been taken, the same core group would have iteratively re-evaluated options until reaching group consensus.

If a more quantitative approach was undertaken, the outcomes should be shared with the Working Group in order to finalise the details of the harvest strategy. If the approach taken yields “acceptable” trade-offs or frontiers, or identifies target reference points, further consultation will still be required to finalise where within the “acceptable” space stakeholders wish to land, and the choice and magnitude of the associated harvest control rules.

If the approach is a simulation model that produces management recommendations that have been optimised across objectives and stakeholder groups, the focus in revisiting the approach is obviously not on the value of the management lever itself, but rather on the data limitations and model assumptions, and the sensitivities to these.

In the case of the CRFFF, this re-engagement has yet to occur. The final harvest strategy for the CRFFF was ultimately more policy driven with less attention to the TBL in terms of its formulation, given the legislative timeframes and policy constraints. The changing membership of the Working Group was a challenge throughout, but this is a common reality.

11. Repeat the preference weighting elicitation over time

The stakeholder preference function may be nonlinear. That is, some of the groups would have non-static weightings: these may be context-specific. We are assuming that the marginal rate of substitution is constant, when they may instead be a continuous function of the harvest strategy.

This speaks to weightings depending on context. For example, at low stock levels, conservation groups may want a fishery to be closed, but when the stocks are healthy, they are happy to consider non-zero social and economic weightings. As such, preferences may change depending on the state of the ecosystem.

More generally, we are assuming our weightings are stable through time. In reality, preference weightings within interest groups may change on any given day, under a politically changing environment, or according to the representativeness of the sample (who's in the room).

As such, it is worthwhile periodically repeating the Analytical Hierarchical Process to obtain preference weightings.

In the case of the CRFFF, this repeated preference weighting would likely be with different people, which adds a layer of complexity. This speaks to the process of TBL HS development ideally requiring continuity and a longer-term commitment.

12. Review the harvest strategy

As per the National Harvest Strategy Guidelines (Sloan et al., 2014), irrespective of the amount of prior testing of a harvest strategy, global experience shows that periodic amendments are likely to be necessary (Smith et al., 2007). Amendments may be made in response to, for example, new information or data that substantially changes the understanding of the status of a fishery, when problems are identified in application of the harvest strategy, or when uncertainties that were not previously understood arise (Australian Government 2007). Moreover, even without a change in the status quo, formal reviews are critical in order to determine whether the harvest strategy is performing as anticipated (in terms of the preferred/optimised approach). The development and implementation of TBL HSs are expensive, and there will be a reluctance to invest unless there is ongoing assurance that they are delivering their purported benefits.

A formal review of any harvest strategy should be planned and undertaken on an agreed time frame (for example, every 3-5 years). Critically, any harvest strategy must be allowed to evolve to address deficiencies, unforeseen circumstances and to allow for improvements, whether during a routine review, or as pre-specified overrides permit (Walters and Hilborn, 1978). However, a harvest strategy should not be changed to relax or vary its specifications when the decisions are not suitable to some, or all, stakeholders.

In the case of the CRFFF, this has yet to occur (the first formal review of harvest strategies is scheduled for 2024). Expectation management will be critical during such a review, particularly around the point that outcomes that are not suited to stakeholders are not, in themselves, a justification to change the harvest strategy.

Discussion

A Multi-Criteria Decision Analysis approach to determining and evaluating TBL harvest strategies

While the Working Group agreed on a set of objectives that the harvest strategy should address, not surprisingly the importance weight given to these objectives varied between stakeholder groups. All groups tended to rank the ecological sustainability objectives the highest. The relative weight of the economic objectives varied between stakeholder groups, with the commercially oriented groups (fishers, quota owners and buyers) generally weighting economic objectives similar to the ecological sustainability objectives, while the other groups ("managers", "scientists" and "other") tended to weight the economics objectives much lower. Social objectives were generally given a low weight by all stakeholder groups, with the exception of recreational fishers and the "other" group which consisted predominantly of conservation group and industry association representatives.

These different weightings had an impact on how the different harvest strategy options were potentially perceived by the different stakeholder groups. For example, the use of environmental overrides was considered to result in benefits in terms of expected ecological outcomes but at a higher economic cost. For the commercially oriented groups with similar economic and ecological sustainability objective weights, these effects largely cancelled out, resulting in a slight negative expected benefit. For the other groups with the lower economic weighting, these options were seen to result in a positive outcome.

The apparent trade-offs between ecological and economic outcomes also reflects a tendency for most respondents to discount future economic and ecological impacts (both positive and negative) differently. For example, harvest strategies that performed well against the objective of achieving MEY-level stocks should also result in longer term economic benefits. However, the impacts estimated by the stakeholders were much more short term in respect to economic outcomes than ecological outcomes. This may reflect the state of economic resilience in the fishery – where a reduction in economic performance due to higher operating costs involved in achieving higher ecological performance may result in greater short-term economic stress. It could also reflect a greater emphasis on short-term profit, possibly due to a lack of trust in government not to make decisions that may compromise future profits, or their lived experience as fishers. To some extent this is also reflected in the relative weighting by the groups, with those sectors with a financial stake in the industry placing greater importance on economic outcomes, potentially not so much to only improve economic performance, but to also ensure it does not deteriorate.

While we did not test every combination, the analyses demonstrate how triple bottom line outcomes can be affected by relatively small changes in the harvest strategy and through combining approaches targeted at particular aspects. For example, the addition of spatial consideration to the environmental override option was introduced to improve both ecological outcomes and social outcomes, with spatial management allowing inter-regional equity considerations to be addressed. Although social objectives had a low weighting for many groups, the additional social benefits from the introduction of spatial considerations were sufficient to change the sign on the average net score from negative to positive for some groups, and reduce the magnitude of the negative sign in the others.

The outcomes from the process are not necessarily an endpoint in terms of determining an optimal harvest strategy, since managers will need to consider these findings in light of the agreed strategic direction, their legislative mandate and feasibility. The perceived relative benefits of alternative harvest strategy options were opinion-based: harvest strategy evaluation against triple bottom line objectives, acknowledging alternative sector weightings, can also be undertaken quantitatively using simulation modelling. While acceptability to all groups may not be required given the constraints under which managers must operate and also the diversity of views across and within stakeholder groups, the process identified which areas of a potential option may need further consideration to be more acceptable to a greater proportion of stakeholders. For example, reducing the costs (or the perception of costs) associated with the environmental overrides and spatial management options would result in benefits being realised by all groups. Similarly, further investigation as to why separate OS quota was seen as desirable, while separate CT quota was not may lead to improvements in the management of both sets of species. As a consequence, these results provide valuable insights which can inform the subsequent consultation to refine the harvest strategies. While considered separately, the options are not mutually exclusive. For example, a charter sector allocation could be implemented in addition to the other options.

A key feature of the process used in this study is that it allows managers to integrate all dimensions of sustainability into the harvest strategy development process and also provides an explicit role for stakeholder engagement. Industry and other groups were directly involved in identifying the objectives, weighting the objectives, identifying the potential harvest strategies and providing input (based on expert knowledge) as to how these strategies are likely to perform. The process provides a formal framework in which this consultation can take place. As such, it directly addresses the key impediments to developing effective EBFM (Stephenson et al., 2017a) by providing a suite of stakeholder developed ecological, economic, and social objectives as well as a general process to integrate these into harvest strategies that consider the three dimensions of sustainability.

A quantitative simulation model approach to evaluating TBL harvest strategies

Our goal is to provide a tool for managers, fishery management councils, advisory bodies, scientists, and stakeholders to consider a richer range of trade-offs 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 trade-offs 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 (Symes and Phillipson, 2009; Vieira et al., 2009; Triantafillos et al., 2014; Brooks et al., 2015; Pascoe et al., 2017). 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.

Comparing the MCDA and simulation approaches

This section considers the similarities and differences between the two approaches and what is required to operationalise them. Both approaches are ultimately able to operationalise TBL harvest strategies, however the semi-quantitative MCDA approach already has a successfully precedent in several cases as noted in the literature review . A fully conditioned and optimised TBL assessment model for practical implementation is further away. However, several other approaches could have been applied. In terms of methods, therefore, new fisheries science method paradigms are probably not required. The emphasis is much more likely to be on refinement.

The MCDA semi-quantitative 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 harvest strategies 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, the quantitative approaches can use similar products from the semi-quantitative approach, but the quantitative 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 these approaches share the need for stakeholder input to determine objectives, weights and harvest strategies and as such the Working Groups should have greater diversity of representation for TBL approaches than 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 would not normally have been available). As such, stakeholder groups may need reformulating in the future to cope with the demands of TBL approaches.

The semi-quantitative 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 simulation model 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 suffers from a lack of detailed data: it was not conditioned on historical data. 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). While the simulation provides an impartial and replicable platform for operationalising the TBL approach, this comes at a high information premium. The simulation approach also has less direct stakeholder involvement (rather, it is very "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 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 on which managers wanting to operationalise the TBL wish to accept. The MCDA semi-quantitative approach carries the inherent risk associated with qualitative expert opinion. The objective-function simulation 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 semi-quantitative approach may, to some extent, avoid liability around "getting it wrong".

More broadly, incorporating important external drivers like climate change and policy implementation is going to be difficult within both the simulation and MCDA approaches. For both, these have to be treated as alternative scenarios whose effects on the population, and on the fleet dynamics, are assumed to be known. While the simulation then analytically adjusts the recommended TACs within the optimisation process, expert judgement, as used in the MCDA approach, would be challenging, as the experts would have to reconcile the assumed nature and impact of the driver, with its effect on HS performance.

Boundaries around TBL harvest strategy development

There were some boundaries around developing a TBL HS placed upon the project by the jurisdiction, fishery and scope chosen. These include legislated & policy boundaries that are hard wired, and which must be accounted for when embarking on the development of a TBL HS. Such boundaries are by no means specific to our case study fishery, and stakeholder expectations must be tempered in the context of these. Queensland's target reference point of 60% of target stock unexploited biomass (Department of Agriculture and Fisheries 2017) is such a boundary: optimisation of the objectives may have enabled other choices if this mandated target was set at an alternative level.

A related issue pertaining to hard wired boundaries is the use of harvest strategies as a de facto allocation tool. Queensland's target reference point is such an example as it will override individual preferences to rule out harvest levels that may optimise commercial fishing catch and revenue. The mere setting of a target reference point of 60% of target stock unexploited biomass is not only biologically conservative but may also favour recreational and indigenous interests over commercial. Regardless, all stakeholders will consciously and unconsciously be guarding their interests in the context of such constraints, which can result in biases in their weighting of objectives. How to deal with pre-weighted objectives that already favour some groups over others requires more explicit consideration. The MCDA approach is particularly prone to such biases and this can lead to perverse outcomes overall as a result (e.g. far from optimal for the Queensland community at large). In this sense independent leadership of the MCDA process is critical to both point out and reduce bias.

Another boundary was that the direct stakeholder engagement and representation was limited to the existing Working Group, which comprised mainly harvesters, managers, scientists and environmental non-governmental organisation representatives. The broader involvement of, for example, retailers or consumers may have substantively changed objective weightings. Future TBL HS development should consider the extent of scope or representation; that is, how far down the seafood supply chain are we accounting for in determining a TBL HS? How this is answered can change the TBL HS substantially. There are some limitations to the use of expertise which, if not handled carefully, can be reduced to a popular vote by the (often unintentionally) biased and partially informed.

One option to address this may be in terms of weighting those participating in preference weighting and TBL HS development. Potential participants could first complete a questionnaire rating their current knowledge and expertise across social, management, ecological and economic. The results could be taken into account later in the process to weight individual responses. However, this can be politically fraught and sensitive, and the process of undertaking such weightings should be defensible and transparent.

Including "all environmental aspects"

The project has taken a comprehensive definition of "harvest strategies that include all environmental aspects":

- In defining objectives that explicitly address target, bycatch, and threatened and protected species abundance, localised depletion due to fishing and environmental events, discard mortality, and broader ecological risks.
- In weighting these objectives according to (informed) stakeholder preference.
- By identifying alternative types of harvest strategies that explicitly account for environmental concerns (e.g. having environmental overrides, having spatially-explicit TACs; pulling out separate TACs for specific species).
- By acknowledging the importance of having environmental science expertise within the Working Group.

- In the quantitative simulation, setting target and limit reference points that are precautionary and that use buffers (penalties) on estimates of biomass.
- In the quantitative simulation, optimising the total allowable catch over all of the TBL objectives and over all stakeholder preference weighting profiles.
- In the quantitative simulation, by explicitly considering simplified “climate change” and “cyclone” conditions as scenarios. That said, there is currently a poor quantitative understand of the relationship between acute (cyclones) and chronic (climate change) environmental events and the abundance or availability of key species groups. As such, these scenarios are likely to be gross over-simplifications.

The challenges of acknowledging all environmental and social aspects in a TBL HS

The challenges of developing triple bottom line harvest strategies that include all environmental and social aspects are multi-faceted, and include data collection, evaluating the harvest strategy’s performance, and designing the harvest strategy.

In a broad sense, data collection around environmental and social metrics is often not tailored to the needs of a harvest strategy, in terms of there being an identified relationship between the indicators and fishery catch or effort (or other fishery-related metrics directed adjusted by the harvest strategy). For example, one of the elicited social objectives was “Through sound fishing practices, minimise adverse public perception around discard mortality (compliance with size limits, environmental sustainability, and waste)”. In discussions within the project team and DAF, it was suggested that possible performance indicators against this objective could include either the number of letters of complaint received from the public and social media, or from scored perceptions using an engagement monitoring survey of the local community. However, it is unclear how either of these metrics would vary with adjustments to fishing mortality made under a harvest strategy – that is, the relationship between these metrics and the fishery is not defined.

On the other hand, one of the environmental objectives was to minimise broader ecological risks. In considering suitable corresponding performance indicators, the project team acknowledged the JCU NERP study (http://www.nerptropical.edu.au/project/marine-reserves-contribute-biodiversity-and-fishery-sustainability) on effects of fishing line on the reef, which attempted to develop a fishing impact index as a function of amount of gear left behind that could be monitored over time (Williamson et al., 2014; Russ and Williamson, 2015). However, even if this relationship was defined, what remains unclear is the flow-on effect to the species of interest, and how (or whether) the amount of gear left behind is related to the magnitude of management adjustments. Further, the effect of gear is presumably only one type of “broader ecological risk”. We ultimately made the very loose assumption that the performance indicator was a weak linear function of effort between target and limit reference levels for the latter. Yet another environmental objective pertained to minimising the discard mortality of undersize fish. This perhaps has a more intuitive relationship with effort, but, as discarding is rarely reported, understanding the form of this relationship is difficult.

That is, even when environmental and social data are collected, relating these to their relationship with the target species, and to potential management adjustments, is difficult. A coordinated approach should be taken to ensure that environmental and social data collection programs are tailored to the needs of the harvest strategy.

Quantitative TBL harvest strategy evaluation is made difficult by the demands of this requirement for a detailed understanding of the relationships between performance indicators and the management adjustments. In our simulation, many of these relationships were gross oversimplifications. However, as discussed above, to define these defensibly is difficult even if data are available. Table 15 captures an early dialogue across project team and DAF undertaken when considering how to quantify the elucidated environmental objectives. This illustrates the complexity of obtaining meaningful data and relating this to management controls.

In terms of evaluating TBL harvest strategies, therefore, it may be that the semi-quantitative approaches involving trade-off evaluations via expert judgment is more achievable. Similarly, using Bayesian Belief Networks to capture non-quantitative objectives in a probability-based manner may also be a more pragmatic approach. However, policy and legislation demand that we need to continue to move to a quantitative approach for reconciling TBL objectives and operationalising these defensibly within harvest strategies.

One way to simplify the incorporation of “all environmental and social aspects” is to do so via fewer, possibly broader, objectives, and then agreeing on a representative performance indicator for each. This would reduce the volume of data required, and has the added benefit of lessening the “dilution effect” of stakeholder preferences showing little difference across large numbers of objectives. However, it may be difficult to argue that this approach is capable of truly embracing all of the relevant environmental or social aspects.

One positive attribute of both approaches (semi-quantitative MCDA, and quantitative simulation) is that stakeholder preferences across the (often, conflicting) TBL objectives are formally integrated into the analyses. This circumvents the need for this to be done conceptually, which often leads to tension and management paralysis, or important objectives being effectively downweighted.

The triple bottom line may not only be embraced by the objectives and performance indicators against which the harvest strategy is evaluated, but explicitly by the design of the harvest strategy itself. For example, harvest strategy designs that were intended to directly address environmental concerns included pulling out separate TACs for species deemed to be at risk, having spatially explicit harvest control rules, and having overrides under exceptional circumstances, such as during cyclones. However, in a practical sense, the more complex and nuanced a harvest strategy, the more difficult and costly it is to manage and enforce. The extent of buy-in from fishers can also be compromised if they perceive the harvest strategy as being overly complex.

This raises the question as to whether fishery managers are better to be pragmatic and assume various environmental and social aspects are vicariously managed via simpler approaches. For example, it may be that fishery stock assessment focusing on fewer total objectives, a few key species and precautionary harvest control rules, coupled with risk assessments, may be adequate to passively protect the fishery as a whole. The use of “fixed” (or “set and forget”) decision rules (i.e. rules that do not change according to stock status), such as gear controls, or spatial or temporal restrictions can be useful in this context. Of course, this approach would still need to respond to chronic and acute environmental impacts, and consideration of the risk that some species and regions may respond in a different manner to others.

A final challenge in incorporating all environment and social aspects is in accepting what is within the direct control of a harvest strategy, which generally aims to control fishing mortality. There may be valid environmental and social aspects that are outside of this scope. For example, the social objective “Maximise utilisation of the retained catch of target species”, while a worthy objective, is unable to be influenced by harvest control rules, such as a TAC adjustment. Expectation management around what can and cannot be delivered by a TBL harvest strategy is paramount.

Table 15. Notes capturing the early discussion of how to interpret and define the elicited environmental objectives.

Overarching objective	Specific objectives		Description	Performance indicators	Performance indicators available?	Available - what data/information are/is available to inform PIs?	"Wish list" - how could we better measure the PI?
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 is about achieving an ecologically as well as an economically sustainable take for the main target species (noting that B_{MEY} is higher than B_{MSY}). It is also about building or maintaining the ecological resilience of stocks so they are better able to cope with shocks and environmental change. (need to ensure that the terms "MEY", "B_{MEY}", "MSY", "B_{MSY}" are all clearly defined so that all stakeholders understand them)	While biomass estimates may be able to be obtained for coral trout, and possibly RTE, via formal stock assessments, direct estimates of biomass may be more difficult for the other species, and indirect proxies (such as standardised catch rates, or catch relative to some reference level) may have to be used.	Logbooks & Quota reporting: landings per species, % landings live Coral Trout, annual fishing effort (Boat ramp survey info maybe able to be used to guide rec fishing catches) [CD comment: are we going to assume commercial CPUE reflects the stock and therefore will not need recreational CPUE indices? or do we need both which will bring its own issues of course]	comment: We have the Boat Ramp Survey program which collects recreational monthly CPUE data monthly for 45 boat ramps across Qld for 40 species including coral trout. May well avoid hyperstability issues associated with commercial fishery. We intend using it as an index of abundance. **Given this is only 2 years of data, may have to use commercial CPUE data for now, for all species, and augment with assessment estimates of abundance for CT and RTE, as available (ie in multi-indicator decision rules)**	
		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	This is about the gear and fishing activity imposing minimal risk to, or sustainably exploiting species that are not considered under 1.1.1, but are also harvested. See comments under 1.2.3		Logbooks: vessel characteristics, OS species landings [comment Need to also look at whether risk assessment PIs such as overlap of fishery with species distribution are available etc.]	**Combined CPUE from commercial logbook, PLUS/OR species composition ratios, PLUS/OR CPUE of key species, PLUS SAFE (as available)**	
	1.2 Ensure ecosystem resilience	1.2.1 Minimise risk to bycatch species	This is about minimising the impact of fishing on the broader ecosystem.	Performance indicators could be obtained through interviews (if trusted relationships exist) or observer programs. Fishing effort would be an indirect proxy, as discards would inherently scale with this. [comment: Experience is that non-retained bycatch is almost never recorded in logbooks. Even compulsory reporting of SOCI (Species of Conservation Interest) appears to be dodgy. Note definitions: "Byproduct" is retained, "Bycatch" is not. There may be some byproduct species that are not on the Other Species quota list, but uncertain.]	Logbooks: license endorsements [as per comment re. risk assessments. Assume 1.1.2 will try something like SAFE or Catch only and 1.2.1 use PSA like approaches] [comment: Doubt that logbook data will be useful for this.]	**PSA (by end of year); effort as indirect proxy**	**Interviews, observer programs**
		1.2.2 Minimise discard mortality (of undersized target species, or from high-grading of target species)	This is about minimising the mortality of fish that die as a result of having been captured and returned to the ocean. In the CRFF this is partly related to predators preying on stressed fish, and may also include barotrauma or other injuries. Discard mortality is unlikely to be controlled for by decision rules invoked on the basis of an assessment yielding performance indicators. It is more likely to be managed within the harvest strategy using "fixed" decision rules (i.e. rules that do not change according to stock status), such as gear controls, size and possession limits or rules around how discards are released.	Performance indicators could be obtained through interviews (if trusted relationships exist) or observer programs. Fishing effort would be an indirect proxy, as discards would inherently scale with this. [See comments above.]	Logbooks: discard records, and bycatch landings [comment: Doubt that logbook data will elucidate this.]	**Combination of effort and stock status: dsicarding probability scales directly with effort, but probability of discard of undersize increase as population declines. Also catch of undersize This is a very poor, indirect indicator, but this is one PI that should be targeted for the future. ?Check boat ramp surveys for weak estimate of % discard.**	**Interviews, observer programs. We have anecdotal evidence around sharks eating discards, but this doesn't speak to the amount of discarding.**
		1.2.3 Minimise broader ecological risks	This is about the gear and fishing activity having a minimal effect on the broader ecosystem. This includes compromising the food chain via the removal of target species (although this should be handled via 1.1.1 above), but more broadly includes direct interactions with habitat, non-targeted species. Again, this objective is unlikely to be controlled for by decision rules invoked on the basis of an assessment yielding performance indicators. It is more likely to be managed within the harvest strategy using "fixed" decision rules (i.e. rules that do not change according to stock status), such as gear controls, or spatial or temporal restrictions. That stated, it follows that such risks would directly correlate, and therefore scale with, with fishing effort.	Performance indicators of such interactions could be obtained through formal risk assessments, which would rely on logbook data, interviews or observer programs.	Spawning fishery closures	**JCU NERP study effects of fishing line on reef? Was successful. If ongoing, they were trying to develop a fishing impact index as a function of amount of gear left behind that could be monitored over time. Look at this and see what's coming out that could be useful and if study is ongoing. Otherwise, have to rely on risk assessment.**	
		1.2.4 Minimise risk to TEPS	This is about the gear and fishing activity avoiding unacceptable levels of risk to threatened, endangered or protected (TEP) species. Again, this objective is unlikely to be controlled for by decision rules invoked on the basis of an assessment yielding performance indicators. It is more likely to be managed within the harvest strategy using "fixed" decision rules (i.e. rules that do not change according to stock status), such as gear controls, or spatial or temporal restrictions. That stated, it follows that such risks would directly correlate, and therefore scale with, with fishing effort.	Performance indicators of such interactions could be obtained through formal risk assessments, which would rely on logbook data, interviews or observer programs.	Logbooks: TEPS interactions	**TEP interactions in logbooks; but beware increased reporting as a function of education and awareness. C'wealth found using camera validation that TEPs were worst reported in logbooks. Augment by PSA (given that a risk assessment will be undertaken anyway).**	
	1.3. Minimise risk of localised depletion	1.3.1. Due to fishing	This is about avoiding local or regional overfishing, regardless of overall stock status. Localised depletion occurs through effort being overly concentrated in a given area, either because the area is (for a range of possible reasons) deemed to be particularly attractive, or because effort is being displaced into the area from elsewhere.	This requires spatially-specific indicators. These may be indirect proxies –e.g. effort is moving further offshore due to suspected localised inshore depletion.	comment: Difficult one, but perhaps catch distribution relative to some reference years?	comment: BRS program provides regional CPUE index. **Catch and/or CPUE by region, relative to a reference year. Produce heat maps relative to reference heat map ito intensity (how hot?) and spatial extent (number of cells)**	
		1.3.2. In response to environmental event (e.g. cyclone, climate change)	This is about building in contingencies to minimise adverse effects around events over which there exist no local control. This may pertain more to the decision rule component of the harvest strategy, where measures such as stricter controls in response to environmental effects may come into play (while noting the possible risks associated with fishers relocating from affected areas).	The detection of effects of such perturbations could be through fishery-dependent indicators such as localised catch rates (in the case of single catastrophic events, such as cyclones), or, in the case of progressive effects such as climate change, through standardised fishery independent surveys of relative abundance	BoM wind-speed estimates, map of impact area, Qld Government wave-height measurements, water temperature measurements	**For isolated catastrophic event, can catch hot spot or cyclone as single event and flag management strategy - big event has happened and this area is declared as is on watching brief. Use previous indicators under 1.3 (CPUE by region - moving away from areas and into new areas) to look for localised depletion and monitor that area explicitly. In terms of gradual effect of climate change, then would just be a matter of monitoring target species indicators and reacting ito decision rules being more precautionary in context of climate climate.**	

Experience in engaging with DAF given major management reforms

The project was limited in its ability to provide a recommended, operational TBL harvest strategy in time for the Queensland DAF deadline at the end of 2018.

The project team led the process of eliciting, identifying, and fleshing the details of, alternative harvest strategy options, and the MCDA approach was completed in time to assist the selection of the initial harvest strategy of choice.

Ideally, the outcomes and recommendations from this project will be able to more comprehensively assist DAF, and the CRFFF in particular, at the point of the first formal review of the harvest strategies. For the CRFFF, the first formal review is scheduled for 2024.

Conclusion

A literature review of (existing work around) multi-objective management systems and associated assessment approaches underpinned the choice of approaches taken for this project. The results highlighted that, while triple bottom line harvest strategies have become mainstream in science and policy, they have yet to be routinely operationalised. However, to date this has been largely limited to a conceptual treatment or semi-quantitative approaches. The review of technical approaches to evaluate TBL harvest strategies is summarised in the General Methodology such that the trade-offs between them are transparent.

We interpreted “triple bottom line harvest strategies” as an obligation to evaluating harvest strategy performance in the context of all objectives, as well as being cognisant of TBL issues when designing harvest strategies. (As opposed to e.g. monitoring environmental aspects, putting in indirect control rules like buffers to vicariously protect these).

We formally appraised a short-list of fishery options to select the Queensland Coral Reef Finfish Fishery as the choice of Queensland state-based multi-sector case study fishery, and applied both approaches to develop triple bottom line harvest strategy frameworks.

The inventory of current environmental, economic and social objectives formed the basis of our approach to elicit TBL objectives for the CRFFF. This inventory will hopefully be a useful and comprehensive reference and starting point for identifying a relevant subset for specific fisheries. It may assist in translating conceptual management objectives to operational objectives for multi-sector fisheries. The project team presented a “pared down” version of 75 objectives which formed the basis for engagement with the CRFFF WG. Through an iterative process of consultation, 21 TBL harvest strategy objectives, as well as numerous, “management regime” objectives (those that were outside of the direct control of the harvest strategy) were identified.

To identify priorities for environmental, economic and social objectives, we undertook a modified Analytic Hierarchy Process (AHP) based on a series of pair-wise comparisons. This enabled individual preference weightings to be obtained from over 100 stakeholders. These were aggregated according to the group with which the stakeholder most closely identified. All groups tended to rank the ecological sustainability objectives the highest. However, the weightings on the lower level objectives appeared fairly similar across the different stakeholder groups. This is an artefact to some extent of the “dilution” effect of distributing the higher level objective weights over many sub-objectives. This suggests that there is there is a trade-off between capturing all TBL concerns using very explicitly defined sub-objectives, and having too many such that preferences are diluted.

In direct consultation with the CRFFF WG, we identified six alterative “harvest strategies” that built on the current management arrangements to explicitly address key environmental, economic and social concerns. To the extent possible, details were fleshed out around each option, while clearly identifying outstanding issues or decisions associated with each. The “modified status quo” was accepted by Fisheries Queensland as the interim harvest strategy for the CRFFF given the December 2018 deadline.

Based on the results of the literature review and the collective expertise of the project team, we developed two alternative approaches with which to evaluate trade-offs between triple bottom line objectives and stakeholder preferences: a semi-quantitative multi-criteria decision analysis framework, and a quantitative simulation model approach.

The MCDA approach:

- Derived relative impacts against each objective from expert opinion
- Required iterative consultation with members of the CRFFF WG
- Applied weights for the individual stakeholders
- Conceptually considered the TBL trade-offs they perceived to be associated with the alternative harvest strategies identified by the WG
- Derived a subjective probability distribution of net benefits of each option, and
- Gave mean outcome by stakeholder group.

The quantitative simulation approach:

- Used a spatially-, species group- and fishery sector- explicit model to estimate outcomes
- Was data-hungry and requires the ability to quantitatively define each performance indicator
- Assumed perfect knowledge (i.e. there were no assessment model) and no random errors
- Obtained optimised TACs for each year based on maximising a weighted value function based on stakeholder group weights applied to all objectives, and obtaining an overall optimum across stakeholder groups
- Considered some, but not all, of the harvest strategy options identified by the WG
- Made strong assumptions that social outcomes would be linked to catch and effort
- Evaluated impacts based on deviations from status quo scenarios
- Explicitly considered cyclones and climate change effects in a simplified manner in specific scenarios, and
- Considered 25 years of projections.

Both approaches were able to operationalise triple bottom line harvest strategies.

The MCDA semi-quantitative approach carries the inherent risk associated with qualitative expert opinion. The quantitative simulation model has inherent uncertainties associated with data gaps and assumptions. In particular, the simulated performance indicators corresponding to many of the objectives, and especially the social objectives, were crude and highly simplified guesses that assumed that these metrics were directly related to catch, effort or CPUE.

We concluded that the quantitative simulation can only currently be considered as an illustrative proof-of-concept. While novel in its ability to consider multiple objectives in terms of their (often, assumed) relationship to harvest controls (and hence, catch effort, or CPUE), and its ability to optimise across the range of difference stakeholder group weightings, it is highly data-hungry and requires the ability to quantitatively define each performance indicator. While uncertainties and assumptions can be objectively explored via sensitivity analyses, right now, our quantitative approach is not ready to be used for management advice. To do so requires a clear review and evaluation of the kinds of information required and how stakeholders wish to quantitatively translate their TBL objectives into operational objectives and performance indicators.

On the other hand, qualitative/semi-quantitative approaches have demonstrated that managers can “have it all” in terms of operationalising TBL harvest strategies. While they are based on expert opinion, the level of stakeholder engagement and the sense of ownership and accountability conferred by such approaches may, to some extent, avoid the liability around “getting it wrong”.

That stated, policy and legislation demand that we need to continue to move to a quantitative approach for reconciling TBL objectives and operationalising these defensibly within harvest strategies. The quantitative approach provides a means forward, albeit one that will force managers and stakeholders to confront the substantive associated data requirements.

Based on the case study application, and acknowledging that, per the literature review, a range of theoretical approaches to TBL evaluation is available, we developed a concise and accessible General Methodology for harvest strategy development against the triple bottom line for multi-sector fisheries. The General Methodology summarises our learnings in such a way that practitioners can apply the process of developing TBL harvest strategies to other fisheries.

Implications

As ecosystem-based fisheries management becomes more common, as fisheries management reforms in Queensland progress, and for many fisheries confronted directly by environmental impacts, decision makers and managers increasingly face triple bottom line trade-offs. To date, TBL harvest strategy evaluation has been largely limited to a conceptual treatment (e.g. Stephenson et al. 2017) or semi-quantitative approaches (Dichmont et al., 2012).

In terms of Queensland fisheries, this project has:

- Demonstrated a clear process to identifying TBL objectives and weight these across stakeholders
- Assisted with developing harvest strategy options for the CRFFF
- Demonstrated two different approaches to operationalising TBL harvest strategies
- Provided advice and guidance to the management reform process
- Provided a comprehensive inventory of TBL objectives, and
- Provided a General Methodology that provides accessible guidance to managers faced with developing TBL harvest strategies.

The process of eliciting and weighting TBL objectives has the added benefit of providing an impartial and standardised means to avoid different stakeholders or stakeholder groups pushing for their own agendas.

This project has embraced two alternate approaches to operationalising the triple bottom line within fishery harvest strategies, using the Coral Reef Finfish Fishery as a case study.

1. The use of a semi-quantitative multi-criteria decision analysis (MCDA) and expert opinion.

2. A quantitative simulation with a multi-objective value function that inherently optimises over both the objectives as well as different stakeholder group weightings/preferences.

Both approaches are ultimately able to operationalise TBL harvest strategies. However, in terms of implications, what is more important are the similarities and differences between the two approaches and what is required to operationalise them.

MCDA approaches offer a pragmatic approach to assess management options from a multi-objective perspective. 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. They introduce additional uncertainty through different subjective cognitive models and biases. However, these cognitive models may better reflect the combination of drivers of behaviour and outcomes in the fishery. Differences in opinion may be considered as uncertainty, and probabilistic outcomes determined. The semi-quantitative MCDA approach already has a successful precedent in several cases as noted in the literature review.

On the other hand, the quantitative simulation approach explicitly advises on the levels of management control(s) that yield optimal TBL outcomes. It can also consider multiple objectives in terms of their relationship to harvest controls and provide optimal strategies both across the range of objectives, and alternative stakeholder group objective weightings. That stated, a fully conditioned and optimised TBL management strategy evaluation model for practical implementation is further away, due to:

- The data required to inform the performance indicators corresponding to each objective
- The assumptions and proxies around the nature of the relationship between each performance indicator and the harvest strategy
- The uncertainties around the parameterisation of these relationships (in terms of targets, limits, and threshold values, weightings, rate parameters), and
- The fact that management changes also can affect fisher behaviour, and these are also difficult to capture in a model.

The data required for a quantitative approach makes extra demands on existing monitoring programs. Multi-sector, multi-species fisheries such as the CRFFF are most directly confronted with the TBL; yet their data quantity and quality are often mixed; reference points and performance indicators vary between them; and environmental, economic, and social information for both sectors is often limited.

Quantitative TBL approaches also require a coordinated approach to monitoring and data collection, particularly for environmental and social data, because this needs to be in such a currency that can relate to the harvest strategy. Attempting to incorporate all environmental aspects into TBL harvest strategies is challenging because, while environmental data may exist, this may be unable to be readily related to how the environmental aspect impacts with the fish stock, and hence how this interacts with harvest control rules. Similarly, existing social metrics may be difficult to relate back to how they are impacted by adjustments to fishing mortality.

On the other hand, semi-quantitative approaches cannot explicitly specific management lever (in this case, TAC) values and inter-annual adjustments. Rather, these are agreed upon through a process of expert judgement. However, policy and legislation demand that we need to continue to move to a quantitative approach for reconciling TBL objectives and operationalising these defensibly within harvest strategies.

Several other approaches were reviewed within the project and could have been applied (Table 14). As such, the emphasis of future work is likely to be on refinement of existing approaches, rather than on new approaches.

Overall, operationalising the TBL in a harvest strategy is demonstrably possible, but it is clearly early days and much work remains to be done.

Recommendations

Given that the CRFFF harvest strategy was required to have been in place by the end of 2018, the first formal review of the harvest strategy in 2024 will be the first opportunity to consider incorporating the TBL objectives and approaches described within this project.

TBL harvest strategy development is non-trivial. Regardless of the approach taken (semi-quantitative, or quantitative simulation model), this work has 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. Such teams may be economically and logistically challenging to organise. A key factor in the choice of the CRFFF as the case study fishery for this project was the existence of an active fishery Working Group. Regardless, a "bottom-up" style of engagement, whereby stakeholders are an integral part of the harvest strategy development process from the point of inception, is critical to success. That stated, stakeholders require careful direction in order to effectively elicit and weight objectives and alternate harvest strategies. The expert input required to any triple bottom line approach requires the ongoing commitment of a dedicated identified expert panel.

The number of TBL objectives should be reviewed with the aim of capping these at a lower number than the 21 considered here (say, 10). This is both to reduce both the "dilution effect" of preference weightings not showing strong difference across objectives, and to make the harvest strategy practical and workable in terms of its monitoring requirements and evaluation. The sensitivity analyses undertaken as part of the quantitative simulation can provide guidance as to which performance indicators may be more robust and therefore may be able to be excluded.

The maximum number of objectives is currently arbitrary, but should balance the observed need for more than one environment, economic and social objective, against the observed "dilution effect" that resulted from having 21 objectives. While an optimal number of objectives could theoretically be formally tested, this would require multiple rounds of stakeholder questionnaires and was beyond the scope of this project.

In terms of the CRFFF, data and stock assessments suggest that all three species groups are above biomass levels corresponding to 60% of the unfished biomass. This was reflected in the simulations with the corresponding performance indicators at or close to their maximum values. Additionally, related sustainability objectives will always have strong weightings as these are bound by legislative requirements to utilise fishery resources in an ecologically sustainable manner. Indeed, the environmental indicators consistently had the highest overall weightings across the four TBL pillars, for all stakeholder groups surveyed.

For the CRFFF, the TBL emphases are therefore more likely to be on the economic, social and institutional/management pillars, but also on external environmental factors. The impact of key environmental aspects, such as climate change and cyclones, on the species of interest needs to be more directly quantified, to permit better evaluation of how control rules may best be able to mitigate against adverse effects.

Additionally, the relationships between catch, effort and CPUE and the various social performance indicators need to be better defined. Where social data is collected (such as employment levels within a community), it is difficult to understand how these may be affected in response to harvest strategy management controls, such as an increase or decrease in the TAC.

This points to the need for a coordinated approach to monitoring, centred on data that ultimately enables the calculation of the performance indicators corresponding to all TBL objectives.

The use of Bayesian Belief Networks (BBNs) to capture non-quantitative objectives may be a sensible compromise that avoids the need to define explicit relationships between performance indicators and management controls whilst still evaluating objectives in a semi-quantitative probability-based manner (i.e. the outputs of a quantitative model would feed into a BBN model). This approach warrants further investigation.

Importantly, TBL HS evaluation needs to consider longer-term evaluation, as opposed to the short-term optimisation undertaken using the simulation modelling approach. Within the simulation, a multi-year forward optimisation process would have been preferable. This should be a priority for the future development and improvement of the simulation approach.

More generally, there must be a balance between developing TBL-defensible harvest strategies and what is practical, cost-effective and achievable. A key priority should be to identify which of the TBL objectives (which can directly include institutional objectives such as the cost of management) and which external environmental factors are most critical, not only in terms of stakeholder weightings, but in terms of their perceived sensitivity (for the former) and impact (for the latter), and to focus efforts around these.

Due to their costs and data requirements, TBL harvest strategies are unlikely to be able to be developed for small-scale, low-value fisheries, many of which lack adequate data to undertake a stock assessment on the main target species. Cost-benefit trade-offs must be considered in the context of developing a TBL harvest strategy.

Finally, this work needs to be seen to be used. The early work of (Fletcher et al., 2002) set out an Environmentally Sustainable Design (ESD) framework which most fisheries jurisdictions have adopted in various forms. This included ecological, economic and social aspects. Western Australia implemented a form of TBL ESD framework (Caputi et al., 2018) wherein managers weighted TBL objectives. The approaches taken in this project to directly incorporate the triple bottom line in harvest strategies should form the basis for further, coordinated approaches moving forward.

Further development

While we have shown an approach to eliciting and weighing TBL objectives, and have outlined a General Methodology, as well as alternative harvest strategy options for the case study fishery, we have not advocated any one approach to TBL HS evaluation.

For now, the semi-quantitative MCDA approach is pragmatic, achievable, and has a successful precedent in a wide range of fisheries, natural resource and environmental management applications. However, the simulation model addresses the need to move to a quantitative approach for reconciling TBL objectives and operationalising these defensibly within harvest strategies, but there are substantive associated data and analytical requirements.

Moreover, this was primarily a methodological project focused on providing proof of concept around alternative approaches to operationalising TBL harvest strategies. Both approaches are ultimately able to operationalise TBL HSs. The semi-quantitative MCDA approach already has a successful precedent in many cases, including applications in Australia, and, pending the identification and commitment of a dedicated Working Group with broader representation, and the ability to meet the associated costs, be able to be implemented. A fully conditioned and optimised TBL assessment simulation model for practical implementation is further away. The following points highlight key areas for further development:

- The preference weighting exercise should be periodically repeated, as preference profiles can change over time and in different contexts.

- The various harvest strategy options still require significant work to be implementable. In particular, the form of the control rules, including the magnitude and nature of their needs to be fully articulated.

- The quantitative 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.

- If a quantitative TBL evaluation approach is to be pursued, then this needs to be evaluated in the context of a full MSE that integrates the existing available stock assessments for CT and RTE, and likely a more comprehensive operating model that can better account for spatial, fleet and population dynamics. The ELFSim operating model would be ideal, but this is currently limited to Coral Trout.

- The translation of conceptual to operational TBL objectives and performance indicators was done at the discretion of the project team. If a quantitative TBL evaluation is to be undertaken, stakeholders need to review and approve the translation of TBL conceptual objectives to operational objectives

- The performance indicators corresponding to each of the operational objectives are fraught with assumptions and uncertainties, both about the nature of the relationship between the indicator and the catch, effort or fish population size, and about their parameterisation.

- If a quantitative TBL evaluation is pursued, consider developing a Bayesian Belief Network (BBN) to capture non-quantitative objectives in a probability-based manner. The outputs of the operating model would feed into the BBN model to quantify the social components.

Extension and Adoption

Two fishery managers and one scientist from DAF were directly involved as part of the project team, resulting in a strong collaboration with this agency.

The project team worked closely with the Coral Reef Finfish Fishery Working Group, and DAF generally, during the course of the project. The project team was granted dedicated time with the Working Group at all meetings during 2017 and 2018, with the Working Group proving key inputs around objective elicitation, objective preference weighting, and harvest strategy shortlisting, in particular. The Working Group of 18 members included a good cross section of key science, management, conservation, and industry stakeholders involved in harvest strategy development in Queensland. During 2018, DAF managers also made in-person petitions to industry stakeholders at various key ports to complete the objective preference weighting survey. We ultimately received 110 responses, of which around half were from commercial fishers.

The key outcomes of the project, structured around the General Methodology, were presented to managers, scientists and stakeholders nationally as a series of two, two-hour online interactive “webinars” in September 2020. Each live session attracted between 30 and 40 participants from across state and Commonwealth jurisdictions. The presentations were recorded and at least 15 downloads have occurred to date. NSW FRI have expressed an interest in the project outcomes as they move to formal harvest strategy development for their fisheries. NT Fisheries were also well represented at the live webinars. A key part of the webinar presentation was the representation of the General Methodology as a cartoon flowchart, developed by Dr. Sue Pillans (Figure 31). This assisted with the dissemination of key points, and to maximise the visual impact of the online presentation. Queensland DAF will play a key role in promoting the adoption of the TBL methodology and approaches developed as part of this project. As outlined above under “Further Development”, there is a gap of work between this project and adoption. This includes scoping the associated costs of the alternative approaches and determining whether these are financially feasible. TBL approaches may be adopted when the initial harvest strategies that were established in response the Queensland Fishery Management Reforms are formally reviewed in 2024

From the Queensland CRFFF fishery manager's perspective, the case study demonstrated the need to be in the right space at the right time for an engagement of the type required to develop of TBL HSs. Conditions that should be met include stability in management, where no reforms are underway, and no significant reforms are needed to implement the harvest strategy. This provides less distractions and increases the likelihood that stakeholders will be engaging in a proactive rather than a reactive manner. This project coincided with the new policy framework of the Sustainable Fisheries Strategy 2017-2027. This introduced: i) harvest strategies (plus overarching policy/guidelines), ii) vessel tracking, iii) regulatory reforms; iv) new monitoring (of boat ramps and species, a recreational survey, and economic/social data collection); v) a framework for consultation - working groups, expert panel; and vi) a policy impetus of "60% biomass target". This made stakeholder engagement somewhat difficult as their focus was primarily on these new policy issues, and resulted in some lack of understanding of the end point of the process (e.g. of how were all sectors were to be included). Critically, the discussion and implementation of the new reforms imposed significant constraints on the CRFFF working group time, that could otherwise have been more focused on the TBL HS engagement.

As such, the Queensland CRFFF fishery manager's recommendations included that the existing management framework should, ideally, be mature (as the existing management framework influences the available harvest control rule levers). It was therefore also recommended that a fully explicit TBL harvest strategy should not be the first harvest strategy a fishery should ever have, but rather, a second or third generation of harvest strategy where the stakeholders and the management framework are very settled. In this context, it is also helpful if the fishery is in a healthy, non-contentious space, with the stocks to be managed being already at or near their sustainable targets. Fisheries Queensland also agreed that a high number of objectives is fraught, because it becomes difficult to keep stakeholder expectations in check. The need for quality data, including economic, social and recreational data, was also acknowledged.

Ultimately, the implemented harvest strategy objectives were driven by:

- the timeframe for delivering a harvest strategy (versus the longer timeline of the project)
- the need for consistency across all Queensland fisheries, and the legislated target reference point of 60% of the unfished stock biomass.
- the fact that it was more practical to adopt a hierarchical set of objectives, where the stock status was paramount
- the limited ability to pull management levers relating to social/economic measures.

Early in the project, the project team provided general guidance and feedback in the drafting stages of the Queensland Fisheries Sustainable Fisheries Strategy 2017-2027. The team gave presentations of suites of tools that may have been of interest, and the PI gave a dedicated workshop on the FishPath harvest strategy decision support software and engagement process.

The following conference presentations have been given:

- Hjort Symposium: Dichmont, C.M., Dowling, N.A., Pascoe, S., Cannard, T., Pears, R.J., Breen, S., Roberts, T., Leigh, G.M., Mangel, M. Operationalising Triple Bottom Line Harvest Strategies.

- Pascoe, S., Cannard, T., Dowling, N., Dichmont, C.M., Hutton, T. 2019. Integrating economic and social objectives in marine resource management: Australian experiences. Invited PICES presentation

The main results from the project have been published as follows:

- Pascoe, S., Cannard, T., Dowling, N.A., Dichmont, C.M., Breen, S., Roberts, T., Pears, R.J., Leigh, G.M. (2019). Developing Harvest Strategies to Achieve Ecological, Economic and Social Sustainability in Multi-Sector Fisheries. Sustainability doi:10.3390/su11030644

- Dichmont, C.M., Dowling, N.A., Pascoe, S., Cannard, T., Pears, R.J., Breen, S., Roberts, T., Leigh, G.M., Mangel, M. (2020). Operationalising Triple Bottom Line Harvest Strategies. ICES Journal of Marine Science https://doi.org/10.1093/icesjms/fsaa033

- Dowling, N.A., Dichmont, C.M., Leigh, G.M., Pascoe, S., Pears, R.J., Roberts, T., Breen, S., Cannard, T., Mamula, A., Mangel, M. 2020). Optimising triple bottom line harvest strategies over multiple objectives and stakeholder preferences. Ecological Modelling 435 https://doi.org/10.1016/j.ecolmodel.2020.109243

Project coverage

Not applicable.

References

Aguarón, J., and Moreno-Jiménez, J. M. a. 2000. Local stability intervals in the analytic hierarchy process. European Journal of Operational Research, 125: 113-132.

Aguarón, J., and Moreno-Jiménez, J. M. a. 2003. The geometric consistency index: Approximated thresholds. European Journal of Operational Research, 147: 137-145.

Ahn, B., Lee, B., and Shafer, C. S. 2002. Operationalizing sustainability in regional tourism planning: an application of the limits of acceptable change framework. Tourism Management, 23: 1-15.

Andalecio, M. N. 2011. Multi-Criteria Decision Models for Management of Tropical Coastal Fisheries. *In* Sustainable Agriculture Volume 2, pp. 251-280. Ed. by E. Lichtfouse, M. Hamelin, M. Navarrete, and P. Debaeke. Springer Netherlands, Dordrecht.

Anderson, J. L., Anderson, C. M., Chu, J., Meredith, J., Asche, F., Sylvia, G., Smith, M. D., et al. 2015. The Fishery Performance Indicators: A Management Tool for Triple Bottom Line Outcomes. PLoS ONE, 10: e0122809.

Auster, P. J., Fujita, R., Kellert, S. R., Avise, J., Campagna, C., Cuker, B., Dayton, P., et al. 2009. Developing an Ocean Ethic: Science, Utility, Aesthetics, Self-Interest, and Different Ways of Knowing. Conservation Biology, 23: 233-235.

Baby, S. 2013. AHP Modeling for Multicriteria Decision-Making and to Optimise Strategies for Protecting Coastal Landscape Resources. International Journal of Innovation, 4: 218-227.

Bebbington, J., Brown, J., and Frame, B. 2007. Accounting technologies and sustainability assessment models. Ecological Economics, 61: 224-236.

Benlian, A. 2011. Is traditional, open-source, or on-demand first choice? Developing an AHP-based framework for the comparison of different software models in office suites selection. European Journal of Information Systems, 20: 542-559.

Benson, A. J., and Stephenson, R. L. 2018. Options for integrating ecological, economic, and social objectives in evaluation and management of fisheries. Fish and Fisheries, 19 (1): 40-56.

Bernstein, G. A., and Cetron, M. J. 1969. SEER: A Delphic approach applied to information processing. Technological Forecasting, 1: 33-54.

Bodin, L., and Gass, S. I. 2003. On teaching the analytic hierarchy process. Computers & Operations Research, 30: 1487-1497.

Bond, A. J., Dockerty, T., Lovett, A., Riche, A. B., Haughton, A. J., Bohan, D. A., Sage, R. B., et al. 2011. Learning How to Deal with Values, Frames and Governance in Sustainability Appraisal. Regional Studies, 45: 1157-1170.

Bond, A. J., and Morrison-Saunders, A. 2011. Re-evaluating Sustainability Assessment: Aligning the vision and the practice. Environmental Impact Assessment Review, 31: 1-7.

Bottomley, P. A., and Doyle, J. R. 2001. A comparison of three weight elicitation methods: good, better, and best. Omega, 29: 553-560.

Bottomley, P. A., Doyle, J. R., and Green, R. H. 2000. Testing the reliability of weight elicitation methods: Direct rating versus point allocation. JMR, Journal of Marketing Research, 37: 508-513.

Bousquet, F., and Le Page, C. 2004. Multi-agent simulations and ecosystem management: a review. Ecological Modelling, 176: 313-332.

Brooks, K., Schirmer, J., Pascoe, S., Triantafillos, L., Jebreen, E., Cannard, T., and Dichmont, C. M. 2015. Selecting and assessing social objectives for Australian fisheries management. Marine Policy: 111–122.

Buckworth, R. C., Deng, R. A., Plagányi, E. E., Punt, A., Upston, J., Pascoe, S., Miller, M., et al. 2015. Northern Prawn Fishery RAG Assessments 2013-15. Final Report to the Australian Fisheries Management Authority, Research Project 2013/0005, June 2015. CSIRO, Brisbane, 177 pp.

Bureau of Meteorology 2019. The Australian Tropical Cyclone Database http://www.bom.gov.au/cyclone/history/index.shtml, Accessed on 10 August 2019.

Burgman, M., and Possingham, H. 2000. Population viability analysis for conservation: the good, the bad and the undescribed. *In* Genetics, Demography and Viability of Fragmented Populations, pp. 97-112. Ed. by A. G. Young, and G. M. Clarke. Cambridge University Press, Cambridge.

Butterworth, D. S., and Punt, A. 2003. The role of harvest control laws, risk and uncertainty and the precautionary approach in ecosystem-based management. *In* Responsible fisheries in the marine ecosystem, pp. 311-319. Ed. by M. Sinclair, and G. Valdimarsson. FAO and CABI Publishing, Rome and New York.

Caddy, J. 2009. Practical issues in choosing a framework for resource assessment and management of Mediterranean and Black Sea fisheries. Mediterranean Marine Science, 10: 83-120.

Caddy, J. F. 2004. Current usage of fisheries indicators and reference points, and their potential application to management of fisheries for marine invertebrates. Canadian Journal of Fisheries and Aquatic Sciences, 61: 1307-1324.

Caddy, J. F., Wade, E., Surette, T., Hebert, M., and Moriyasu, M. 2005. Using an empirical traffic light procedure for monitoring and forecasting in the Gulf of St. Lawrence fishery for the snow crab, Chionoecetes opilio. Fisheries Research, 76: 123-145.

Caputi, N., de Lestang, S., How, J., Trinnie, F., and Fletcher, W. 2018. Ecosystem-based fisheries management (or 'triple bottom line') assessments of the western rock lobster resource: Is there an optimal target for fishing? Marine Policy, 94: 264-274.

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Diaz, S., Dietz, T., et al. 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proceedings of the National Academy of Sciences, 106: 1305-1312.

Cegan, J. C., Filion, A. M., Keisler, J. M., and Linkov, I. 2017. Trends and applications of multi-criteria decision analysis in environmental sciences: literature review. Environment Systems and Decisions, 37: 123-133.

Charles, A. T. 1989. Bio-socio-economic fishery models: labour dynamics and multi-objective management. Canadian Journal of Fisheries and Aquatic Sciences, 46: 1313-1322.

Chesson, J., Clayton, H., and Whitworth, B. 1999. Evaluation of fisheries-management systems with respect to sustainable development. ICES Journal of Marine Science, 56: 980-984.

Chesson, J., and Whitworth, B. 2004. A Stocktake of Environmental, Socio-economic and Governance Performance Assessment Systems in use by Australian Governments. *In* Final report to the National Oceans Office, p. 199 pp. Bureau of Rural Sciences, 199 pp., Canberra, Australia.

Cinner, J. 2014. Coral reef livelihoods. Current Opinion in Environmental Sustainability, 7: 65-71.

Clark, C. W., and Munro, G. R. 2017. Capital Theory and the Economics of Fisheries: Implications for Policy. Marine Resource Economics, 32: 123-142.

Cochrane, K. L. 2000. Reconciling sustainability, economic efficiency and equity in fisheries: the one that got away? Fish and Fisheries:, 1(1): 3-21.

Cochrane, K. L. 2002. ed. A fishery manager's guidebook: management measures and their application (No. 424). . Food & Agriculture Organisation, Chapter 1.

Colapinto, C., Jayaraman, R., and Marsiglio, S. 2017. Multi-criteria decision analysis with goal programming in engineering, management and social sciences: a state-of-the art review. Annals of Operations Research, 251: 7-40.

COMLAW 2008. Great Barrier Reef Marine Park Act 1975 Act Compilation - C2008C00633 Attorney-General's Department, Canberra, Australia. Available online via http://www.austlii.edu.au/au/legis/cth/consol_act/gbrmpa1975257/

Commonwealth of Australia 2007. Environment Protection and Biodiversity Conservation Act 1999. Act No. 91 of 1999 as amended: 353 pp.

Costello, C., Ovando, D., Clavelle, T., Strauss, C. K., Hilborn, R., Melnychuk, M. C., Branch, T. A., et al. 2016. Global fishery prospects under contrasting management regimes. Proceedings of the National Academy of Sciences, 113: 5125.

Coulthard, S. 2012. What does the debate around social wellbeing have to offer sustainable fisheries? Current Opinion in Environmental Sustainability, 4: 358-363.

Courtney, A. J., Spillman, C. M., Lemos, R. T., Thomas, J., Leigh, G. M., and Campbell, A. B. 2015. Physical Oceanographic Influences on Queensland Reef Fish and Scallops. FRDC Final Report project no. 2013/020.

Cowx, I. G., and Van Anrooy, R. 2010. Social, economic and ecological objectives of inland commercial and recreational fisheries. Fisheries Management and Ecology, 17: 89-92.

Crawford, G., and Williams, C. 1985. A note on the analysis of subjective judgment matrices. Journal of Mathematical Psychology, 29: 387-405.

DAFF 2007. Australian Fisheries Harvest Policy: Policy and Guidelines. Department of Agriculture, Fisheries and Forestry (DAFF), Canberra.

Davis, A., and Wagner, J. 2006. A right to fish for a living? The case for coastal fishing people's determination of access and participation. Ocean & Coastal Management, 49: 476-497.

Dawes, J. 2008. Do data characteristics change according to the number of scale points used? An experiment using 5-point, 7-point and 10-point scales. International journal of market research, 50: 61-104.

De Boni, A., Roma, R., and Ottomano Palmisano, G. 2018. Fishery policy in the European Union: A multiple criteria approach for assessing sustainable management of Coastal Development Plans in Southern Italy. Ocean & Coastal Management, 163: 11-21.

de la Mare, W. K. 1998. Tidier Fisheries Management needs a new MOP (Management Oriented Paradigm). Reviews in Fisheries and Fish Biology, 8: 349-356.

De Vaus, D. 2013. Surveys in Social Research, Routledge, Crows Nest, Australia.

Department of Agriculture and Fisheries. 2017. Queensland Sustainable Fisheries Strategy 2017-2027.

DEWHA 2003. Triple Bottom Line Reporting in Australia: A Guide to Reporting Against Environmental Indicators Environment Australia, June 2000. ISBN 0 642 54937 0 Available at: http://www.environment.gov.au/archive/settlements/industry/finance/publications/indicators/pubs/indicators.pdf.

Dichmont, C., Pascoe, S., Jebreen, E., Pears, R., Brooks, K., and Perez, P. 2012a. Providing social science objectives and indicators to compare management options in the Queensland trawl planning process.

Dichmont, C., Pascoe, S., Jebreen, E., Pears, R., Brooks, K., and Perez, P. 2012b. Providing social science objectives and indicators to compare management options in the Queensland trawl planning process. CSIRO. Brisbane.

Dichmont, C. M., Dutra, L. X. C., Owens, R., Jebreen, E., Thompson, C., Deng, R. A., van Putten, E. I., et al. 2016. A generic method of engagement to elicit regional coastal management options. Ocean & Coastal Management, 124: 22-32.

Dichmont, C. M., Dutra, L. X. C., van Putten, I., Deng, R. A., Owens, R., Jebreen, E., Thompson, C., et al. 2014. Design and implementation of Management Strategy Evaluation for the Great Barrier Reef inshore (MSEGBR). Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns: 284pp.

Dichmont, C. M., Ellis, N., Bustamante, R. H., Deng, R., Tickell, S., Pascual, R., Lozano-Montes, H., et al. 2013a. Evaluating marine spatial closures with conflicting fisheries and conservation objectives. Journal of Applied Ecology, 50: 1060-1070.

Dichmont, C. M., Pascoe, S., Jebreen, E., Pears, R., Brooks, K., and Perez, P. 2013b. Choosing a fishery's governance structure using data poor methods. Marine Policy, 37: 123-131.

Dichmont, C. M., Pascoe, S., Kompas, T., Punt, A. E., and Deng, R. 2010. On implementing maximum economic yield in commercial fisheries. Proceedings of the National Academy of Sciences, 107: 16-21.

Dichmont, C. M., Punt, A. E., Dowling, N., De Oliveira, J. A., Little, L. R., Sporcic, M., Fulton, E., et al. 2015. Is risk consistent across tier-based harvest control rule management systems? A comparison of four case-studies. Fish and Fisheries.

Dickey-Collas, M. 2014. Why the complex nature of integrated ecosystem assessments requires a flexible and adaptive approach. ICES Journal of Marine Science: Journal du Conseil.

Doyle, J. R., Green, R. H., and Bottomley, P. A. 1997. Judging Relative Importance: Direct Rating and Point Allocation Are Not Equivalent. Organizational Behavior and Human Decision Processes, 70: 65-72.

Dutra, L. X. C., Dichmont, C. M., van Putten, I. E., Thébaud, O., Deng, R. A., Pascual, R., Owens, R., et al. 2016. How important is the coast? A survey of coastal objectives in an Australian regional city. Marine Policy, 71: 229-241.

Dutra, L. X. C., Thébaud, O., Boschetti, F., Smith, A. D. M., and Dichmont, C. M. 2015. Key issues and drivers affecting coastal and marine resource decisions: Participatory management strategy evaluation to support adaptive management. Ocean & Coastal Management, 116: 382-395.

Elkington, J. 1997. Cannibals with Forks: The Triple Bottom Line of 21st Century Business, New Society Publishers, 416 pp, Gabriola Island, Canada.

Elkington, J. 2006a. Governance for Sustainability. Corporate Governance: An International Review, 14: 522-529.

Elkington, J. 2006b. Governance for sustainability. Corporate Governance-an International Review, 14: 522-529.

Elkington, J. 2018. 25 years ago I coined the phrase "Triple Bottom Line." Here's why it's time to rethink it. Harvard Business Review.

Emery, T. J., Gardner, C., Hartmann, K., and Cartwright, I. 2017. Incorporating economics into fisheries management frameworks in Australia. Marine Policy, 77: 136-143.

Enríquez-Andrade, R. R., and Vaca-Rodríguez, J. G. 2004. Evaluating ecological tradeoffs in fisheries management: a study case for the yellowfin tuna fishery in the Eastern Pacific Ocean. Ecological Economics, 48: 303-315.

FAO 1995. Code of Conduct for Responsible Fisheries. Available at: ftp://ftp.fao.org/docrep/fao/005/v9878e/v9878e00.pdf.

Farmery, A. K., Ogier, E., Gardner, C., and Jabour, J. 2019. Incorporating ecologically sustainable development policy goals within fisheries management: An assessment of integration and coherence in an Australian context. Journal of environmental management, 249: 109230.

Firestone, J., Hadders, H., and Cavaleri, S. 2009. Measuring Organizational Sustainability Performance: the Adaptive Quadruple Bottom Line Scorecard. Proceedings of the 6th International Conference on Intellectual Capital, Knowledge Management & Organisational Learning: 124-131.

Fletcher, W., Chesson, J., Fisher, M., Sainsbury, K., Hundloe, T., Smith, A., and Whitworth, B. 2002. National ESD reporting framework for Australian fisheries: the 'How to' guide for wild capture fisheries, . FRDC project 2000/145, Canberra.

Fletcher, W. J., Wise, B. S., Joll, L. M., Hall, N. G., Fisher, E. A., Harry, A. V., Fairclough, D. V., et al. 2016. Refinements to harvest strategies to enable effective implementation of Ecosystem Based Fisheries Management for the multi-sector, multi-species fisheries of Western Australia. Fisheries Research, 183: 594-608.

Freese, S., Glock, J., and Squires, D. 1995. Direct allocation of resources and cost-benefit analysis in fisheries: an application to pacific whiting. Marine Policy, 19: 199-211.

Fulton, E. A., Punt, A. E., Dichmont, C. M., Gorton, R., Sporcic, M., Dowling, N., Little, L. R., et al. 2016. Developing risk equivalent data-rich and data-limited harvest strategies. Fisheries Research, 183: 574-587.

Fulton, E. A., Smith, A. D. M., Smith, D. C., and Johnson, P. 2014. An Integrated Approach Is Needed for Ecosystem Based Fisheries Management: Insights from Ecosystem-Level Management Strategy Evaluation. PLoS ONE, 9: e84242.

Fulton, E. A., Smith, A. D. M., Smith, D. C., and van Putten, I. E. 2011. Human behaviour: the key source of uncertainty in fisheries management. Fish and Fisheries, 12: 2-17.

Gaichas, S. K., Fogarty, M., Fay, G., Gamble, R., Lucey, S., and Smith, L. 2017. Combining stock, multispecies, and ecosystem level fishery objectives within an operational management procedure: simulations to start the conversation. ICES Journal of Marine Science, 74: 552-565.

Garcia, S. M., Zerbi, A., Aliaume, C., Do Chi, T., and Lasserre, G. 2003. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper, No. 443: 71 pp.

Glaser, M., and Diele, K. 2004. Asymmetric outcomes: assessing central aspects of the biological, economic and social sustainability of a mangrove crab fishery, Ucides cordatus (Ocypodidae), in North Brazil. Ecological Economics, 49: 361-373.

Global Reporting Initiative 2004. Sustainability Reporting Guidelines on Economic, Environmental and Social Performance. www.globalreporting.org/ReportingFramework/G3Guidelines/.

Goldberg, E. 2001. Profit, People, Planet: Policy Issues and Options Around Triple Bottom Line Reporting. NZ Department for the Environment, 52pp.

Gosling, E., Reith, E., Knoke, T., and Paul, C. 2020. A goal programming approach to evaluate agroforestry systems in Eastern Panama. Journal of environmental management, 261: 110248.

Gourguet, S., Macher, C., Doyen, L., Thébaud, O., Bertignac, M., and Guyader, O. 2013. Managing mixed fisheries for bio-economic viability. Fisheries Research, 140: 46-62.

Gourguet, S., Thébaud, O., Jennings, S., Little, L. R., Dichmont, C. M., Pascoe, S., Deng, R. A., et al. 2016. The cost of co-viability in the Australian northern prawn fishery. Environmental Modeling & Assessment, 21: 371-389.

Graham, S., Barnett, J., Fincher, R., Hurlimann, A., and Mortreux, C. 2014. Local values for fairer adaptation to sea-level rise: A typology of residents and their lived values in Lakes Entrance, Australia. Global Environmental Change, 29: 41-52.

Griffin, W. L., and Woodward, R. T. 2011. Determining policy-efficient management strategies in fisheries using data envelopment analysis (DEA). Marine Policy, 35: 496-507.

Group of 100 Incorporated 2003. Sustainability: A guide to triple bottom line reporting. An Association of Australia's senior Finance Executives from the nation's business enterprises: 52 pp.

Guillen, J., Macher, C., Merzéréaud, M., Bertignac, M., Fifas, S., and Guyader, O. 2013. Estimating MSY and MEY in multi-species and multi-fleet fisheries, consequences and limits: an application to the Bay of Biscay mixed fishery. Marine Policy, 40: 64-74.

Gupta, S., Fügenschuh, A., and Ali, I. 2018. A multi-criteria goal programming model to analyze the sustainable goals of India. Sustainability, 10: 778.

Halpern, B. S., Klein, C. J., Brown, C. J., Beger, M., Grantham, H. S., Mangubhai, S., Ruckelshaus, M., et al. 2013. Achieving the triple bottom line in the face of inherent trade-offs among social equity, economic return, and conservation. Proceedings of the National Academy of Sciences.

Hanley, N., and Barbier, E. B. 2009. Pricing nature: Cost-benefit analysis and environmental policy. Cheltenham: Edward Elgar.

Hanley, N., and Spash, C. 1993. Cost-Benefit Analysis and the Environment. (Edward Elgar Publishing Ltd: Cheltenham).

Hayashi, K. 2000. Multicriteria analysis for agricultural resource management: A critical survey and future perspectives. European Journal of Operational Research, 122: 486-500.

Head, L., Trigger, D., and Woodward, E. 2004. Nature, culture and the challenges of environmental sustainability: Bridging the science/humanities divide: An emerging research network. Australian Research Council Special Research Initiative ID: SR0354620: 86 pp.

Healey, M. C. 1984. Multiattribute Analysis and the Concept of Optimum Yield. Canadian Journal of Fisheries and Aquatic Sciences, 41: 1393-1406.

Henry, G. W., and Lyle, J. M. 2003. National Recreational Fishing Survey. In The National Recreational and Indigenous Fishing Survey Canberra: Australian Government Department of Agriculture, Fisheries and Forestry, FRDC Project No. 99/158, 27-97.

Hilborn, R. 2007. Defining success in fisheries and conflicts in objectives. Marine Policy, 31: 153-158.

Himes, A. H. 2007. Performance Indicator Importance in MPA Management Using a Multi-Criteria Approach. Coastal Management, 35: 601 - 618.

Hobday, A. J., Ogier, E., Fleming, A., Hartog, J., Thomas, L., Stobutzki, I., and Finn, M. 2016. Fishery Status Reports: Healthcheck for Australian Fisheries, FRDC Final Report 2014/008.

Huang, I. B., Keisler, J., and Linkov, I. 2011. Multi-criteria decision analysis in environmental sciences: Ten years of applications and trends. Science of The Total Environment, 409: 3578-3594.

Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Dietzel, A., Eakin, C. M., Heron, S. F., et al. 2018. Global warming transforms coral reef assemblages. Nature, 556: 492-496.

Hummel, J. M., Steuten, L. G. M., Groothuis-Oudshoorn, C. J. M., Mulder, N., and Ijzerman, M. J. 2013. Preferences for Colorectal Cancer Screening Techniques and Intention to Attend: a Multi-Criteria Decision Analysis. Applied Health Economics and Health Policy, 11: 499-507.

Jennings, S., Pascoe, S., Hall-Aspland, S., LeBouhellec, B., Norman-Lopez, A., Sullivan, A., and Pecl, G. 2016a. Setting objectives for evaluating management adaptation actions to address climate change impacts in south-eastern Australian fisheries. Fisheries Oceanography, 25: 29-44.

Jennings, S., Pascoe, S., Hall-Aspland, S., Le Bouhellec, B., Norman-Lopez, A., Sullivan, A., and Pecl, G. 2016b. Setting objectives for evaluating management adaptation actions to address climate change impacts in south-eastern Australian fisheries. Fisheries Oceanography, 25: 29-44.

Katsikopoulos, K. V., Durbach, I. N., and Stewart, T. J. 2018. When should we use simple decision models? A synthesis of various research strands. Omega, 81: 17-25.

Kenter, J. O., O'Brien, L., Hockley, N., Ravenscroft, N., Fazey, I., Irvine, K. N., Reed, M. S., et al. 2015. What are shared and social values of ecosystems? Ecological Economics, 111: 86-99.

Kim, S. W., Sampayo, E. M., Sommer, B., Sims, C. A., Gómez-Cabrera, M. d. C., Dalton, S. J., Beger, M., et al. 2019. Refugia under threat: Mass bleaching of coral assemblages in high-latitude eastern Australia. Global change biology, 25: 3918-3931.

Kimani, P., Wamukota, A., Manyala, J. O., and Mlewa, C. M. 2020. Analysis of constraints and opportunities in marine small-scale fisheries value chain: A multi-criteria decision approach. Ocean & Coastal Management, 189: 105151.

Kjaersgaard, J. 2007. Multiple criteria decision making in fisheries management: Experiences and implications. Food Economics - Acta Agriculturae Scandinavica, Section C, 4: 49 - 64.

Kjærsgaard, J., Andersen, J., and Mathiesen, C. 2007. Multiple objectives and perceptions of optimal management: The Danish industrial fishery in the North Sea. European Review of Agricultural Economics, 34: 181-208.

Klein, C., McKinnon, M. C., Wright, B. T., Possingham, H. P., and Halpern, B. S. 2015. Social equity and the probability of success of biodiversity conservation. Global Environmental Change, 35: 299-306.

Kleine, A., and von Hauff, M. 2009. Sustainability-Driven Implementation of Corporate Social Responsibility: Application of the Integrative Sustainability Triangle. Journal of Business Ethics, 85: 517-533.

Koschke, L., Fürst, C., Frank, S., and Makeschin, F. 2012. A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecological Indicators, 21: 54-66.

Lane, D. E. 1989. Operational research and fisheries management. European Journal of Operational Research, 42: 229-242.

Leigh, G. M., Campbell, A. B., Lunow, C. P., and O'Neill, M. F. 2014. Stock assessment of the Queensland east coast common coral trout (Plectropomus leopardus) fishery. http://era.daf.qld.gov.au/4547/ [Accessed 30 June 2015].

Leigh, G. M., Williams, A. J., Begg, G. A., Gribble, N. A., and Whybird, O. J. 2006. Stock assessment of the Queensland East Coast red throat emperor (Lethrinus miniatus) fishery. Queensland Department of Primary Industries and Fisheries, Brisbane. https://www.daf.qld.gov.au/fisheries/monitoring-our-fisheries/data-reports/sustainability-reporting/stock-assessment-reports/red-throat-emperor-fishery-assessment [Accessed 9 November 2016].

Lenton, T. M. 2013. Environmental Tipping Points. Annu. Rev. Environ. Resour. 38:1-29.

Leung, P. 2006. Multiple-criteria decision-making (MCDM) applications in fishery management. International Journal of Environmental Technology and Management, 6: 96-110.

Leung, P., Muraoka, J., Nakamoto, S. T., and Pooley, S. 1998. Evaluating fisheries management options in Hawaii using analytic hierarchy process (AHP). Fisheries Research, 36: 171-183.

Levin, K., Cashore, B., Bernstein, S., and Auld, G. 2010. Playing it forward: path dependency, progressive incrementalism, and the 'super wicked' problem of global climate change. Presented at the introduced, and defined, the term "super wicked" was presented at the International Studies Association Convention Chicago, February 28th – March 3, 2007, Available at http://environment.research.yale.edu/documents/downloads/0-9/2010_super_wicked_levin_cashore_bernstein_auld.pdf.

Little, L., Punt, A., Mapstone, B., Pantus, F., Smith, A., Davies, C., and McDonald, A. 2007. ELFSim—A model for evaluating management options for spatially structured reef fish populations: An illustration of the "larval subsidy" effect. Ecological Modelling, 205: 381-396.

Little, L. R., Kerrigan, B., Thébaud, O., Campbell, A., Norman-López, A., Innes, J., Cameron, D., et al. 2015. Evaluating candidate monitoring strategies, assessment procedures and harvest control rules in the spatially complex Queensland Coral Reef Fin-fish Fishery. 198 pp.

Little, L. R., Kerrigan, B., Thébaud, O., Campbell, A., Norman-López, A., Innes, J., Cameron, D., et al. 2016. Evaluating candidate monitoring strategies, assessment procedures and harvest control rules in the spatially complex Queensland Coral Reef Fin-fish Fishery. Fisheries Research and Development Corporation, Canberra. http://frdc.com.au/research/Final_Reports/2011-030-DLD.pdf [Accessed 18 May 2017]. 198.

Liu, S., Crossman, N. D., Nolan, M., and Ghirmay, H. 2013. Bringing ecosystem services into integrated water resources management. Journal of environmental management, 129: 92-102.

Long, R. D., Charles, A., and Stephenson, R. L. 2017. Key principles of ecosystem-based management: the fishermen's perspective. Fish and Fisheries, 18: 244-253.

Magnuson-Stevens Fishery Conservation and Management Act 1976. 16 U.S. Code § 1851. https://www.govinfo.gov/content/pkg/PLAW-109publ479/html/PLAW-109publ479.htm.

Magnuson-Stevens Fishery Conservation and Management Act 2006. Magnuson-Stevens Fishery Conservation and Management Reauthorization Act.

Malczewski, J., and Rinner, C. 2015. Multicriteria decision analysis in geographic information science*,* Springer, New York.

Mangel, M., and Dowling, N. A. 2016. Reference Points for Optimal Yield: A Framework for Assessing Economic, Conservation, and Sociocultural Tradeoffs in Ecosystem-Based Fishery Management. Coastal Management, 44: 517-528.

Mangel, M., and Levin, P. S. 2005. Regime, phase and paradigm shifts: making community ecology the basic science for fisheries. Philosophical Transactions of the Royal Society B: Biological Sciences, 360: 95-105.

Mapstone, B. D., Little, L. R., Punt, A. E., Davies, C. R., Smith, A. D. M., Pantus, F., McDonald, A. D., et al. 2008. Management strategy evaluation for line fishing in the Great Barrier Reef: Balancing conservation and multi-sector fishery objectives. Fisheries Research, 94: 315-329.

Mardle, S., and Pascoe, S. 1999. A review of applications of multiple criteria decision making techniques to fisheries. Marine Resource Economics: 41-63.

Mardle, S., Pascoe, S., Boncoeur, J., Le Gallic, B., Garcia-Hoyo, J. J., Herrero, N., Jimenez-Toribio, R., et al. 2002. Objectives of fisheries management: case studies from the UK, France, Spain and Denmark. Marine Policy, 26: 415-428.

Mardle, S., Pascoe, S., and Herrero, I. 2004. Management objective importance in fisheries: An evaluation using the analytic hierarchy process (AHP). Environmental Management, 33: 1-11.

Mardle, S., Pascoe, S., Tamiz, M., and Jones, D. 2000. Resource allocation in the North Sea demersal fisheries: A goal programming approach. Annals of Operations Research, 94: 321-342.

Marre, J.-B., Pascoe, S., Thébaud, O., Jennings, S., Boncoeur, J., and Coglan, L. 2016a. Information preferences for the evaluation of coastal development impacts on ecosystem services: A multi-criteria assessment in the Australian context. Journal of environmental management, 173: 141-150.

Marre, J.-B., Thébaud, O., Pascoe, S., Jennings, S., Boncoeur, J., and Coglan, L. 2016b. Is economic valuation of ecosystem services useful to decision-makers? Lessons learned from Australian coastal and marine management. Journal of environmental management, 178: 52.

Marshall, N. A., and Marshall, P. A. 2007. Conceptualizing and operationalizing social resilience within commercial fisheries in northern Australia." Ecology and Society, 12.

Martin, T. G., Burgman, M. A., Fidler, F., Kuhnert, P. M., Low-Choy, S., Mcbride, M., and Mengersen, K. 2012. Eliciting Expert Knowledge in Conservation Science. Conservation Biology, 26: 29-38.

Mascia, M. B. 2003. The Human Dimension of Coral Reef Marine Protected Areas: Recent Social Science Research and Its Policy Implications. Conservation Biology, 17: 630-632.

McDonald, A. D., Little, L. R., Gray, R., Fulton, E., Sainsbury, K. J., and Lyne, V. D. 2008. An agent-based modelling approach to evaluation of multiple-use management strategies for coastal marine ecosystems. Mathematics and Computers in Simulation, 78: 401-411.

Millennium Ecosystem Assessment 2005. Ecosystems and Human Well-being: Wetlands and Water (Synthesis). Island Press, Washington, DC.: 68 pp.

Munch, S. B., Poynor, V., and Arriaza, J. L. 2017. Circumventing structural uncertainty: A Bayesian perspective on nonlinear forecasting for ecology. Ecological Complexity, 32: 134-143.

Murphy, P. 2001. Affiliation Bias and Expert Disagreement in Framing the Nicotine Addiction Debate. Science, Technology, & Human Values, 26: 278-299.

Nielsen, J. R., and Mathiesen, C. 2006. Stakeholder preferences for Danish fisheries management of sand eel and Norway pout. Fisheries Research, 77: 92-101.

Nielsen, K. N., Baudron, A. R., Fallon, N. G., Fernandes, P. G., Rahikainen, M., and Aschan, M. 2019. Participatory planning and decision support for ecosystem based fisheries management of the west coast of Scotland. Fisheries Research, 211: 59-68.

Nunan, F. 2014. Wealth and welfare? Can fisheries management succeed in achieving multiple objectives? A case study of Lake Victoria, East Africa. Fish and Fisheries, 15: 134-150.

O'Neill, M. F., Leigh, G. M., Martin, J. M., Newman, S. J., Chambers, M. S., Dichmont, C. M., and Buckworth, R. C. 2011. Sustaining productivity of tropical red snappers using new monitoring and reference points. FRDC Project no. 2009-037. Department of Employment, Economic Development and Innovation, Brisbane. https://www.frdc.com.au/project/2009-037.

Ogier, E., Emery, T., Farmery, A., Flood, M., Gardner, C., Jabour, J., Nicol, S., et al. 2020. Meeting Sustainability Expectations: Policy Translation, Objective Setting and Reporting for Australian Fisheries. FRDC project 2013-204. Institute for Marine and Antarctic Studies, University of Tasmania: 141 pp.

Pascoe, S., Brooks, K., Cannard, T., Dichmont, C. M., Jebreen, E., Schirmer, J., and Triantafillos, L. 2014a. Social objectives of fisheries management: What are managers' priorities? Ocean & Coastal Management, 98: 1-10.

Pascoe, S., Bustamante, R., Wilcox, C., and Gibbs, M. 2009a. Spatial fisheries management: A framework for multi-objective qualitative assessment. Ocean & Coastal Management, 52: 130-138.

Pascoe, S., Bustamante, R., Wilcox, C., and Gibbs, M. 2009b. Spatial fisheries management: a framework for multi-objective qualitative assessment. Ocean and Coastal Management, 52: 130-138.

Pascoe, S., Cannard, T., Dowling, N. A., Dichmont, C. M., Breen, S., Roberts, T., Pears, R. J., et al. 2019. Developing Harvest Strategies to Achieve Ecological, Economic and Social Sustainability in Multi-Sector Fisheries. Sustainability, 11: 644.

Pascoe, S., Cannard, T., Jebreen, E., Dichmont, C. M., and Schirmer, J. 2015. Satisfaction with fishing and the desire to leave. AMBIO, 44: 401-411.

Pascoe, S., and Dichmont, C. M. 2017. Does membership matter? Individual influences in natural resource management decision making. Marine Policy, 83: 48-54.

Pascoe, S., Dichmont, C. M., Brooks, K., Pears, R., and Jebreen, E. 2013a. Management objectives of Queensland fisheries: putting the horse before the cart. Marine Policy, 37: 115-122.

Pascoe, S., Dichmont, C. M., Brooks, K., Pears, R., and Jebreen, E. 2013b. Management objectives of Queensland fisheries: Putting the horse before the cart. Marine Policy, 37: 115-122.

Pascoe, S., and Doshi, A. 2018. Estimating coastal values using multi-criteria and valuation methods.

Pascoe, S., Hutton, T., van Putten, I., Dennis, D., Skewes, T., Plagányi, É., and Deng, R. 2013c. DEA-based predictors for estimating fleet size changes when modelling the introduction of rights-based management. European Journal of Operational Research, 230: 681-687.

Pascoe, S., and Mardle, S. 2001. Optimal fleet size in the English Channel: a multi-objective programming approach. European Review of Agricultural Economics, 28: 161-185.

Pascoe, S., Mary Dichmont, C., Brooks, K., Pears, R., and Jebreen, E. 2013d. Management objectives of Queensland fisheries: Putting the horse before the cart. Marine Policy, 37: 115-122.

Pascoe, S., Proctor, W., Wilcox, C., Innes, J., Rochester, W., and Dowling, N. 2009c. Stakeholder objective preferences in Australian Commonwealth managed fisheries. Marine Policy, 33: 750-758.

Pascoe, S., Tobin, R., Windle, J., Cannard, T., Marshall, N., Kabir, Z., and Flint, N. 2014b. Piloting of social, cultural and economic indicators for the Gladstone Healthy Harbour Partnership Report Card. 182 pp.

Pascoe, S., Tobin, R., Windle, J., Cannard, T., Marshall, N., Kabir, Z., and Flint, N. 2016a. Developing a social, cultural and economic report card for a regional industrial harbour. PLoS ONE, 11: e0148271.

Pascoe, S., Tobin, R., Windle, J., Cannard, T., Marshall, N., Kabir, Z., and Flint, N. 2016b. Developing a Social, Cultural and Economic Report Card for a Regional Industrial Harbour. PLoS ONE, 11.

Pascoe, S., Van Putten, I., Hoshino, E., and Vieira, S. 2020. Determining key drivers of perceptions of performance of rights-based fisheries in Australia using a Bayesian belief network. ICES Journal of Marine Science, 77: 803-814.

Pascoe, S. D., Plagányi, É. E., and Dichmont, C. M. 2017. Modelling multiple management objectives in fisheries: Australian experiences. ICES Journal of Marine Science, 74: 464-474.

Pashaei Kamali, F., Borges, J. A. R., Meuwissen, M. P. M., de Boer, I. J. M., and Oude Lansink, A. G. J. M. 2017. Sustainability assessment of agricultural systems: The validity of expert opinion and robustness of a multi-criteria analysis. Agricultural Systems, 157: 118-128.

Péreau, J. C., Doyen, L., Little, L. R., and Thébaud, O. 2012. The triple bottom line: Meeting ecological, economic and social goals with individual transferable quotas. Journal of Environmental Economics and Management, 63: 419-434.

PFMC 2013. Report of the Pacific Sardine Harvest Parameters Workshop. Pacific Fishery Management Council, Portland, OR. Available at: http://www.pcouncil.org/ wp-content/uploads/I1b_ATT1_SARDINE_WKSHP_RPT_APR2013BB.pdf (accessed 26 June 2014).

Pitcher, T. J., and Cheung, W. W. L. 2013. Fisheries: Hope or despair? Marine Pollution Bulletin, 74: 506-516.

Pitcher, T. J., and Preikshot, D. 2001. RAPFISH: a rapid appraisal technique to evaluate the sustainability status of fisheries. Fisheries Research, 49: 255-270.

Plagányi, É., Deng, R., Dennis, D., Hutton, T., Pascoe, S., van Putten, I., and Skewes, T. 2012a. An Integrated Management Strategy Evaluation (MSE) for the Torres Strait tropical rock lobster *Panulirus ornatus* fishery. 233 pp.

Plagányi, É., Deng, R., Dennis, D., Hutton, T., Pascoe, S., van Putten, I., and Skewes, T. 2012b. An Integrated Management Strategy Evaluation (MSE) for the Torres Strait tropical rock lobster *Panulirus ornatus* fishery. CSIRO/AFMA Final Project Report. AFMA Project number 2009/839: 233 pp.

Plagányi, É. E., van Putten, I., Hutton, T., Deng, R. A., Dennis, D., Pascoe, S., Skewes, T., et al. 2013. Integrating indigenous livelihood and lifestyle objectives in managing a natural resource. Proceedings of the National Academy of Sciences, 110.

Pöyhönen, M., and Hämäläinen, R. P. 2001. On the convergence of multiattribute weighting methods. European Journal of Operational Research, 129: 569-585.

Punt, A. E. 2015. Strategic management decision-making in a complex world: quantifying, understanding, and using trade-offs. ICES Journal of Marine Science, 74: 499-510.

Punt, A. E., Butterworth, D. S., de Moor, C. L., De Oliveira, J. A. A., and Haddon, M. 2016. Management strategy evaluation: best practices. Fish and Fisheries, 17: 303-334.

Punt, A. E., Deng, R. A., Dichmont, C. M., Kompas, T., Venables, W. N., Zhou, S., Pascoe, S., et al. 2010. Integrating size-structured assessment and bioeconomic management advice in Australia's northern prawn fishery. ICES Journal of Marine Science, 67: 1785-1801.

Punt, A. E., Smith, A. D. M., and Cui, G. 2002. Evaluation of management tools for Australia's South East Fishery 3. Towards selecting appropriate harvest strategies. Marine and Freshwater Research, 53: 645-660.

Queensland Government 2019. Wave monitoring. https://www.qld.gov.au/environment/coasts-waterways/beach/monitoring Accessed on 10 August 2019.

Read, A. D., and West, R. J. 2010. Qualitative risk assessment of multiple-use marine park effectiveness – A case study from NSW, Australia. Ocean & Coastal Management, 53: 636-644.

Ren, C. F., Li, R. H., Zhang, L. D., and Guo, P. 2016. Multiobjective Stochastic Fractional Goal Programming Model for Water Resources Optimal Allocation among Industries. Journal of Water Resources Planning and Management, 142: 04016036.

Rice, J., Moksness, E., Attwood, C., Brown, S. K., Dahle, G., Gjerde, K. M., Grefsrud, E. S., et al. 2012. The role of MPAs in reconciling fisheries management with conservation of biological diversity. Ocean & Coastal Management, 69 217-230.

Richerson, K., Levin, P. S., and Mangel, M. 2010. Accounting for indirect effects and non-commensurate values in ecosystem based fishery management (EBFM). Marine Policy, 34: 114-119.

Rindorf, A., Dichmont, C. M., Levin, P. S., Mace, P., Pascoe, S., Prellezo, R., Punt, A. E., et al. 2016. Food for thought: pretty good multispecies yield. ICES Journal of Marine Science, 74: 475-486.

Rindorf, A., Dichmont, C. M., Thorson, J., Charles, A., Clausen, L. W., Degnbol, P., Garcia, D., et al. 2017. Inclusion of ecological, economic, social, and institutional considerations when setting targets and limits for multispecies fisheries. ICES Journal of Marine Science, 74: 453-463.

Roberts, R., and Goodwin, P. 2002. Weight approximations in multi-attribute decision models. Journal of Multi-Criteria Decision Analysis, 11: 291-303.

Roman, G. S. J., Dearden, P., and Rollins, R. 2007. Application of Zoning and “Limits of Acceptable Change” to Manage Snorkelling Tourism. Environmental Management, 39: 819-830.

Russ, G. R., and Williamson, D. 2015. NERP TE Project 8.2 - Assessing the ecological effects of management zoning on inshore reefs of the Great Barrier Reef Marine Park (JCU). National Environmental Research Program.

Saaty, T. L. 1980a. The Analytic Hierarchy Process. McGraw-Hill, New York.

Saaty, T. L. 1980b. The Analytic Hierarchy Process*,* McGraw-Hill, New York.

Saaty, T. L. 1986. Axiomatic Foundation of the Analytic Hierarchy Process. Management Science, 32: 841-855.

Sainsbury, K. J., Punt, A. E., and Smith, A. D. M. 2000. Design of operational management strategies for achieving fishery ecosystem objectives. ICES Journal of Marine Science, 57: 731-741.

Samvedi, A., Jain, V., and Chan, F. T. S. 2013. Quantifying risks in a supply chain through integration of fuzzy AHP and fuzzy TOPSIS. International Journal of Production Research, 51: 2433-2442.

Sara, J., Stikkelman, R. M., and Herder, P. M. 2015. Assessing relative importance and mutual influence of barriers for CCS deployment of the ROAD project using AHP and DEMATEL methods. International Journal of Greenhouse Gas Control, 41: 336-357.

Scandol, J. 2005. Use of quality control methods to monitor the status of fish stocks. Kruse, GH, Gallucci, VF, Hay, DE, Perry, RI, Peterman, RM, Shirley, TC, Spencer, PD, Wilson, B. et Woodby, D., éditeurs: Fisheries Assessment in Data-Limited Situations: 213-234.

Schwermer, H., Barz, F., and Zablotski, Y. 2020. A Literature Review on Stakeholder Participation in Coastal and Marine Fisheries. *In* YOUMARES 9 - The Oceans: Our Research, Our Future: Proceedings of the 2018 conference for YOUng MArine RESearcher in Oldenburg, Germany, pp. 21-43. Ed. by S. Jungblut, V. Liebich, and M. Bode-Dalby. Springer International Publishing, Cham.

Senner, R. 2011. Appraising the sustainability of project alternatives: An increasing role for cumulative effects assessment. Environmental Impact Assessment Review, 31: 502-505.

Sherrouse, B., and Semmens, D. 2013. Demonstration of Social Values for Ecosystem Services (SolVES) by Ben Sherrouse and Darius Semmens of USGS. by Webinar ID: 438-764-008. Questions, comments and feedback to: sarah_carr@natureserve.org, accessed May 2013.

Simon, H. A., Smithburg, D. W., and Thompson, V. A. 1950. Public administration. New York: Knopf.

Sloan, S., Smith, T., Gardner, C., Crosthwaite, K., Triantafillos, L., Jeffriess, B., and Kimber, N. 2014. National Guidelines to Develop Fishery Harvest Strategies, Fisheries Research & Development Corporation, Adelaide South Australia, Project No. 2010/061, pp. 1-70. .

Smith, A. D. M., Fulton, E. J., Hobday, A. J., Smith, D. C., and Shoulder, P. 2007. Scientific tools to support the practical implementation of ecosystem-based fisheries management. ICES Journal of Marine Science, 64: 633-639.

Soma, K. 2003. How to involve stakeholders in fisheries management - a country case study in Trinidad and Tobago. Marine Policy, 27: 47-58.

State of Queensland 2017a. Fisheries Act 1994, Current as at 3 July 2017 accessed at https://www.legislation.qld.gov.au/view/pdf/inforce/2017-07-03/act-1994-037. 155 pp.

State of Queensland 2017b. Queensland Sustainable Fisheries Strategy 2017-2027. Queensland Department of Agriculture and Fisheries, Brisbane. https://www.daf.qld.gov.au/business-priorities/fisheries/sustainable/sustainable-fisheries-strategy/sustainable-fisheries-strategy-overview [Accessed 19 December 2019].

Stephenson, R. L., Benson, A. J., Brooks, K., Charles, A., Degnbol, P., Dichmont, C. M., Kraan, M., et al. 2017a. Practical steps toward integrating economic, social and institutional elements in fisheries policy and management. ICES J Mar Sci, Handling editor: Linwood Pendleton.

Stephenson, R. L., Benson, A. J., Brooks, K., Charles, A., Degnbol, P., Dichmont, C. M., Kraan, M., et al. 2017b. Practical steps toward integrating economic, social and institutional elements in fisheries policy and management. ICES Journal of Marine Science, 74: 1981–1989.

Stouten, H., Heene, A., Gellynck, X., and Polet, H. 2011. Policy instruments to meet fisheries management objectives in Belgian fisheries. Fisheries Research, 111: 8-23.

Suggett, D., and Goodsir, B. 2002. Triple Bottom Line Measurement and Reporting in Australia - Making it Tangible. 116 pp. pp.

Sylvia, G., and Enríquez, R. R. 1994. Multiobjective Bioeconomic Analysis: An Application to the Pacific Whiting Fishery. Marine Resource Economics, 9: 311-328.

Symes, D., and Phillipson, J. 2009. Whatever became of social objectives in fisheries policy? Fisheries Research, 95(1).

Taylor, S., Webley, J., and McInnes, K. 2012. 2010 Statewide Recreational Fishing Survey. State of Queensland, Department of Agriculture, Fisheries and Forestry, Brisbane.

TEEB 2010. Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of The Economics of Ecosystems and Biodiversity (TEEB). 36 pp.

Thadsin, K., George, H., and Stanley, M. 2012. Introduction of AHP Satisfaction Index for workplace environments. Journal of Corporate Real Estate, 14: 80-93.

Thébaud, O., Innes, J., Norman-López, A., Slade, S., Cameron, D., Cannard, T., Tickell, S., et al. 2014. Micro-economic drivers of profitability in an ITQ-managed fishery: An analysis of the Queensland Coral Reef Fin-Fish Fishery. Marine Policy, 43: 200-207.

Tobin, R. C., Sutton, S. G., Penny, A., Williams, L., Maroske, J., and Nilsson, J. 2010. Baseline socio-economic data for Queensland east-coast inshore and rocky reef fishery stakeholders. Part A: Commercial inshore and rocky reef fishers. Fishing and Fisheries Research Centre Technical Report No 5, Townsville: James Cook University.

Tozer, P. R., and Stokes, J. R. 2002. Producer Breeding Objectives and Optimal Sire Selection. Journal of Dairy Science, 85: 3518-3525.

Triantafillos, L., Brooks, K., Schirmer, J., Pascoe, S., Cannard, T., Dichmont, C., Thebaud, O., et al. 2014. Final Report - 2010-040-DLD e Developing and Testing Social Objectives for Fisheries Management. Fisheries Research and Develoment Corporation.

Tullberg, J. 2012. Triple bottom line - a vaulting ambition? Business Ethics-a European Review, 21: 310-324.

Turner, K. G., Anderson, S., Gonzales-Chang, M., Costanza, R., Courville, S., Dalgaard, T., Dominati, E., et al. 2015. A review of methods, data, and models to assess changes in the value of ecosystem services from land degradation and restoration. Ecological Modelling, 319: 190-207.

Urquhart, J., Acott, T., Reed, M., and Courtney, P. 2011. Setting an agenda for social science research in fisheries policy in Northern Europe. Fisheries Research, 108: 240-247.

Van Ittersum, K., Pennings, J. M. E., Wansink, B., and van Trijp, H. C. M. 2007. The validity of attribute-importance measurement: A review. Journal of Business Research, 60: 1177-1190.

van Putten, I., Lalancette, A., Bayliss, P., Dennis, D., Hutton, T., Norman-López, A., Pascoe, S., et al. 2013. A Bayesian model of factors influencing indigenous participation in the Torres Strait tropical rocklobster fishery. Marine Policy, 37: 96-105.

van Putten, I. E., Dichmont, C. M., Dutra, L. X. C., Thébaud, O., Deng, R. A., Jebreen, E., Owens, R., et al. 2015. Objectives for management of socio-ecological systems in the Great Barrier Reef region, Australia. Regional Environmental Change: 1-15.

Vergara-Solana, F., Araneda, M. E., and Ponce-Díaz, G. 2019. Opportunities for strengthening aquaculture industry through multicriteria decision-making. Reviews in Aquaculture, 11: 105-118.

Vieira, S., Schirmer, J., and Loxton, E. 2009. Social and economic evaluation methods for fisheries: a review of the literature. Fisheries Research Contract Report No. 21. Department of Fisheries, Western Australia. 94p.

Voss, R., Quaas, M. F., Schmidt, J. O., Tahvonen, O., Lindegren, M., and Möllmann, C. 2014. Assessing Social – Ecological Trade-Offs to Advance Ecosystem-Based Fisheries Management. PLoS ONE, 9: e107811.

Walters, C. J., and Hilborn, R. 1978. Ecological Optimization and Adaptive Management. Annual Review of Ecology and Systematics, 9: 157-188.

Wang, J.-J., Jing, Y.-Y., Zhang, C.-F., and Zhao, J.-H. 2009. Review on multi-criteria decision analysis aid in sustainable energy decision-making. Renewable and Sustainable Energy Reviews, 13: 2263-2278.

Wattage, P., and Mardle, S. 2005. Stakeholder preferences towards conservation versus development for a wetland in Sri Lanka. Journal of environmental management, 77: 122-132.

Webley, J. A. C., McInnes, K., Teixeira, D., Lawson, A. and Quinn, R. 2015. Statewide Recreational Fishing Survey 2013-14. Department of Agriculture and Fisheries, Brisbane. http://era.daf.qld.gov.au/id/eprint/6513/1/2013-14SRFS%20Report.pdf [Accessed 19 December 2019].

Weninger, Q. 2001. An analysis of the efficient production frontier in the fishery: implications for enhanced fisheries management. Applied Economics, 33: 71-79.

Whitmarsh, D., and Wattage, P. 2006. Public attitudes towards the environmental impact of salmon aquaculture in Scotland. European Environment, 16: 108-121.

Wilcoxon, F. 1945. Individual comparisons by ranking methods. Biometrics Bulletin, 1: 80-83.

Williamson, D. H., Ceccarelli, D. M., Evans, R. D., Hill, J. K., and Russ, G. R. 2014. Derelict fishing line provides a useful proxy for estimating levels of non-compliance with no-take marine reserves. PLoS ONE, 9: e114395-e114395.

Wills, I. 2006. Economics and the Environment: A signalling and incentives approach*,* Allen & Unwin, Crows Nest, NSW, Australia.

Wilson, D., Curtotti, R., and Begg, G. 2010. Fishery status reports 2009: status of fish stocks and fisheries managed by the Australian Government.

World Values Survey 2015. Data & Documentation. http://www.worldvaluessurvey.org/WVSContents.jsp, Accessed on 11 October.

Yang, M., Qian, X., Zhang, Y., Sheng, J., Shen, D., and Ge, Y. 2011. Spatial multicriteria decision analysis of flood risks in aging-dam management in China: A framework and case study. International journal of environmental research and public health, 8: 1368-1387.

Young, E. G., Melnychuk, M. C., Anderson, L. E., Hilborn, R., and Hyder, K. 2019. The importance of fishing opportunity to angler utility analysis in marine recreational fisheries. ICES Journal of Marine Science, doi:10.1093/icesjms/fsz234.

Zimmermann, F., and Yamazaki, S. 2017. Exploring conflicting management objectives in rebuilding of multi-stock fisheries. Ocean & Coastal Management, 138: 124-137.

Project materials developed

Conference presentations have been given at:

- Hjort Symposium: **Dichmont, C.M**., Dowling, N.A., Pascoe, S., Cannard, T., Pears, R.J., Breen, S., Roberts, T., Leigh, G.M., Mangel, M. Operationalising Triple Bottom Line Harvest Strategies.
- PICES Conference Pascoe, S., Cannard, T., **Dowling, N.**, Dichmont, C.M., Hutton, T. 2019. Integrating economic and social objectives in marine resource management: Australian experiences. Invited PICES presentation, Victoria, Canada.
- **Pascoe, Sean**; Dowling, Natalie A.; Dichmont, Catherine M.; Cannard, Toni; Pears, Rachel J.; Breen, Sian; Roberts, Tom; Leigh, George M. 2020 Use of Data Envelopment Analysis (DEA) to assess fisheries management alternatives in the presence of multiple objectives. World Fisheries Congress, Adelaide. (Note: Delayed until 2021 due to COVID19).

The main results from the project have been, or are being published as:

- Dichmont, C.M., Dowling, N.A., Pascoe, S., Cannard, T., Pears, R.J., Breen, S., Roberts, T., Leigh, G.M., Mangel, M. (2020). Operationalizing Triple Bottom Line Harvest Strategies. ICES Journal of Marine Science https://academic.oup.com/icesjms/advance-article/doi/10.1093/icesjms/fsaa033/5812750.
- Pascoe, S., Cannard, T., Dowling, N.A., Dichmont, C.M., Breen, S., Roberts, T., Pears, R.J., Leigh, G.M. (2019). Developing Harvest Strategies to Achieve Ecological, Economic and Social Sustainability in Multi-Sector Fisheries. Sustainability, 11 (3), 644, http://www.mdpi.com/2071-1050/11/3/644
- Dowling, N.A., Dichmont, C.M., Leigh, G.M., Pascoe, S., Pears, R.J., Roberts, T., Breen, S., Cannard, T., Mangel, M. (2020). Optimising harvest strategies over multiple objectives and stakeholder preferences. ICES Journal of Marine Science, 435, 109243, https://www.sciencedirect.com/science/article/pii/S0304380020303136?via%3Dihub.

Appendices

List of researchers

Natalie Dowling, Sean Pascoe, Toni Cannard (CSIRO Oceans & Atmosphere)

Catherine Dichmont (Cathy Dichmont Consulting)

Rachel Pears (GBRMPA)

George Leigh, Tom Roberts, Sian Breen (DAF)

Appendix A: Achieving the triple bottom line in fishery harvest strategies: challenges identified in the literature

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). The implementation of TBL, however, remains problematic and it has not been operationalised with fishery harvest strategies (Mangel and Dowling, 2016). Indeed, Elkington (2018) sought to recall and rethink the concept, stating that it has “failed to bury the single bottom line [economic] paradigm”. To date, consideration of the TBL has been largely limited to conceptual treatment (Stephenson et al., 2017a) or intuitive forecasting methods using expert opinion (Bernstein and Cetron, 1969; Dichmont et al., 2012b; Dichmont et al., 2014; Pascoe et al., 2019).

Pascoe et al. (2009a) 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 (AHP, Saaty, 1980) to determine which options performed best, taking into account the multiple objectives inherent in fisheries management. Read and West (2010) used an alternative approach and 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.

The more recently prioritised social objectives (needed to balance the triple bottom line) are usually more difficult to define and quantitatively measure, which presents further challenges, including the need to link economic and environmental objectives with social objectives in fisheries models (Pascoe et al., 2017). Alternatively, it may be possible to determine economic based non-market values. Several valuation tools have been developed using the standpoint of explicit valuations of natural capital. These include the InVEST tool[12] which is grounded in the Natural Capital Project[13] concepts. This is a suite of software models that map and value the goods and ecosystem services that are vital to humanity, including food, life-enabling processes (e.g., water purification), and even ‘life-fulfilling’ conditions (e.g., experience of naturally beautiful landscapes and opportunities for recreation (TEEB, 2010). InVEST assists the managers of natural resources for multiple uses to evaluate trade-offs. However, this and other tools do not include the fine-scale stock assessment detail that fisheries managers require.

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

---

[12] InVEST software details available at https://naturalcapitalproject.stanford.edu/software/invest

[13] The Natural Capital Project, see http://www.naturalcapitalproject.org/

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 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. 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 only 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., Plaganyi et al., 2012) 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. (2017a) identified three key impediments to embracing TBL objectives in a full quantitative analysis: lack of explicit social, economic and institutional objectives; lack of process for routine integration of all four pillars of sustainability; and bias towards biological considerations. While international agreements and legislation call for incorporation of four pillars of sustainability, the social (including cultural), economic and institutional aspects (the 'human dimension') have been relatively neglected to date. 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. (2012b) and Plagányi et al. (2013) used a suite of integrated models to capture these 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., 2013c). The economic implications of the fleets' effort levels were assessed using a bioeconomic model (Plagányi et al., 2012b).

Earlier 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. (2013a) use an MSE that included a bio-economic and ecosystem model to evaluated marine spatial closures with conflicting fisheries and conservation objectives.

Beyond the explicit incorporation of all TBL objectives, formal methods that have attempted acknowledge the TBL result in discrete strategies do not consider stakeholder's preferences (weightings) across the range of objectives, and provide no formal means of determining the optimal solution given these weightings. Pascoe et al. (2013d) 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 Eastern 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.

Thus, operationalising the triple bottom line, beyond a simple conceptualisation is complex. To embed the TBL in formal management, 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.

Appendix B: The process of objective elicitation: a brief literature review

The eliciting of objectives has been conducted in a participatory manner in both commercial and artisanal fisheries settings (Plagányi et al., 2013; Dutra et al., 2015; van Putten et al., 2015; Dichmont et al., 2016). When eliciting objectives, it is important to create equity within disparate groups to encourage both mutual understanding of often conflicting desires. Fundamental tenets of participatory practice encourage open and overt discussion of issues. Behaviour science studies and game theory describe the various modes of stakeholder's actions (as actors or agents) in the objective elicitation process (Bousquet and Le Page, 2004; Fulton et al., 2011), especially when multi-criteria weighting is used as the major trade-off mechanism. In these cases, when a group of stakeholders are required to cooperatively decide on the objectives, it is ethical and advisable to explain the whole purpose of the objective setting exercise in the full context of whichever process is at hand, for example, a harvest strategy, report card, or Management Strategy Evaluation (MSE).

In the MSE context, Punt (2015) and Punt et al. (2016) explain that part of the MSE development is to identify strategic and conceptual objectives and performance measures that are able to determine status and trends those objectives. Management objectives are likely to be conflicting. Almost by definition, objectives stated by decision makers cannot be "wrong", and should be given serious consideration even if there is no consensus among decision-makers regarding the appropriateness of some of the objectives. Nevertheless, the process of elucidating objectives should emphasise that they be quantifiable through the operating models that are part of the MSE. Punt (2015) provide a list of single species and multispecies MSE plus a listing of whole of ecosystem MSE alongside typical performance measures and a list of MSE cases where these measures are used.

Mapstone et al. (2008) provide a "gold standard" for eliciting objectives and using performance measures to evaluate of closure regimes for Australia's Great Barrier Reef by identifying places linked to values. In their study, ecological objectives included places that have or protect natural ecosystem values, (e.g. bushland, beaches, sea, rivers, wetlands), economically important places where people either earnt or spend money (noting this encompassed whole of the life cycle for production, consumption, waste disposal and associated technology) (e.g. tourist operations, factories, landfill and waste recycling sites, major infrastructure), socially important places that people gather, and organise to meet particular needs (e.g. clubs, libraries, schools, cafés, and public amenities such as playgrounds and picnic areas), and culturally important places important to people for making or expressing meaning (e.g. indigenous sacred sites, cultural heritage sites, museums). Representatives of the Mapstone et al. (2008) research team conducting the MSE met separately with each stakeholder group several times over two years, then held workshops that brought all the stakeholders together to ensure that all objectives were collectively understood (though perhaps not agreed). These workshops also reviewed how objectives were to be expressed as performance measures that could be output by the MSE. However, Punt (2015) proposed that the resource intense nature of Mapstone et al. (2008) 'gold standard' approach could be the reason that it has seldom been adopted in practice.

A more common approach to identifying objectives and performance indicators is to separate the process of identifying management objectives (which can be broad, vague, and likely inconsistent) from the process of translating those objectives into specific performance indicators. This approach taken by the International Whaling Commission Scientific Committee (IWC SC) identified and ranked objectives, and then developed quantitative performance measures (the value of performance indicators relative to some (typically, target or limit) reference level) to report against the objectives (Punt et al., 2016).

Another approach, adopted for the MSE for Pacific sardine (*Sardinops sagax*) off the US west coast, recognised that management objectives are largely “prespecified” through National Standards that are part of the US *Magnuson-Stevens Fishery Conservation and Management Act* (1976) and (2006), with guidelines provided by the National Marine Fisheries Service (Punt et al., 2016). Here, the choice of performance measures involved an iterative process whereby an initial set of performance measures was selected by analysts conducting the MSE (PFMC, 2013), and those performance measures were modified based on input from decision-makers (the PFMC), their scientific and policy advisors, as well as members of stakeholder groups (fisher and environmental non-governmental organisations).

More recently in Australia, Pascoe et al. (2013d) developed an objectives hierarchy. A preliminary hierarchy was drawn from a comprehensive review of natural resource management objectives, and was cross-referenced to policy documents related to the fishery and the GBR marine park, and to key legislation by interdisciplinary researchers. Subsequently, the Scientific Advisory Group (scientists, fisheries managers and industry members from catching and processing sectors) agreed on the final hierarchy by consensus and this was adjusted slightly with minor additions by the government department responsible for the management of the fishery (the Department of Employment, Economic Development and Innovation[14]), (Figure B1).

[14] Now split into several Departments, including DAF and the Department of Environment and Science

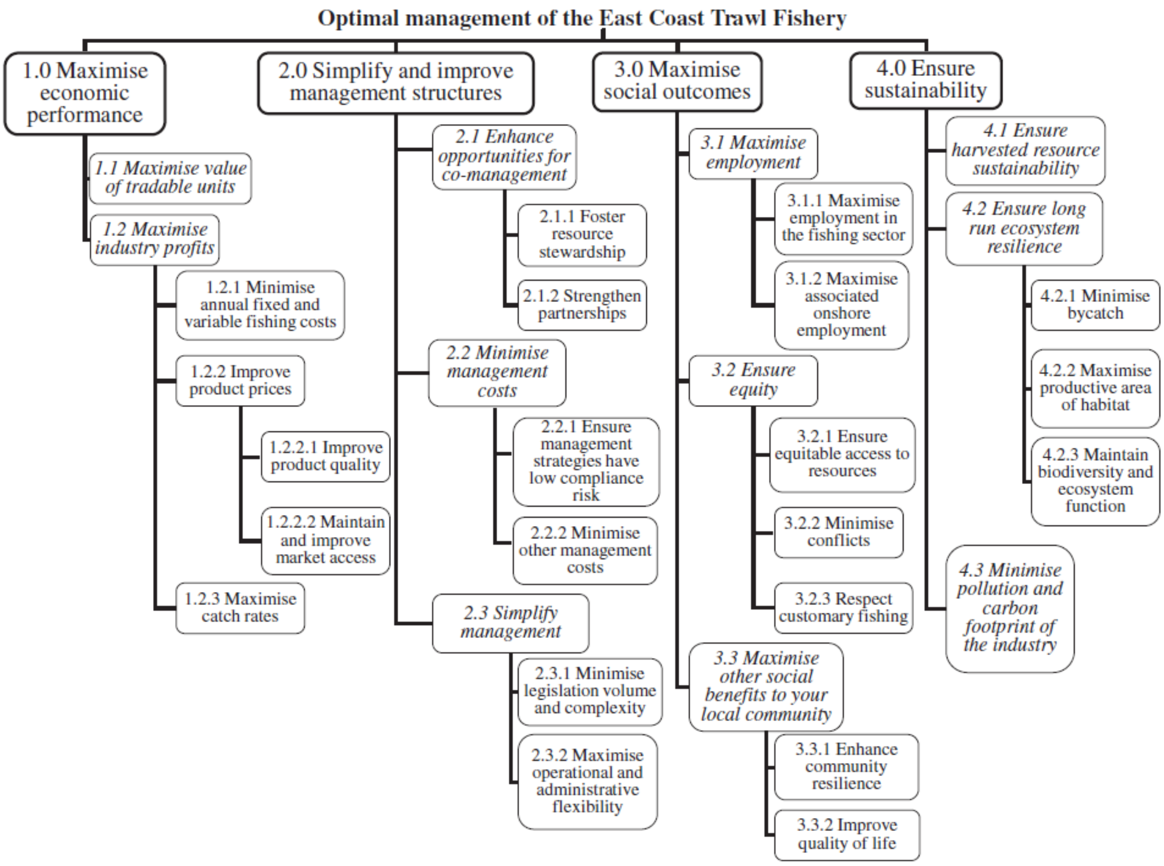


Figure B1. Objectives hierarchy developed for the Queensland East Coast Trawl Fishery (Source: Pascoe et al. (2013d))

Dichmont et al. (2016) applied Pascoe et al.'s (2013d) hierarchy in a broader regional coastal management context in Queensland. They developed a pragmatic, bottom-up decision support system to elicit objectives, using a hierarchal engagement model of local stakeholders, regional and senior coastal managers. A review of existing objectives from government organisations, NGOs and Natural Resource Management (NRM) bodies that were directly or indirectly relevant to the region was undertaken. This was then combined into a hierarchical tree format using input from a series of workshops attended by a Reference Group (RG) and the Local Marine Advisory Committee (LMAC) (van Putten et al., 2015; Dichmont et al., 2016). A survey of the RG, LMAC and local public was then undertaken to ascertain the relative importance of different objectives. Two approaches were undertaken: the recommended Analytical Hierarchical Process (Saaty, 1980a; Pascoe et al., 2013d) and a new Point Allocation method at each level of the objective tree and called the Hierarchical Point Allocation method (Dichmont et al., 2014). This semi-quantitative generic elicitation framework provided a prioritised list of management options in the context of clearly articulated management objectives.

The following process of eliciting objectives for coastal management was taken by van Putten et al. (2015):

- Literature review of existing objectives across the region and identification higher-level objectives for fisheries and natural resources,

- Existing objectives listed or categorised according to level (high, medium, low),
- Objectives list provided to LMAC workshop or during interviews with individuals, to determine abbreviated list of critical objectives and seek feedback.

Appendix C: Pilot study: reconciling different stakeholder weightings under TBL management

Natalie A. Dowling, Cameron Speir, Aaron Mamula, Marc Mangel

Introduction

The triple bottom line (TBL), of reconciling economic, ecological (environmental) and social objectives, and reporting performance in this context, was originally attributed to Elkington (1997) The triple bottom line was conceived of as a tool for influencing a single decision maker (a firm) to explicitly value non-financial objectives. These 'non-financial' objectives have been the subject of robust debate, with issues including:

- Whether companies care about their reputation or standing in the community because they value these things independently from their profits, or whether corporate goodwill reduces to a form of profit maximisation. For example, encouraging employees to volunteer in the community during work time may, through goodwill, increase the market size or willingness to pay for company products. From a technical standpoint (even if not from a standpoint of understanding motivations), it matters as to whether the objective of the firm is profit maximization, or whether companies have a private, independent, purely altruistic value for social/ecological outcomes that is unrelated to long-term profitability.
- Top-down or legislative mandate/incentives: in the environmental economics literature, Porter (and others) argued that companies should pursue pollution abatement, despite this being costly, because pollution represents productive inefficiency. Here, the triple bottom line is important because companies are not traditionally 'wired' to consider non-financial motives, yet, if they are incentivised to do so (through regulation perhaps) it ends up helping their 'bottom line.'

Although the triple bottom line rightly gets attributed to Elkington (1997), TBL thinking has a very strong intellectual connection to Porter and Van der Linde (1995). The latter considered how the Dutch flower industry has responded to its environmental problems, by developing a closed-loop system, which also reduced variation in growing conditions, as well as handling costs. The net result was not only dramatically lower environmental impacts but also lower costs, better product quality, and enhanced global competitiveness. Focusing only on the static cost impacts of environmental regulation ignores the more important offsetting productivity benefits from innovation. As a result, policy makers, business leaders and environmentalists have acted in ways that unnecessarily drive up costs and impede progress on environmental issues. This static mind-set has thus created a self-fulfilling prophecy leading to ever more costly environmental regulation. Regulators tend to set regulations in ways that deter innovation. Companies, in turn, oppose and delay regulations instead of innovating to address them Porter and Van der Linde (1995) did not feel it naïve to expect that reducing pollution will often enhance competitiveness, because pollution often is a form of economic waste.

Economists have largely clung to the black-and what 'are regulations good or bad?' angle of Porter and Van der Linde's work. Yet there are strong undercurrents of TBL thinking in what is usually referred to as "The Porter Hypothesis" and also in the many papers it has spawned: specifically, the normative idea that if industries can realise efficiencies through technology adoption forced on them by environmental regulation, then they should have been pursuing a TBL all along.

In the corporate context we generally think of TBL concepts relative to a single decision maker (purportedly) optimizing over three different objectives. Halpern et al. (2013) note that maximizing conservation goals and achieving equity in social outcomes, while minimizing overall costs, is the ideal triple bottom line outcome. Stephenson et al. (2017) purport a quadruple bottom line for fisheries, or four "pillars of sustainability" that, in addition to economical, ecological and social "pillars", also includes institutional aspects. Institutional or managerial objectives of "simplifying and improving management structures" were also considered by Pascoe et al. (2013).

In a fisheries context, the Food and Agriculture Organization of the United Nations (FAO) developed a Code of Conduct for Responsible Fishing which addresses issues of bycatch and responsible fisheries management (FAO 1995). 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).

While these are excellent objectives, they have not been operationalised with fishery harvest strategies (Mangel and Dowling, 2016). More broadly, as Ecosystem Based Fishery Management (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 to intuitive forecasting methods, such as a Delphic approach (e.g. Dichmont et al. 2016 – Mackay-Buderim), which is a polling technique employed for the systematic solicitation of expert opinion (Bernstein and Cetron, 1969). Additionally, arbitrary increases in catch often become a proxy for socio-economic considerations.

Benson and Stephenson (2017) reviewed TBL methods, finding that two of seven proposed tools to support decision making in the management system could provide tactical advice, but only one (management strategy evaluation, MSE) provided advice that consistent with their criteria for generation, transmission, and use of scientific information in management advisory processes. Furthermore, formal methods that acknowledge the TBL (Table C1, from Benson and Stephenson 2017) 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 MSE (e.g. Plaganyi's Tropical Rock Lobster model) is conditioned on how TBL objectives are weighted, and there is no means to formally make recommendations that reconcile interest groups.

Stephenson et al. (2017) affirm that, while international agreements and legislation call for incorporation of four pillars of sustainability, the social (including cultural), economic and institutional aspects (the 'human dimension') have been relatively neglected to date. They identify three key impediments: a relative lack of explicit social, economic and institutional objectives; a general lack of process (frameworks, governance) for routine integration of all four pillars of sustainability; and a bias towards biological considerations.

On the other hand, (Costello et al., 2016) argues that if fisheries are "properly" managed, there are no trade-offs between objectives. In practice, this is difficult to achieve. There is a difference between the overriding long-term improvements experienced when formal management is introduced to a depleted, previously unmanaged stock, and balancing economic, environmental, and social objectives, that are often directly at odds with each other. Some progress has been made: Anderson et al. (2015) present a Fishery Performance Indicators (FPI) tool for assessing performance in individual fisheries, and for identifying the links between enabling conditions, fisheries management strategies and triple bottom line outcomes. Rindorf et al. (2017) describe a "Pretty Good Multispecies Yield" space, and extend this to a "pretty good multidimensional yield" that accommodates situations where the yield from a stock affects the ecosystem, economic and social benefits, or sustainability. The approach takes combinations of fishing mortalities that provide >95% of the yield in a single stock analysis, and then excludes combinations that are i) undesirable in an ecosystem context; ii) incompatible with technical interactions; and iii) incompatible with economic and social objectives. While both of these approaches attempt to account for TBL objectives, neither consider differences in weightings between stakeholders or stakeholder groups.

At the same time, work has been done on weighting stakeholder preferences against TBL objectives: Pascoe et al. (2013) assessed the relative importance of the different objectives to different stakeholder groups in the Queensland Eastern Trawl Fishery, Australia, using the Analytic Hierarchy Process (Table C2). This is based upon the construction of a series of pairwise comparison matrices which compare sub-objectives to one another, and a key advantage is that only two elements or objectives are being compared at any one time rather than all objectives having to be compared with each other simultaneously. Across stakeholder interest groups, Pascoe's (2013) East Coast Trawl weightings show a 4-fold difference in economic outcomes, 2-fold in social outcomes, and almost 2-fold in environmental outcomes. This motivates the need to reconcile weightings, and therefore, triple bottom line harvest strategies, across interest groups.

Thus, the question remains as to how to optimise a TBL value function, given a set of weightings, across a range of scenarios, across 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 commensurate. Mangel and Dowling (2016) (Plummer paper) demonstrated a more fundamental way of interpreting weightings for various stakeholder groups, in the form of a single, TBL value function. We here generate a Pareto frontier over which a given strategy can be optimised for any combination of weightings, against which trade-offs can be assessed.

Our approach is consistent with the "efficiency frontier" presented by Halpern et al. (2013), whereon optimal solutions lie, and represent different importance (weight) given to conservation versus equity goals. As opposed to the approach of Rindorf (2017) that takes a suite of fishing mortalities corresponding to sustainable yield, and progressively refines this, we consider the TBL objective weighting profile for given stakeholder groups as an integrated value function that is optimised across a suite of catch levels.

While our Pareto frontier can find an optimal strategy for a given set of preferences, it cannot, however, directly reconcile among different stakeholders with different sets of weightings. The question then becomes how to make sense of, and seek an overall optimum solution among differing sets of stakeholder preferences.

We here provide a rational formal means to reconcile the stakeholder preferences. That is, we illustrate a formal way in which to trade off the values across the various sets of weightings, where these show a lack of agreement among stakeholders. Our work can alternatively be seen as a demonstration of a rational approach to "mutually disagreeing". The main contribution of the paper is to apply a TBL-type value function to stakeholder groups using weightings loosely based on a real fishery– the East Coast Trawl Fishery in Queensland, Australia, using empirically estimated value functions for each of the three TBL components and weightings from each from 5 interest groups (including fishery managers) (Table C3).

Even if everyone agrees on weights (preferences), our Pareto frontier provides a more elegant way of optimising over multiple strategies, and multiple indicators (e.g. Mapstone et al. 2008) comprising a value function. But where there are a range of difference preferences (e.g. Pascoe et al. 2014), we apply a single value function and determine the cost to each group of what is being lost, if their optimal strategy is not adopted. This can be considered analogous to game theory, or Nash equilibrium – the best overall compromise will see everyone sacrificing a little bit, but by moving too far away from this optimal point of compromise, someone will do worse.

Our work is motivated by the increasing acknowledgement of the need to directly acknowledge and operationalise the triple bottom line within fishery harvest (management) strategies. Ecosystem-based fisheries management has been considered in a quantitative sense by Richerson et al. (2010) sandeels (Ammodytes spp.) in the Shetland Islands. Many fisheries in Queensland, Australia are directly confronted with the need for triple bottom line management, given their recent fishery management reforms, their multi-sector nature of many of their fisheries, and their interaction with the Great Barrier Reef Marine Park. In particular, the Coral Reef Finfish Fishery and the East Coast Trawl Fishery are the subject of a project to operationalise TBL harvest strategies. Haida Gwai herring in Hawaii are also faced with similar multi-sector and environmental issues.

We choose a simple model with a limited number of components to each of the TBL values, and base our value function around a simple biological model, so that the concepts are explicit and to facilitate the clarity of the ideas.

Methods and Results

The basic purpose of the exercise is to choose a policy option (level of total *catch* in a fishery) that maximises social welfare. We propose a total social value function that consists of three additive components: economic benefits (net *revenue* to commercial fishery participants), conservation benefits (population level of a *predator* species, which is a function of the abundance of the targeted species), and community/social benefits (defined in this example as effort applied to the fishery by an *artisanal* fishing fleet).

$V = \rho_r V_r(C) + \rho_p V_p(C) + \rho_a V_a(C)$

where

Vr(C) is a net *revenue* function that is increasing in *C*

Vp(C) is stock dynamics curve for a *predator* species that is decreasing in *C*

Va(C) is an effort function for the *artisanal* fleet that is increasing in *C*.

Population model and value function

Our approach is to optimise, over a range of possible catch levels, a value function for a given set of stakeholder group weightings. We consider a fishery with both commercial and artisanal (recreational, subsistence) components. In doing so, we attempt to determine both the optimal catch level across the triple bottom line, and the relative allocation of catch to each of the commercial and artisanal sectors.

Taking the same line as Richerson et al. (2010) and Munch et al. (2017), we define a value(s) for each of: i) economic, ii) environmental (biological), and iii) social objectives, each of which is some function (directly or indirectly) of catch. Each of the three TBL components of the total social value function are denominated in different units (money, predator population, effort levels for the artisanal fleet) and can be scaled 0 to 1.We apply a corresponding weight to each value, and sum to obtain an overall value function, that can be maximised over the range of possible catches. Rather than thinking about optimizing over welfare functions for different groups, we suppose that each stakeholder has a welfare function that is defined along three axes (economic profit, environmental sustainability (using population size of dependent predator as a proxy, and a combination of artisanal effort and the level of “anti-social impact” inflicted by commercial effort). The latter speaks to users such as recreational divers, or the tourism industry, whose utility is proportional to the state of preservation of marine habitat.

We define the value function for the TBL as a function of management strategy, M. We here assume the management strategy equates to a total allowable catch, bit it could alternatively equate to management by fishing mortality, F, such as in an effort-quota fishery.

For purposes of example, we assume that

i) The economic metric equates to the net commercial revenue, being maximised at the maximum economic yield, MEY .
ii) The conservation metric equates to the. population size of some dependent predator of the target species, and that this is decreasing function of management strategy, being maximised at zero take.
iii) The social metric equates to the level of artisanal effort relative to that of the commercial fleet, and to some "carbon footprint"-style "anti-social impact" function associated with the level of commercial effort. Respectively, these are maximised and minimised at some $M*$.

The value function is expressed as

$$V = \rho_r V_r(\bar{C}) + \rho_p V_p(\bar{C}) + \rho_s V_s(\bar{C}) \qquad (1)$$

where

$V_r(\bar{C})$ is the economic value as a function of catch, $\bar{C}$, with corresponding weighting ρ_r,

$V_p(\bar{C})$ is the environmental value as a function of catch, $\bar{C}$, with corresponding weighting ρ_p, where

ρ_p has maximum value $(1 - \rho_r)$, and

$V_s(\bar{C})$ is the social value as a function of catch, $\bar{C}$, with corresponding weighting ρ_s, where

$\rho_s = 1 - (\rho_r + \rho_p)$.

For any set of weightings, can find the catch that maximises the value function.

The key points with this approach are that: i) each value metric can be quantified in a relative manner, such that it can be standardised to range between 0 and 1, ii) values are all some function of the strategy to be employed (in this case, a catch limit) and ii) weightings can be specified, through some form of revealed preference analysis, such as an analytical hierarchical method (per Pascoe et al. 2013).

In a more traditional firm-level analysis a firm's preferences would be captured by a profit function. If one were to aggregate up to the industry level then the industry's preferences would be captured by the sum of all the individual profit functions. In asking how individuals weight revenue versus environmental sustainability versus artisanal fishing and extent of commercial "anti-social impact", we are making the case that these metrics are composite commodities.

The concept of value is rooted in the concept of maximum willingness to pay for something. The fishing industry's maximum willingness to pay for a catch that generates π_1 is π_1 so the fishing industry's value function for profits is the profit function. In asking how the fishermen weight revenue versus dependent predators versus artisanal effort/commercial "anti-social impact", we are saying, "revenue is an indicator for industry success/sustainability," and "birds is an indicator of ecosystem health", and "artisanal effort is an indicator for fishing community health". As such, asking for weights is analogous to asking, "In your meta decision process, what weight do you give to industry success versus ecosystem health, versus social and broader community values?"

Acknowledging the more standard economics approach, an alternative way to express the above value function is to start with an arbitrary individual i and express their utility function as a function of catch ($), dependent predator abundance ($Pred$), and artisanal effort combined with level of "anti-social impact" ($Art_E + Dam$).

$$u^i(c) = f(\$, Pred, Art\_E + Dam)$$

Per equation (1), one can rewrite $u^i(c)$

$$u_i(c) = U^1(V_r(c)) + U^2(V_p(c)) + U^3(V_s(c))$$

That is, there is:

- A technical function (V_r) that maps fishing effort/catch into dollars. So this is a production function
- A subjective function (U^1) that maps this financial outcome to welfare for individual i (the economic) weighting)
- Another "technical" function (V_p) that defines the population dynamics of the dependent predator
- Another subjective function (U^2) the maps dependent predator outcomes to welfare for individual i (the environmental sustainability weighting)
- A third technical function (V_s) that maps catch and effort into the social indicator
- A third subjective function U^3 that expresses this social outcome in welfare terms (the social weighting).

If we want to think about the change in individual $i's$ welfare with a change in catch, this is

$$\frac{\partial u}{\partial c} = \frac{\partial U^1}{\partial V_r}\frac{\partial V_r}{\partial c} + \frac{\partial U^2}{\partial V_p}\frac{\partial V_p}{\partial c} + \frac{\partial U^3}{\partial V_s}\frac{\partial V_s}{\partial c}$$

This is analogous to what the weights in equation (1) are doing:

- $\frac{\partial U^1}{\partial V_r}$ is the marginal utility of fishing revenue to individual i
- $\frac{\partial V_r}{\partial c}$ is the marginal revenue of the harvest function
- $\frac{\partial U^2}{\partial V_p}$ is the marginal utility of the dependent predator population to individual i
- $\frac{\partial V_p}{\partial c}$ is the contribution/degradation of an additional unit of catch to the dependent predator population function.
- $\frac{\partial U^3}{\partial V_s}$ is the marginal utility of the artisanal effort and commercially unfished habitat to individual i

- $\frac{\partial V_s}{\partial c}$ is the contribution/degradation of an additional unit of catch to the social value function.

Using this form of equation (1), there are some quantities that are analogous to standard production economics quantities. For example, in a standard profit analysis the value of an additional unit of input (x) is

$$\frac{\partial \pi}{\partial x} = p\frac{\partial y}{\partial x}$$

where $y = f(x)$ is the production function that transforms inputs to outputs and p is the price of the output.

If we think about catch as the primary input into the fishing industry profit function then, from the bulleted list above, we have $\frac{\partial V_r}{\partial C}$ which is the marginal contribution of an additional unit of catch to industry profits. Then $\frac{\partial U^1}{\partial V_r}$ is playing the role of p in that it determines how the output from an additional unit of input is being valued. This is consistent with some standard results of public goods provision. For instance, the Samuelson Condition for efficient provision of public goods is:

$$\sum_i MB_i = MC$$

which says that public goods should be provided up to the level where the sum of each individual's marginal benefit equals the marginal cost of provision.

If the dependent predator population size can be considered the "public good" and their population is inversely related to catch, then the marginal cost of predator provision is the foregone profits from however many units of catch are required to get one more predator.

Equation (1) replaces the $U(\,)$ functions above with single value parameters (ρ). As such, it is important to acknowledge that non-constant marginal rates of substitution cannot currently be accommodated.

For simplicity, we assume a steady state Schaefer model:

$$\frac{dN}{dt} = r\bar{N} * \left(1 - \frac{\bar{N}}{K}\right) - \bar{C} \qquad (2)$$

We take fixed values for the intrinsic population growth rate, r, and carrying capacity, K:

- $r = 0.2$
- $K = 10000$

We assume that fishers catch only the surplus production:

$$\bar{C} = r\bar{N} * \left(1 - \frac{\bar{N}}{K}\right) \qquad (3)$$

For any given value of $\bar{C}$, we can rearrange (3) to obtain a quadratic solution for $\bar{N}$:

$$\bar{N} = \frac{K}{2} \pm \frac{\sqrt{r^2 - \frac{4r\bar{C}}{K}}}{\frac{-2r}{K}} \qquad (4)$$

We take the maximum value of the two possible solutions.

We further assume that catch is some function of effort, E

$$\bar{C} = \left(1 - e^{-q\bar{E}}\right)\bar{N} \qquad (5)$$

with catchability, $q = 0.002$

Equation (5) can be rearranged to give

$$\bar{E} = \frac{-ln\left(1 - \frac{\bar{C}}{\bar{N}}\right)}{q} \qquad (6)$$

We assume a commercial and an artisanal fleet. Each of these has separate catchabilities, but we assume the artisanal catchability q_{art} is some fixed fraction of the commercial catchability $q_{com}[i]$:

$$q_{art} = 0.2 * q_{com}[i] \qquad (7)$$

It follows that:

$$q_{com}E_{com} + q_{art}(q_{com})E_{art} = -ln\left(1 - \frac{\bar{C}}{\bar{N}}\right) \qquad (8)$$

We assume that the maximum artisanal effort is that corresponding to the artisanal sector taking the entire catch, $\bar{C}$:

$$\overline{E_{art\_max}} = \frac{-ln\left(1 - \frac{\bar{C}}{\bar{N}}\right)}{q_{art}(q_{com})} \qquad (9)$$

Across a set of values, $\overline{E_{art0}}$ to $\overline{E_{art\_max}}$, the share of the total catch taken by the artisanal sector is

$$\overline{C_{art}} = \bar{N}\left(1 - e^{-q_{art}(q_{com}) * \overline{E_{art}}}\right) \qquad (10)$$

Thus the catch remaining for the commercial sector will be

$$\overline{C_{com}} = \bar{C} - \overline{C_{art}} \qquad (11)$$

The corresponding effort for the commercial sector, under catchability q_{com} is therefore

$$\overline{E_{com}}(\overline{C_{com}}) = \frac{-ln\left(1 - \frac{\overline{C_{com}}}{\bar{N} - \overline{C_{art}}}\right)}{q_{com}} \qquad (12)$$

Calculating economic value:

We assume that the economic value equates to the revenue from the commercial fleet, scaled relative to that at MEY, $Rev(MEY)$

$$V_r(\overline{C_{com}}) = \frac{p\overline{C_{com}} - c_0 \bar{E}_{com}(\overline{C_{com}})}{Rev(MEY)} \qquad (13)$$

Where p is the unit price, and c_0 is the cost per unit of effort.

We then calculated MEY analytically:

$$\frac{dN}{dt} = r\bar{N} * \left(1 - \frac{\bar{N}}{K}\right) - qE\bar{N} \qquad (14)$$

At steady state, $\frac{dN}{dt} = 0$, so $\bar{N} = K\left(1 - \frac{qE}{r}\right)$, and therefore revenue, Rev, equals

$$Rev = p \cdot qEK\left(1 - \frac{qE}{r}\right) - c_0 E \qquad (15)$$

We solve this quadratic for effort, which gives the maximum revenue, and hence MEY.

Calculating the environmental value:

We assume the environmental value, $V_e(\bar{C})$, is represented by the abundance of a next-order predator, which is assumed to be a logarithmic function of the target species:

$$V_e(\bar{C}) = \frac{\alpha \ln(\bar{N}(\bar{C})) - \beta}{\alpha \ln(\bar{N}(0)) - \beta} \qquad (16)$$

where the value is scaled relative to the predator abundance under zero fishing.

For the sets of 1000 random weightings, and the 11x11 crosses of fixed weightings, alpha=0.1, beta = 0.8 (to give a greater range of V_e values.

For the fixed set of stakeholder group weightings, alpha=0.1, beta = 0.05 (as otherwise, V_e didn't tease out but clumped across groups).

Calculating social value:

We assume that there are two components to the social value:

a) artisanal effort relative to that of commercial fleet, calculated as:

$$V_a(\bar{C}(q_{com}), \overline{E_{art}}) = \frac{\overline{E_{art}}}{\overline{E_{art}} + \overline{E_{com}}} \qquad (17)$$

Since

$$\overline{E_{com}}(\overline{C_{com}}) = \frac{-ln\left(1 - \frac{\overline{C_{com}}}{\bar{N}}\right)}{q_{com}} \qquad (18)$$

it follows that

$$\overline{E_{art\_max}} = \frac{-ln\left(1-\frac{\bar{C}}{\bar{N}}\right)}{q_{art}(q_{com})} \qquad (19)$$

We calculate the value function (eqn (17) over a range of artisanal effort values, from 0 to $\overline{E_{art\_max}}$, where

$$\overline{C_{com}} = \bar{C} - \overline{C_{art}} \qquad (20)$$

and

$$\overline{C_{art}} = \bar{N}\left(1 - e^{-q_{art}(q_{com})*\overline{E_{art}}}\right) \qquad (21)$$

The second social value component is

b) a habitat/carbon footprint of "anti-social impact", that is assumed to be directly proportional to the level of commercial effort:

$$V_d(\overline{C_{com}}) = \frac{d_0E(MSY) - d_0E(\overline{C_{com}})}{d_0E(MSY)} = 1 - \frac{E(\overline{C_{com}})}{E(MSY)} \qquad (22)$$

where

$$E(MSY) = -log(1 - (rK/4)/\overline{N(MSY)})/q_{com} \qquad (23)$$

Both components are equally weighted to give the total overall social value function:

$$V_s(\bar{C}) = \alpha V_a(\bar{C}(q_{com}), \overline{E_{art}}) + (1-\alpha)V_d(\overline{C_{com}})\ ; \alpha = 0.5 \qquad (24)$$

Each of the triple bottom line value functions, together with steady-state abundance and commercial effort, is plotted as a function of total steady-state catch in Figure C1. Note that performance is an artefact of how the triple bottom line objectives are constructed.

The overall weighted value function is:

$$V = \rho_r V_r(\bar{C}) + \rho_e V_e(\bar{C}) + \rho_s V_s(\bar{C}) \qquad (25)$$

where ρ_e has maximum value $(1 - \rho_r)$

and $\rho_s = 1 - (\rho_r + \rho_e)$

For any set of weightings, we can find the catch that maximises the value function.

Systematic example

Looking across a broad range of weightings (rather than specific, discrete sets) is important, because of the ability this confers to identify sensitivities on the Pareto curves/frontiers

Sensitivity of the optimised catch to the weightings was investigated by undertaking the value function optimisation for 1000 randomly generated weightings. Figure C1 shows that, while there is a gradual decrease in commercial catch for the optimised value function with increased environmental and decreased economic weightings, there is a clear frontier of pairwise weighting combinations below which the commercial catch is negligible. As such, for the value functions as currently articulated, there is strong sensitivity when weighting profile are close to this frontier.

Contour plots show the optimal total values (Figure C2) across the weighting surface (recalling that the social weighting is (1-(economic + environmental weighting). Figure C2 shows clearly that the overall value is optimised when no economic weighting is assigned, or, alternatively, along a linear frontier of trade-offs between the levels of economic and environmental weightings. Overall value is minimised at the lowest environmental weightings, for intermediate economic weighting.

To examine the overall behaviour of the value function for the assumed fixed inputs, we optimised over the following range of possible total catch:

$$\bar{C} = (10,25,50,75,100,125,150,175,200,225,250,275,300,235,350,375,400,425,450,500)$$

For the following combinations of weightings:

$\rho_r = c(0,0.1,0.2,0.3,0.4,0.5,0.6,0.7,0.8,0.9,1.0)$ crossed with

$\rho_e = c(0,0.1,0.2,0.3,0.4,0.5,0.6,0.7,0.8,0.9,1.0)$,

for all combinations whose sum did not exceed 1.0.

A profile of values corresponding to the overall maximised value function, across the range of weightings considered (Figure C3) shows vertical cascading due to the range of artisanal effort values considered, but showed a negative correlation between economic and environmental values, and between economic and social values, with positive correlation between environmental and social values. While the scaled commercial revenue ranged between 0 and 1, the scaled environmental value ranged between 0.95 and 1.0, such that predator numbers, as articulated, were not strongly affected by the values of catch considered.

It is the overall value function (equation 25) that is maximised for any given set of weightings, Figure C4 shows the corresponding levels of commercial catch, and economic, environmental and commercial value as a function of the randomly selected combinations of economic and environmental weightings. Predictably, catch and economic value were highest for larger economic weighting values, and were negligible below economic weights of ~0.4. Regardless of the environmental weighting, environmental and social value were highest below this ~0.4 economic weight threshold (Figure C4).

<u>Practice problem application</u>

We now consider an example based on the weightings elicited from the East Coast Trawl Fishery in Queensland, Australia (Pascoe et al. 2013), where an international review of natural resource management objectives was used to develop a candidate list. Objectives most relevant to the fishery were short-listed by a scientific advisory group. Additional objectives specific to Queensland fisheries management also identified and incorporated. The relative importance of the different objectives to different stakeholder groups was then assessed using the Analytic Hierarchy Process. We re-normalised the three weightings corresponding to the three main TBL objectives, to obtain weightings for 6 sets of stakeholder groups. We used these to guide our choice of simplified weightings for 5 of these groups (omitting “onshore industry”) (Table C3).

We assume that each individual belongs to only 1 group, that individuals within a group have identical utility function, and that every individual in the system (not every group) gets equal weight. Thus we are abstracting from the individuals that comprise these groups and just considering the groups in the aggregate.

As with the systematic example (of which this is a subset), the environmental value was not compromised greatly using current parameterisation and value functions, while the commercial revenue spans the full range (Figure C5; Table C3). Stakeholder groups clustered as expected. The conservation and recreational fishery groups, predictably, shared an optimal solution where the environmental and social value components were maximised, and commercial revenue was zero. The commercial industry and fishery managers had similar weighting profiles, and their value functions were optimised at high levels of commercial catch and minimal social and environmental values. The “local communities” group was intermediate between these extremes.

The total catch levels that correspond to the optimised value functions are shown in Table C3. Generally, these are either close to their maximum or minimum values: there are few instances where the level of catch is intermediate. It was generally seen that when the economic weighting was less than 0.15, the catch level was minimised, and all of this was allocated to the artisanal sector. Moreover, there is a fine balance regarding the level of commercial catch: the recreational fishery and local community stakeholder groups had similar weighting profiles, but that corresponding to the local community stakeholder group (economic weighting 0.2) resulted in a substantial level of commercial catch, whereas that for the recreational fishery (economic weighting 0.15) did not.

While the sensitivity to weighting profiles is a function of how the value function is articulated, this nonetheless speaks to “pivot points”, or frontier edges – combination of weightings about which the catch profile dramatically changes – that can occur regardless of how the value functions are specified. As illustrated for the East Coast Trawl Fishery example, it is important to identify these frontiers and where they overlap with stakeholder group weighting profiles, as areas of sensitivity.

Reconciling TBL values among stakeholder groups

Again using the simple set of 5 stakeholder group weightings loosely based on Pascoe et al (2013) – Table C3, we now consider 10 discrete strategies: 5 levels of overall total catch, and two allocation ratios (0.3, and 0.7) between the commercial and artisanal sectors.

For each of the two allocation ratios, we attempt to determining the overall optimal set of stakeholder group weightings (TBL value profile) that minimises the trade-off in optimal performance, given the optimal strategy (level of catch) for any given stakeholder group. That is, given one stakeholder group's TBL values, we can identify those values corresponding to the optimal strategy for any other stakeholder group (according to their own TBL values). We are trying to formally identify an overall optimal strategy as that which affords the least amount of loss in optimal value across all groups.

Figure C6a and b shows the value profiles as a function of the strategies for each stakeholder group, and also shows the optimal strategies for each stakeholder group, transposed on to the fishery managers' value profile. Considering Figure C6a (0.3 allocation ratio), it can be seen that the optimal strategy (level of catch) from the fishery managers' perspective (i.e. that for which the value function is optimised) is 450t of catch. The conservation stakeholder group's weighting profile shows their value being optimised at 175t of total catch. If we assume this level of catch and apply it to the fishery managers' value profile, per the right-hand panel of Figure C6a, it can be seen that this level of catch equates to a trade-off in value for the managers of only 6%.

These trade-offs are presented numerically in Table C4, the rows of which are the TBL values for one of each of the stakeholder groups (according to their weightings), at which the strategy (catch, with %art to the artisanal sector) is maximised according to each of the stakeholder group weightings, in each column. The columns are the TBL value by stakeholder group.

- Diagonals are the same values relative to themselves; hence the relative and absolute differences of 1 and 0, respectively. The upper matrix shows the absolute values, the middle the differences relative to that corresponding to the group's optimal strategy, while the lower matrix shows absolute differences.

For a set of stakeholder group weightings to be overall optimal, or, conversely, the least favourable (such that a "minimum whinge" level can be considered (Hilborn 2007)), we consider two metrics:

i) Average values:
   a. that the average relative trade-off in value (according to that stakeholder group) across optimal strategies for each of the stakeholder groups, be closest to 1 (or zero, if considering absolute differences) – that is, managers' weightings across stakeholder groups DO matter.
   b. Similarly, the least favourable group is that for which the average relative-trade-off is closest to 1, and the absolute difference is largest in magnitude. A "minimum whinge" criterion can be considered

ii) "Maximin" approach:
   a. that the minimum relative (or maximum absolute) trade-off in value (according to that stakeholder group) across optimal strategies for each of the stakeholder groups, is maximised (or minimised, if considering absolute differences) ("maximin" approach) – that is, managers' weightings across stakeholder groups DO NOT matter.

b. Similarly, the least favourable group is that for which the minimum relative-trade-off is the smallest, and the absolute difference is largest in magnitude

Alternatively, for a stakeholder group's strategy to be overall optimal, the same criteria can be applied, except the average and minimum relative values are taken across all stakeholder groups for each given strategy.

Using the simplified set of weightings, the above criteria are suggesting that we should defer to the **recreational group** set of *weightings* using the "average" criterion, or **conservation or local community group** sets of *weightings*, using the" maximin" criterion, as being optimal across all stakeholder groups. The criteria generally suggest that the overall optimum *strategy* is that corresponding to the fishery managers

However, the fact that these recommendations differ depending on the criteria used to determine the optimal overall strategy indicates that there is sensitivity to the between-group weightings. The least desirable set of weightings was those of the fishing industry, with the exception of the 30% allocation ratio, for which the "average" criterion suggested that the fishery managers' weighting profile was the least desirable.

For example (considering 30% artisanal catch) this is saying the, for the commercial fishing industry, their optimal value (~0.75) for their set of weightings occurs at a strategy taking 450 units of catch. On the TBL value curve for the recreational group weightings, 450 catch units equates to a value of ~0.76. Relative to the optimal value for the recreational group, which occurs at a catch of 250 and a value of 0.79, this is only a 0.3% compromise in community group optimal value.

Under the recreational weighting profile, the commercial fishery would be the worst off, but would still be within 97% of their optimal value (cell H26, which shows the relative commercial fishery TBL value given the optimal recreational strategy - the latter of which would be adopted if it were taken that the recreational weightings were overall optimal.

An alternative metric is to consider the overall level of satisfaction with any given strategy. This is obtained using the same "average" and "maximin" criterion as before, but taking these down the columns (i.e. for any given strategy) as opposed to across the rows (i.e. for any given group). The optimal strategy was generally that of the fishery managers or the community group, which is to be expected, given that their weighting profile in intermediate to that of the other groups. The conservation group's optimal strategy was consistently the least desirable, skewed largely by the compromise felt by the fishing industry group if adopting this strategy.

It is a concern that conflicting recommendations emerge when considering the relative loss in value to a group by assuming the optimal strategy of another, across the strategies as opposed to across the groups.

REFERENCES

Anderson, J. L., C. M. Anderson, J. Chu, J. Meredith, F. Asche, G. Sylvia, M. D. Smith, D. Anggraeni, R. Arthur, A. Guttormsen, J. K. McCluney, T. Ward, W. Akpalu, H. Eggert, J. Flores, M. A. Freeman, D. S. Holland, G. Knapp, M. Kobayashi, S. Larkin, K. MacLauchlin, K. Schnier, M. Soboil, S. Tveteras, H. Uchida, and D. Valderrama. 2015. The Fishery Performance Indicators: A Management Tool for Triple Bottom Line Outcomes. PLoS ONE 10:e0122809.

Benson, A. and Stephenson, R.L. 2017. Options for integrating ecological, economic, and social objectives in evaluation and management of fisheries. Fish and Fisheries 2017: 1-17. DOI: 10.1111/faf.12235

Elkington, J. 1997. Cannibals With Forks: The Triple Bottom Line of 21st Century Business. Capstone, Oxford, 1997, 402 pp. ISBN 1-900961-27-X.

FAO. 1995. Code of Conduct for Responsible Fisheries. Available at: ftp://ftp.fao.org/docrep/fao/005/v9878e/v9878e00.pdf.

Goldberg, E. 2001. Profit, People, Planet: Policy Issues and Options Around Triple Bottom Line Reporting. NZ Department for the Environment, 52pp.

Head, L., D. Trigger, and E. Woodward. 2004. Nature, culture and the challenges of environmental sustainability: Bridging the science/humanities divide: An emerging research network. Australian Research Council Special Research Initiative ID: SR0354620:86 pp.

Hilborn, R. 2007. Defining success in fisheries and conflicts in objectives. Marine Policy 31(2): 153-158.

Magnuson-Stevens Act, 16 U.S. Code § 1851 (1996)

Mangel, M., and Dowling, N.A. (2016). Reference Points for Optimal Yield: A Framework for Assessing Economic, Conservation, and Sociocultural Tradeoffs in Ecosystem-Based Fishery Management, Coastal Management, DOI: 10.1080/08920753.2016.1208884

Porter, M. E., & Van der Linde, C. (1995). Green and competitive: ending the stalemate. Harvard business review, 73(5), 120-134.

Richerson K, Levin PS, Mangel M (2010) Accounting for indirect effects and non-commensurate values in ecosystem based fishery management. Marine Policy 34:114–119

Pascoe et al. 2013 Management objectives of Queensland fisheries: Putting the horse before the cart. Marine Policy 37 (2013) 115–122

Rindorf, A., Dichmont, C. M., Levin, P. S., Mace, P., Pascoe, S., Prellezo, R., Punt, A. E., Reid, D. G., Stephenson, R., Ulrich, C., Vinther, M.,and Clausen, L. W. Food for thought: pretty good multispecies yield. – ICES Journal of Marine Science, 74: 475–486.

Stephenson, R. L., Benson, A. J., Brooks, K., Charles, A., Degnbol, P., Dichmont, C. M., Kraan, M., Pascoe, S., Paul, S. D., Rindorf, A., and Wiber, M.2017. Practical steps toward integrating economic, social and institutional elements in fisheries policy and management. – ICES Journal of Marine Science, doi:10.1093/icesjms/fsx057.

Suggett, D., and B. Goodsir. 2002. Triple Bottom Line Measurement and Reporting in Australia - Making it Tangible. 116 pp.

Table C1: A summary of TBL methods reviewed by Benson and Stephenson (2017)

Method	Description	Reference
Ecological risk assessment for the effects of fishing (ERAEF)	is a method for planning fisheries research and management activities based on estimates of relative risk	(Hobday et al., 2011; Smith et al., 2007)
Management strategy evaluation (MSE)	uses simulation models to compare alternative harvest management strategies under a variety of assumptions about the dynamics of fish and fisheries. The typical goal of MSE is to identify harvest management strategies that are robust to uncertainty for a particular fishery system	(Smith, Sainsbury, & Stevens, 1999)
Descriptive multivariate models (DMVM)	are methods that measure the status of a community, food web, or ecosystem using commonly available data	(Link et al., 2002).
Dynamic multispecies models	address problems of interacting species or fisheries that are not typically included in single-species models. They address two distinct problems: biological interactions in the form of predation and competition (trophic dynamics), and technical interactions, which address issues of joint capture of species by multiple fishing units in time and space	(Murawski, 1984; Pope, 1979).
Aggregate species (“production”) models represent the	relationship between fishing effort and yield for aggregate species groups and fish communities	(Gaichas et al., 2012; Pope et al., 2006).
Minimum realistic models (MRM) and models of intermediate complexity for ecosystem assessments (MICE)	reduce the structure of ecosystem models to the minimum required to inform management	(Butterworth & Punt, 1999; FAO 2008; Plagányi, 2007; Plagányi et al., 2014).

Bayesian belief networks (BBNs) (McCann, Marcot, & Ellis, 2006).	graphically depict causal relationships among key components of a management system. They represent uncertainty about both the natural resource and its response to management intervention by probabilistically representing relationships between system variables	(Cain, 2001).

Table C2: Overall weightings by stakeholder group for the Queensland, Australia, East Coast Trawl Fishery, summarised based on the results of Pascoe et al. (2013)

	Fishing industry	Onshore industry	Fisheries managers	Conservation	Recreational fishing	Local communities
Maximise economic performance	0.44	0.46	0.31	0.11	0.12	0.20
Maximise social outcomes	0.16	0.12	0.16	0.18	0.34	0.21
Ensure sustainability	0.40	0.42	0.53	0.70	0.54	0.59

Table C3: Optimised catches and values by stakeholder weighting set, for the five hypothetical stakeholder groups detailed in Pascoe et al. (2013).

	weightings			**catch**			**value**			
Stakeholder group	economic	environmental	social	total	artisinal	commercial	economic	environmental	social	overall
	roRev	roEnv	roSoc	Cbar	C_art	C_com	Vr_ana	Ve	Vs	Value_ana
Fishing industry	0.45	0.4	0.15	500	0	500	0.997	0.920	0	0.817
Recreational fishery	0.15	0.5	0.35	10	10	0.000	0.000	0.999	1	0.850
Conservation	0.1	0.7	0.2	10	10	0.000	0.000	0.999	1	0.900
Local communities	0.2	0.6	0.2	400	122.397	277.603	0.563	0.963	0.627	0.816
Fishery managers	0.3	0.5	0.2	450	46.448	403.552	0.815	0.952	0.351	0.791

Table C4: Determining the overall optimal set of stakeholder group weightings (TBL value profile), that minimises the trade-off in optimal performance given the optimal strategy (level of catch) for any of the 4 given stakeholder group. Rows are the TBL values for one of each of the stakeholder groups (according to their weightings), at which the strategy (catch, with %art to the artisanal sector) is maximised according to each of the stakeholder group weightings, in each column. Columns are the TBL value by stakeholder group.

For a set of stakeholder group weightings to be overall optimal, we consider two criteria: i) that the average relative trade-off in value (according to that stakeholder group) across optimal strategies for each of the stakeholder groups, be closest to 1 (or zero, if considering absolute differences); ii) that the minimum relative (or maximum absolute) trade-off in value (according to that stakeholder group) across optimal strategies for each of the stakeholder groups, is maximised (or minimised, if considering absolute differences).

ROWS are the TBL values for one of each of the stakeholder groups (according to their weightings), at which the strategy (catch, with %art to the artisanal sector) is maximised according to each of the stakeholder group weightings, in each column.

ROWS: **Stakeholder group weightings defining the strategy**

Ref = value for the stakeholder group whose weightings are defining the strategy and the TBL value

COLUMNS:			roRev	roEnv	roSoc	%art	Fish_ind_strat	Rec_fish_strat	Conserv_strat	Local_com_strat	Fish_mgrs_strat	TBLval_ref
TBL value	Fish_ind_TBLval	1	0.45	0.4	0.15	0.3	0.75	0.65	0.62	0.73	0.75	0.75
by stakeholder group	Rec_fish_TBLval	2	0.15	0.5	0.35	0.3	0.77	0.80	0.79	0.79	0.77	0.80
	Conserv_TBLval	3	0.1	0.7	0.2	0.3	0.85	0.87	0.87	0.86	0.85	0.87
	Local_com_TBLval	4	0.2	0.6	0.2	0.3	0.81	0.80	0.79	0.82	0.81	0.82
	Fish_mgrs_TBLval	5	0.3	0.5	0.2	0.3	0.78	0.74	0.72	0.78	0.78	0.78
	Fish_ind_TBLval	6	0.45	0.4	0.15	0.7	0.63	0.57	0.55	0.61	0.62	0.63
	Rec_fish_TBLval	7	0.15	0.5	0.35	0.7	0.81	0.83	0.83	0.83	0.82	0.83
	Conserv_TBLval	8	0.1	0.7	0.2	0.7	0.86	0.89	0.89	0.88	0.87	0.89
	Local_com_TBLval	9	0.2	0.6	0.2	0.7	0.79	0.80	0.79	0.80	0.80	0.80
	Fish_mgrs_TBLval	10	0.3	0.5	0.2	0.7	0.73	0.70	0.69	0.72	0.73	0.73

COLUMNS: **TBL value by stakeholder group**

ROWS: **Stakeholder group weightings defining the strategy**

Criteria: closest to optimum TBL value across all stakeholder groups; robustness

relative difference

	#	roRev	roEnv	roSoc	%art	Fish_ind_strat	Rec_fish_strat	Conserv_strat	Local_com_strat	Fish_mgrs_strat	TBLval_ref	average ratio closest to 1/0	row for which minimum value is largest/smallest
Fish_ind_TBLval	1	0.45	0.4	0.15	0.3	1.00	0.87	0.83	0.98	1.00	1.00	0.933926125	**0.825820545**
Rec_fish_TBLval	2	0.15	0.5	0.35	0.3	0.97	1.00	1.00	0.99	0.97	1.00	**0.991838035**	0.972772994
Conserv_TBLval	3	0.1	0.7	0.2	0.3	0.98	1.00	1.00	0.99	0.98	1.00	0.992348559	**0.975190761**
Local_com_TBLval	4	0.2	0.6	0.2	0.3	1.00	0.98	0.97	1.00	1.00	1.00	0.990391678	0.972264858
Fish_mgrs_TBLval	5	0.3	0.5	0.2	0.3	1.00	0.94	0.92	0.99	1.00	1.00	**0.971548329**	0.92096884
average ratio closest to 1/0						**0.98898**	0.95887	**0.94359**	**0.98862**	**0.98898**	1.00000		
row for which minimum value is largest/smallest						0.97277	0.86857	**0.82582**	**0.97524**	0.97277	1.00000		
Fish_ind_TBLval	6	0.45	0.4	0.15	0.7	1.00	0.90	0.87	0.97	0.99	1.00	**0.945346486**	**0.867891502**
Rec_fish_TBLval	7	0.15	0.5	0.35	0.7	0.98	1.00	1.00	1.00	0.99	1.00	**0.996040055**	0.985714228
Conserv_TBLval	8	0.1	0.7	0.2	0.7	0.97	1.00	1.00	0.99	0.98	1.00	0.993651781	0.979482989
Local_com_TBLval	9	0.2	0.6	0.2	0.7	0.99	0.99	0.99	1.00	1.00	1.00	0.995179646	**0.986349292**
Fish_mgrs_TBLval	10	0.3	0.5	0.2	0.7	1.00	0.97	0.95	0.99	1.00	1.00	0.982152483	0.95115637
average ratio closest to 1/0						0.9869	0.9716	**0.9609**	0.9890	**0.9909**	1.0000		
row for which minimum value is largest/smallest						0.9695	0.9005	**0.8679**	0.9663	**0.9795**	1.0000		

absolute difference

	#	roRev	roEnv	roSoc	%art	Fish_ind_strat	Rec_fish_strat	Conserv_strat	Local_com_strat	Fish_mgrs_strat	TBLval_ref	average diff closest to 0/largest magnitude	row for which minimum value is smallest/largest
Fish_ind_TBLval	1	0.45	0.4	0.15	0.3	0.00	-0.10	-0.13	-0.02	0.00	0.00	**-0.049632723**	**-0.130838408**
Rec_fish_TBLval	2	0.15	0.5	0.35	0.3	-0.02	0.00	0.00	-0.01	-0.02	0.00	**-0.006490795**	-0.021652251
Conserv_TBLval	3	0.1	0.7	0.2	0.3	-0.02	0.00	0.00	-0.01	-0.02	0.00	-0.006633371	-0.021508222
Local_com_TBLval	4	0.2	0.6	0.2	0.3	0.00	-0.01	-0.02	0.00	0.00	0.00	-0.007837645	**-0.022623951**
Fish_mgrs_TBLval	5	0.3	0.5	0.2	0.3	0.00	-0.04	-0.06	-0.01	0.00	0.00	-0.022219519	-0.061719902
average ratio closest to 0/largest magnitude						-0.0091	-0.0315	**-0.0432**	**-0.0090**	-0.0091	0.0000		
row for which minimum value is smallest/largest						-0.0217	-0.0987	**-0.1308**	**-0.0186**	-0.0217	0.0000		
Fish_ind_TBLval	6	0.45	0.4	0.15	0.7	0.00	-0.06	-0.08	-0.02	0.00	0.00	**-0.034324159**	**-0.082968371**
Rec_fish_TBLval	7	0.15	0.5	0.35	0.7	-0.02	0.00	0.00	0.00	-0.01	0.00	**-0.003301976**	-0.011912106
Conserv_TBLval	8	0.1	0.7	0.2	0.7	-0.03	0.00	0.00	-0.01	-0.02	0.00	-0.005651479	-0.018265197
Local_com_TBLval	9	0.2	0.6	0.2	0.7	-0.01	-0.01	-0.01	0.00	0.00	0.00	-0.003865276	**-0.010946031**
Fish_mgrs_TBLval	10	0.3	0.5	0.2	0.7	0.00	-0.02	-0.04	0.00	0.00	0.00	-0.012977052	-0.035514537
average ratio closest to 0/largest magnitude						-0.0112	-0.0189	**-0.0260**	-0.0078	**-0.0075**	0.0000		
row for which minimum value is smallest/largest						-0.0271	-0.0625	**-0.0830**	**-0.0212**	**-0.0183**	0.0000		

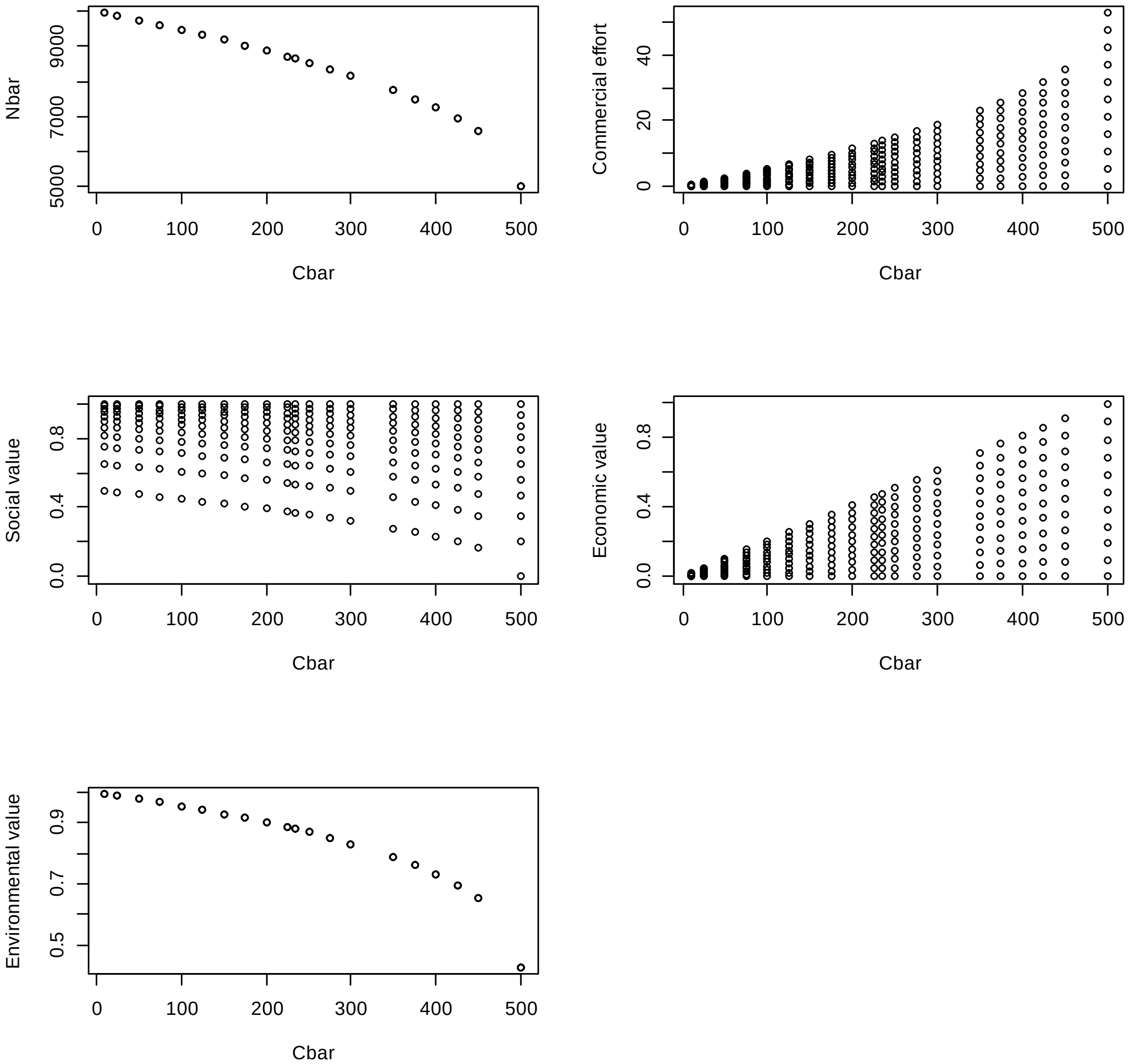


Figure C1 Scatterplots of: i) steady state abundance (Nbar), ii) commercial effort, iii) social value, iv) economic value, and v) environmental value, versus steady state catch. The three panels with cascading points correspond to the 11 different levels of artisanal effort considered.

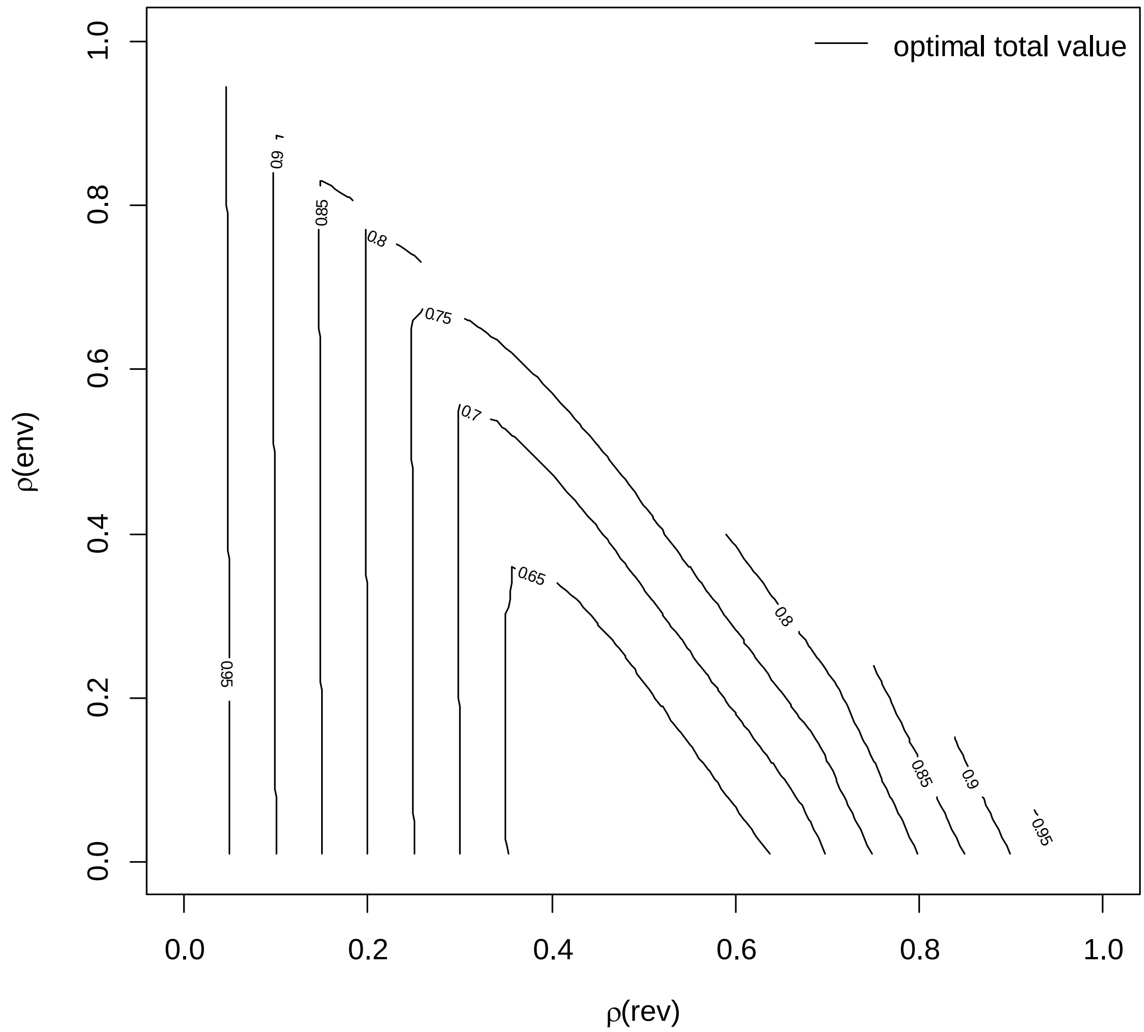


Figure C2: Contour plot of optimal values across the weighting surface, defined by the environmental (roEnv) and economic (roRev) weightings

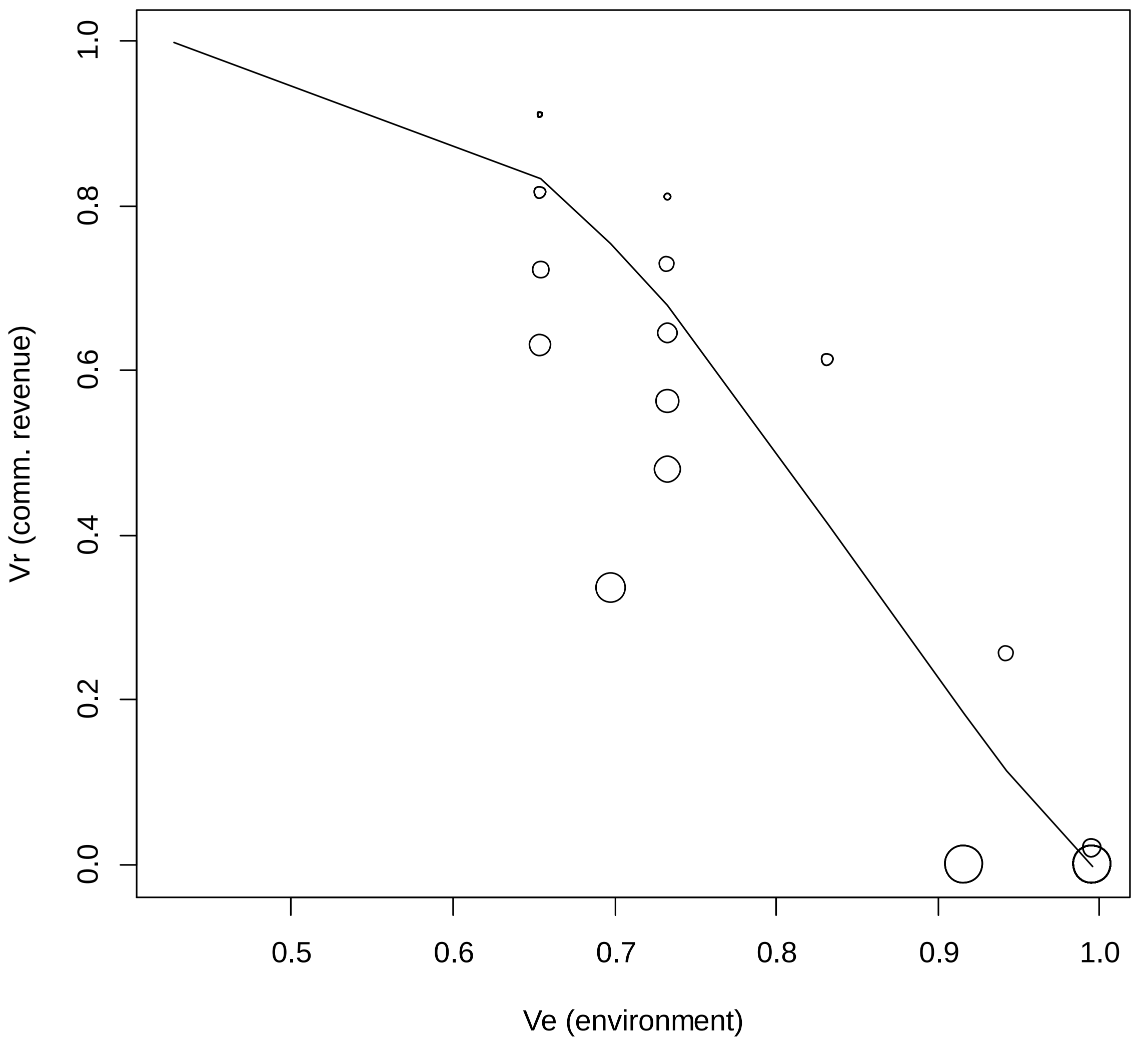


Figure C3: Profile of values corresponding to the overall maximised value function, across the range of 11 fixed weightings considered. The economic value is on the y-axis, environmental value on the x-axis, and the size of circles corresponds to the social value. A Loess smoother line has been added; this is analogous to a preference curve.

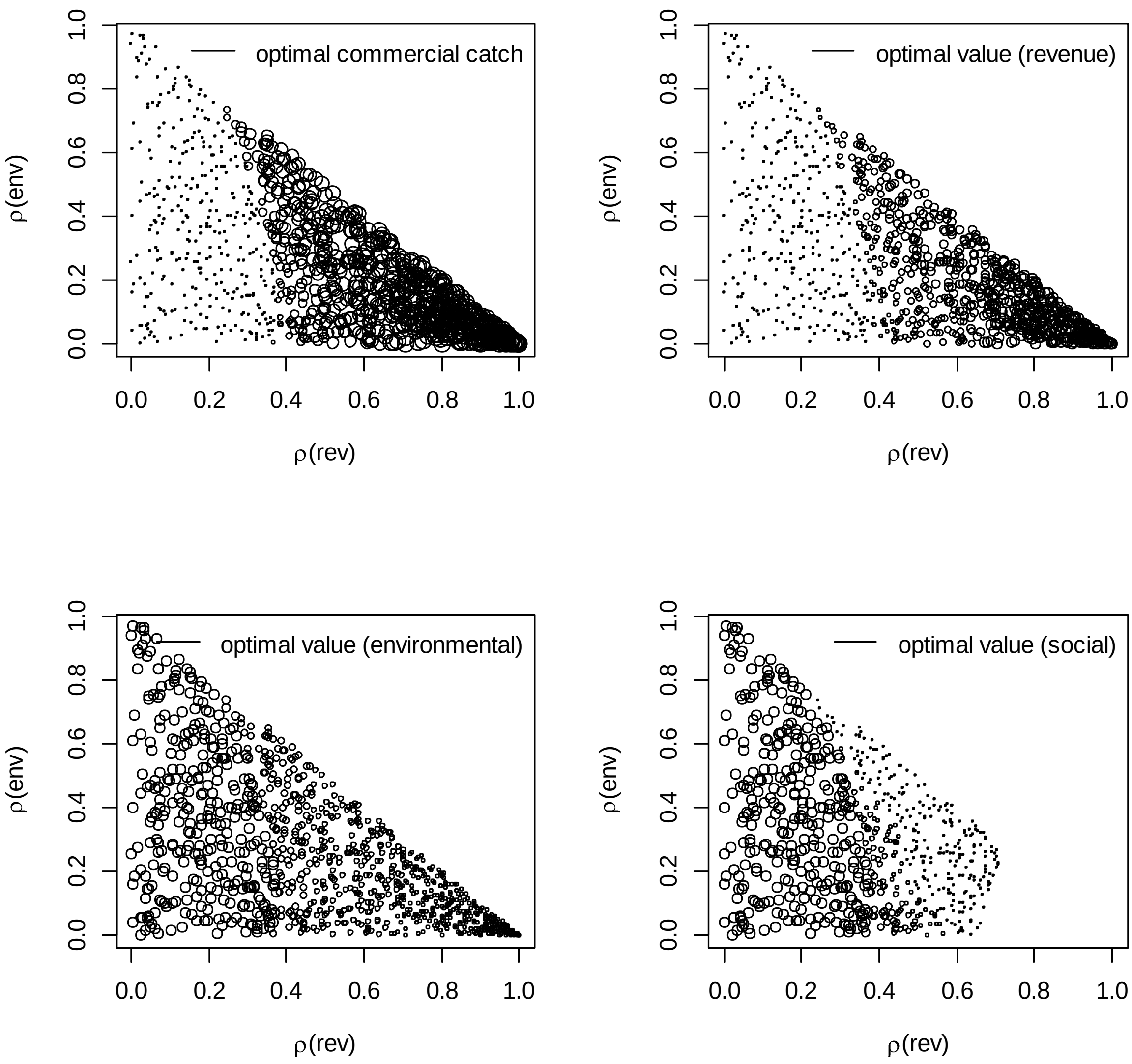


Figure C4: Plot of optimal i) commercial catch, ii) economic value, iii) environmental value, and iv) social value (indicated by size of circles) as a function of the environmental weighting (ρ(env), y-axis), and the economic weighting (ρ (rev), x-axis), for 1000 randomly selected combinations of economic and environmental weightings (recalling that the social weighting is simply 1-([sum of economic and environmental weightings]).

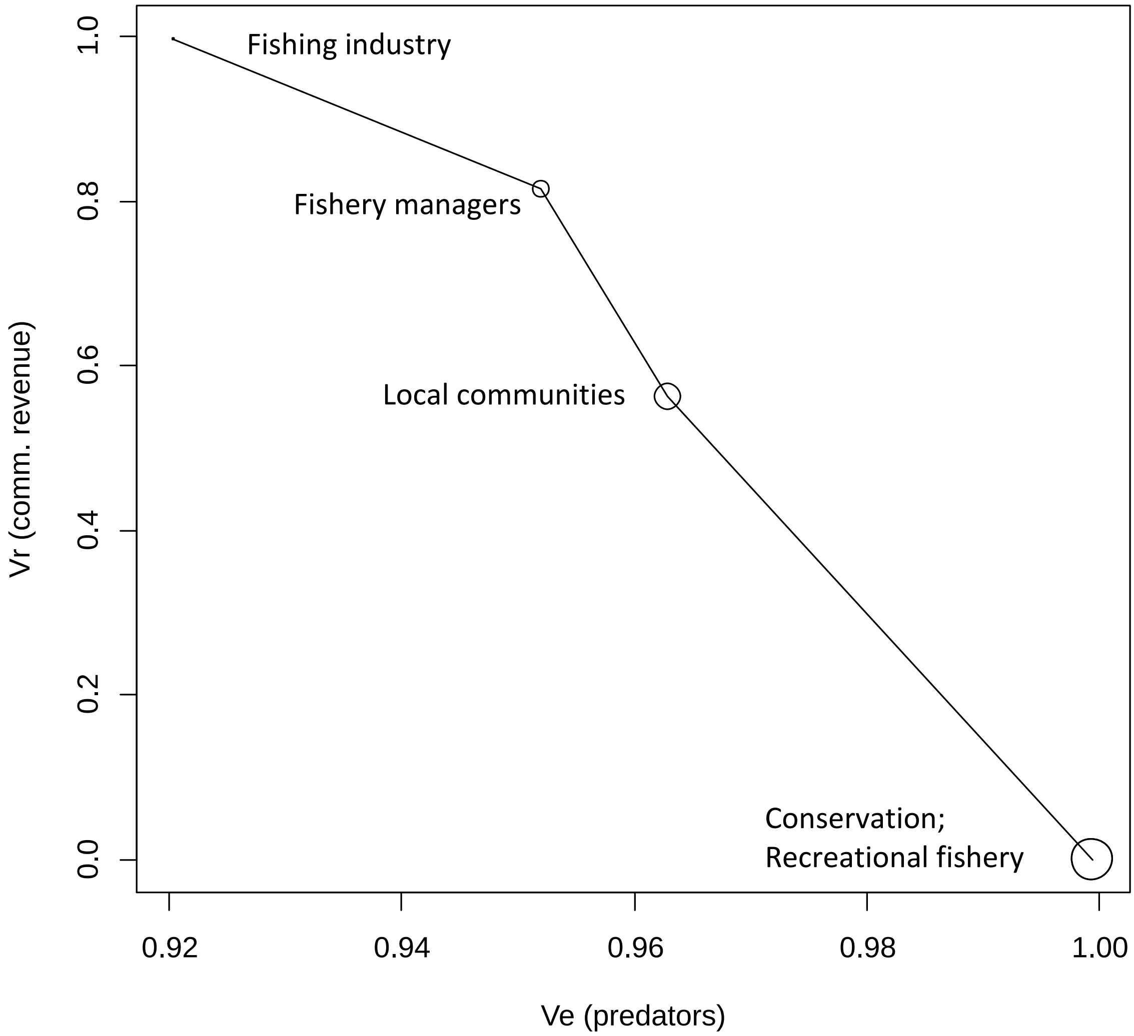


Figure C5: Profile of values corresponding to the overall maximised value function, for 5 sets of simple stakeholder group weightings based on the Queensland East Coast Trawl Fishery example of Pascoe et al. (2013). The economic value is on the y-axis, environmental value on the x-axis, and the size of circles corresponds to the social value. Although there were 5 stakeholder groups, there are only 4 points, since there are 2 shared optimal solutions. The line linking the points could be construed as defining emerging preferences.

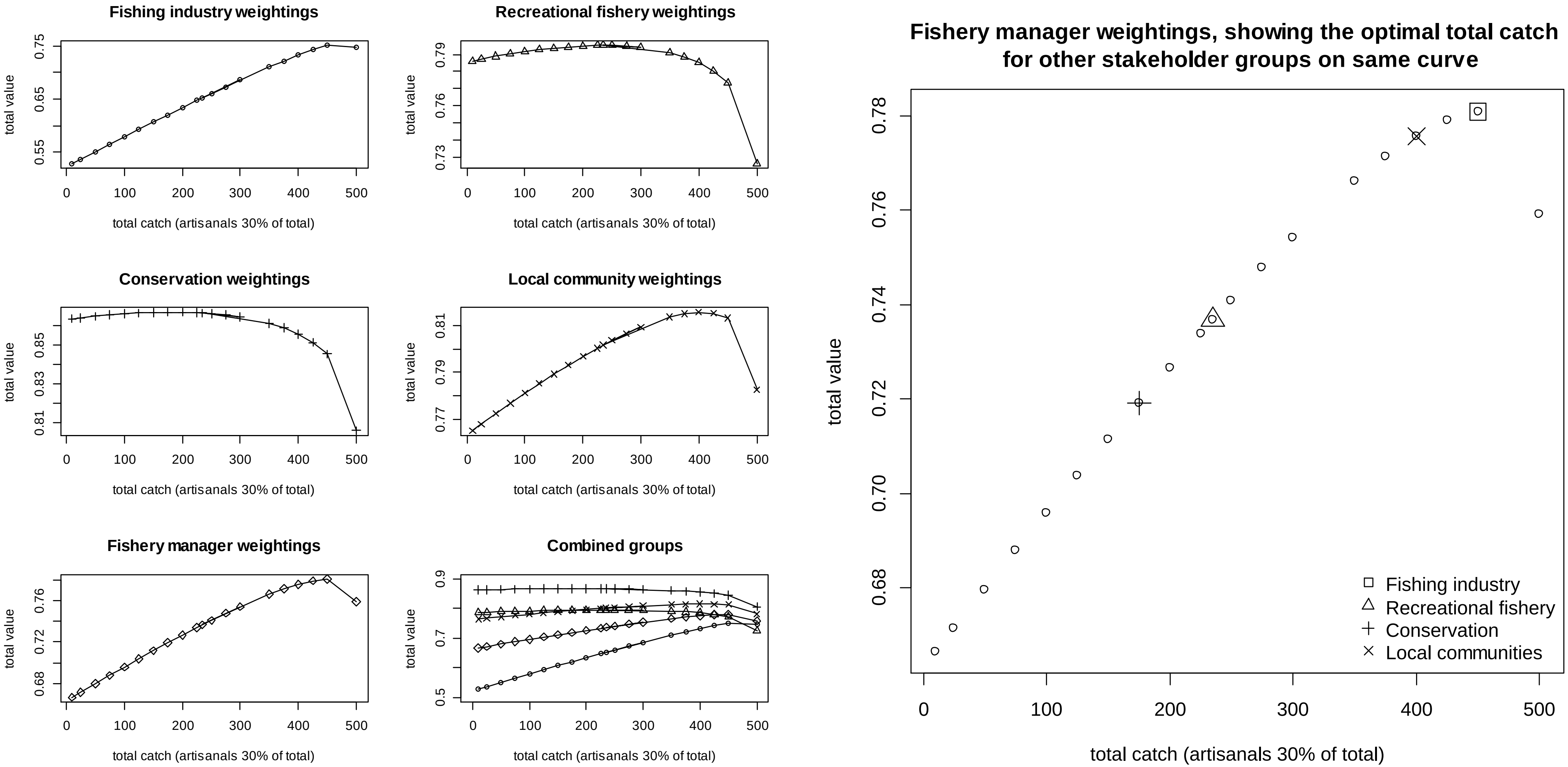


Figure C6a: Total value for varying levels of total catch (assuming artisanal effort takes ~30%), for the 4 sets of simple stakeholder group weightings based on Pascoe et al. (2013). The right panel shows the profile for the fishery manager weightings, with the optimal total catch for the other stakeholder groups shown on the same curve

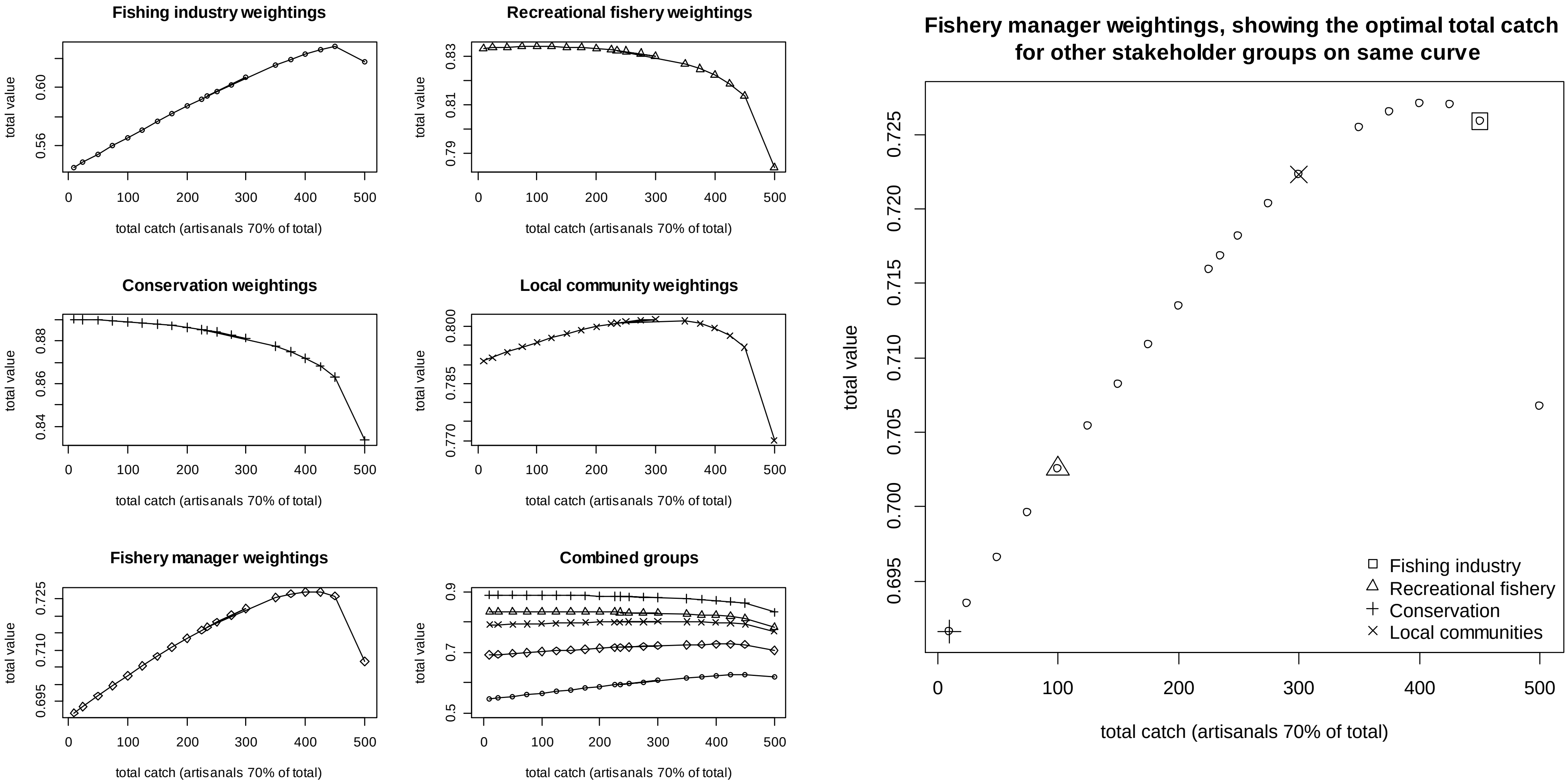


Figure C6b: Total value for varying levels of total catch (assuming artisanal effort takes ~70%), for the 5 sets of simple stakeholder group weightings based on Pascoe et al. (2013). The right panel shows the profile for the fishery manager weightings, with the optimal total catch for the other stakeholder groups shown on the same curve.

Appendix D: Simulation and Performance Indicator Specifications

We simulate the 3 main species groups in the Coral Sea Finfish Fishery: Coral Trout (CT), Red Throat Emperor (RTE), and the “other species” collective (OS).

We do not fit the model to data and assume perfect knowledge of stock sizes, environmental parameters, and fishing mortality. That is, there is no stock assessment or sampling model estimating underlying biomass. We also assume that the set 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).

1 Historical: Setting up equilibrium structure

We determined the unfished age structure assuming equilibrium dynamics, with natural mortality acting alone upon constant average unfished levels of recruitment:

$$V_{a,s} = \begin{cases} 1 & a = 0 \\ V_{a-1,s} \cdot e^{-M_{a,s}} & a < a_{max_s} \\ V_{a-1,s} \cdot e^{-M_{a,s}}/(1 - e^{-M_{a,s}}) & a = a_{max_s} \end{cases} \quad (1)$$

where

$V_{a,s}$ is the proportion of the population at age *a* of species *s*

a_{max} is the maximum age modelled (the plus-group) of species *s*

M is the age-specific instantaneous rate of natural mortality of species *s*

The spawner biomass per recruit, *SBR*, used in the stock-recruitment function, is

$$SBR_s = \sum_{a=1}^{a_{max,s}} V_{a,s} \cdot p_{a,s} \cdot m_{a,s} \quad (2)$$

where

$m_{a,s}$ is the average individual mass for fish of age *a* of species *s*

$p_{a,s}$ is the proportion of mature fish of age *a* of species *s*

a_{max} is the maximum age for species *s*

Mass (in kg) is calculated from length according to the power relationship

$$m_{a,s} = 0.001 \cdot i_s \cdot L_{a,s}{}^{j_s}$$

where species-specific length-at-age $L_{a,s}$, with parameters i_s and j_s, is calculated from the von Bertalanffy growth equation and is assumed to be deterministic:

$$L_{a,s} = L_{\infty s} \cdot \left(1 - e^{-k_s(a - t0_s)}\right)$$

The initial numbers-at-age in each region are:

$$N_{a,s,A,1} = T_s \cdot V_{a,s} \cdot Frac_{s,A} \quad (3)$$

where

$N_{a,s,A,1}$ is the number of fish of species *s* of age *a* in region *A* in year 1

T_s is the initial seeding number for fish of species *s*.

$Frac_{s,A}$is the proportion of species *s* expected in region *A*. We approximate this using the initial, region-specific biomass estimates for CT and RTE. We assume the OS are equally distributed spatially.

2 Historical: Population dynamics

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 thus

$$N_{a,s,A,y(mid-year)} = N_{a,s,A,y-1} \cdot e^{-M_{a,s}/2} \quad (4)$$

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

$$F_{f,s,A,y} = \frac{C_{obs\ F,s,A,y}}{\sum_{a=1}^{a_{max,s}} S_{a,F,s} \cdot N_{a,s,A,y} \cdot m_{a,s}} \quad (5)$$

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*.

We assume that selectivity for RTE is age-based, but for CT is length-based, which can be converted to selectivity-at-age using the length-age relationship.

We update abundance by applying the mortality due to catch, and finally the remainder of the natural mortality, to the interim (mid-year) numbers to obtain

$$N_{a,s,A,y} = N_{a,s,A,y(mid-year)} \cdot \left(1 - S_{a,f,s} \cdot \sum_f F_{f,s,A,y}\right) e^{-M_{a,s}/2} \quad (6)$$

We assume no migration between regions: CT show site-fidelity to the reefs on which they settle as larvae. Williams et al. (2010) hypothesised that RTE move more than CT, but such movement would still not be on the scale of our modelled regions. We make the same assumption for OS.

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

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

$$N_{a_{max},s,A,y+1} = N_{a_{max},s,A,y} + N_{a_{max}-1,s,A,y} \quad (7)$$

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

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

$$B_{s,y} = \sum_{a=1}^{a_{max,s}} m_{a,s} \cdot \sum_{i=1}^{Narea} N_{a,s,i,y} \quad (8)$$

For each species, we assume annual recruitment, R_y follows a Beverton-Holt stock-recruitment relationship with process uncertainty $E_{y,s}$

$$R_{y,s} = \frac{B_{sp\ s,y}}{\alpha_s + \beta_s \cdot B_{sp\ s,y}} \cdot e^{E_{y,s}} \quad (9)$$

where h_s is the steepness for species s and

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

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

(10)

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

We then distributed recruits in space according to

$$N_{0,s,A,y} = Frac_{s,A} \cdot R_{y,s} \quad (11)$$

3 Calculation of catchability

We assume a total allowable catch (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.

In principle, it is possible to calculate targeted and bycatch catchabilities (Somers and Wang 1997), at least for the commercial sector, where there are dedicated fishers for each of CT, RTE, and OS species groups. However, i) targeting behavior is not recorded with frequency or consistency within the commercial fleet, ii) targeting behavior has changed over time in the commercial fleet, with formerly “dead boats” re-gearing as live Coral Trout vessels, without this change being explicitly reported, and iii) there is no recreational effort time series. Thus, we estimate catchability for each species and fleet (sector) assuming that any day of effort on which one of the three species groups was reported in the catch, would contribute to the catchability of that species group.

For the three species groups, we define the overall catchability on species *i*, q_i, following Mapstone et al.'s (2008) equation 18a. We use historical data of targeted catch and effort, and historically modelled biomass

$$q_{i,f} = \exp\left(\frac{\sum_y \ln\left(C_{f,i,y}/B_{i,y}\right)}{\sum_y \ln\left(E_{f,i,y}\right)}\right) \qquad (12)$$

$$q_{i,f} = \exp\left(\frac{\sum_y \ln\left(C_{f,i,y}/B_{i,y}\right)}{\sum_y \ln\left(E_{f,i,y}\right)}\right) \qquad (12)$$

4 Projections: The harvest strategy

The harvest strategy is a system of Total Allowable Catches (*TAC*s), 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 allocating *TAC* allocated between the commercial and charter sectors only, we assume 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, we assumed they took a fixed amount for each species group, based on the averages over the final three historical years.

In each year, the *TAC*s 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.

5 Projections: Fleet dynamics

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

Otherwise, we distribute the fishing mortality per equation 23 of Little et al. (2007), for the commercial and charter sectors (*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

$PropF_{f,s,A,y}$ is the proportion of fishing mortality for fleet *f* on species *s* in region *A* and year *y*

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

If species *s* in region *A* was not fished by fleet *f* in the previous year

$$PropF_{f,s,A,y} = \left[\frac{\sum_{y'=1}^{y-1} C_{f,s,A,y'}}{yrsfished_{f,s,A}}\right] \Big/ \sum_A \left[\frac{\sum_{y'=1}^{y-1} C_{f,s,A,y'}}{yrsfished_{f,s,A}}\right] \qquad (13a)$$

We assume the recreational fishing effort is distributed equally between the two regions

$$PropF_{rec,s,A,y} = \frac{1}{Narea} \qquad (13b)$$

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.

6 Projections: Fishing mortality

As above, we assume perfect knowledge and that the species-specific *TAC*s are achieved each year. Species, and when appropriate region-specific, TACs will be achieved both via targeted and non-targeted fishing.

Fishing mortality by species, *s*, (and region, *A*) is determined by dividing the fleet-specific TAC by the biomass, as per equation (5). That is, when the TAC is specified globally, as $TACglob_{f,s,y}$, the fishing mortality is

$$F_{f,s,A,y} = \frac{TACglob_{f,s,y} \cdot PropF_{f,s,A,y}}{\sum_{a=1}^{a_{max,s}} m_{a,s} \cdot S_{a,f,s} \cdot N_{a,s,A,y}} \qquad (14a)$$

When the TAC is spatially explicit, the fishing mortality is

$$F_{f,s,A,y} = \frac{TAC_{f,s,A,y}}{\sum_{a=1}^{a_{max,s}} m_{a,s} \cdot S_{a,f,s} \cdot N_{a,s,A,y}} \qquad (14b)$$

We obtain the effort associated with the given *TAC* (and the catch by targeting practice) using catchability

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

7 Projections: Population dynamics

As with the historical period, we assume that in any year, *y*, fish undergo half of natural mortality prior to being fished, are fished, and then experience the remaining natural mortality. Mid-year abundance is calculated using equation (4), as for the historical period.

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

$$C_{f,s,A,y} = F_{f,s,A,y} \cdot \sum_{a=a_{legal,s}}^{a_{max,s}} \cdot S_{a,f,s} \cdot N_{a,s,A,y} \qquad (15)$$

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;* and

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

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

$$UC_{f,s,A,y} = F_{f,s,A,y} \cdot \sum_{a=1}^{a_{MLL,s}} m_{a,s} \cdot S_{a,f,s} \cdot N_{a,s,A,y} \qquad (16)$$

is discarded.

Catch in mass is obtained by multiplying equation (16) by the species-specific mass-at-age, $m_{a,s}$.

We update abundance by applying the mortality due to catch, and finally the remainder of the natural mortality, to the interim (mid-year) numbers: to

$$N_{a,s,A,y} = N_{a,s,A,y(mid-year)} \cdot \left(1 - \sum_f PropF_{f,s,A,y} \cdot F_{f,s,y} \cdot S_{a,f,s}\right) e^{-M_{a,s}/2} \qquad (17)$$

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

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

$$N_{a+1,s,A,y+1} = N_{a,s,A,y} \qquad (18)$$

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

$$B_{sp\ s,y+1} = \sum_{a=1}^{a_{max,s}} \sum_{i=1}^{Narea} p_{a,s} \cdot m_{a,s} \cdot N_{a,s,i,y+1} \quad (19)$$

As above, we determine recruitment, R_y using a Beverton-Holt stock-recruitment relationship, and recruits are distributed among the regions, as per the historical period (equations (9)-(11)). Here we set the process stochasticity in the Beverton- Holt stock recruitment relationship to 0.

8 Projections: Performance indicators

In each projection year, we calculate the performance indicators (PIs). Each PI corresponds to a single TBL or governance objective, as elicited from stakeholders (Pascoe et al., 2019).

In principal, we seek the maximum value for each PI in each year, $PI_{i,yr}$.

1.1.1 Maintain target species (CT and RTE) biomass at optimal sustainable levels.

This PI applies only to Coral Trout and Red Throat Emperor.

We use a truncated dome-shape for this PI (Figure D1.8.1). We assume that the target reference point ranges from 40%-60% of the unfished biomass, although this may be higher from a conservation standpoint. The broad target (plateau for the dome) encompasses the range from biomass at maximum sustainable yield (traditionally assumed to be 0.4B_0) and biomass at maximum economic yield (traditionally assumed to be 0.48B_0), as well as the Queensland specified target of 0.6 B_0. Having this broad target allows for some flexibility when trading off with the economic objectives.

In the dome specification, if the relative biomass is within 10% of the target range, the score of the PI for that species *s* is 1:

$$Score_{s,y} = 1; 0.36 \le \frac{B_{s,y}}{B_{s,0}} \le 0.66$$

Below the limit of 20% of the unfished biomass, the score of the PI 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:

$$Score_{s,y} = \frac{1}{(0.4 - 0.2)} \cdot \frac{B_{s,y}}{B_{s,0}} + \left(1 - \frac{0.4}{(0.4 - 0.2)}\right); 0.2 \le \frac{B_{s,y}}{B_{s,0}} \le< 0.36$$

.

Above the upper target value + 10%, the score decreases linearly from the Target Reference Point TRP) to unfished biomass, down to a minimum of (currently) (set as variable) 0.5 (i.e. we're half as happy as at the target level):

$$Score_{s,y} = \frac{(0.5 - 1.0)}{(1.0 - 0.6)} \cdot \frac{B_{s,y}}{B_{s,0}} + \left(0.5 - \frac{(0.5 - 1.0)}{(1.0 - 0.6)}\right); \frac{B_{s,y}}{B_{s,0}} > 0.66$$

If the relative biomass of any one species is below its limit reference point, then the overall PI is zero. Otherwise, for each of the alternative specifications, we obtain the overall PI is taken by averaging across species so that

$$PI_{1,y} = \frac{\sum_{s=1}^{2} Score_{s,y}}{2}$$

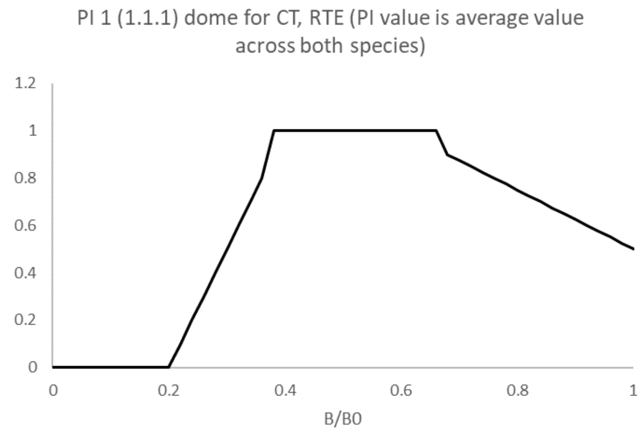


Figure D1.8.1: Functional form of performance indicator 1.1.1

1.1.2 Risk to Other Species (that are harvested, per the "Other Species" list) in the fishery which are not included in 1.1.1

The TRP is 0.4 of the unfished “other species” biomass, as a proxy for MSY, and the limit is 0.2 of the unfished biomass.

The PI follows a hockey-stick rule (Figure D1.8.2), where the PI is 1 above a biomass of 0.4 B_0, 0 below a biomass of 0.2 B_0, and tracks linearly with relative biomass between these values:

$$PI_{2,y} = \frac{1}{(0.4-0.2)} \cdot \frac{B_y}{B_0} + \left(1 - \frac{0.4}{(0.4-0.2)}\right); 0.2 \le \frac{B_y}{B_0} \le 0.4$$

We chose the hockey stick for this reason. When both performance indicators are dome-shaped, these may be in contradiction if one group of species is above its target (and being pulled back), and the other is below its target (and being pulled up).

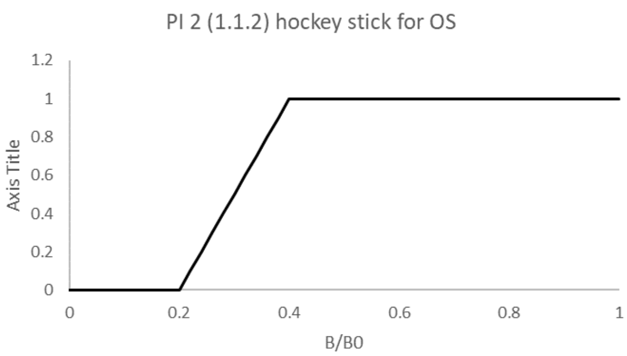


Figure D1.8.2: Functional form of performance indicator 1.1.2

From a conservation standpoint, a target of 0.6 B_0 and a limit of 0.3 B_0 may be more aligned with this objective.

1.2.1 Risk to bycatch species

This refers to generic bycatch, as opposed to specific species. It does not include undersize discarding, or high grading, since these are covered in separate PIs below. However, almost all catch is sold in the fishery and the gears are relatively clean, so that bycatch is not a critical issue in this fishery.

We assume that this PI is a linear function of effort, normalised to 1.5 the maximum historical effort (this does efficient fishers a disservice). A weighting by region could be added, if certain regions are considered to induce more bycatch (Figure D1.8.3).

To determine the score associated with this PI, we calculate, for each target species, fleet and region, the effort relative to the historical high, setting the score equal to 1 if the effort is greater than 1.5 times the historical high. We then average to obtain a single value and subtract the mean value from 1.

$$ByCatRisk_{f,s,A,y} = \frac{E_{f,A,y}}{1.5 \cdot \max(E_{f,A,y=1:Histyr})}; \frac{E_{f,A,y}}{1.5 \cdot \max\,(E_{f,A,y=1:Histyr})} < 1$$

$$ByCatRisk_{f,s,A,y} = 1\ otherwise$$

$$PI_{3,y} = 1 - \frac{\sum_{f,s,A} ByCatRisk_{f,s,A,y}}{(Nfleet * Nspecies \cdot Narea)}; \frac{E_{f,A,y}}{1.5 \cdot \max\,(E_{fA,y=1:Histyr})} < 1$$

$$PI_{3,y} = 0\ otherwise$$

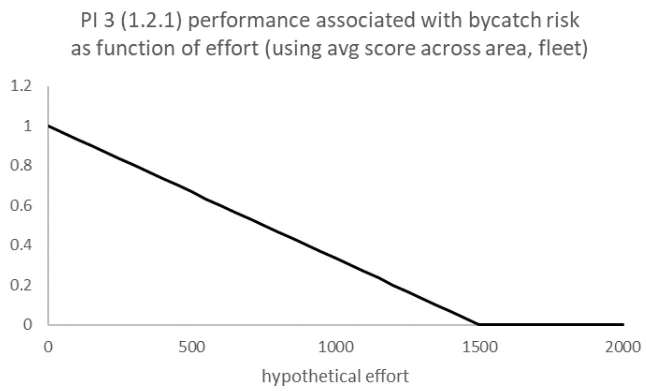


Figure D1.8.3: Functional form of performance indicator 1.2.1

1.2.2 Discard mortality (of undersized target species, or from high-grading of target species)

As described above, given a minimum legal length *MLL* , we compute the undersize catch, $UC_{f,s,A,y}$, from equation (16). We assume that the minimum legal length for each species group is length at maturity.

Given i) the fishery's history of not exceeding the Coral Trout TAC, ii) that the commercial fishery prefers plate-size fish, iii) the cost of fishing is high such that the fishery becomes uneconomic as catch rates decrease, and iv) that the recreational sector does not high grade, we assume no high grading. Furthermore, 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.

We calculate the total proportion of discards by fleet, species, region and year, $D_{f,s,A,y}$ by standardising the undersize catch relative to the total (legal and undersize) take

$$D_{f,s,A,y} = \frac{UC_{f,s,A,y}}{(F_{f,s,A,y} \cdot \sum_{a=1}^{maxage} m_{a,s} \cdot S_{a,f,s} \cdot N_{a,s,A,y})}$$

We then average over fleet, species and region to yield a mean overall discard, $meanD_y$.

To find the PI for discarding, we normalise according to the worst possible expected discard percentage (0.5, as above) (Figure D1.8.4)

$$PI_{4,y} = 1 - \frac{meanD_y}{0.5}$$

Values of $PI_{4,y} < 0$ are set to 0, and values of $PI_{4,y} > 1$ are set to 1.

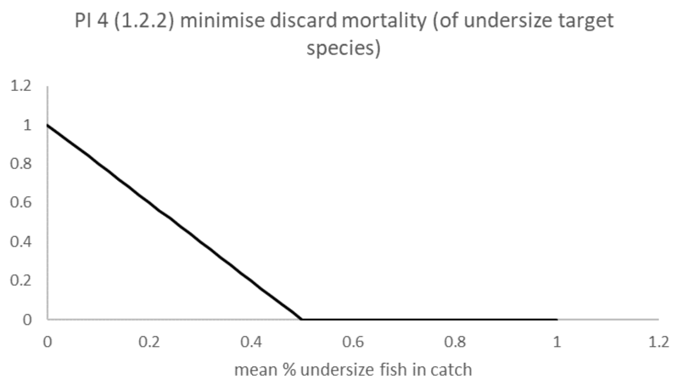


Figure D1.8.4: Functional form of performance indicator 1.2.2

1.2.3 Broader ecological risks; and 1.2.4 Risk to Threatened, Endangered, and Protected Species (TEPS)

We assume broader ecological risk ($PI_{5,y}$) is a function of effort. We set the PI to 1 when effort is 0, and let it linearly decrease to 0.8 between 0 and a target effort level. Between the target and limit effort, we let this PI value linearly decrease from 0.8 to 0; it is 0 when effort exceeds the limit. (Figure D1.8.5).

We set target effort to be half of the effort averaged over the last 5 years of the historical time series, and limit effort to be the historical high effort. Even though TEP interactions appear to be infrequent, there is the concern that these are not reported, so the historical high effort is probably an appropriate limit.

Effort is summed over the two regions and three fleets to obtain total effort for the year, $TotE_y$

$$TotE_y = \sum_A \sum_f E_{f,A,y}$$

We thus set target and limit effort levels as

$$TargetE_y = 0.5 \cdot \frac{\sum_{z=Histyr-4}^{z=Histyr} TotE_z}{5}$$

$$LimitE = 0.8 \cdot max\left(\sum_A \sum_f E_{f,A,y=1:Histyr}\right)$$

We determine the PI between the target and the limit effort, assuming linear decline

$$PI_{5,y} = \left(\frac{0.8(LimitE - TotE_y)}{(LimitE - TargetE)}\right); TargetE \leq TotE_y \leq\ LimitE$$

Below the target, we use another straight line:

$$PI_{5,y} = \left(\frac{(0.8 - 1.0) \cdot TotE_y}{(TargetE)} + 1\right); TotE_y < TargetE$$

$$if\ PI_{5,y} < 0,\ PI_{5,y} = 0$$

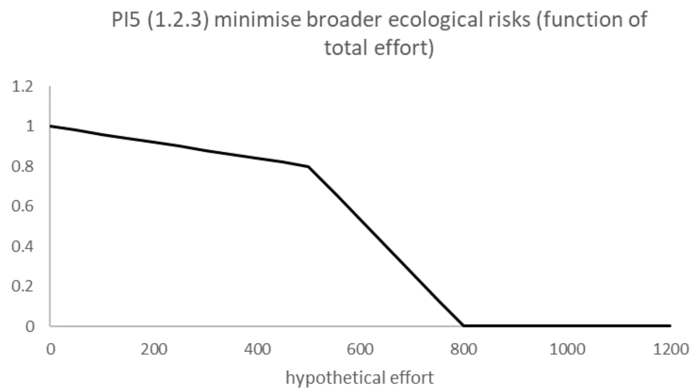


Figure D1.8.5: Functional form of performance indicator 1.2.3

We formulate the TEP risk ($PI_{6,y}$) in a similar manner, except that between the target and limit effort, the PI value is a weak inverse exponential function of effort (Figure D1.8.6).

For the TEP risk, we use an exponential function between the target and limit effort:

$$PI_{6,y} = \left(1.8 * \frac{1}{1.8}^{\left((1-TotE_y)-(1-TargetE_y)\right)/\left((1-LimitE)-(1-TargetE_y)\right)}\right) - 1$$

Below the target, the same straight-line equation as for the broader ecological risk applies:

$$PI_{6,y} = \left(\frac{(0.8-1.0) \cdot TotE_y}{(TargetE)} + 1\right); TotE_y < TargetE$$

$$if\ PI_{6,y} < 0,\ PI_{6,y} = 0$$

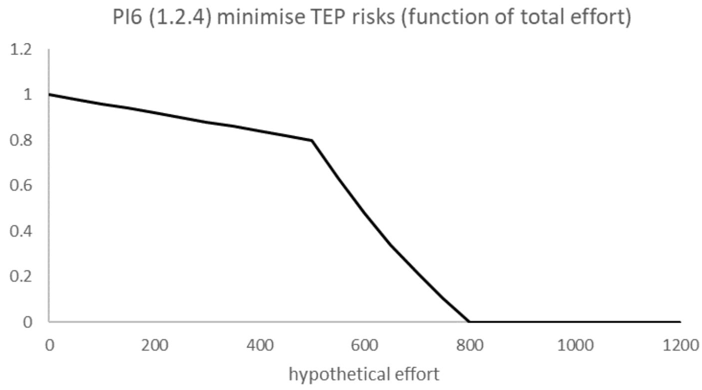


Figure D1.8.6: Functional form of performance indicator 1.2.4

1.3. Risk of localised depletion

We separate risks due to fishing and those due to environmental variation (cyclones and climate change).

1.3.1 Localised depletion due to fishing

We calculate this risk only for CT and RTE that is, for *s* = 1 and 2.

We compute biomass by region, relative to that region's unfished biomass

$$RelBio_{s,A,y} = TotBioM_{s,A,y} / TotBioM_{s,A,1}$$

and assume that the PI is 1 above a relative region-specific biomass of 0.5, 0 below a relative region-specific biomass of 0.2, and tracks linearly with relative biomass between these values (Figure D1.8.7).

$$Score_{s,A,y} = \frac{1}{(0.5-0.2)} \cdot RelBio_{s,A,y} + \left(1 - \frac{0.5}{(0.5-0.2)}\right); 0.2 \leq RelBio_{s,A,y} \leq 0.5$$

$$if\ RelBio_{s,A,y} < 0.2\,, \qquad PI_{7,y,1} = 0$$

$$if\ RelBio_{s,A,y} > 0.5\,, \qquad PI_{7,y,1} = 1$$

The PI is the minimum across the species and regions:

$$PI_{7,y} = minimum_{s,A}(Score_{s,A.y})$$

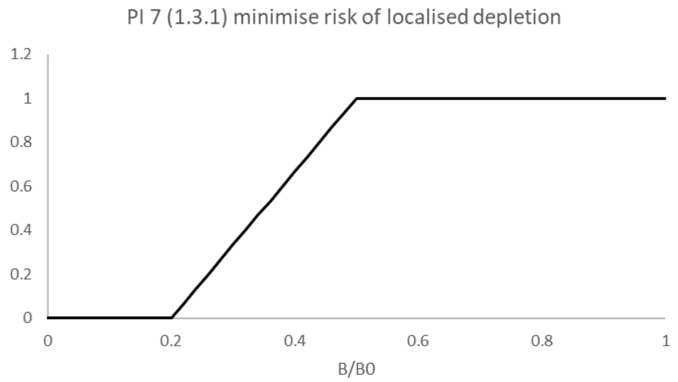


Figure D1.8.7: Functional form of performance indicator 1.3.1

1.3.2 Localised depletion due to environmental events (e.g. cyclone, climate change)

As described above, we treat cyclones and climate change as separate model scenarios (with accompanying relevant fleet dynamics, and the perceived positive and negative impacts on each species group).

However, the PI for localised depletion must reflect the need to be conservative and precautionary given that availability is reduced as a result of environmental perturbations. To do so, we apply a 20% penalty to the target 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.

That is, if the relative biomass is within 10% of the target range, the PI is 1:

$$PI_{8,y} = 1; 0.45 \leq \frac{B_y}{B_0} \leq 0.825$$

Below the penalised limit of (0.2/0.8=) 25% of the unfished biomass, the PI is 0. Between the lower end of the 10% tolerance around the lower penalised target value, and the limit of 0.25, the PI tracks linearly with relative biomass:

$$PI_{8,y} = \frac{1}{(0.5 - 0.25)} \cdot \frac{B_y}{B_0} + \left(1 - \frac{0.5}{(0.5 - 0.25)}\right); 0.25 \leq \frac{B_y}{B_0} < 0.45$$

.

Above the upper target value + 10%, the PI decreases linearly from TRP to the unfished biomass, down to a minimum of (currently) (set as variable) 0.5 (i.e. we're half as happy as at target):

$$PI_{8,y} = \frac{(0.5 - 1.0)}{(1.0 - 0.75)} \cdot \frac{B_y}{B_0} + \left(0.5 - \frac{(0.5 - 1.0)}{(1.0 - 0.75)}\right); \frac{B_y}{B_0} > 0.825$$

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.

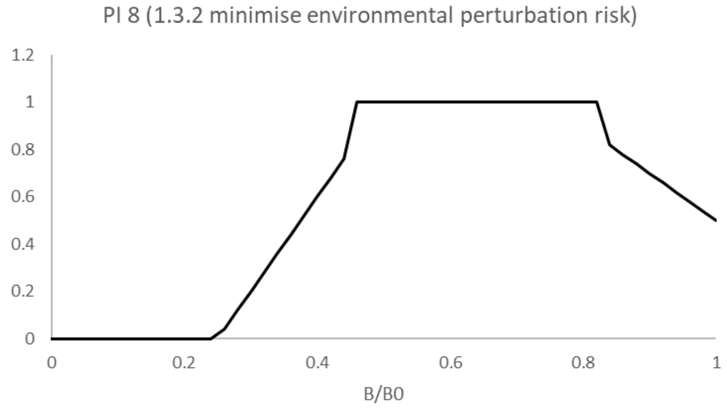


Figure D1.8.8: Functional form of performance indicator 1.3.2

2.1.1 Commercial fishing industry profits

We calculate this PI for the commercial sector, fleet 1, as price multiplied by catch, minus costs, which, for each region, *A*, and year, *y*,are

$$Cost_{1,A,y} = Fuelcost_{1,A} \cdot E_{1,A,y} + GearUnit_1 \cdot E_{1,A,y} + CatchUnit_1 \cdot C_{1,A,y}$$

where

$GearUnit_1$ is the cost of commercial gear associated with one day's commercial effort, set to 0.1

$CatchUnit_1$ is the cost associated with one unit of commercial catch, set to 0.1.

FuelCost is the cost associated with one unit of catch, set to 20 for the northern region, and 10 for the southern region.

Commercial profit is then

$$Profit_{1,y} = \sum_A (\sum_i (price_{i,y} \cdot CatM_{i,1,A,y}) - Cost_{1,A,y})$$

where the commercial fleet is indexed as fleet 1, and unit *Price* is 5 for CT, 2 for RTE and 1 for OS.

We compute the value of the PI (Figure D1.8.9) by taking the ratio of profit to that at MEY, approximated by taking the simulated historical high profit for the commercial sector (this corresponds to about $0.6B_0$ for the CT species group)

$$PI_{9,y} = \sum_i (Profit_{1,y} \ / \ ProfitMEY_1)$$

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

$$PI_{9,y} = -2 \cdot (Profit_{1,y} \ / \ ProfitMEY_1) + 3; 1.5 \leq (Profit_{1,y} \ / \ ProfitMEY_1) < 1.0$$

If the current profit exceeds 1.5 time the approximation for profit at MEY, the performance indicator is 0.

$$PI_{9,y} = 0; (Profit_{1,y} \ / \ ProfitMEY_1) > 1.5$$

In addition, if the biomass of any one species is less than the limit reference point of 0.2B0, the PI = 0

$$PI_{9,y} = 0; \frac{B_{s,y}}{B_{s,0}} < 0.2$$

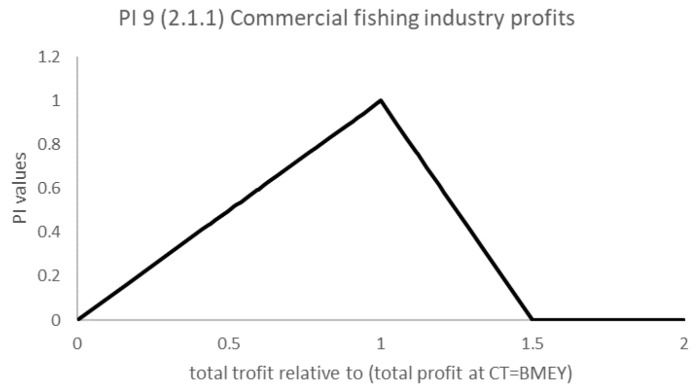


Figure D1.8.9: Functional form of performance indicator 2.1.1

2.1.2 Charter sector profits

We assume that gross profit for charter operators is the product of effort in days (as a proxy for the number of people fishing per day), multiplied by the charter price per day.

As with the commercial sector, costs are calculated for each region, *A*, and year, *y*,

$$Cost_{2,A,y} = Fuelcost_{2,A} \cdot E_{2,A,y} + GearUnit_2 \cdot E_{2,A,y} + CatchUnit_2 \cdot C_{2,A,y}$$

where

GearUnit is the cost of gear associated with one day's effort, here set to 0.1

CatchUnit is the cost associated with one unit of catch, here set to 0.05.

FuelCost is the fuel cost associated with one day's effort, here set to 10

Charter profit is then

$$Profit_{2,y} = \sum_A price_y \cdot E_{2,A,y} - Cost_{2,A,y}$$

where

$price_y$ is the price charged by charter operators for one day of effort.

As with the commercial sector profit (Figure D1.8.9), we compute the PI is by taking the ratio of profit to that at MEY, approximated by test simulations projecting forward so that the charter profit stabilised, noting that this corresponded approximately to 0.5B_0 for the CT group, and to 0.55 for the RTE group.

$$PI_{10,y,} = Profit_{2,y} \,/\, ProfitMEY_2$$

As with the commercial profit, 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

$$PI_{10,y} = -2 \cdot (Profit_{2,y} \,/\, ProfitMEY_2) + 3; 1.5 \leq (Profit_{2,y} \,/\, ProfitMEY_2) < 1.0$$

If the current profit exceeds 1.5 time the approximation for profit at MEY, the performance indicator is zero:

$$PI_{10,y} = 0; (Profit_{2,y} \,/\, ProfitMEY_2) > 1.5$$

In addition, if the biomass of any one species is less than the limit reference point of 0.2B0, PI = 0

$$PI_{10,y} = 0; \frac{B_{s,y}}{B_{s,0}} < 0.2$$

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 PI.

2.2. Value of recreational and charter fisher experience (direct to participant)

We assume the value of recreational fishing and charter experiences to the participants is a weighted function of catch, catch-per-unit-effort (*CPUE*), and effort. 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).

We assume the following weights on catch, *CPUE*, and effort, respectively:

Recwts = (0.4,0.3,0.3)

We assume the following weights on the catch of each species group (CT, RTE, OS), respectively:

RecCwts = (0.4,0.3,0.3)

We apply the species weightings to the catch by weight (as opposed to catch-by-numbers, since trophy fish are more highly valued):

$$WtRecC_{A,y} = \sum_i RecCwts_i \sum_{f=2}^{3} CatM_{i,f,A,y}$$

$$RecCPUE_{A,y} = \sum_i \sum_{f=2}^{3} C_{i,f,A,y} \Big/ \sum_{f=2}^{3} E_{f,A,y}$$

The recreational utility is then the weighted sums of recreational catch, CPUE, and effort, where each region's utility is, in turn, weighted according to the proportion of recreational effort in that region:

$$RecUtil_{A,y} = \frac{\sum_{f=2}^{3} E_{f,A,y}}{\sum_A \sum_{f=2}^{3} E_{f,A,y}} \left(Recwts_1 \cdot WtRecC_{A,y} + Recwts_2 \cdot RecCPUE_{A,y} + Recwts_3 \cdot \sum_{f=2}^{3} E_{f,A,y}\right)$$

We then average over all regions

$$AvgRecUtil_y = \sum_A RecUtil_{A,y} \Big/ Narea$$

and compute the PI by standardising this average by the maximum historical recreational utility:

$$PI_{12y} = \frac{AvgRecUtil_y}{HistMax(AvgRecUtil)}$$

where the denominator is the maximum historical *AvgRecUtil*.

$$if\ PI_{12,y} > 1\ ,\ PI_{12,y} = 1$$

2.3 Flow-on economic benefits to local communities

This is a function of the commercial and charter profits by region from PIs 2.1.1 and 2.1.2, and recreational effort by region.

We turn recreational effort by region into a dollar value (related to expenditure on fuel, bait, and accommodation) by applying a scalar.

The average benefit is calculated as:

$$AvgBenefit_y = \sum_{A=1}^{Narea} \left(\sum_{f=1}^{2} Profit_{f,y} + RecEff\_dollar\_scalar_A \cdot E_{3,A,y} \right) / Narea$$

where *RecEff_dollar_scalar* is the dollar value of one unit of recreational effort, by region, currently set to 10.0.

We obtain the PI by normalising relative to the historical maximum value:

$$PI_{13,y} = AvgBenefit_y / \max\left(AvgBenefit_{hist}\right)$$

$$if\ PI_{13,y} > 1\ ,\ PI_{13,y} = 1$$

2.4 Short term (inter-annual) economic risk

We approximate short-term risk as the interannual percent variability in profit, assessed by the coefficient of variation in profit (CV) 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 optimal and equates to a PI value of 1, and that +/- 25% is the limit below which the PI score value is 0 (Figure D1.8.10).

We calculate this PI for the commercial and charter fleets only:

$$CVprofit_f = \frac{stdev(\sum_i Profit_{i,f,yr-9:yr})}{mean(\sum_i Profit_{i,f,yr-9:yr})}$$

$$if\ CVprofit_f < 0.1, CVscore_f = 1.0$$

$$CVscore_f = \frac{-1}{(0.25-0.1)} \cdot CVprofit_f + \left(1 + \frac{0.1}{(0.25-0.1)}\right); 0.1 \le CVprofit_f \le 0.25$$

If the CV for any one fleet is below the Limit Reference Point, then whole score for this objective is zero:

$$if\ CVprofit_f > 0.25, PI_{14,y} = 0.0$$

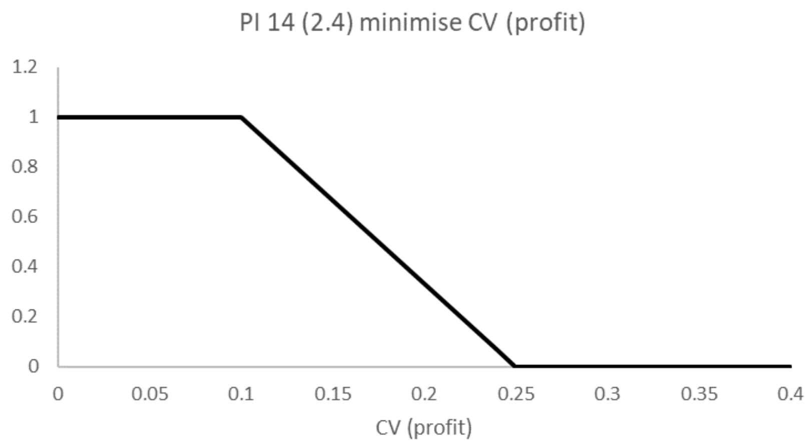


Figure D1.8.10: Functional form of performance indicator 2.4, for one fleet

Otherwise, the PI is the mean of the CV scores across the commercial and charter fleets

$$PI_{14,y,} = \sum_{f=1}^{2} CVscore_f / 2$$

2.5 Costs of management associated with the harvest strategy: monitoring, undertaking assessments, adjusting management controls

As a starting point, we assume that if the TAC for each species group exceeds 1.5 times 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 PI is the average of the species group scores:

$$Score_i = \begin{cases} 0; TAC_{i,y} > 1.5(\max(C_{i,hist})) \\ 1;\ TAC_{i,y} \leq 1.5(\max(C_{i,hist})) \end{cases}$$

$$PI_{15,y} = \frac{\sum_i Score_i}{Nspecies}$$

3.1 Willingness to comply with the harvest strategy

We assume that willingness to comply with the harvest inversely scales with the complexity of management; that is, the more management controls, the higher the lack of compliance.

Conditioned on the TAC by species *i* and region *A*, $TAC_{i,A}$, divided among each of the sectors (fleets), the maximum number of management controls is:

$$MaxMgmtControls\ = Narea \cdot Nspecies \cdot Nfleet$$

The actual number of management controls is

$$MgmtControls\ = Nsector \cdot TACarea \cdot Nspecies$$

Where *Nsector* is the number of sectors (fleets) receiving a TAC (as opposed to a static quota), and *TACarea* is the number of regions to which separate TACs apply.

The possibility of failure of the harvest strategy due to its complexity is:

$$ComplexFail = MgmtControls/MaxMgmtControls$$

We also assume 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 (Figure D1.8.11). We assume a target combined TAC of 4,500t and a standard deviation of 1000t:

$$normcoeff = 1/(StDevTAC \cdot \sqrt{(2 \cdot \pi)})$$

$$TargetRef = normcoeff \cdot e^{-0.5 \cdot (0.0/StDevTAC)^2}$$

$$DisagreeFail = \frac{normcoeff \cdot e^{-0.5 \cdot \left(\frac{\sum_{i,A}(TAC_{i,A}) - TargetTotTAC}{stDevTAC}\right)^2}}{TargetRef}$$

.

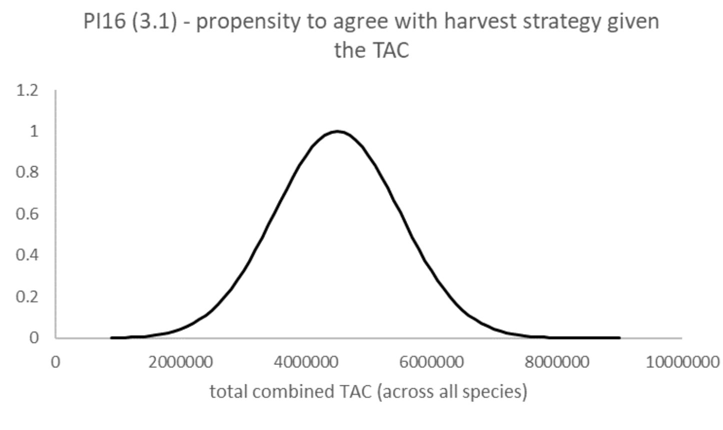


Figure D1.8.11: Functional form of the "disagree fail" component of performance indicator 3.1

We compute the PI value by adding each of these two weighted terms and subtracting from 1:

$$PI_{16,y} = 1.0 - (wt1 \cdot ComplexFail + wt2 \cdot DisagreeFail)/2$$

where *wt1* =0.4*, wt2* = 0.6 currently.

The first term on the right hand side pertains to inadvertent mistakes; the second term is an active disregard due to disagreeing with regulations

4.1 Equity between recreational, charter, indigenous and commercial fishing

For this PI, we consider equitable access to the resource and social/public perceptions of the fishery.

4.1.1 Equitable access to the resource

We approximate equitable access by the extent to which the end of year catch proportion by sector (fleet) *f* and species *i* conformed to the specified (fixed) allocation fraction $AllocFrac_{i,f}$:

$$TotCatM_{i,f} = \sum_{A} CatM_{i,f,A,y}$$

$$AllocDev_{i,f} = abs\left(\left(\frac{TotCatM_{i,f}}{\sum_f \sum_A CatM_{i,f,A,y}} - AllocFrac_{i,f}\right) / AllocFrac_{i,f}\right)$$

where *AllocFrac* is currently assumed to be 0.6, 0.2 and 0.2 for each of the commercial, charter and recreational fleets, respectively, for each of the three species groups.

The deviation from equitable access follows another hockey stick relationship (Figure D1.8.12):

$$Dev_{i,f} = \frac{1.0}{(AllocThresh - AllocTol)} \cdot AllocDev_{i,f} + \left(1 - \frac{AllocThresh}{(AllocThresh - AllocTol)}\right);\ AllocTol \le AllocDev_{i,f} \le AllocThres$$

$$Dev_{i,f} = \frac{1.0}{(AllocThresh - AllocTol)} \cdot (AllocDev_{i,f} - AllocTol);\ AllocTol \le AllocDev_{i,f} \le AllocThres$$

$$Dev_{i,f} = 1.0;\ AllocDev_{i,f} > AllocThres$$

$$Dev_{i,f} = 0;\ AllocDev_{i,f} < AllocTol$$

where

AllocThresh is the deviation threshold above which the fleets are dissatisfied, set at 20%

AllocTol is the deviation tolerance below which the fleets are satisfied, set at 2%

We determine the PI is by the average deviation across species groups and sectors:

$$PI_{17,yr} = 1 - \sum_i \sum_f Dev_{i,f} / (Nspecies \bullet Nfleet)$$

$if\ TotCatM_{i,f} = 0.0\ , \qquad PI_{17,yr} = 0$

Given that the TAC is divided according to these allocation fractions, and we assume perfect information, there should not be deviations, at least for the commercial sector.

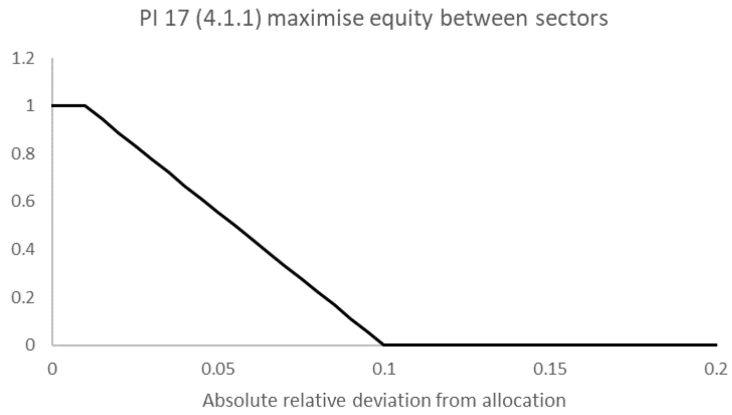


Figure D1.8.12: Functional form of performance indicator 4.1.1

4.2 Social perceptions of the fishery

In this case, we use PIs that capture public perception around environmental damage caused by the fishery.

4.2.1 Public perception around discard mortality (compliance with size limits, environmental sustainability, and waste)

We already have indicators of minimising the risk associated with discarding ($PI_{4,y,1}$), and Threatened, Endangered, and Protected Species (TEPS) ($PI_{6,y,1}$).

We recast these PIs so that the higher their value, the lower the risk:

$$Risk_{Discards} = 1 - PI_{4,y,1}$$

$$Risk_{TEP} = 1 - PI_{6,y,1}$$

For the TEPS risk, we assume that the perception is 0 when the risk is 0, and rises linearly with risk to be 0.2 when the risk is 10%:

$$Percept_{TEP} = 2 \cdot Risk_{TEP};\ \ Risk_{TEP} < 0.1,$$

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:

$$Percept_{TEP} = \frac{0.8}{(0.5 - 0.1)} \cdot Risk_{TEP} + \left(1 - \frac{0.8 \cdot 0.5}{(0.5 - 0.1)}\right); 0.1 \leq Risk_{TEP} \leq 0.5$$

$$if\ Risk_{TEP} > 0.5, Percept_{TEP} = 1.0$$

For the discarding risk, we assume a saturating relationship, where there is no concern below 50% risk, with a linear increase in perception (concern) above this.

$$if\ Risk_{Disc} < 0.5, Percept_{Disc} = 0.0$$

$$Percept_{Disc} = \frac{1}{(1.0 - 0.5)} \cdot Risk_{Disc} + \left(1 - \frac{1}{(1.0 - 0.5)}\right); 0.5 \leq Risk_{Disc} \leq 1.0$$

We then weight the two perceptions and subtract this from 1 to obtain the PI (Figure D1.8.13):

$$PI_{18,yr} = 1 - (0.7 \cdot Percept_{Disc} + 0.3 \cdot Percept_{TEP})$$

The stronger weighting on discarding is due to a greater public awareness of this relative to any awareness of the fishery interacting with TEPS.

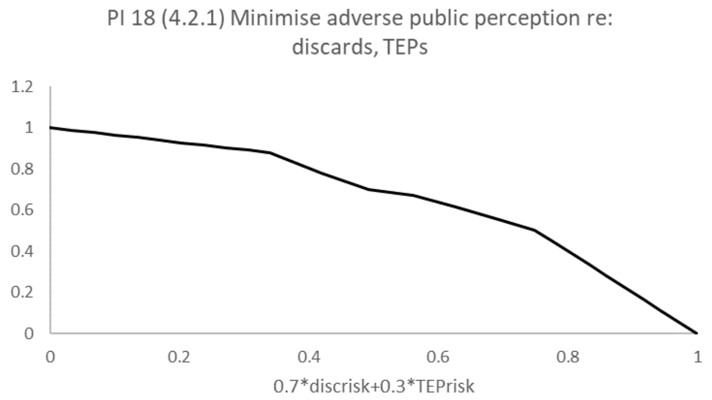


Figure D1.8.13: Functional form of performance indicator 4.2.1

4.2.2 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 (CT and RTE sustainability), 1.1.2 (OS sustainability), and 2.3 (flow-on economic benefits), and take a (non-weighted) average across them:

$PI_{19,yr}$=$\left(PI_{1,yr} + PI_{2,yr} + PI_{13,yr}\right)/3.0$

4.3 Net social value to the local community from use of the resource

These performance indicators include access to local seafood, and spatial (community) equity.

4.3.1 Access to local seafood (all species)

We assume that this PI is a function of the non-exported landings (= dead CT, plus all RTE and OS) that applies to the commercial and charter sector catches (that is, fleet f = 1 and 2).

We assume some fixed proportion of live to dead CT (currently, that 10% of CT catch is non-live) ($PercDeadCT = 0.1$).

We assume the PI is 0 if the local available domestic percentage is <20%, and 0.8 if the local available domestic percentage achieves a historical proportion, which we assume to be 0.5 ($PastDomProp = 0.5$).

We assume a hockey stick relationship for values between these two thresholds (Figure D1.8.14). If the local available percentage exceeds that from the past, the PI value increases linearly from 0.8 to 1.0 when the local available percentage is 100%.

The total domestic percentage catch is:

$$TotDomPerc = \left(PercDeadCT \cdot \sum_{f=1}^{2} \sum_{A} TotCatM_{1,f,A,y} + \sum_{f=1}^{2} \sum_{s=2}^{3} \sum_{A} TotCatM_{i,f,A,y} \right) \Bigg/ \sum_{f=1}^{3} \sum_{s=1}^{3} \sum_{A} TotCatM_{i,f,A,y}$$

where s=1 is CT, and s=2 and 3 are RTE and OS respectively.

If the total catch is zero, $\sum_{f=1}^{3} \sum_{i=1}^{3} \sum_{A} TotCatM_{i,f,A,y} = 0$, then $TotDomPerc = 0$.

The PI is then

$$if\ TotDomPerc < 0.2, PI_{20,y} = 0$$

$$PI_{20,y} = \frac{0.2}{(1.0 - PastDomProp)} \cdot TotDomPerc + \left(1.0 - \frac{0.2}{(1.0 - PastDomProp)}\right); TotDomPerc > PastDomProp$$

$$PI_{20,y} = \frac{0.8}{(PastDomProp - 0.2)} \cdot TotDomPerc + \left(0.8 - \frac{0.4}{(PastDomProp - 0.2)}\right);\ 0.2 \leq TotDomPerc \leq PastDomProp$$

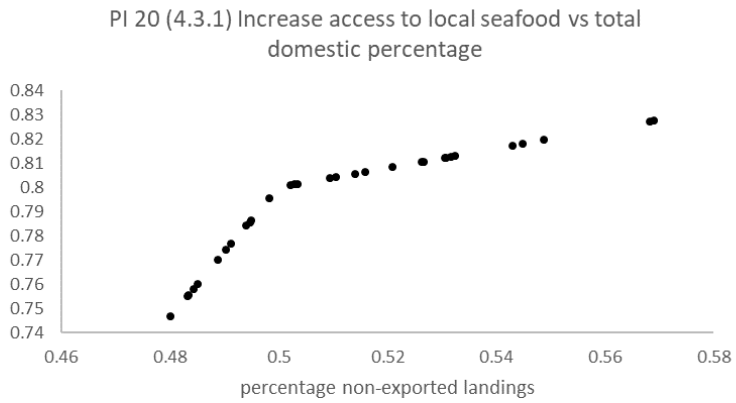


Figure D1.8.14: Functional form of performance indicator 4.3.1

4.3.2 Equity between regions and local communities

We assume the equitable proportions of catch (by weight) by region are those of the relative average biomass across species groups:

$SpatialFrac_A = \sum_1^{Nspecies} AreaFracN_{S,A} / Nspecies$

where $AreaFracN_{S,A}$ is the fixed proportion of species *s* in each region *A*.

The deviation threshold, above which the region is “unhappy”, is set at 20%:

$$SpAllocThres = 0.2$$

The deviation tolerance, below which the region is “happy”, is set at 5%:

$$SpAllocTol = 0.05$$

The absolute percent difference between the relative catch by region and the equitable proportion is calculated:

$$SpAllocDev_A = abs(TotCatArea_A / TotCat - SpatialFrac_A)$$

We assume a hockey stick relationship for values between the two thresholds (Figure D1.8.15).

$$SpDevPerf_A = 0;\ SpAllocDev_A < SpAllocTol$$

$$SpDevPerf_A = 1;\ SpAllocDev_A > SpAllocThres$$

$$PI_{21,y} = \frac{1}{(SpAllocThres - SpAllocTol)} \cdot SpAllocDev_A + \left(1.0 - \frac{SpAllocThres}{(SpAllocThres - SpAllocTol)}\right);\ SpAllocTol \leq SpAllocDev_A \leq SpAllocThres$$

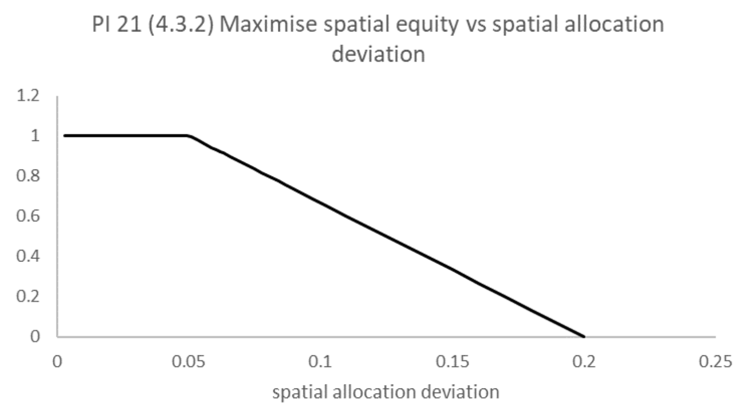


Figure D1.8.15: Functional form of performance indicator 4.3.2

If at least one region yields no catch, the value of the PI is 0

$if\ TotCatArea_A = 0.0, PI_{21,y} = 0$

Otherwise, the PI is the one minus the region-averaged spatial allocation deviation:

$$PI_{21,y} = 1.0 - \left(\frac{\sum_A SpAllocDev_A}{Narea}\right)$$

9 The TBL value function

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

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

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

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

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 calculate the value function for every other set of stakeholder group weightings, *k*:

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

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).

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:

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

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 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.

Appendix E: Detailed summaries of sensitivity tests undertaken on each performance indicator, and additional model scenarios

Sensitivity tests

Here we present the detailed results for each of the 17 performance-indicator-specific sensitivity tests undertaken, per Table E.1. These are summarised in Figure E.1, as boxplots showing the distribution of the impact of the test, relative to the reference Scenario 2a (non-spatial, dynamic TACs for both the commercial and charter sector). Scenario numbers are those given in Table 13.

Scenario 13: Here, we changed the target and limit biomasses to make the PIs around the target and OS species more conservative (PI1 TargBio1 = 0.85, TargBio2 = 0.6; PI2 TargBio = 0.6, LimBio = 0.3). This resulted in an increased CT biomass with generally much lower CT catches (results not shown). However, RTE catch increased, with catches, as observed earlier, inversely correlated to those of CT (results not shown). This correlation relates to the profitability objectives, and the increased RTE catch in response to CT catch would have maintained the commercial and charter profit. OS catch became highly interannual variable (results not shown). This was possibly due an "impossible conflict" between the catches of this species group required to achieve the optimal PI value, and those of the other two species groups required to achieve their optimal PI value, or due to the trade-off between biomass level and profitability within each year. The higher resultant overall effort led to increased risk to bycatch species (objective 1.2.1) and discarding (objective 1.2.2), and worse performance of the dependent social perception indicator (objective 4.2.1). Charter profit was increased (objective 2.1.2), presumably due to an increase in effort, as was the sustainability indicator related to environmental perturbations (objective 1.3.2), presumably due to the increased CT biomass.

Scenario 14: The risk to bycatch species is specified by defining some maximum effort threshold, above which bycatch is unacceptably high. Doubling this from 1.5 times the historical high effort to 3.0 times correspondingly reduced the risk to bycatch species (objective 1.2.1).There was little impact on the overall catch or biomass time series for any species group (results not shown), except that the magnitude of the interannual variability in catch for each was slightly increased (though this remained inversely corelated between species groups, such that the interannual variability in profit was unaffected).

Scenario 15: When the worst discarding reference threshold was set at a more conservative 20%, as opposed to 50%, this, obviously, increased the discarding risk (objective 1.2.2). This consequently increased the adverse public perception around discard mortality (objective 4.2.1). There was little overall impact on the catch or biomass trajectories, though the magnitude of the interannual variability in catch was substantially reduced for CT and RTE (results not shown).

Scenario 16: Reducing the threshold levels of desirable effort associated with broader ecological risk (objective 1.2.3) and risk to TEPs (objective 1.2.4) to more conservative values resulted in no changes to performance indicators, since the effort levels were always in excess of the limit threshold (results not shown).

Scenario 17: The risk of localised depletion due to fishing events (objective 1.3.1) was higher when the target and limit reference points were made more conservative. More variability over the projection years occurred for the bycatch and discarding risks (objectives 1.2.1, 1.2.2), and three of the 5 social performance indicators. There was little discernible effect on the catch or biomass trajectories (results not shown).

Scenario 18: The proxy used to account for risk of localised depletion in response to environmental events was to penalise the biomass estimate, so as to be more precautionary. Here the penalty was to assume the biomass was 60% as opposed to the baseline 80%. This resulted in a higher level of risk for this performance indicator (objective 1.3.2), but improved discarding risk, possibly due to reduced effort resulting in a higher proportion of larger fish in the catch. Again, more variability over the projection years occurred for the bycatch and discarding risks (objectives 1.2.1, 1.2.2), and three of the 5 social performance indicators. There was little discernible effect on the catch or biomass trajectories (results not shown).

Scenario 19: For both the commercial and charter profit indicators, costs were increased by 50%, and the reference profit at MEY was also increased by 20%. This reduced the interannual variability in catch for all three species groups and led to a linear increase in RTE catch over the projection period (results not shown). The catch of OS stabilised at lower levels, resulting in an increase in OS biomass (results not shown). Performance indicators for commercial, and the directly related indigenous profit (objectives 2.1.1, 2.1.3) were minimised, which was presumably due to the increased costs, since commercial catches were not adversely affected, but the charter profitability (objective 2.1.2), as a function of effort, was relatively unaffected by the increased costs. Spatial equity between regions (objective 4.3.2) was slightly adversely affected, presumably as spatial effort patterns varied in response to the increased costs. The bycatch and discarding risks reduced (objectives 1.2.1, 1.2.2) and the corresponding public perception increased (objective 4.2.1).

Scenario 20: In determining the value of recreational and charter fishing (direct to the participant), we changed the weights to emphasis catch and CPUE, and CT catch. This had no effect, as the performance indicator was at its optimal value (results not shown).

Scenario 21: When considering the flow-on economic benefit to local communities (objective 2.3), estimated as a function of commercial and charter profits, and recreational effort, increasing the recreational dollar scalar had very little effect, as the performance indicator was almost always at its optimal value. There was minimal effect on the other performance indicators, though there was increased variability about many, and, particularly, the bycatch and discarding risks (objectives 1.2.1, 1.2.2) and the associated public perception (objective 4.2.1). There was little effect on the catch and biomass trajectories (results not shown).

Scenario 22: Changing the target and limit coefficient of variation on inter-annual profit (objective 2.4) to make them more conservative resulted in minimal change to the inter-annual variability in profit. This is because in the early years of the projection, the performance indicator jumped from its minimal value of 0, to, in later years, its optimal value of 1 (results not shown). The threshold values had little impact since, for the majority of years, the performance indicator value was either well below the limit, or well above the target. There was some increased variability about other performance indicator values, but overall little change.

Scenario 23: Reducing the catch threshold (to the historical high catch) above which the associated costs of management are unacceptably high, reduced the interannual variability in catch in the later years of the projections (for all species groups), and resulted in slightly higher levels of biomass (results not shown), presumably because high catches were penalised by this indicator. Correspondingly, bycatch and discarding risks (objectives 1.2.1, 1.2.2) and the associated public perception (objective 4.2.1) were improved, while charter profit (objective 2.1.2) was adversely affected, due to a reduction in effort levels.

Scenario 24: The willingness to comply with the harvest strategy was altered by changing the weighting on "disagree fail" term from 0.6 to 0.7. This actually improved this performance indictor (objective 3.1). The “complexity fail” term of the performance indicator is constant between this and the reference scenario (they have the same number of TACs), so the “disagree fail” term, which is a function of the TAC, must have been more favourable, resulting in an improved performance indicator value when it was more strongly weighted. No discernible changes in the time series of catch or biomass were evident (results not shown), while slight improvements in the discarding risk (objective 1.2.2) and associated public perception (objective 4.2.1) occurred, and equity between sectors was slightly compromised (objective 4.1).

Scenario 25: Equitable access to the resource was defined based on the relative historical catches by species group and sector. When the deviation threshold was changed to 10% (from 20%) and tolerance to 1% (from 2%), the perceived access equity (objective 4.1) was correspondingly reduced. Bycatch and discarding risks (objectives 1.2.1, 1.2.2) and the associated public perception (objective 4.2.1) were slightly improved, possibly due to slightly reduced effort (though charter profitability (objective 2.1.2), which is a function of effort, showed only an increase in variability) while the variability in the spatial equity performance indicator (objective 4.3.2) increased. For all species groups, the interannual variability in catch was reduced in the later years of the time series, but there were no discernible impacts on the time series of biomass (results not shown).

Scenario 26: The extent of adverse public perception around discard mortality and TEPS was changed so that the risk thresholds for each were more conservative, which reduced the perceived performance of this performance indicator (objective 4.2.1). The discarding risk itself (objective 1.2.2) was also slightly worsened under this scenario. Time series of catch and biomass were relatively unaffected, though catch oscillated interannual between the extremes seen in the reference scenario, with fewer intermediate values (results not shown).

Scenario 27: The perception of fishing as a positive activity is modelled as a function of the CT and RTE relative biomass (objective 1.1.1), the OS biomass (objective 1.1.2), and the flow-on economic benefits (objective 2.3). Changing the weightings on each from equal to 0.5, 0.3 and 0.2 made almost no difference to the overall indicator (objective 4.2.3), nor to any others or to the catch and biomass time series.

Scenario 28: To alter the “increase access to local seafood” performance indicator (objective 4.3.1), the assumed proportion of dead-landed (locally available) CT was increased from 10% to 30%, past local availability of catch was increased from 50% to 70%, and the threshold local availability was increased from 0.2 to 0.4. This slightly reduced the modelled extent of access to local seafood (objective 4.3.1), but had little effect on other indicators, with the exception of some negative bias on equitable access between sectors (objective 4.1). There was little discernible effect on the catch or biomass time series (results not shown).

Scenario 29: Here we changed the definition of an equitable spatial catch allocation (objective 4.3.2) from that reflecting the relative abundance to 30%/70% (north/south), as well as tightening the deviation tolerance and threshold. This resulted in a lower extent of spatial equity (objective 4.3.2). The model responded by reducing effort in the south, to try to achieve this ratio, which resulted in improved (lessened) risk of bycatch and discarding (objectives 1.2.1, 1.2.2) , and hence the public perception against the latter (objective 4.2.1), as well as reduced charter profit (objectives 2.1.2). The interannual variability in CT and RTE catch was greatly reduced, and CT biomass reached its target level more rapidly (results not shown).

TABLE E.1: Descriptive summary of conceptual objectives together with their translation into operational objectives, or PIs, 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 Appendix D)	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 (CT, RTE and key other species yet to be identified)**	We use a dome-shaped specification (Figure D1.8.1). 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).	Scenario 13: 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 Scenario 13 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	Scenario 14: Changed effort threshold to 3x

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
			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.	historical high, as opposed to 1.5x
		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).	Scenario 15: 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)	Scenario 16: 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 Scenario 16 above

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
	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 species and areas.	Target and limit relative biomass reference points are set at 0.5 and 0.2.	Scenario 17: Target and limit reference points are changed to 0.6 and 0.3
		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.	Scenario 18: 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	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.	Scenario 19: Costs are multiplied by 1.5 AND ProfitMEY by 1.2, both for this and 2.1.2 below

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
			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.		
		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 was approximated by taking the historical high profits for each fishing sector, noting that these corresponded approximately to 0.6B0 for CT.	As for Scenario 19 above
		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.	Scenario 20: 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

Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
	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.	Scenario 21: 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 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.	The target and limit reference values are assumed.	Scenario 22: Changed target from +/-10%CV to +/- 5%CV, and limit from +/-25%CV to +/-20%CV
	2.5 Minimise costs of management associated with the harvest strategy: monitoring, undertaking assessments, adjusting		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.	Scenario 23: Changed threshold from 1.5x to 1.0x historical high catch

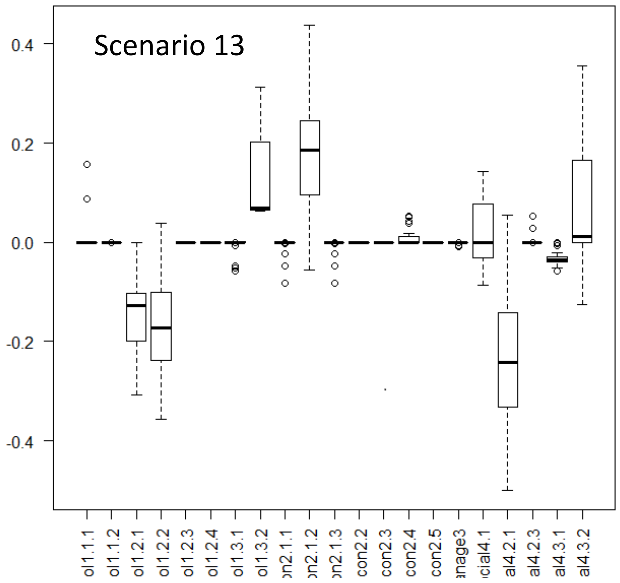
Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
	management controls				
3. Enhance management performance	**3.1 Maximise willingness to comply with the harvest strategy**		We assume that willingness 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.	Scenario 24: 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)
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.	Scenario 25: Deviation threshold changed to 10% and tolerance to 1%
	4.2 Improve social	**4.2.1. Through sound fishing practices, minimise**	We already have indicators of discarding (1.2.2) and TEPS (1.2.4). We recast these performance	The nature of the perception relationships, together with their	Scenario 26: Changed TEP risk

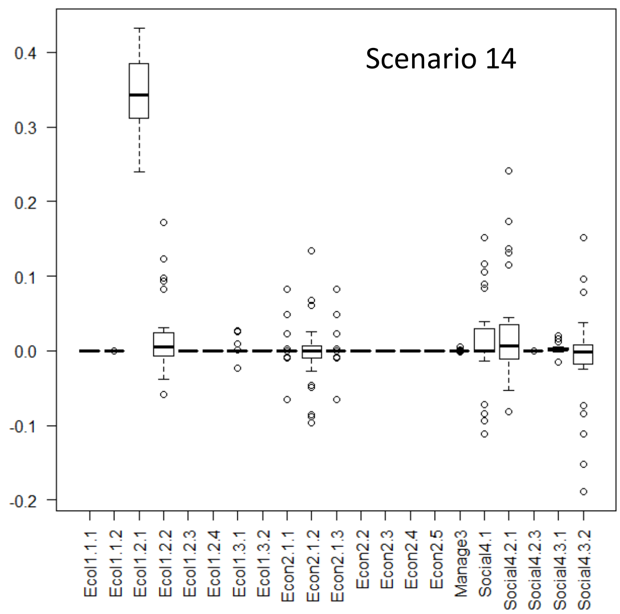
Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
	perceptions of the fishery (social licence to operate) (rec, commercial, charter, indigenous)	**adverse public perception around discard mortality (compliance with size limits, environmental sustainability, and waste)**	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.	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.	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 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 (CT and RTE sustainability), 1.1.2 (OS sustainability), and 2.3 (flow-on economic benefits), and take an average across them.	Each of the three contributing performance indicators is currently equally weighted.	Scenario 27: Changed weights from equal, to 0.5 CT & RTE, 0.3 OS, 0.2 flow-on economic benefits
	4.3 Enhance the net social value to the local community	**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 nature of the relationship, together with their threshold values, are assumed, as is the percentage of dead CT.	Scenario 28: Changed to assume 30% dead CT (rather than 10%), a past local

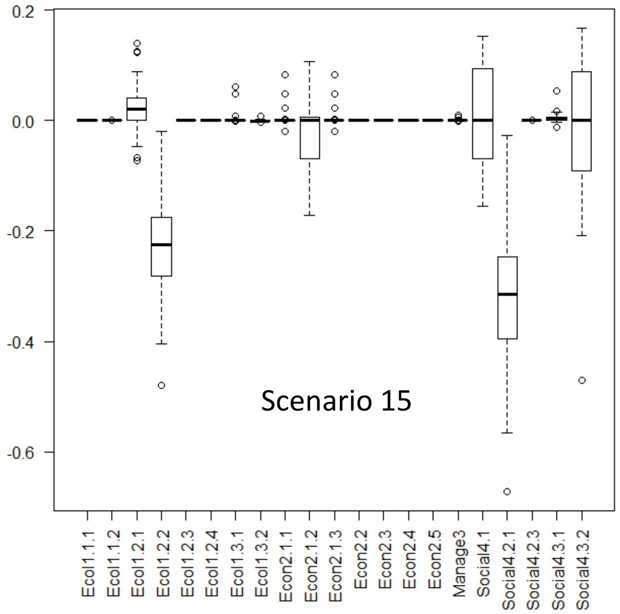
Overarching objective	Sub-objectives	Specific objectives	Operational objective (descriptive; full equations in Appendix D)	Assumptions	Sensitivity analysis
	from use of the resource		the local available domestic percentage is <20%, and 1 if the local available domestic percentage achieves that from the past, assumed to be equal to 0.5. We assume a "hockey stick" relationship between these two thresholds.		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.	Scenario 29: 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%

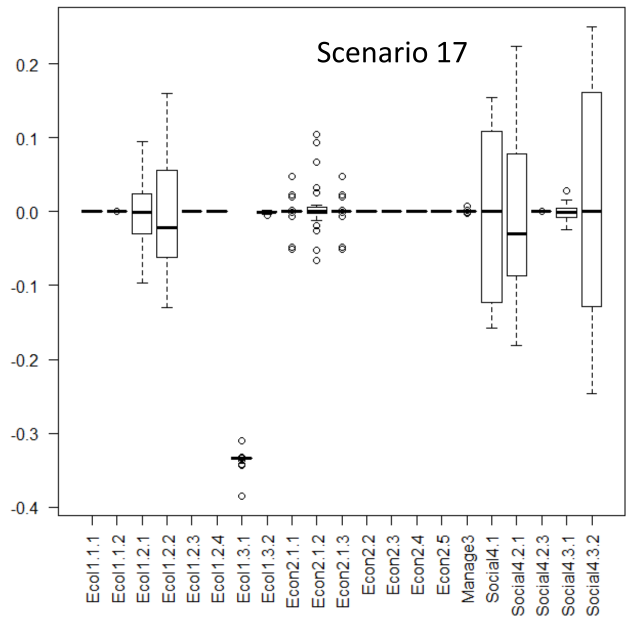
TABLE E.2: Summary of ALL model scenarios explored. Yellow rows indicate the scenarios included in the main body of the report.

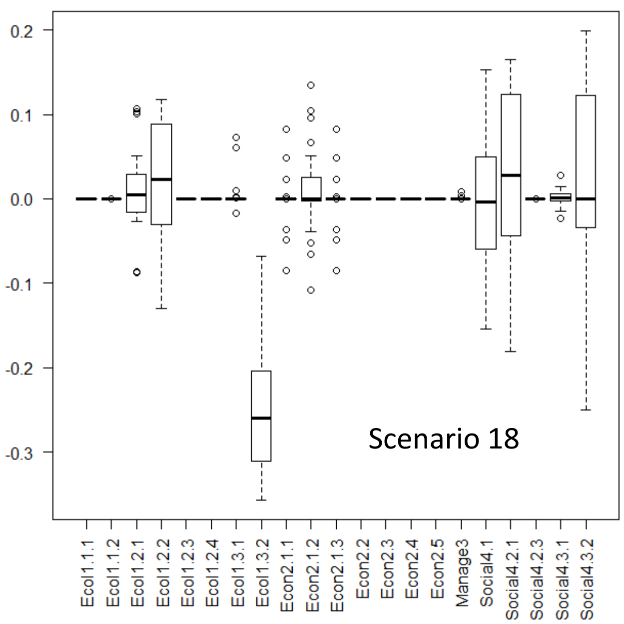
Theme	Scenario	Relative to reference scenario	Description	Highest Avg (1) or MaxiMin (2)	Area-specific TAC	Sectors subject to dynamic TAC	"Climate Change"	"Cyclone"	PI sensitivity index	TAC Constrained Metarule	Nelder-Mead or L-BFGS-B optimisation algorithm	"Fishdown": additional years with heavy fishing	
Reference	0		Commercial TAC only	1	FALSE	0 = Commercial only	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
	1	0	Commercial TAC optimised with Maximin criteria (as opposed to Highest Average)	2	FALSE	0 =Commercial only	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
	2	2a	Commercial and charter TAC with metarule of constrained interannual TAC threshold	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	0	TRUE	Nelder-Mead	""	0
	3	2a	All sectors receive TAC	1	FALSE	2 = all sectors get TAC	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
	2a		Reference: commercial and charter TAC	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
	2b	2FD	Per scenario 2a, but with historically "fished down" populations	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	0	FALSE	Nelder-Mead	"_fishdown2"	10
	2c	2a	Per scenario 2a, but with L-BFGS-B optimiser as opposed to Nelder-Mead	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	0	FALSE	L-BFGS-B	""	0
Area-specific TACs	4	0	Commercial only; region-specific TACs	1	TRUE	0 =Commercial only	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
	5	2a	Commercial and charter TACs; region-specific TACs	1	TRUE	1 =Commercial & charter only	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
	6	3	All sectors receive TAC; region-specific TACs	1	TRUE	2 = all sectors get TAC	FALSE	FALSE	0	FALSE	Nelder-Mead	""	0
External environmental perturbations	7	2a	"Climate change"	1	FALSE	1 =Commercial & charter only	TRUE	FALSE	0	FALSE	Nelder-Mead	""	0
	8	2a	"Cyclone"	1	FALSE	1 =Commercial & charter only	FALSE	TRUE	0	FALSE	Nelder-Mead	""	0
	9	2a	"Climate change" and "cyclone"	1	FALSE	1 =Commercial & charter only	TRUE	TRUE	0	FALSE	Nelder-Mead	""	0
	10	5	"Climate change"; region-specific TACs	1	TRUE	1 =Commercial & charter only	TRUE	FALSE	0	FALSE	Nelder-Mead	""	0
	11	5	"Cyclone"; region-specific TACs	1	TRUE	1 =Commercial & charter only	FALSE	TRUE	0	FALSE	Nelder-Mead	""	0
	12	5	"Climate change" and "cyclone"; region-specific TACs	1	TRUE	1 =Commercial & charter only	TRUE	TRUE	0	FALSE	Nelder-Mead	""	0
Performance indicator sensitivity	13	2a	PI 1.1.1, 1.1.2 sensitivity (CT/RTE and OS relative biomass target and limit reference points)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	1	FALSE	Nelder-Mead	""	0
	14	2a	PI 1.2.1 sensitivity (threshold value - multiplier of historical high catch, above which bycatch is unacceptably high)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	3	FALSE	Nelder-Mead	""	0
	15	2a	PI 1.2.2 sensitivity (threshold value = "worst possible" discard percentage)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	4	FALSE	Nelder-Mead	""	0
	16	2a	PI 1.2.3, 1.2.4 sensitivity (TEP risk target and limit reference point values = multiples of recent average, and historical high, effort, respectably)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	5	FALSE	Nelder-Mead	""	0
	17	2a	PI 1.3.1 sensitivity (area-specific relative biomass target and limit reference point values)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	6	FALSE	Nelder-Mead	""	0
	18	2a	PI 1.3.2 sensitivity (penalty value on 1.1.1 target and limit reference points, to be precautionary in face of environmental events)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	7	FALSE	Nelder-Mead	""	0
	19	2a	PI 2.1.1. 2.1.2 sensitivity (multipliers on costs and profit target reference point, for both commercial and charter profit)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	8	FALSE	Nelder-Mead	""	0
	20	2a	PI 2.2 sensitivity (weightings on relative contribution of catch, CPUE and effort relative contribution to recreational profit)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	10	FALSE	Nelder-Mead	""	0
	21	2a	PI 2.3 sensitivity (recreational dollar scalar value= dollar value of one unit of recreational effort, to the local community)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	11	FALSE	Nelder-Mead	""	0
	22	2a	PI 2.4 sensitivity (target and limit reference point values for coefficient of variation on interannual profit)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	12	FALSE	Nelder-Mead	""	0
	23	2a	PI 2.5 sensitivity (threshold value = multiplier of maximum historical catch, above which management costs are unacceptably high)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	13	FALSE	Nelder-Mead	""	0
	24	2a	PI 3.1 sensitivity (weightings on relative contribution of management complexity, and disagreement with management, to willingness to comply)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	14	FALSE	Nelder-Mead	""	0
	25	2a	PI 4.1.1 sensitivity (deviation threshold and tolerance values for catch proportion by sector)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	15	FALSE	Nelder-Mead	""	0
	26	2a	PI 4.2.1 sensitivity (threshold and limit reference values for perceived risk to TEPS, and asymptotic saturation point for perceived discard risk)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	16	FALSE	Nelder-Mead	""	0
	27	2a	PI 4.2.3 sensitivity (weightings on each of the three performance indicators 1.1.1, 1.1.2, 2.3 (sustainability, and flow-on community benefits) on perception of fishing as a positive activity)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	17	FALSE	Nelder-Mead	""	0
	28	2a	PI 4.3.1 sensitivity (assumed dead CT, assumed past local availability, threshold value of locally available % of fish)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	18	FALSE	Nelder-Mead	""	0
	29	2a	PI 4.3.2 sensitivity (tolerance and threshold values associated with equitable spatial allocation %)	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	19	FALSE	Nelder-Mead	""	0
Stocks more heavily expolited historically	2 FD	2	Per scenario 2, but with historically "fished down" populations	1	FALSE	1 =Commercial & charter only	FALSE	FALSE	0	TRUE	Nelder-Mead	"_fishdown2"	10
	5 FD	2b	Per scenario 5, but with historically "fished down" populations	1	TRUE	1 =Commercial & charter only	FALSE	FALSE	0	FALSE	Nelder-Mead	"_fishdown2"	10
	9 FD	2b	Per scenario 9, but with historically "fished down" populations	1	FALSE	1 =Commercial & charter only	TRUE	TRUE	0	FALSE	Nelder-Mead	"_fishdown2"	10


Scenario 13
0.4
0.2
0.0
-0.2
-0.4
ol1.1.1 ol1.1.2 ol1.2.1 ol1.2.2 ol1.2.3 ol1.2.4 ol1.3.1 ol1.3.2 on2.1.1 on2.1.2 on2.1.3 con2.2 con2.3 con2.4 con2.5 nage3 cial4.1 al4.2.1 al4.2.3 al4.3.1 al4.3.2


Scenario 14
0.4
0.3
0.2
0.1
0.0
-0.1
-0.2
Ecol1.1.1 Ecol1.1.2 Ecol1.2.1 Ecol1.2.2 Ecol1.2.3 Ecol1.2.4 Ecol1.3.1 Ecol1.3.2 Econ2.1.1 Econ2.1.2 Econ2.1.3 Econ2.2 Econ2.3 Econ2.4 Econ2.5 Manage3 Social4.1 Social4.2.1 Social4.2.3 Social4.3.1 Social4.3.2


Scenario 15
0.2
0.0
-0.2
-0.4
-0.6
Ecol1.1.1 Ecol1.1.2 Ecol1.2.1 Ecol1.2.2 Ecol1.2.3 Ecol1.2.4 Ecol1.3.1 Ecol1.3.2 Econ2.1.1 Econ2.1.2 Econ2.1.3 Econ2.2 Econ2.3 Econ2.4 Econ2.5 Manage3 Social4.1 Social4.2.1 Social4.2.3 Social4.3.1 Social4.3.2


Scenario 17
0.2
0.1
0.0
-0.1
-0.2
-0.3
-0.4
Ecol1.1.1 Ecol1.1.2 Ecol1.2.1 Ecol1.2.2 Ecol1.2.3 Ecol1.2.4 Ecol1.3.1 Ecol1.3.2 Econ2.1.1 Econ2.1.2 Econ2.1.3 Econ2.2 Econ2.3 Econ2.4 Econ2.5 Manage3 Social4.1 Social4.2.1 Social4.2.3 Social4.3.1 Social4.3.2


Scenario 18
0.2
0.1
0.0
-0.1
-0.2
-0.3
Ecol1.1.1 Ecol1.1.2 Ecol1.2.1 Ecol1.2.2 Ecol1.2.3 Ecol1.2.4 Ecol1.3.1 Ecol1.3.2 Econ2.1.1 Econ2.1.2 Econ2.1.3 Econ2.2 Econ2.3 Econ2.4 Econ2.5 Manage3 Social4.1 Social4.2.1 Social4.2.3 Social4.3.1 Social4.3.2

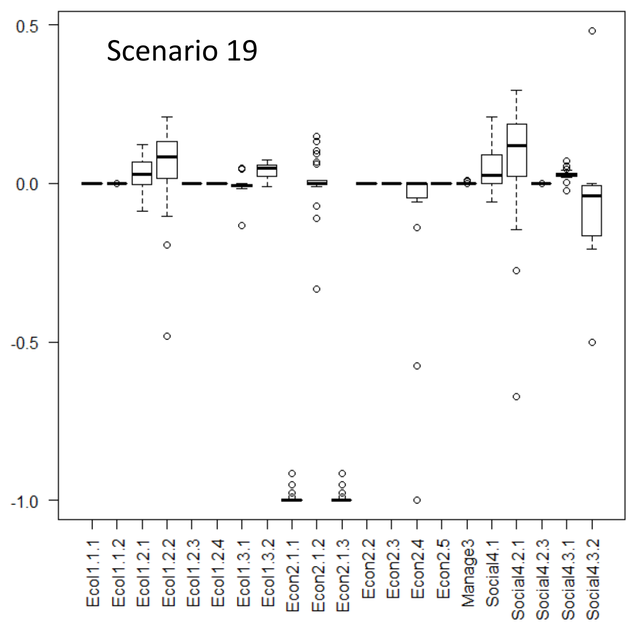
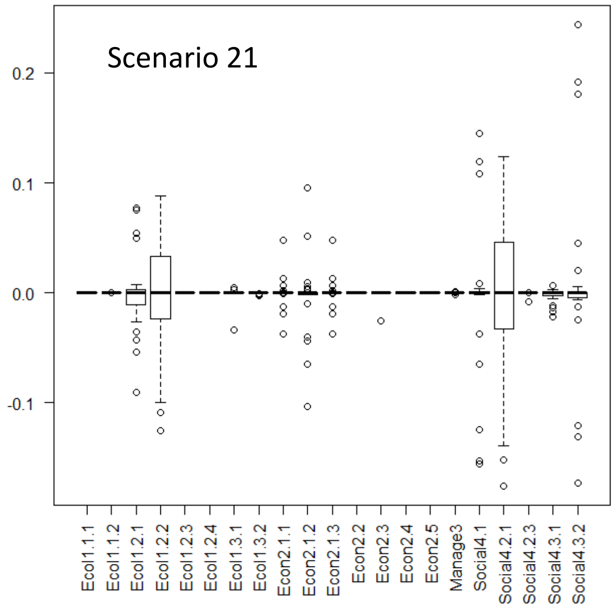
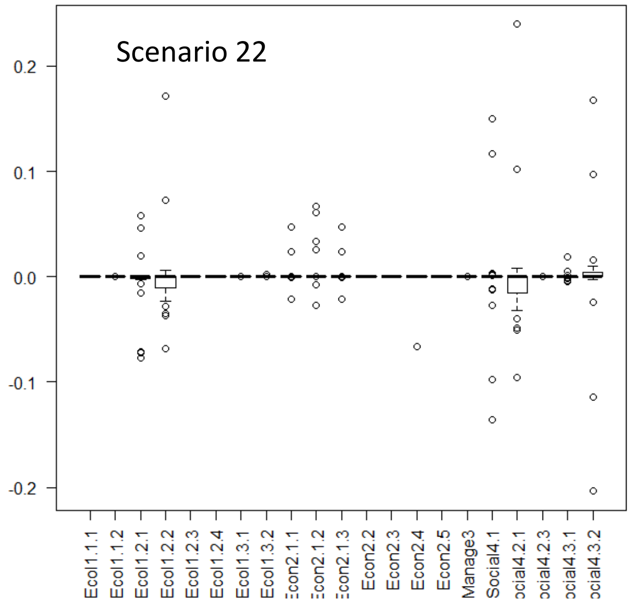
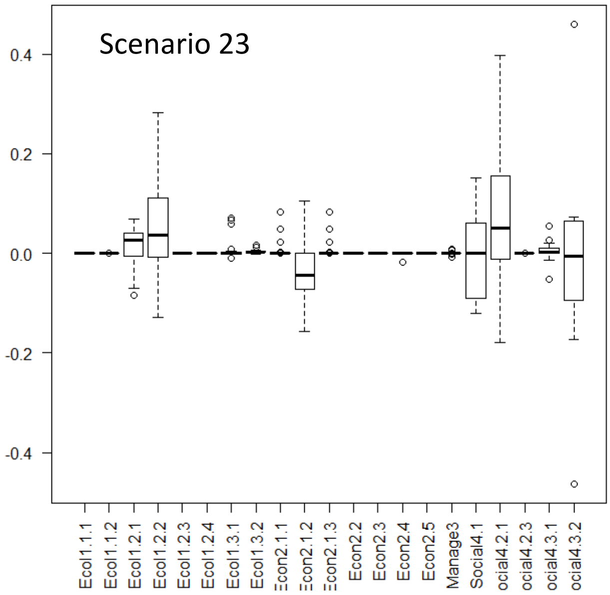
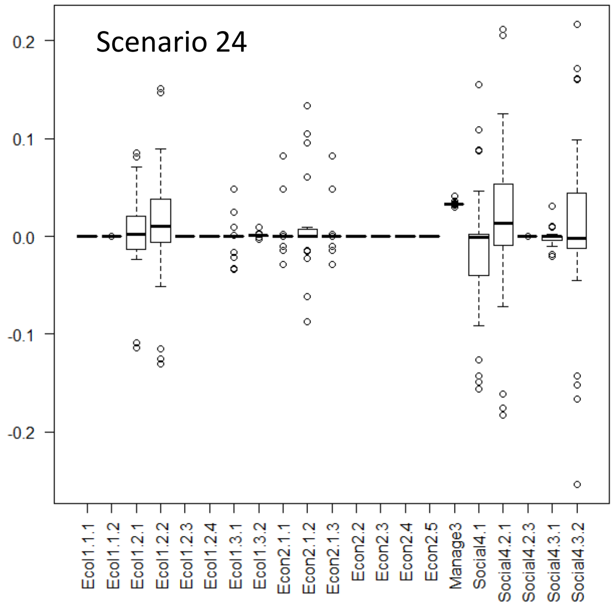

Scenario 19
0.5
0.0
-0.5
-1.0
Ecol1.1.1 Ecol1.1.2 Ecol1.2.1 Ecol1.2.2 Ecol1.2.3 Ecol1.2.4 Ecol1.3.1 Ecol1.3.2 Econ2.1.1 Econ2.1.2 Econ2.1.3 Econ2.2 Econ2.3 Econ2.4 Econ2.5 Manage3 Social4.1 Social4.2.1 Social4.2.3 Social4.3.1 Social4.3.2

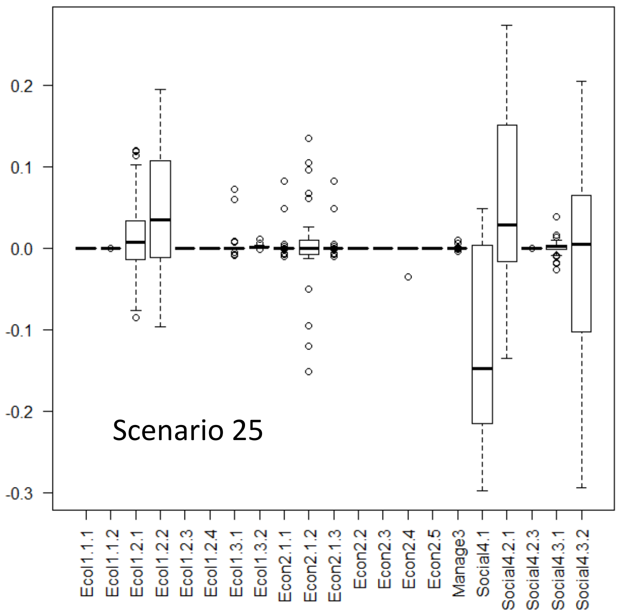
Figure E.1: Distribution of the impacts of Scenarios 13-30 (sensitivity analyses around performance indicators, per Table 13) relative to Scenario 2a, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.


Scenario 21
0.2
0.1
0.0
-0.1
Ecol1.1.1
Ecol1.1.2
Ecol1.2.1
Ecol1.2.2
Ecol1.2.3
Ecol1.2.4
Ecol1.3.1
Ecol1.3.2
Econ2.1.1
Econ2.1.2
Econ2.1.3
Econ2.2
Econ2.3
Econ2.4
Econ2.5
Manage3
Social4.1
Social4.2.1
Social4.2.3
Social4.3.1
Social4.3.2


Scenario 22
0.2
0.1
0.0
-0.1
-0.2
Ecol1.1.1
Ecol1.1.2
Ecol1.2.1
Ecol1.2.2
Ecol1.2.3
Ecol1.2.4
Ecol1.3.1
Ecol1.3.2
Econ2.2
Econ2.3
Econ2.4
Econ2.5
Manage3
Social4.1


Scenario 23
0.4
0.2
0.0
-0.2
-0.4
Ecol1.1.1
Ecol1.1.2
Ecol1.2.1
Ecol1.2.2
Ecol1.2.3
Ecol1.2.4
Ecol1.3.1
Ecol1.3.2
Econ2.2
Econ2.3
Econ2.4
Econ2.5
Manage3
Social4.1


Scenario 24
0.2
0.1
0.0
-0.1
-0.2
Ecol1.1.1
Ecol1.1.2
Ecol1.2.1
Ecol1.2.2
Ecol1.2.3
Ecol1.2.4
Ecol1.3.1
Ecol1.3.2
Econ2.1.1
Econ2.1.2
Econ2.1.3
Econ2.2
Econ2.3
Econ2.4
Econ2.5
Manage3
Social4.1
Social4.2.1
Social4.2.3
Social4.3.1
Social4.3.2


Scenario 25
0.2
0.1
0.0
-0.1
-0.2
-0.3
Ecol1.1.1
Ecol1.1.2
Ecol1.2.1
Ecol1.2.2
Ecol1.2.3
Ecol1.2.4
Ecol1.3.1
Ecol1.3.2
Econ2.1.1
Econ2.1.2
Econ2.1.3
Econ2.2
Econ2.3
Econ2.4
Econ2.5
Manage3
Social4.1
Social4.2.1
Social4.2.3
Social4.3.1
Social4.3.2

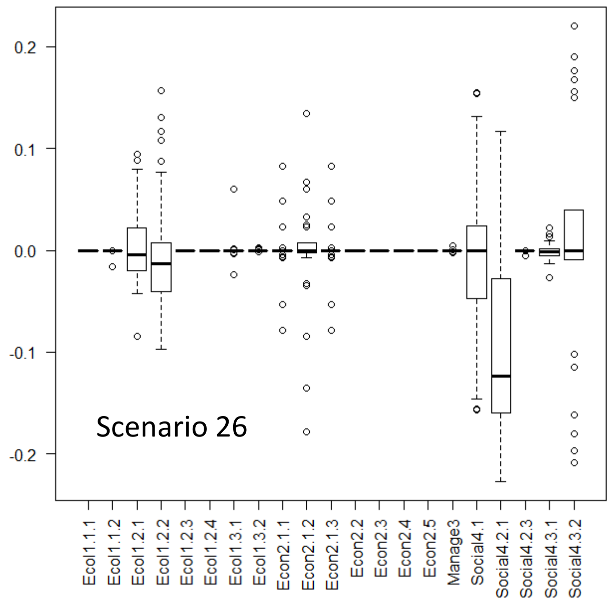

Scenario 26
0.2
0.1
0.0
-0.1
-0.2
Ecol1.1.1
Ecol1.1.2
Ecol1.2.1
Ecol1.2.2
Ecol1.2.3
Ecol1.2.4
Ecol1.3.1
Ecol1.3.2
Econ2.1.1
Econ2.1.2
Econ2.1.3
Econ2.2
Econ2.3
Econ2.4
Econ2.5
Manage3
Social4.1
Social4.2.1
Social4.2.3
Social4.3.1
Social4.3.2

Figure E.1 cont'd. Distribution of the impacts of Scenarios 13-30 (sensitivity analyses around performance indicators, per Table 13) relative to Scenario 2a, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

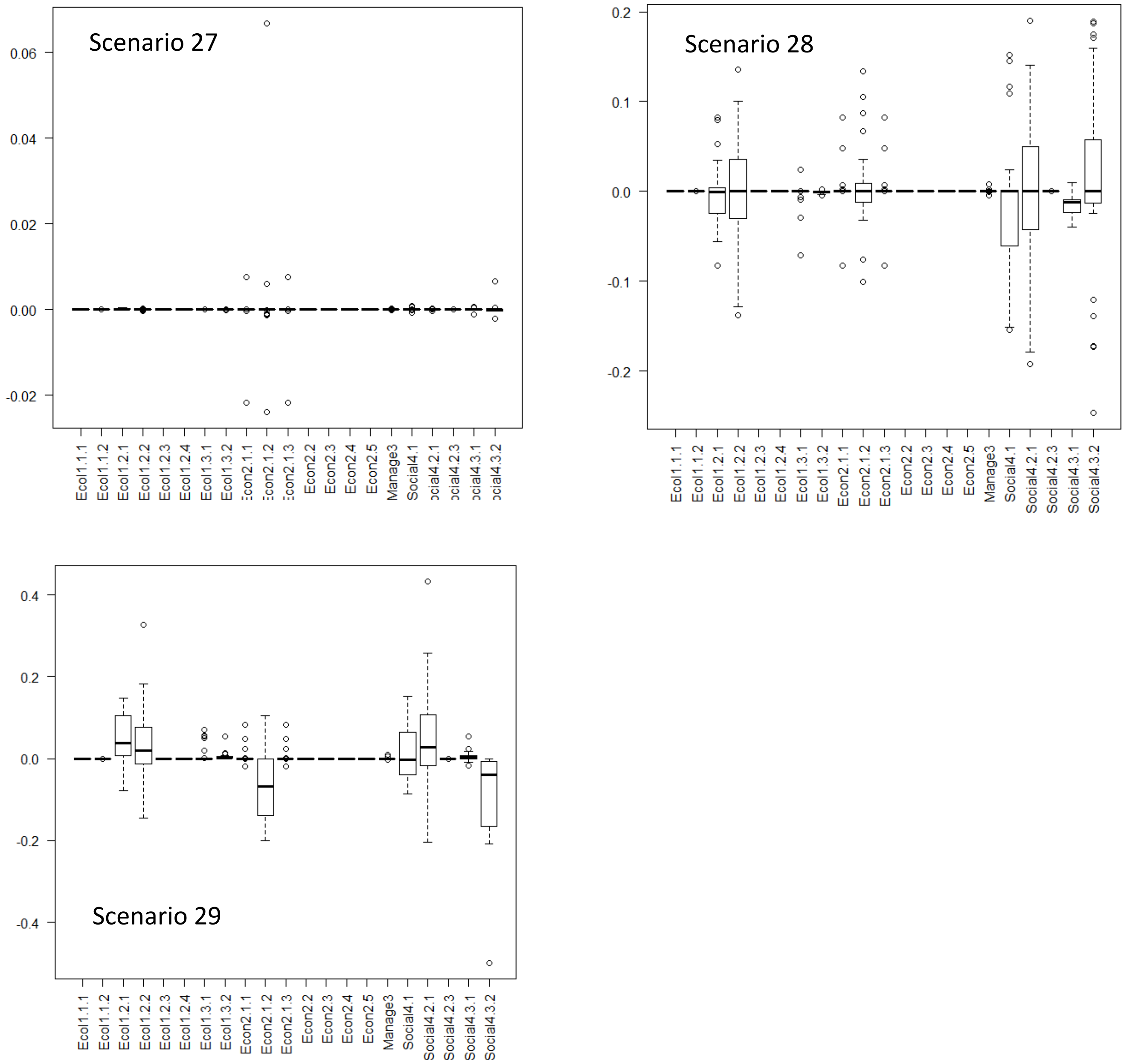


Figure E.1 cont'd. Distribution of the impacts of Scenarios 13-30 (sensitivity analyses around performance indicators, per Table 13) relative to Scenario 2a, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

Additional model scenarios

To explore the possible effects of external environmental perturbations in greater detail, we used Scenario 2a as the reference scenario (non-spatial, dynamic TACs for both the commercial and charter sector). Results are summarised in Figure E.2, as boxplots showing the distribution of the impact of the test, relative to the reference Scenario 2a. Scenario numbers are those given in Table 13.

Under the simplified climate change scenario (Scenario 7), there was little effect on overall catch or biomass, nor most of the performance indicators. However, slightly lower catches of CT toward the end of the time series, and an associated slight increase in CT biomass (results not shown), resulted in slight improvement in the conservative sustainability indicator pertaining to mitigation against long-term change (objective 1.3.2). There was a slight relative increase in discarding (a reduction in performance indicator 1.2.2), as well as a worsening of the associated social perception indicator (objective 4.2.1) as a result of increased relative proportions of undersized fish in the catch. This was likely due to the overall reduction of abundance of all species by 0.7% per year, which would also have been responsible for the increase in the risk to bycatch species (objective 1.2.1).

A "cyclone" was simulated in the 5th 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 projection years 5–8 (Scenario 8). This could be seen as temporary decreases and increases in CT and RTE catches, respectively (results not shown). The interannual variability in species-group-specific catches was also reduced for these two species groups (results not shown). There was a slight relative increase in discarding (a reduction in performance indicator 1.2.2, as well as a worsening of the associated social perception indicator 4.2.1) as a result of increased relative proportions of undersized fish in the catch, possibly as a result of the temporary reduction in availability of CT. Across all performance indicators, the main difference was a reduction in the charter sector profitability (objective 2.1.2). This appears incongruous given that commercial profitability was unaffected, but as opposed to commercial profitability, charter profitability is simulated as a function of effort. Effort is simulated for each sector in each year as the catch divided by the product of the (time invariant) catchability, and the fishable biomass. As the available biomass of RTE increased during the simulated "cyclone", for the same catch, this implied a reduction in effort, and hence, a reduction in charter sector profitability.

We also considered simulated "climate change" and "cyclone" effects when TACs were set specific to region. For these, we used Scenario 5 (region-specific, dynamic TACs for both the commercial and charter sector) as the reference scenario. Results are summarised in Figure E.3, as boxplots showing the distribution of the impact of the test, relative to the reference Scenario 2a. Scenario numbers are those given in Table 13.

Relative to scenario 5, there was little effect of simulated climate change with region-specific TACs (Scenario 10) on the overall catch or biomass trajectories, with the exception that OS biomass showed a more gradual decrease to its target level, reflecting lower OS catches earlier in the projection years. These lower OS catches occurred in the northern region, which is to be expected given the simulated southern migration under "climate change" (results not shown). In direct opposition to the analogous "climate change" scenario 7 (where TACs were not region specific), there was a slight relative decrease in discarding (an increase in performance indicator 1.2.2), as well as an increase in the associated social perception indicator (4.2.1), relative to scenario 5, as a result of lower relative proportions of undersized fish in the catch. There was also a slightly lower bycatch risk (1.2.1) suggesting slightly less effort. As other performance indicators were barely affected, it would appear that, regional-specific TACs under "climate change" resulted in reduced effort and reduced early OS catches in the north, with the former being responsible for the lower discarding and both resulting in lower proportions of smaller fish in the catch.

With the flexibility afforded by regionally-specific TACs, the simulated cyclone had little effect on performance indicators (Scenario 11). The reductions in charter sector profitability and the increased discarding risk associated with reductions in effort seen in Scenario 8 was less evident here. The reduced availability of CT in the south was not discernible on the overall catch time series, compensated as it was by increases in catch in the north. The correlation in catches between species groups was particularly evident in year 39, when, after the years of increased RTE availability (and hence catches) resulted in a zero catch for this species group in the south (presumably because RTE biomass had by then been driven to the lower end of its target range). In the same year, OS catch peaked (results not shown).

When considering both simulated climate change and a cyclone under region-specific TACs (Scenario 12), the catch and biomass time series are hybrids of those observed in scenarios 10 and 11 – they are almost identical to the latter except for, as with scenario 10, lower OS catches earlier in the projection years (results not shown). Overall, the flexibility afforded by region-specific TACS buffers the impact of climate change and cyclones, such that the performance indicators are not significantly affected.

If TACs are allocated only to the commercial sector (Scenario 4), there is little discernible difference in the catch and biomass trends between Scenario 4, which had region-specific TACs, and Scenario 0, which did not (results not shown). Keeping the charter and recreational catches constant afforded a lack of flexibility that constrained the commercial TAC setting: total catch for each species showed very little variation from the final historical year (results not shown). Relative to Scenario 0, allowing region-specific TACs resulted in slightly improved performance indicators relating to broader ecological and discarding risk (objectives 1.2.1, 1.2.2), and slightly higher biomasses that led to a slight improvement in the performance indicator of sustainability in the face of environmental events (objective 1.3.2) (Figure E.4). Commercial and recreational profits were also slightly improved (objectives 2.1.1, 2.3).

When TAC was allocated to all three sectors, higher catches of CT and lower catches of OS resulted when TACs were also specific to each of the two regions, relative to when TACs were not region specific (results not shown). While this had little effect on most performance indicators, the region-specific TACs did slightly improve the spatial equity between regions (objective 4.3.2), and significantly increased charter profit (objective 2.1.2). This was likely as a result of increased effort, since the risk to bycatch species (objective 1.2.1), inversely proportional to effort, was worsened. The increased effort was likely due to a reduction in biomass, since the risk of undersize discarding (objective 2.2.2) and the related public perception (objective 4.2.1) were worsened. Introducing region-specific TACs also reduced the “willingness to comply” management objective (3.1) due to the increase in the perceived complexity of management (Figure E.4).

We now re-examine the effects of region-specific TACs and external environmental perturbations in the context of stocks having been more heavily historically “fished down”. Having regional-specific TACs affords greater flexibility in rebuilding the “fished down” stocks (Scenario 5FD, considered relative to scenario 2b), such that biomass recovery is more tempered (results not shown). The majority of performance indicators are relatively unaffected, but the spatial flexibility afforded by regional-specific TACs reduced the risk of undersize discarding (thus improving performance indicators 1.2.2 and the related 4.2.1) (Figure E.5).

Aside from slightly suppressing CT catch and increasing that of the OS group during the simulated “cyclone” years, under the “fished down” scenario there is little overall effect of the simulated “climate change” or “cyclone” (Scenario 9FD). This is similar to the response to simulated “climate change” and “cyclone” effects when the stocks were not fished down. Each species group is recovered to its target biomass levels - the CT and RTE stocks had already recovered to their target levels by projection year 5, when the simulated “cyclone” was imposed (results not shown). Relative to Scenario 2b, there is little difference in the performance indicators (Figure E.5), and certainly less difference than for the analogous scenarios where the historical populations were not “fished down”.

Finally, we consider a metarule, whereby the TAC for any species group is only permitted to vary by a maximum of 20%, but, for the charter and recreational sectors, the TAC is not adjusted at all unless the (unconstrained) modelled TAC for the next year exceeds a threshold of 30% of that of the previous year. We consider such a rule because managers often wish to cap the extent of interannual adjustments.

Applying the metarule based on the historical catch patterns, with TACs allocated to the commercial and charter sectors (Scenario 2) does not have a large impact on the Coral Trout catch or biomass trajectories, since the most recent catches place the stock on a path to recovery to target biomass levels. However, the metarule does not permit RTE and OS catches to increase to the extent that the biomass is reduced to target levels over the projection period (results not shown). Consequently, the performance indicator corresponding to CT and RTE sustainability is compromised (objective 1.1.1, as this is penalized when biomass is above target levels. However, the risks related to ecosystem resilience (objectives 1.2.1-1.2.4) are all lower under the metarule, due to the reduced catch levels (Figure E.6). As would be expected, all economic performance indicators are adversely affected under the metarule, except for those relating to reduced interannual variability and reduced costs of management (objectives) 2.4 and 2.5, which are improved. The social performance indicators related to perception of discard activity and of fishing as a positive activity (objectives 4.2.1 and 4.2.3) are improved, as is the spatial equity between regions (objective 4.3.2), the former because of their relationship with the improved ecosystem objectives, and the latter, presumably, because of the relatively lower deviation from the historical status quo.

Applying the same metarule, but in the context of the stocks having been historically “fished down” (Scenario 2FD), resulted in minimal recovery of CT: stock levels were still below target by the end of the projection (results not shown). While CT catches were reduced in the first two years (by the maximum, constrained extent possible in the first year), they showed little change thereafter. In contrast, while the unconstrained model achieved similar levels of CT catch to the metarule-constrained model for most years of the projection, it recovered the CT stock to target levels by dramatically reducing catches to close to zero for a few years early in the projection time series (results not shown). To compensate for the lack of recovery of CT, the metarule simulation drove the RTE population above the target biomass to close to 90% B_0 by the end of the projection (results not shown). This was presumably because the performance indicators related to sustainability (objectives 1.1.1 and 1.3.2) (while the biomass of either CT or RTE was above the limit reference point of 20% B_0) were calculated as the average of the CT and RTE performance "score". Performance indicator 1.3.2 penalised the actual relative biomasses by 20%, and, as the lowest-value (worst) performance indicator (Figure E.6), likely explained the minimal RTE catches and consequent high RTE biomass to offset the below-target CT biomass. Economic performance indicators relating to commercial, charter and indigenous profitability were adversely affected by the low RTE catches (objectives 2.1.1-2.1.3; Figure E6). As a result of the low catch levels of RTE, the performance indicator relating to reduced costs of management (objective 2.5) was improved.

Finally, we considered the L-BFGS-B (Limited-Memory Broyden–Fletcher–Goldfarb–Shanno box constraints) method as an alternate optimisation algorithm to the Nelder-Mead method, for Scenario 2 (TACs allocated to the commercial and charter sectors) (Scenario 2c). The run time was increased from hours to days, and the overall results were generally similar, with the most sensitive performance indicators being those related to broader ecological and discarding risk (objectives 1.2.1, 1.2.2), and the related social perception risk (objective 4.2.1). This shows that, while the model was generally robust to the optimisation algorithm, the complexity of the solution surface does results in minor differences (Figure E7).

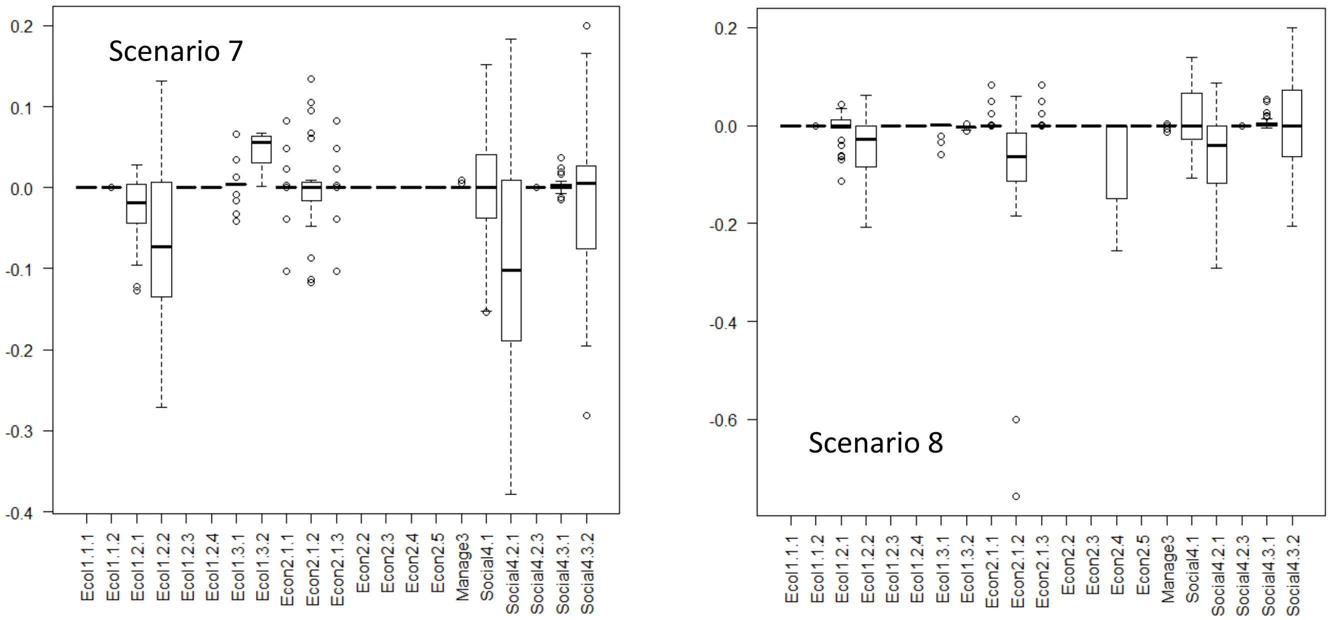


Figure E.2. Distribution of the impacts of Scenarios 7 and 8 (external environmental perturbations, per Table 13) relative to Scenario 2a, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

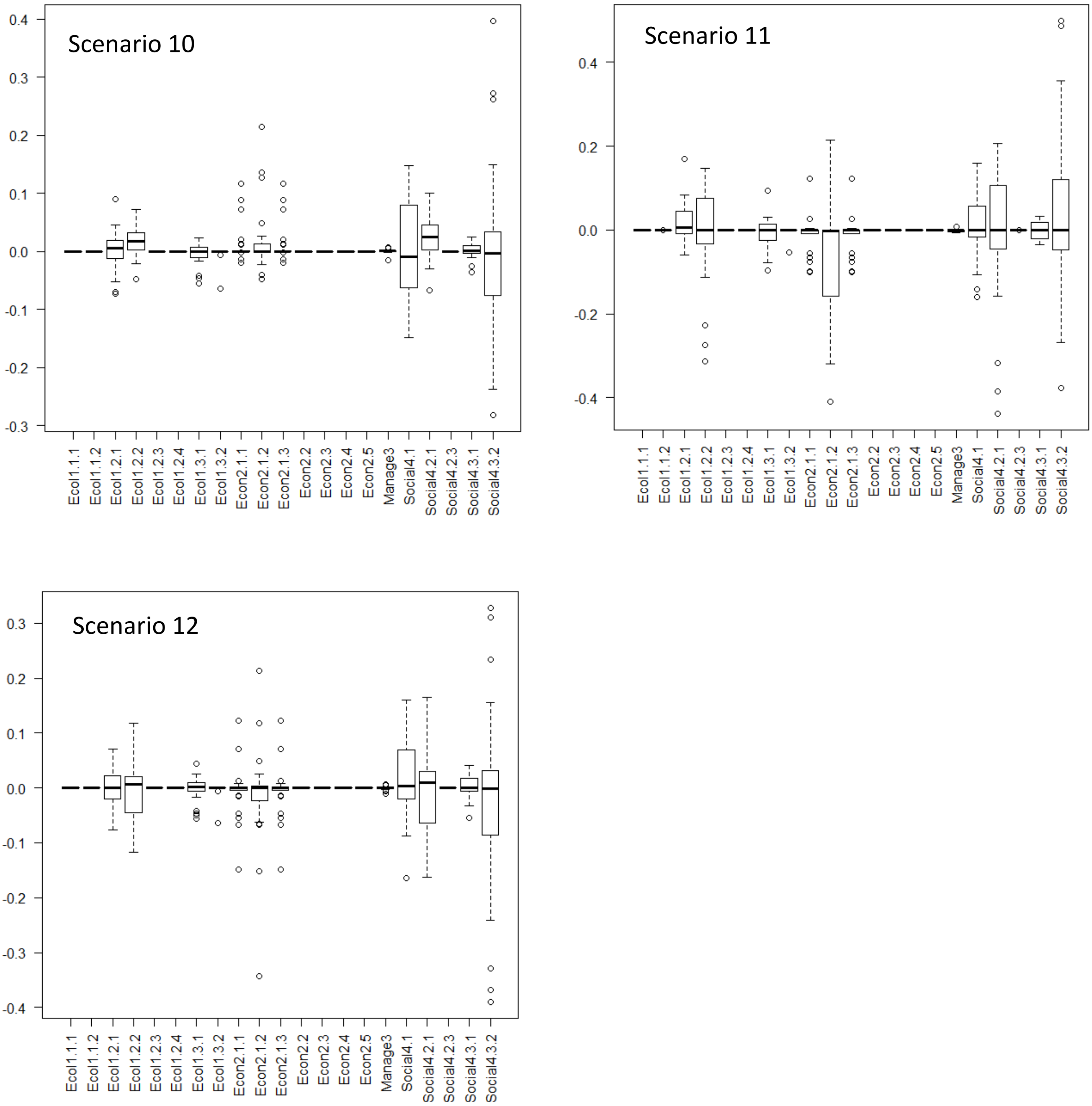


Figure E.3. Distribution of the impacts of Scenarios 10-12 (external environmental perturbations with region-specific TACs, per Table 13) relative to Scenario 5, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

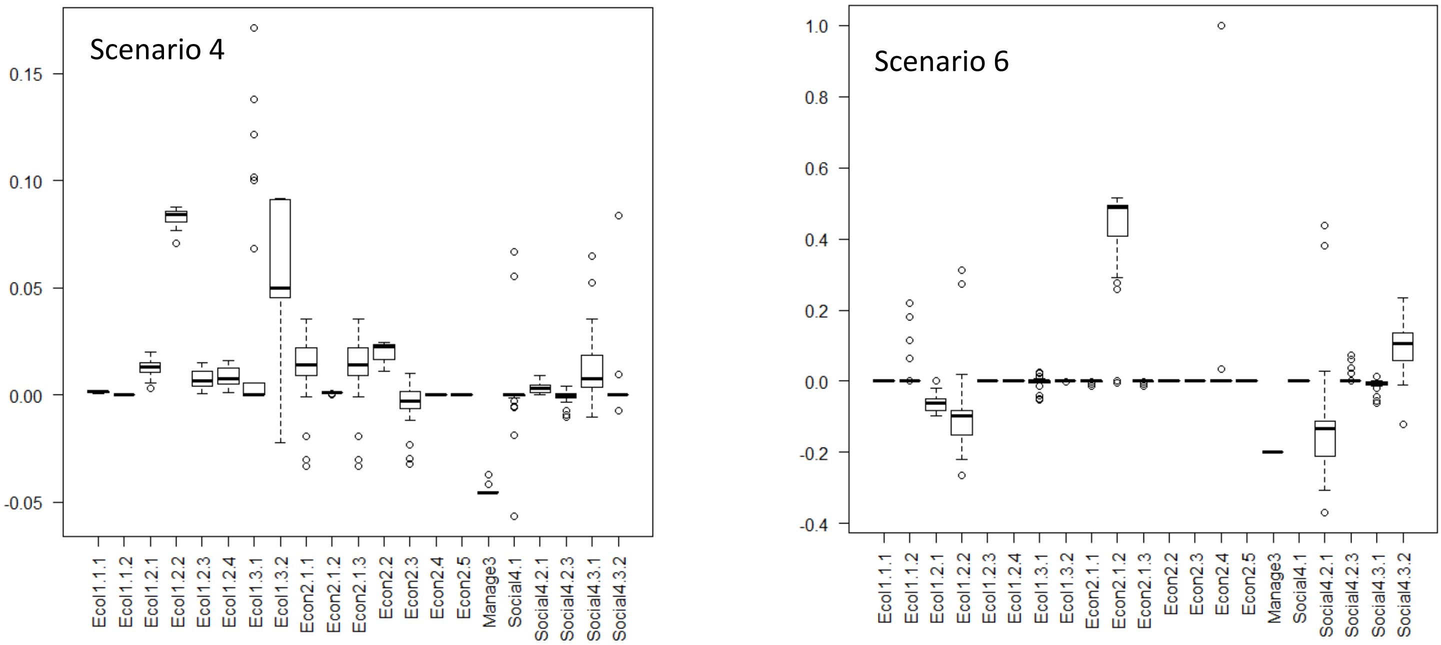


Figure E.4. Distribution of the impacts of Scenarios 4 and 6 (only commercial TAC; all sectors receive TAC, respectively, per Table 13) relative to Scenarios 0 and 3, respectively, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

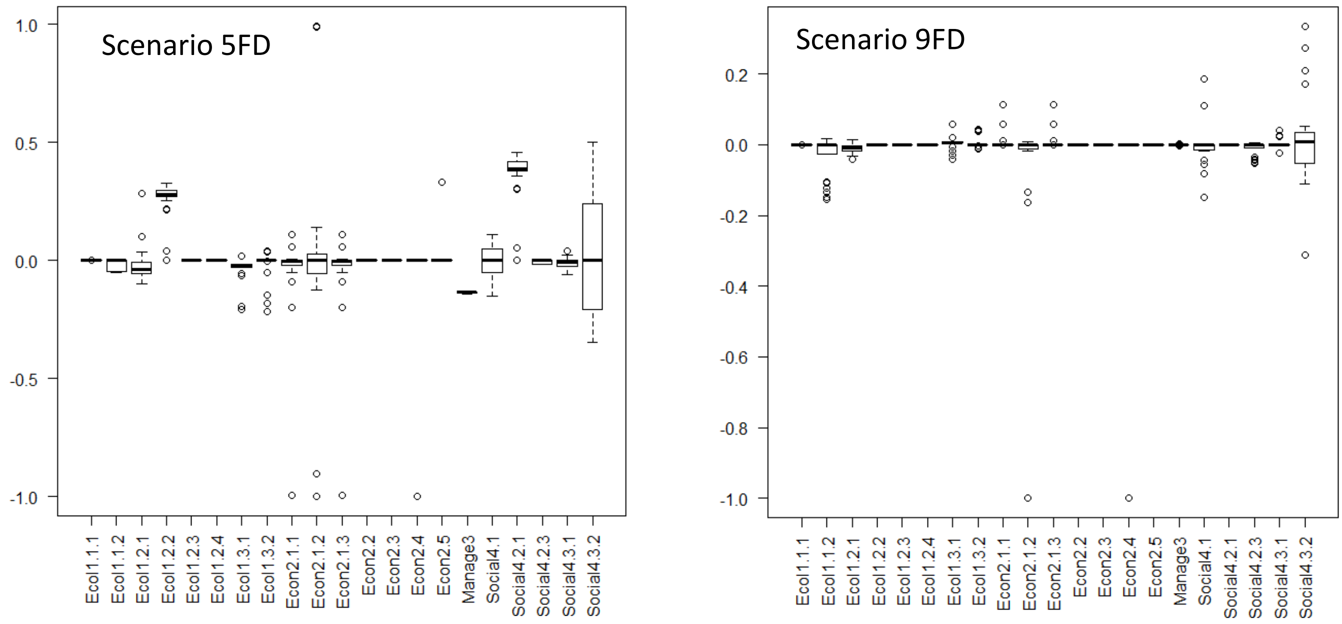


Figure E.5. Distribution of the impacts of Scenarios 5FD and 9FD (region-specific TACs, and “cyclone” and “climate change”, respectively, under historically “fished-down” conditions) (per Table 13), relative to Scenario 2b, by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

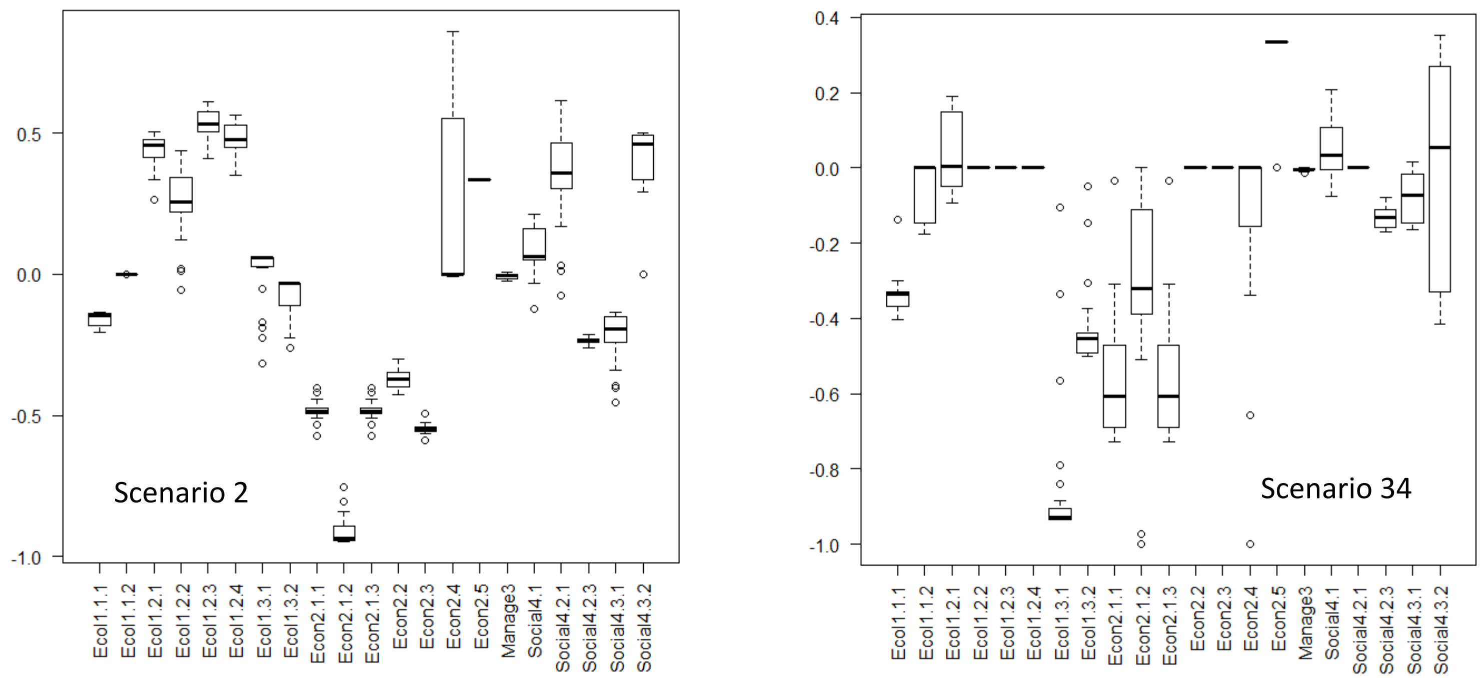


Figure E.6. Distribution of the impacts of Scenarios 2 (relative to Scenario 2a) and 2FD (relative to Scenario 2b) (metarule on TAC interannual adjustment magnitude, under historical and “fished down” conditions, respectively) (per Table 13), by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.

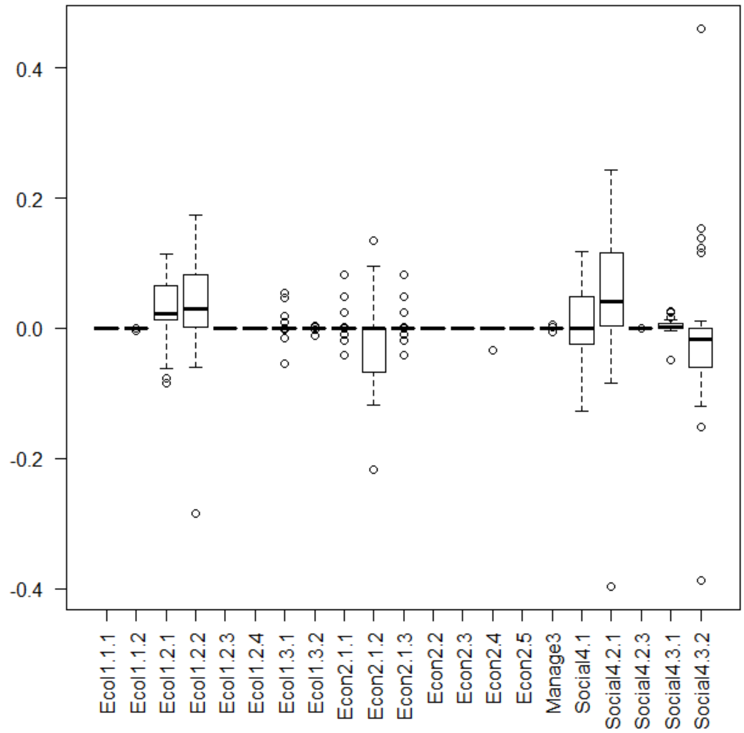


Figure E.6. Distribution of the impacts of Scenario 2c (relative to Scenario 2a) (L-BFGB-B optimisation algorithm as opposed to Nelder-Mead, per Table 13), by performance indicator. A value of 0 indicates no change, while -1.0 and 1.0 are the maximum negative and positive differences, respectively.