Trade-offs and co-benefits between fishing and conservation through area closures and partially protected areas

Final Report

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

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Abbreviations

AMP	Australian Marine Parks
AHP	Analytic Hierarchy Process
CICES	Common International Classification of Ecosystem Services
DEA	Data envelopment analysis
DMU	Decision-making units
EA	Ecosystem Accounts
FPA	Fully protected areas
GCI	Geometric Consistency Index
GDP	Gross domestic product
GMM	Geometric mean method
GVP	Gross value of production
IUCN	International Union for Conservation of Nature
MAHP	Modified Analytic Hierarchy Process
MPAs	Marine protected areas
NTAs	No take areas
PPAs	Partially protected areas
SEEA EA	UN System of Environmental Economic Accounting—Ecosystem Accounting
SNA	System for National Accounts
TEV	Total economic valuation
UNCEEA	United Nations Committee of Experts on Environmental-Economic Accounting

Executive Summary

Internationally, there is a growing push to protect vulnerable marine habitats and species through the establishment of networks of marine protected areas (MPAs). While these have a conservation focus, there is recognition that these benefits can still be achieved while allowing some activities to take place in particular areas with lower risk to conservation. As a result, there has been an increase in the use of partially protected areas (PPAs) in combination with fully protected areas (FPAs), reconstituting what might be considered to be a formal MPA. Spatial and temporal closures have also been used as fisheries management tools in many cases.

In this project, we examine the ecological, economic and social costs and benefits that have been generated through the use of PPAs and fisheries-specific closures. The project is undertaken by a multidisciplinary team from QUT, Natural Capital Economics and CSIRO. The project reviews Australian and international experiences with the use of PPAs and fisheries closures, and considers how an understanding of these benefits and costs can inform management of future PPAs through two case studies developed through workshops with Marine Park managers at the state and Commonwealth level.

Aims/objectives

The overall project has three objectives:

1. Undertake a review of existing literature on values of partially protected areas, including a review of different frameworks in which the environmental, social and economic values can be incorporated for management decision making
2. Assess the key environmental, social and economic values and their trade-offs in the use of partially protected areas, based on two case studies
3. Identify how understanding these values can assist management decision making when implementing marine parks more broadly in Australia.

Methodology

The project involved two main stages. The first stage (addressing objective 1) was divided into two parts. The first part reviewed alternative frameworks for decision support that could aid in marine reserve planning, taking into consideration the ecological, economic and social benefits and costs that may occur under different management scenarios. This included a review of the types of measures, how they might be estimated and how they may be combined to support decision making. The second part reviewed empirical international studies that had identified and/or estimated these benefits and costs in PPAs and fisheries closures. The focus of the review was on the types of impacts observed (and their significance) rather than the values themselves, as these were considered case specific.

The second stage of the project involved an analysis of Australian experiences with marine reserve design to allow for multiple use based on two case studies. As part of this, an assessment of the importance of key considerations when designing a marine reserve was undertaken. The second stage of the project involved workshops with marine park managers for an inshore (Queensland) marine park and offshore (Commonwealth) marine reserve network. The workshops provided an opportunity for the managers to highlight issues they encountered during the zoning process as well as develop a set of objectives that would aid in future marine park design. A survey was subsequently undertaken of marine park and fisheries managers, as well as commercial and recreational fishing organisations, to determine the importance different stakeholders placed on these objectives as an input into potential multicriteria decision analysis approaches in the future. The workshops also supported a review of the ecological, economic and social impacts in each of the case study marine parks.

Results/key findings

The study considered a range of frameworks that may aid future decision making, including benefit-cost analysis based on economic valuation of the ecological benefits as well as the social and economic impacts; ecosystem accounting frameworks that are useful for monitoring changes in these values over time; multicriteria decision analysis frameworks that allow an assessment of potential management options over a range of ecological, economic and social objectives and well-being frameworks and indexes that focused on integrating objective and measurable social outcomes and individual stakeholders' subjective perceptions of wellbeing. Each framework has strengths and weaknesses, and the choice of framework will depend on the availability of relevant information as well as the definition of the key objectives of the MPA/PPA zoning.

The empirical studies reviewed were mixed in their conclusions regarding the impacts of fully protected areas and partially protected areas within a marine park. In many cases, economic impacts arising from effort displacement offset to some extent the impact of the reduced fishing area, although this was at a cost. Locational changes in fishing activity also had implications for onshore economic impacts (i.e., loss of benefits to some regions and associated sectors such as processing) as well as social impacts (i.e., displacement of fishing employment to other regions or loss of employment altogether).

The study also considered the application of fishery-related closures as fisheries management options. These were largely to either protect juvenile or spawning stocks, or limit bycatch. These were generally found to produce economic benefits in the case of closures related to stock protection, although the displaced effort did result in some costs in other parts of the fishery. The displaced effort also had implications for social impacts, similar to those related to marine protected areas for conservation purposes.

The workshops and associated case study reviews found that a common set of objectives of marine park zoning was to maximise conservation outcomes while minimising the impact on resource users such as commercial and recreational fishers. The review of each case study found that impacts on the fishing sectors were generally less than initially believed (although not necessarily zero). However, in the case of commercial fishing, this seemed to be largely due to measures to remove fishing effort before the zoning (i.e., buyback) such that the effects of effort displacement were minimised. While the economic effects were negligible, the social effects related to reduced fishing activity (i.e., flow on effects to the community) were larger.

A range of objectives, that described the "values" that managers aimed to achieve when designing a marine protected area were also elicited from the workshops. A survey was implemented to elicit the relative importance or weight placed on these objectives from across key stakeholder groups: marine parker managers, fisheries managers, commercial and recreational fisher organisations. The results of the survey were largely consistent with a priori expectations, with priorities for sub-objectives and process objectives closely reflecting the interests and perceived responsibilities of each stakeholder group. Both fisheries and MPA manager respondents tended, on average, to place higher weight on ecological outcomes. In contrast, respondents from commercial and recreational fisher organisations tended to place higher weight on economic outcomes.

Implications for relevant stakeholders

Balancing the trade-off between conservation and use of marine resources is complex. The results of the case-study analysis and the review suggest that economic impacts associated with increasing fully closed areas within marine parks can be minimised through planning and also complementary fisheries management such as buyback of potential excess capacity. In this regard, the use of partially protected areas within marine parks offers a means to maximise conservation outcomes and minimise impacts on otherwise displaced users of the resource. The social impacts of these, however, are less avoidable. These

social impacts need to be considered in the context of the benefits the broader society gains through increased protection of the marine environment.

A key output of the study is the identification of a set of operational objectives that include environmental, economic and social outcomes from MPAs and PPAs. These provide a framework against which future MPAs/PPAs can be assessed, aiding in their planning.

Keywords

Marine protected areas, partially protected areas, fishery closures, economic impacts, social impacts.

Introduction

Marine ecosystems provide a variety of environmental, economic, and social benefits. These include food security, livelihoods from commercial activities (e.g., fishing), climate management (e.g., carbon storage), human wellbeing, marine-based transport routes, amenity, recreation, and cultural and spiritual values for local communities. These benefits are often inter-related and competing, which makes management of these environmental assets necessary but difficult. Overuse of marine resources for extractive purposes may result in increased short run economic benefits, but at the expense of environmental and other social benefits, including loss of habitat and biodiversity and the non-use values these generate to the broader community.

Marine protected areas (MPAs) are a management tool that have the potential to address several of the pressures on marine habitats and biodiversity, particularly over-fishing. MPAs can also help ensure the sustainable provision of multiple other marine ecosystem services that are fundamental for human well-being, including for fisheries, coastal protection (buffering against storms and erosion), tourism and recreation (OECD 2017).

While MPAs are a primary tool for conservation and protection of marine habitats and their associated marine life, 69% of MPAs are partially open to some form of fishing activity, either recreational or commercial (Turnbull *et al.* 2021). While these partially protected areas (PPAs) provide economic and social benefits, these may come at the cost of reduced conservation benefits. Hence, the implementation of PPAs as a management tool to safeguard habitat and aquatic resources is underpinned by a tension to balance environmental, economic, and social values of multiple users.

In Australia, MPAs are implemented primarily to ensure long-term conservation of a marine habitat or ecosystem, and generally fall into one of two categories: fully protected areas (FPAs) and partially protected areas (PPAs). In Australia, which has the second largest MPA network in the world, 75% of the total MPA area is partially protected (Roberts *et al.* 2020). FPAs typically focus on a narrower set of benefits (e.g., ecological restoration) through restriction of harvesting and other direct uses. PPAs allow for some continual economic and social benefit from direct use, such as limited commercial or recreational activities or restricted access for Indigenous communities, while still ensuring protection of key marine habitats (providing non-use values). Multiple FPA and PPA can be designated within a marine ecosystem or habitat, creating a protected area network across the ecosystem.

In comparison, area closures are also often implemented for fisheries management purposes, primarily to protect spawning aggregations (e.g., temporal closures) or juvenile habitats (e.g., spatial closures). These may also have broader marine conservation benefits, but this is not their primary purpose. As with PPAs in MPAs, these closures may also permit some recreational or commercial activities that are not expected to impact the main fisheries management objective.

The challenge faced by resource managers implementing PPAs, as compared to FPAs, is the need to balance the competing social and economic benefits from use of the marine resources (harvesting and other direct uses) with the ecological benefits from its protection (i.e., environmental benefits requiring restricted use). The optimal level of restrictions on use depends on the objectives of the PPA, and needs to be assessed across the full set of multiple benefits. This requires an understanding of the multiple values that can be realised through partial, rather than full, protection, and the trade-offs between these values.

Managing this balance requires that resource managers are cognisant of these diverse use (and non-use) values, and have an appropriate management framework to ensure that decision making results in socially optimal outcomes where possible. Decision making that does not consider these multi-user

values can result in undesirable trade-offs, adverse environmental impacts, unnecessary complexity, ineffectiveness of management, inefficiency in resource use, and potentially social conflict.

Access to aquatic resources, guided by good management, is fundamental for the continued delivery of economic and social benefits such as food, income, employment, recreation and cultural identity for all Australians. Optimising these benefits means sharing resources fairly using open and evidence-based processes within the limits of sustainability. As a first step to developing optimal multi-user management, there is a need to first review, and where possible, quantify the multiple values (environmental, economic, and social) that are attained through the implementation of PPAs.

The purpose in this report is to provide an overview of Australian and international experiences with the application of PPAs to achieve social, economic and environmental (conservation) benefits. We examine a range of frameworks with which to assess the efficacy of multi-use marine reserves with respect to these key objectives. We consider also the relative importance different stakeholder groups place on these objectives.

Objectives

The overall project has three objectives:

1. Undertake a review of existing literature on values of partially protected areas, including a review of different frameworks in which the environmental, social and economic values can be incorporated for management decision making
2. Assess the key environmental, social and economic values and their trade-offs in the use of partially protected areas, based on two case studies
3. Identify how understanding these values can assist management decision making when implementing marine parks more broadly in Australia.

Method

Scope of the study

By IUCN's definition, an MPA's primary goal is the conservation of nature (Grorud-Colvert *et al.* 2021). IUCN recognise seven different levels of protection, ranging from fully protected (MPAs) to areas that are multiple use, allowing some extraction of the marine resources. These include (Hayes *et al.* 2021):

1. Sanctuary Zones (IUCN Ia) provide for the conservation of ecosystems, habitats and native species in as natural and undisturbed a state as possible.
2. National Park Zones (IUCN II) provide for the conservation of ecosystems, habitats and native species in as natural a state as possible.
3. National Monument (IUCN III) provide for the protection of natural landforms or geological features (e.g., seamounts)
4. Habitat Protection Zones (IUCN IV) provide for the conservation of ecosystems, habitats and native species in as natural a state as possible while allowing activities that do not harm or cause destruction to seafloor habitats.
5. Recreational Use Zones (IUCN IV) provide for the conservation of ecosystems, habitats and native species in as natural a state as possible while providing for recreational use.
6. Multiple Use Zones (IUCN VI) provide for ecologically sustainable use and the conservation of ecosystems, habitats and native species.
7. Special Purpose Zones (IUCN VI) provide for ecologically sustainable use and the conservation of ecosystems, habitats and native species, while applying special purpose management arrangements for specific activities.

An example of the types of activities permitted in each zone type is given in Table 1**Error! Reference source not found.**.

Table 1. Activities permitted in different PPA zones

	Zone				
Activity	Multiple use	Special purpose	Recreational use	Benthic sanctuary	Sanctuary
	IUCN.VI	IUCN.VI	IUCN.II	IUCN Ia	IUCN Ia
Commercial fishing					
• demersal trawl, Danish seine, gillnetting (below 183m) & scallop dredging	×	×	×	×	×
• pelagic fishing	✓	×	×	✓	×
• other	✓	×	×	×	×
Recreational fishing	✓	✓	✓	✓1	×
Charter fishing	🗒	🗒	🗒	🗒1	×
Research	🗒	🗒	🗒	🗒	🗒
Commercial shipping	✓	✓	✓	✓	✓
Commercial tourism	🗒	🗒	🗒	🗒	🗒
Mining					
• seismic survey and transit	✓	✓	×	×	×
• all other	🗒	🗒	×	×	×

Notes: × not allowed, ✓ allowed, 🗒 authorisation required; 1 Pelagic only

For the purposes of this report, we consider three levels of protection (Figure 1). The first two (MPAs and PPAs) are consistent with the IUCN classification (although simplified), while the third is separate to the IUCN classification**Error! Reference source not found.**. Fisheries closures are fisheries management measures that are aimed directly at limiting fishing mortality on key species either to ensure their protection (e.g., bycatch species) or enhance future production in the fishery (i.e., juvenile area closures, seasonal closures to protect spawning stocks or enable growth to an optimal size). While they may involve a conservation element (i.e., protection of key species), they are of direct relevance to fisheries management rather than marine conservation management in general.

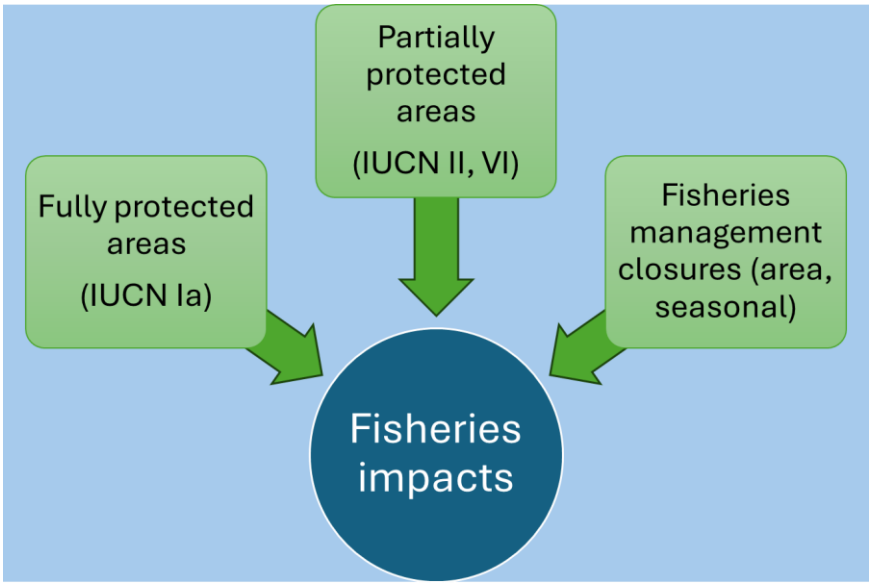


Figure 1. Types of closures considered in this study

Review of alternative frameworks for decision support

The study considered different potential frameworks to capture the different values to support management decision making in regard to MPA, PPAs and spatial and temporal fisheries closures. These included:

(1) ecosystem services
(2) total economic value, where all benefits are measured in terms of economic value measured using money as the numeraire where possible;
(3) multi-criteria decision analysis, which may combine monetary and non-monetary values and preferences;
(4) ecosystem accounting approaches, which capture economic and biophysical measures but potentially less able to capture social values measured as welfare explicitly; and
(5) wellbeing frameworks and indexes.

The advantages and limitations of these difference frameworks and corresponding valuation methodologies, including pragmatic factors such as data availability, their ability to account for public preferences, support social licence to operate and improve the likelihood of attaining beneficial outcomes for all users, are also discussed, aiding resource managers.

Review of empirical international studies

There was a detailed review of previous empirical international studies of MPA, PPAs and spatial and temporal fisheries closures to identify types and potentially magnitude of different environmental, social and economic values. The review considered the types or forms of partial protection currently in use, with the aim of assessing how these management measures influence the type and degree of environmental, social and economic values generated.

Reviews were conducted using Google Scholar as the primary search tool to identify both relevant (e.g., based on number of citations) peered reviewed research papers and the broader "grey" literature. This was then extended utilising a snowballing approach that considered manuscripts either already in the researcher's collection or cited by authors in the Google Scholars searches. Non-English papers were not considered.

The focus was on studies was from developed countries with well-established fisheries management. Case studies from developing countries were considered and used as appropriate. For example, for the purposes of identifying social impacts, studies from developing countries were excluded as these were considered to have substantially different baseline social structures. This exclusion does not invalidate the empirical results of these studies, but underlies that their relevance to the Australian context in this case was not applicable.

Case studies and workshops

Two case studies were undertaken considering the Australian Marine Parks (AMP) (Commonwealth) and Moreton Bay Marine Park (MBMP, Queensland). Both are characterised by multiple use protected area zoning with different levels of use permitted in each zone type.

The two case studies highlighted very different systems of partially closed areas. The Australian Marine Parks are mostly offshore, and while they offer some recreational fishing opportunities, these are limited. Commercial fishing in much of the marine parks is permitted, with the exception of boats using trawl gears. Charter fishing (commercial recreational fishing) is also allowed in most areas. Other activities such as tourism (e.g., whale shark or whale watching) are permitted but the opportunities are limited across the whole network and vary by marine park. Additionally, the reserves also provide cultural values and benefits which may be affected (positively or negatively) through activities in the partially protected areas.

In contrast, the Moreton Bay Marine Park has a substantial recreational fishing component, being adjacent to a major Australian population area (i.e., Brisbane and south-east Queensland). Moreton Bay Marine Park also offers zones with different permitted commercial uses. Tourism benefits are potentially high in the region, with companies offering a range of activities such as charter fishing, whale watching tours, sight-seeing cruises, and kayaking daytrips. Indigenous values associated with the marine park are also likely to be significant. These values are likely to be enhanced through protection. Conversely, social and economic benefits associated with recreational and commercial fishing will be adversely affected though area closures.

With the most recent zoning implemented in 2009, the MBMP offers the opportunity to assess how these values have changed over time. Recreational and commercial fishing information from surveys conducted within the park is available and past analyses of these data have been undertaken. There has also been monitoring undertaken to assess ecosystem change in the protected and partially closed areas. Information on the types of activities permitted to operate in the marine park and compliance, linked to social licence to operate, is also available.

The case study analyses involved first a review of previous studies specific to each of the parks to identify changes in environmental, social and economic values as well as analysis of the potential changes in benefits using existing data. While other potential benefits of the use of the PPAs were identified and considered (i.e., non- use values, social values, other tourism values), the analysis focused on recreational and commercial fisheries as data on these aspects is most readily available.

Workshops with key marine parks managers from each of the case study marine parks were also held to outline the findings of the review and identify additional benefits or impacts not covered in the review. Workshop participants were individuals who are involved in marine park management and/or policy, and were identified by the Department of Environment and Science (Queensland State Government) (workshop 1) and Parks Australia (Commonwealth Government) (workshop 2).

The workshop program included:

- A presentation summarising previous studies undertaken in each of the marine parks to form the basis of identifying the key considerations used in marine park management.
- A facilitated discussion to identify the key values (environmental, economic, and social values of multiple users) that inform the management of the marine park and partially protected areas in particular.
- A discussion on how competing stakeholder needs are prioritised in decision making.

Identification of appropriate objectives for designing partially protected areas

The workshops also focused on how these values may be included in future management decision making. A second key objective of the workshops was to identify a set of generic objectives that can be considered when developing future marine protected areas to allow for economic and social benefits in addition to ecological benefits. Such objectives and their relative importance are necessary for considering the trade-offs between alternative value outcomes from different management options.

Objectives were elicited in the workshops, with the outcomes used to develop an online survey to elicit importance weights associated with each of these objectives. While the workshops focused on marine protected area managers, the online survey included workshop participants and other stakeholders, including recreational and commercial fishers and conservation groups.

Objective classification

To frame the potential set of operational objectives, we build on the framework developed by Keeney (2007) where objectives are grouped into four categories:

1. Fundamental objectives: These are what matters: the ultimate ends objectives. For example, "*maximise ecological, economic and social benefits from marine reserves".*
2. Means objectives: These are the means to achieving the fundamental (ends) objectives, and potentially strategic objectives. For example, *"maximise protection of threatened habitats".*
3. Strategic objectives. These can be higher-level goals of institutions, programs or other initiatives that the fundamental, means and process objectives can help achieve. For example, *"enhance existing benefits by focusing on areas with highest value"*.
4. Process objectives: these involve processes that help support means, ends and strategic objectives. For example, "*Develop greater appreciation of the marine ecosystem through education."*

Some of these types of objectives are nested, hierarchical and thus interdependent (e.g., Fundamental and Means objectives, where the means objectives focus on components of the bigger picture – the fundamental objective), while others may be parallel or independent (e.g., Strategic and Process objectives that indicate how the objectives are to be achieved). By classifying objectives into these categories, we can construct a hierarchy of linked objectives within a framework of linked decisions. It also allows us to identify which objectives are central to further developing partially protected areas, and which objectives are for consideration outside of the process (i.e., strategic objectives).

Key guiding principles in identifying operational objectives

Given the set of potential objectives identified in the broad review and workshops, a set of guiding principles was established for identifying key fundamental and means objectives for inclusion in the decision-making framework (Pascoe *et al.* 2024a):

- Established fundamental objectives must be considered and adopted where relevant.

The legislative and policy framework for the existing marine protected areas may specify fundamental objectives applicable to their designation and management.

- Achieving objectives must be within the control of managers.

If the manager cannot influence the outcome through interventions or governance, then there is no means, process, strategic or fundamental objective that can be considered in the decision-support process (Brooks *et al.* 2015). For example, a social objective of ending regional poverty is outside the control of marine protected area managers so would not be an appropriate social objective (no matter how noble the intentions).

- Objective outcomes must be measurable against a benchmark.

All operational objectives need to be associated with attributes and metrics that are measurable (Dichmont *et al.* 2020). If they cannot be measured or assessed (even qualitatively – yes/no) then the performance of the marine protected area cannot be assessed.

- More is not necessarily better than less.

There is often a tendency to want to consider a wide range of objectives, particularly when many stakeholders are involved, each contributing their own multiple objectives. However, for multi-objective optimization problems, finding a global (or even a local) optimum is difficult. For example, if 30 objectives are identified, the average additive weight of each will be 3.3%, resulting in a relatively "flat" achievement function across a wide range of possible outcomes (Pascoe *et al.* 2024a). Dowling *et al.* (2020b) found that the large number of operational objectives for the Coral Reef Finfish fishery made optimization extremely difficult for this reason, with the optimal outcome highly sensitive to small changes. Similarly, when assessing outcomes against a large array of objectives, the influence of individual objectives becomes limited or diluted. The influence of each objective is roughly inversely related to the number of objectives (Pascoe *et al.* 2024a).

Once a preliminary set of objectives were identified, these were again discussed with the key stakeholders (i.e., previous workshop participants) for validation and modification if necessary.

Elicitation of objective preference weights

Once the final set of operational objectives were identified, an online survey was developed to elicit the preference weights (see Appendix). The survey had three sections:

- Section 1: Provides further information on the study and instructions as to how to complete each set of questions.
- Section 2: Asks you to rate (using a slider) the relative importance of different key ecological, economic and social considerations; and
- Section 3: Asks for information about you and your current role in marine resource management.

Potential survey respondents were identified through internet searches of organisation and government agencies with a potential interest in partially protected areas. Where possible, named individuals were invited to participate, with 217 individuals identified across five different stakeholder groups (Table 2). An additional 35 invitations were sent to "generic" organisational email addresses where individuals could not be identified.

The survey used a Modified Analytic Hierarchy Process approach to elicit the relative preferences (Pascoe et al. 2024b).The MAHP reduces inconsistencies in responses compared with the traditional Analytic Hierarchy Process (AHP) (Saaty 1980) by modifying the way in which the data are collected and analysed, taking into account the symmetry assumption underlying AHP.

Table 2. Invitations to participate in the online survey

Group	Direct invites	Organisational invites
Fisheries Management	61	5
Fishing Industry Association	30	10
Marine conservation	8	3
Marine protected area management	44	11
Recreational fishing association	74	6
Total	217	35

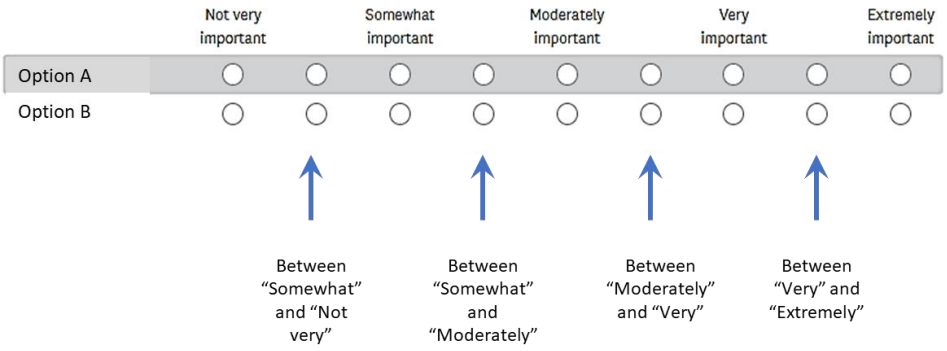


Figure 2. Example of bivariate comparison in the MAHP

In the online survey, respondents were presented with a nine-point importance scale against which they could assess the importance of each objective. A nine-point scale was selected (rather than an "out of 10" as might occur with a Likert scale) as it allows five categories to be defined with mid-points between them, consistent with the traditional AHP approach. The options being assessed are presented together as a choice matrix, with the preference score of each being explicitly compared with the preference scores of the other options (Figure 2**Error! Reference source not found.**) (Pascoe *et al.* 2024b). Additional alternatives are included as additional rows, and all alternatives are compared in the same question.

Following (Pascoe *et al.* 2024b), a separate value is derived for each objective row (e.g., $a_i = 1$ to 9) given the importance level identified in **Error! Reference source not found.**, and the relative score between any two objectives, $a_{i,j}$, is derived from the difference between them, such that:

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

As with the traditional AHP, preferences are assumed to be symmetrical. The use of the difference, rather than nominal value, is to provide a relative preference similar to what is derived from the traditional AHP bivariate comparison. For example, a score of 9 for A and a score of 7 for B is equivalent to an element score of $a_{AB} = 3$ and $a_{BA} = 1/3$ in the traditional AHP bivariate comparison. If both are equal (irrespective of the value assigned to both), then the equivalent element score is $a_{BA} = 1$. The use of the difference also reduces the influence of the initial anchor score (Yang *et al.* 2012; Casper *et al.* 2020). For example, a pair of scores (9,7) and (5,3) will provide the same element score (Pascoe *et al.* 2024b).

Given these element scores, a comparison matrix (*A*) is derived, given by:

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

We use the geometric mean method (GMM) developed by Crawford and Williams (1985), which has been found to be less susceptible to the influence of 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 Geometric Mean Method (GMM) (Crawford and Williams 1985; Crawford 1987) is used to derive the weights for each attribute (ω_i), given by:

$$\omega_i = \left(\prod_{j=1}^{n} a_{i,j} \right)^{1/n} \Bigg/ \sum_i \left(\prod_{j=1}^{n} a_{i,j} \right)^{1/n} \qquad (3)$$

where *n* is the number of comparisons, and $\prod_{j=1}^{n} a_{i,j} = a_{i,1} \times a_{i,2} \times \dots \times a_{i,n}$ (i.e., the product of the elements of row *i*).

The level of inconsistency in the results is given by the Geometric Consistency Index (GCI) is given by:

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

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 to derive a

consistency ratio, CR. Values of CR≤0.1 are generally considered acceptable (Aguarón and Moreno-Jiménez 2003). Given this, the critical values for GCI giving a CR=0.1 are GCI=0.315 for n=3, GCI=0.353 when n=4 and GCI=0.370 for n>4 (Aguarón and Moreno-Jiménez 2003).

Results

Review of previous studies and frameworks

Globally, the number of marine protected areas (MPAs) has grown exponentially over the past decade in order to meet international environmental protection targets, for example protecting and conserving at least 10 percent of the marine and coastal regions under the Convention on Biological Diversity (Jentoft *et al.* 2011). Most of these protected areas allow some extraction of resources in which case they are designated as "partially protected areas" (PPAs) (Zupan *et al.* 2018). The International Union for Conservation of Nature (IUCN) defines seven categories of protection, ranging from strict nature reserve to protected areas with sustainable use of natural resources (Day *et al.* 2019).

Several previous reviews have been undertaken focusing on the ecological performance of fully or partially protected areas in the marine environment (Gaston *et al.* 2008; Sciberras *et al.* 2013; Zupan *et al.* 2018). These have largely been mixed in their assessment. For example, Gill *et al.* (2017) found that 71% of MPAs assessed demonstrated some level of resource improvement, with the results also highly variable (some large some minimal). Gaston *et al.* (2008) concluded that many MPAs were not large enough to maintain viable populations of key species of conservation interest, particularly those that occur at low densities, and hence their usefulness as a conservation tool was limited and dependent also on activities outside the MPA (e.g., habitat loss, poor catchment management etc.). Sciberras *et al.* (2013) found that ecological performance of MPAs and PPAs was variable, with stock rebuilding dependent on the degree of mobility of the species of interest (e.g., sessile animals benefited while stocks of highly mobile species did not).

The designation of MPAs and PPAs also has economic and social consequences. These may be positive (i.e., through increased tourism or improved stock conditions of key commercial species) or negative (e.g., the effects of displaced fishing activities on local communities). In many instances, the use of MPAs and PPAs has been considered a fisheries management tool, with the potential to achieve both fisheries and conservation objectives (Rice *et al.* 2012).

In this review, we focus primarily on the economic and social outcomes associated with PPAs and fisheries-related area and time closures. As the ecological and fisheries benefits generated by these fisheries-focused closures are inextricably linked, these are considered jointly for the analysis of these management measures. We treat these forms of spatial management separately as they have different objectives in terms of the types of benefits generated. The objectives of the PPAs largely focus on conservation benefits while allowing potential benefits also to the recreational and tourism sectors, with the largest costs falling on commercial fisheries. In contrast, fisheries-specific closures (spatial and temporal) are focused more specifically on generating benefits for commercial fisheries, or offsetting externalities that commercial fisheries produce.

Frameworks for assessing conservation and fisheries benefits

The study considered five frameworks for considering the conservation and fisheries benefits:

- The ecosystem services framework;
- The total economic valuation (TEV) framework;
- Multi-criteria decision analysis (MCDA);
- Ecosystem accounting; and
- Wellbeing frameworks and indexes.

All five approaches are anthropocentric; that is, all approaches` considered only flows of benefits to humans have value. The first three of these provide a potential decision-support framework by enabling comparison of management outcomes under different criteria. Ecosystem accounting is not a useful tool for decision support alone but is a formalised monitoring tool that allows changes in key values to be identified and may potentially entice a management response. The wellbeing provides a conceptual framework for examine social impacts, although deriving overall measures of wellbeing to allow comparisons between different management options is not a feature of the approach.

The frameworks are not necessarily mutually exclusive. For example, ecosystem services can provide the basis for economic valuation in the TEV and the ecosystem accounting frameworks, or when economic valuation is not considered appropriate be used as inputs into a MCDA. Similarly, MCDA approaches could be applied in a wellbeing framework to allow some comparisons to be made, although this is not true to the underlying philosophy of the framework itself

The different tools also have different strengths and weaknesses. Some ecosystem services are difficult (some say inappropriate) to value in economic terms, limiting the potential for the TEV framework as a decision support framework to some extent. In contrast, MCDA can readily accommodate measures in different units and potentially include a broader range of options. However, while MCDA can provide a means to identify a management option in terms of a best (relative) outcome under multiple objectives, it cannot provide an overall measure of economic value for comparison to management costs. In contrast, economic valuation approaches can be used to assess both benefits and costs in a common unit.

The above frameworks all include some measure of "value" consistent with the concept of "utility", although how this is measured and used varies across the different frameworks. This contrasts to ecological concepts of "values", which are often considered synonymous with physical features of the biological or geomorphological features of an area. While generally not quantified in a common unit, social "values" are often imbedded in concepts of human wellbeing. From a social perspective, "values" reflect the importance something holds to the individuals and communities associated with the value (e.g., fishers, fishing communities or conservationists) and, in doing so, guide their thinking, feeling, and behaviour to frame the pursuit of their well-being (Hicks *et al.* 2016). These have parallels with many economic values, particularly non-market values as discussed in subsequent parts of the report.

Ecosystem services

Determining the economic values for natural assets, like marine ecosystems and MPAs, involves (1) scoping the values for humans through ecosystem services, (2) identifying the type of value, and (3) quantifying these values using established techniques. This section outlines key concepts relevant to the discussion of these economic values associated with MPAs.

Ecosystem services are the contributions that ecosystems make to human well-being through goods and services. This framework offers a systematic and standardised approach to classify and value the different benefits associated to a natural capital asset such as MPAs. Critically, the tracing of benefits to human well-being allows a theoretical underpinning to estimate economic values.

Economic values are anthropocentric in that these values only manifest if a good or service provides a benefit to human well-being. Identification of the full scope of economic values for natural resources—like marine ecosystems—can be complex as it requires tracing the biophysical asset (and processes) to the resulting value.

Ecosystem services are classified into three broad categories under the Common International Classification of Ecosystem Services (CICES)framework[1]:

1. **Provisioning:** services that describe the material or energy outputs from ecosystems (e.g., commercial or subsistence fishing).
2. **Regulating:** services that the ecosystem provide by acting as regulators through biophysical processes (e.g., climate regulation, water quality management).
3. **Cultural:** non-material services that affect the physical and mental state of people (e.g., recreational activities).

There are several conceptual models for developing an ecosystem service framework (Potts *et al.* 2014; Potschin *et al.* 2016; Marcos *et al.* 2021), with each offering marginal differences in illustrating the flow of values. However, the prevailing best practice model for this framework is outlined in the CICES. The CICES framework utilises a cascade model to scope and classify the final ecosystem services[2] and account for policy responses to anthropogenic environmental stressors (Figure 3). This model demonstrates the relationships between the environment and social and economic systems such that attributes and changes in the environment will flow-on to the specific ecosystem services and corresponding benefits and values experienced by humans. Natural assets like marine ecosystems can produce multiple ecosystem services concurrently.

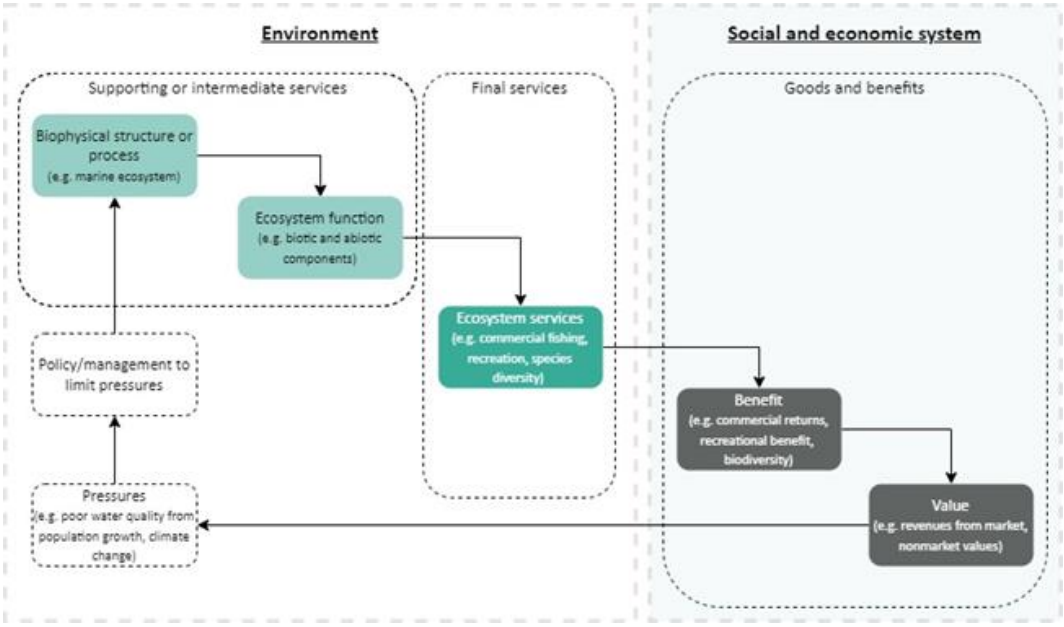


Figure 3. Cascade model for ecosystem services
Derived from Potschin *et al.* (2016)

The use of the cascade model is ideal in considering economic values from ecosystem services because economic value requires tracing the benefits to social wellbeing. Economic value is generated

[1] See https://cices.eu/resources/ for more information.
[2] CICES does not attempt to scope or categorise basic ecological or biophysical processes and services that underpin the final outputs from an ecosystem. These supporting services are considered intermediate and although critical, are not considered services under this framework.

through the supply and demand of goods and services where an economic benefit is an increase in social wellbeing and an economic cost is a loss in wellbeing (Potschin *et al.* 2016)

We can also draw parallels between ecosystem services and the "triple bottom line" of environmental, economic and social benefits (Figure 4). Ecological and economic benefits are largely captured in the provisioning and regulating services, while many cultural services can be considered related to economic and social benefits. The ecosystem services do not map one-to-one to the triple bottom line benefits, as the ecosystem services may capture elements of one or more of these benefits.

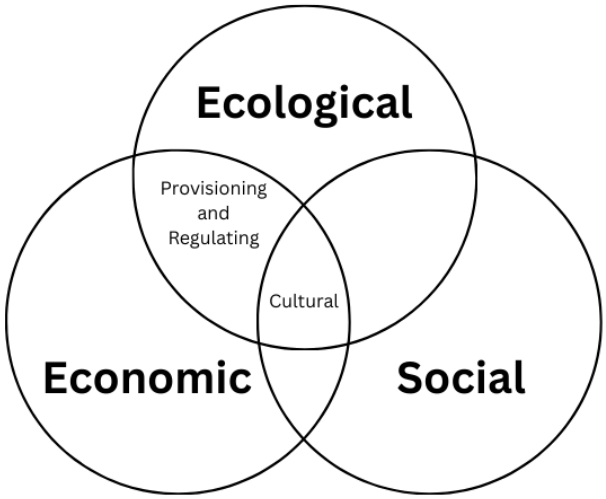


Figure 4. Ecosystem services and the "triple bottom line"

Total economic value

Economic values consist of use and non-use values—with the former relating to more tangible direct and indirect values, and the latter associated with benefits from the maintenance of ecosystem services. As a concept that has been widely researched and reproduced, several reports provide detailed descriptions of each component of the total economic value of an ecosystem service (e.g., Bateman *et al.* 2002; Tapsuwan *et al.* 2009). A schematic and brief descriptions of this widely utilised concept is given in Figure 5.

Economic valuation approaches have been developed to estimate these different values. The most commonly known approach to economic valuation is linked to expenditure on a good or service. Such approaches provide a very basic representation of economic value and may provide a substantial proportion of economic value for a specific ecosystem service. For example, the economic value from commercial fishing and aquaculture is often largely represented by revenue-based metrics of catch (e.g., gross value of production or GVP). However, expenditure alone may not always provide a useful illustration of economic value given it only captures the direct use value of an ecosystem service. Furthermore, the economic values for many ecosystem services are not explicitly traded in markets rendering a dependence on expenditure-based approaches insufficient to provide an understanding of the total economic value from an ecosystem service (Coglan *et al.* 2020).

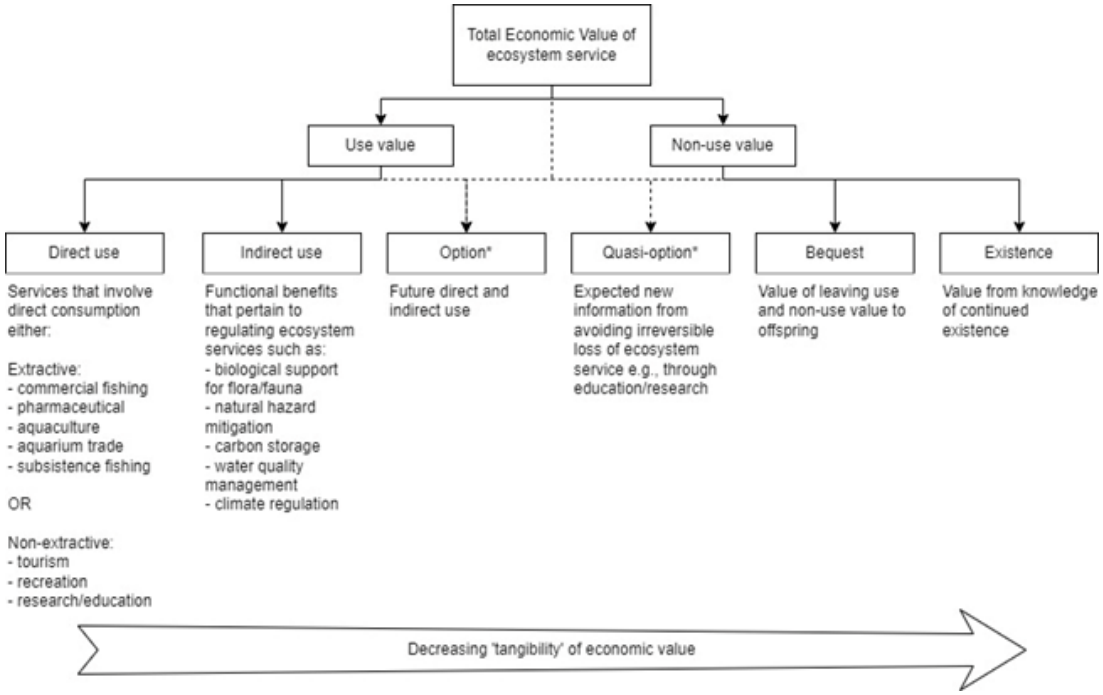


Figure 5. Total economic value of marine ecosystem services

Derived from Scholz et al. (2001)

Various approaches have been developed to quantify economic values. Economic valuation approaches generally fall within two categories: revealed preference and stated preference methods. The former is based on observable behaviour in existing markets, while the latter utilises consumer survey-based techniques to elicit values through hypothetical markets (Figure 6). Where there are constraints (time or resource) in undertaking a dedicated revealed or stated preference study into an economic value, a benefit transfer approach could be applied (Johnston *et al.* 2015). This approach involves transferring values from an existing study, in a comparable context, to a new study. Several matters need to be considered in order to perform benefit transfer robustly (Morrison 2001).

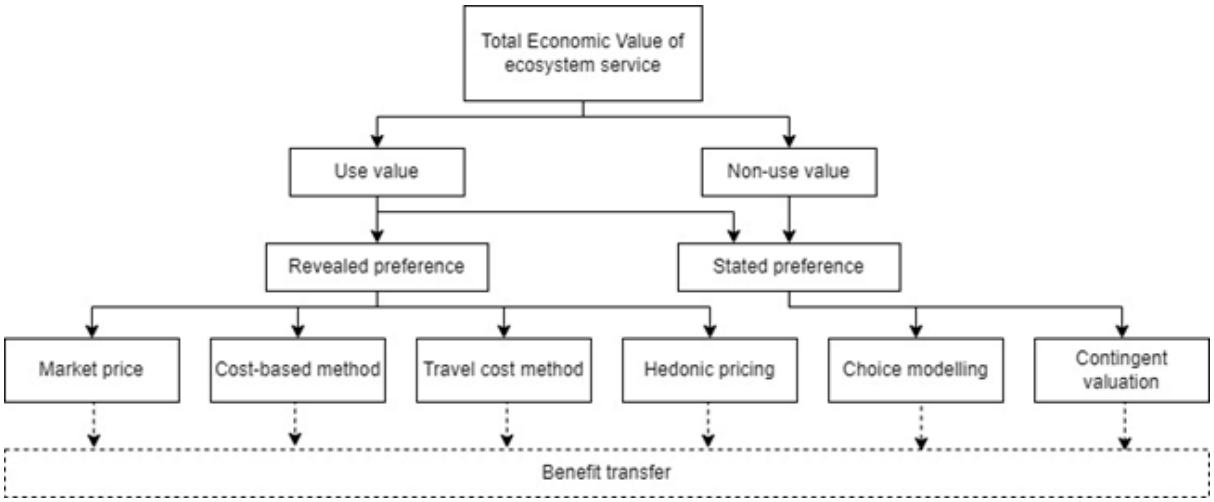


Figure 6. Economic valuation methods

Derived from Bateman *et al.* (2002)

A summary of the key economic valuation methods to estimate use and non-use values from ecosystem services is provided in Table 3. This includes the underlying basis, examples of uses and corresponding strengths and weaknesses of each approach. A common (potential) weakness of all

methods is that they generally only capture value for a specific point in time, not accounting for temporal or condition changes, unless repeated valuation studies are undertaken (or a time series of data are collected).

Each approach is limited in its scope and typically focused on one component of the total economic value for an ecosystem service and thus, using a single approach for each service would likely underestimate the total economic value. It is not uncommon for a variety of values and approaches to be utilised to provide a complete understanding of the total economic value for ecosystem services. These values can be aggregated as the total economic value from an ecosystem service or asset, although consideration should be given to avoid overestimation through double counting.

Estimation of economic use and non-use values is different to estimating the contribution a sector makes to the local economy. Estimating the contribution of e.g., recreational fishing on the ecosystem service captures the direct expenditure on ecosystem services and the indirect and flow-on benefits for the local and wider economies. Whilst this is useful information for policy makers to estimate impacts of change on a local community, it is not a proxy for total economic value of the ecosystem service (Scheufele and Pascoe 2023).

Table 3. Summary of economic valuation approaches

Method	Based on...	Examples	Strengths	Weaknesses
Revealed preference methods—using observed behaviour in existing markets				
Market price	Market transactions	Market value of commercial fishing	• Well-defined representation of direct use value • Methodologies are easy to implement and data relatively easy to obtain	• Market data are available for limited set of goods and services • Impacted by seasonality and other price factors • Difficult to measure the value of larger scale changes that are likely to affect supply or demand
Cost-based methods	Market value as a proxy	Value of heat management through avoided cooling costs Value of coastal protection through avoided replacement costs from damage	• Adequate representation of physical protection values • Consistent with some industry approaches (e.g., insurance) • Moderately data/resource-intensive	• May not represent actual benefit from protection • Does not capture social values of service that may be above costs • Does not recognise that a cost is not a benefit
Travel cost method	Non-market values from analysis of cost/expenditure for visiting a site or participating in an activity	Value of recreational activities like recreational fishing	• Based on actual behaviour and thus, defined value of service • Approach is established and generally well-understood	• Added costs from data collection and user participation • Method requires specific expertise • Debate on assumptions used in methodology • May not reflect true value for all visitors (e.g., may be underestimated if travel expenditure is low)
Hedonic pricing	Non-market values from analysis of market prices for differentiated goods, based on characteristics of corresponding services	Amenity values based on residential prices	• Based on actual behaviour and thus, defined value of service • Some market values (e.g., property values) are efficient in responding to information on value of services • The method is versatile and can be adapted to consider several possible interactions between market goods and environmental quality • Data is typically reliable and relatively easy to obtain	• Scope for services is limited to related market goods • Only captures value for perceived benefits and not actual benefits/consequences • Method requires specific expertise • Resource intensive and costly

Table3 continued. Summary of economic valuation approaches

Method	Based on...	Examples	Strengths	Weaknesses
Stated Preference Methods—using survey-based methods in hypothetical markets				
Choice modelling	Consumer values for changes to attributes of an environmental asset	Value of biodiversity/species within a natural asset	• Can be used to estimate use and non-use values • Considers individual socio-demographics and preference factors in values • Allow estimation of hypothetical or ex-ante changes	• Survey-design and analysis is resource intensive, costly and requires specific expertise • High risk of biases including hypothetical bias • Estimation of non-use values may not be robust if the survey not properly conducted
Contingent valuation	Consumer values for changes to an environmental asset	Value of changes to biodiversity within a natural asset		
Other techniques				
Benefit transfer	Corresponding values estimated in similar locations using other techniques	Transfer of use values for a natural asset or non-market values for changes to attributes of natural asset	• Less resource intensive and costly • Requires less expertise to implement • Can provide a basis to suggest primary study	• May not provide accurate reflection of values, given original studies tend to be policy, user or site-specific • Reference study may be difficult to find • Relevant studies for the policy/issue in question may not be available

Multicriteria decision analysis

Multi-criteria decision analysis (MCDA) is a means of aggregating outcomes over multiple objectives to provide a ranking of management options given the potential trade-offs in outcomes under each. These outcomes, for example, may relate to ecological objectives such as reduced bycatch or habitat damage, economic objectives such as improved commercial fisheries profitability, or social objectives such as improved regional economies. The method can also accommodate ecosystem services as previously defined, as well as other measures that may fall outside the normal scope of ecosystem services (e.g., local economic development).

An advantage of the approach is that it may include different stakeholder preferences for different outcomes of management (Dowling *et al.* 2020a). MCDA approaches are based on the link between preferences for an ecosystem service and the utility it provides. The link between preferences and utility have been well defined in economic theory (Arrow 1963), and the relative preferences for different outcomes is analogous to their relative price in a formal benefit cost analysis.

The use of MCDA to support management decisions has been well established in resource and environmental management (e.g., Kiker *et al.* 2005; Huang *et al.* 2011; Martínez-García *et al.* 2018), and has been used to assess management options and harvest strategies in both commercial fisheries (e.g., Mardle and Pascoe 1999; Leung 2006; Andalecio 2010; Stewart *et al.* 2010; Dichmont *et al.* 2013b; Williams *et al.* 2018; Elvarsson *et al.* 2020) and recreational fishing (e.g., Gao and Hailu 2012; Gao and Hailu 2013; Williams *et al.* 2018). Fisheries offer an ideal testing ground for the approach as they are characterised by multiple objectives and multiple stakeholder groups, and the development of harvest strategies requires assessment of trade-offs between these objectives and groups.

A variety of MCDA methods exist, none of which can claim to be superior to others in all circumstances (Guitouni and Martel 1998). However, each has a similar underlying approach requiring several stages. First, in the context of natural resource management, the objectives that the management option aims to achieve need to be determined, largely through stakeholder engagement, and the relative importance of achieving each of these objectives to different stakeholder groups or decision makers need to be assessed. This can be through a variety of approaches such as the Analytic Hierarchy Process (AHP) (Saaty 1980), ranking-based approaches (e.g., Alfares and Duffuaa 2008), point allocation (e.g., Doyle *et al.* 1997) or the use of choice experiments (e.g., Wattage *et al.* 2005). Next, outcomes of different management options need to be assessed and combined with the importance weights to determine the option that performs "best" across all the objectives, allowing for potential trade-offs between objectives. This may be undertaken quantitatively using a model (e.g., Dichmont *et al.* 2020), qualitatively using expert opinion (e.g., Pascoe *et al.* 2019a), or a combination of the two. Given that the importance weights reflect relative utility of each criteria, the weighted overall outcome reflects the relative utility of each option; an approach with the generic label of multi-attribute utility theory (MAUT) (Dyer 2016). There are a range of other MCDA approaches and tools which largely differ in how they treat the outcomes being combined or measured (e.g., Olson 2001; Gao and Hailu 2013; Mustajoki and Marttunen 2017). These all generally require substantial stakeholder interactions to determine preference structures.

More recently, the use of a data driven approach, data envelopment analysis (DEA), has been considered as an alternative to the commonly applied MCDA approaches to assess alternative marine resource management options (Pascoe *et al.* 2023). DEA derives from the assessment of productivity in an industry and is more broadly used to estimate technical efficiency and capacity utilisation of individual decision-making units (DMUs). Efficiency in this context is the extent to which a set of outputs is being maximised given a set of inputs (an output-oriented approach), or input use is minimised to achieve a given level of output (an input-oriented approach). This has been extended in an environmental context, for example to assess efficiency of countries in protecting biodiversity (Halkos

and Tzeremes 2010) and improving water quality (Macpherson *et al.* 2013), where outputs are expressed in terms of biodiversity or water quality indexes respectively.

As with "traditional" MCDA, DEA can be used to rank different DMUs based on how well they perform (measured as their efficiency in maximising outputs or minimising inputs). From a decision support perspective, DMUs could be considered as different fisheries management options, while the outputs are the expected set of outcomes under each management option. While the similarities between DEA and MCDA have been long established (Belton 1992; Stewart 1996; André *et al.* 2010; Dyckhoff and Souren 2021), the use of DEA to rank different management options has been more sparsely applied. Applications of DEA to assess management alternatives have been undertaken in several industries, such as agriculture (André *et al.* 2010), energy (Galán-Martín *et al.* 2016), railways (Azadeh *et al.* 2008) and assessing research and development applications (Lee *et al.* 2009). However, applications of DEA in an environmental or natural resource management decision support context are rare (Dyckhoff and Souren 2021).

In the context of assessing marine resource management, and fisheries management options in particular, DEA has been applied to assess management options impacting commercial and recreational fisheries in the Gulf of Mexico, comparing economic consumer and producer surplus with stock status (Griffin and Woodward 2011), and in the Australian Southern and Eastern shark and finfish fishery, comparing economic profits, stock status, fleet size and discarding (the latter as an undesirable outcome) (Pascoe *et al.* 2020). Pascoe *et al.* (2023) also provides three case studies using DEA to assess management options, encompassing a wide range of ecological, economic and social objectives.

DEA has several advantages over traditional MCDA approaches. In the absence of stakeholder preference information (i.e., importance weights), DEA can provide decision makers with a means to assess the relative performance of different options even when outputs are measured in non-commensurate units. Many management outcomes are often measured in different units, for example biomass tonnage, fleet economic profits and employment levels. Radial DEA models (Charnes *et al.* 1978) in particular are unit invariant (Knox Lovell and Pastor 1995). That is, normalisation to a common scale is not necessary, allowing direct incorporation of different indicators in different objective outcomes. For example, one of the case studies demonstrated by Pascoe *et al.* (2023) involved measures that were orders of magnitude different, and representing very different outcomes (e.g., fishery biomass, catch rates and probabilities of not catching a fish). In contrast, traditional MCDA approaches require all data to be normalised and on a common scale. How this normalisation is done has been shown to influence the outcome in some cases (Steele *et al.* 2009), so avoiding the need to normalise the data is a substantial benefit of DEA over traditional MCDA.

Ecosystem accounting

The development of Ecosystem Accounts (EA) in Australia is an emerging field in terms of both research and application. As with establishing conservation targets in both the marine and terrestrial environments, Australia has undertaken an international commitment to develop a series of Ecosystem Accounts for key environmental assets.

In April 2018, the Australian Government and all states and territory governments agreed on a National Strategy and Action Plan to implement EA across Australia. This was to be based on the internationally agreed standard: the UN System of Environmental Economic Accounting—Ecosystem Accounting (SEEA EA) (Committee of Experts on Environmental-Economic Accounting 2021). The first national set of ecosystem accounts, including fisheries provisioning services, were produced in Australia in February 2025 (Australian Bureau of Statistics 2025).

The SEEA EA is a framework to capture changes in natural capital as a component of economic activity, and the contribution of ecosystem assets and services to standard measures of economic activity such as gross domestic product (GDP). It is itself an evolving framework for measuring and valuing the links between ecosystems, the services they provide, and human well-being for national planning (Bordt 2018). As with the above approaches (i.e., ecosystem services and TEV), the SEEA EA classifies ecosystem services into three categories: provisioning services, regulating and maintenance services, and cultural services.

The SEEA EA provides a framework to organise and aggregate information on ecosystem assets, ecosystem services and the associated benefits enjoyed through household consumption and sale. The "extent – condition (capacity) – services and benefits" framework (Figure 7, (United Nations Committee of Experts on Environmental-Economic Accounting (UNCEEA) 2017)) relates to both measuring and valuing (in both biophysical and monetary terms) the natural capital assets and the associated flows of services they produce.

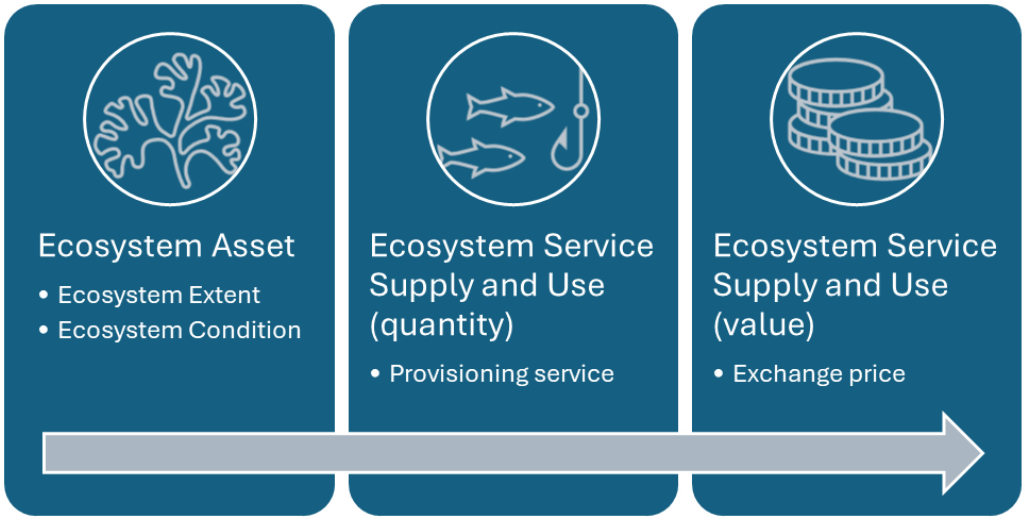


Figure 7. Broad steps in ecosystem accounting and the flow of ecosystem services

As with the previous approaches, marine environmental assets are considered to provide a wide range of ecosystem services that generate direct benefits to humans. For example, protected areas may provide nursery areas for the protection and development of juvenile fish species that later join the adult population and form part of the larger fish stock. Fish stock biomass then provides ecosystem services through the provision of inputs into the production of fish for sale, human consumption or recreation. Given this, nursery areas may be considered intermediate services, while 'fish available for harvesting' is a final service, as it is the latter that provides the service to humans (Luisetti *et al.* 2011). In this regard, the value of the intermediate services provided by nursery areas are captured within the value of the final services provided by the fish stock.

A simplified conceptual model of the relationship between coastal marine reserves, fish biomass and the ecosystem services they generate is illustrated in terms of fisheries related benefits in Figure 8 (Pascoe *et al.* 2019c). In this example, the reserve provides supporting services through providing nursery areas, refuge and (in some cases) feed for associated fish stocks. The fish stocks, in turn, provide fish that are available for harvesting (provisioning service), and fish available for recreational fishing (cultural service).

Supporting services are intermediate services, in that they do not directly provide benefits to humans. While it is possible to include these services in environmental accounts (Obst *et al.* (2016) provides an example of this), they need to be separated from final ecosystem services to avoid double counting.

Hence, intermediate services are generally excluded from ecosystem accounts (Haines-Young and Potschin 2011; Haines-Young and Potschin-Young 2018).

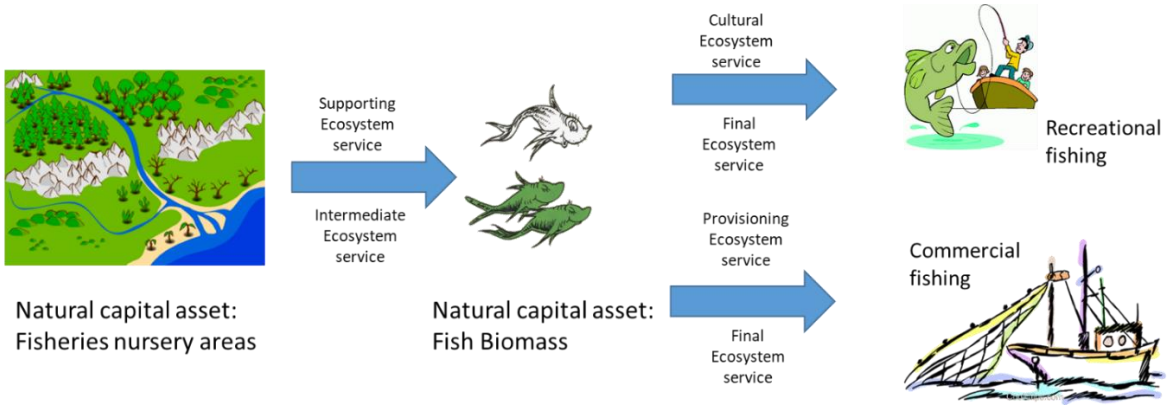


Figure 8. Conceptual relationship between fisheries nursery areas and final ecosystem services

The framework differs substantially from the ecosystem services and TEV approaches (and to some extent the MCDA) in terms of how economic values are applied. The TEV includes largely measure of economic welfare (i.e., consumer surplus) as well as measures of producer surplus. In contrast, the SEEA EA requires the use of exchange values, which reflect an equivalent market price if a market existed for the service. The exchange value represents the contribution of the ecosystem to production of the service. Consumer surplus is excluded. Further, non-use values are not captured in the main ecosystem account use and supply tables but can be recorded in complementary accounts.

In this context, ecosystem accounts relating to MPAs or PPAs would include only fisheries (recreational and commercial) and other recreational-based uses (e.g., scuba diving) in the main supply and use tables. Benefits relating to reduced bycatch, for example, or other non-use conservation-related outcomes are not considered in the main tables, could be recorded in a complementary account but do not formally form part of ecosystem accounts.[3]

A range of different approaches for estimating the exchange values of ecosystem services have been proposed:

- *Resource rent* represents the residual return attributable to the ecosystem service after all other normal returns to capital and labour have been deducted from the operating surplus. Estimates of resource rent only relate to the provision services provided by the fish biomass with the protected area. Measures of resource rent are applicable only for commercial activities, such as fishing, charter fishing or other marine related recreational or tourist activities (e.g., dive boats). Further, as resource rents can also be influenced by management structures (Obst *et al.* 2016), their use as measures of the value of ecosystem services are only appropriate in cases where market structures permit the resource rent to provide reasonable exchange values for the ecosystem service being considered (Hart *et al.* 2018).
- *Quota trading prices* (where available) provide a measure of the discounted future profits expected from harvesting a fisheries resource, while *quota lease prices* provide a short term measure of profits relating to a species within a fishery (Pascoe *et al.* 2019b). These may be considered market-based estimates of resource rent, and have the additional advantage in that they relate to an individual species rather than a fishery as a whole.

[3] Ocean accounts, however, are generally broader than ecosystem accounts, and are based on the SEEA central framework rather than ecosystem account framework. Ocean accounts may also capture non-use values as well as consumer surplus.

- *Production functions* relate the level of fishery output with the level of inputs, including the input of ecosystem services. As noted above, this has primarily been used to estimate changes in the physical quantities of ecosystem services (Barbier 2000), but in some studies this has been extended to consider their contribution to the overall value of the (commercial) fishery and derive the economic value of the natural capital assets (e.g., Taylor and Creighton 2018; Taylor *et al.* 2018). Other studies have developed bio-economic models linking ecosystem service levels to fishery rents to estimate the relative contribution of these services as a measure of their value (Smith and Crowder 2011). While production functions could potentially be estimated for recreational (cultural service) activities, they are more applicable to commercial fisheries.
- *Marginal values from revealed demand functions (i.e., travel cost models)*. Travel cost models have been applied to estimate the welfare value of recreational fishing in a wide range of cases. The contribution of ecosystem services has not generally been assessed using this approach, but some examples exist. While these primarily estimate welfare values, recent research has developed a means to extract exchange values from travel cost willingness-to-pay estimates (Scheufele and Pascoe 2023).
- *Marginal values derived from stated preference methods*. Stated preference approaches can be used to estimate the marginal value of a natural capital asset or the ecosystem service it produces.
- *Consumer Expenditure method*. Consumer expenditure as a measure for exchange value is potentially problematic. First, consumer expenditure is already accounted for in the System of National Accounts (SNA). Hence, the estimated exchange value would require a re-allocation within the accounting systems to avoid double counting. The SEEA EA states that human-made and human inputs need to be deducted to estimate the value-added generated by the ecosystem asset through the provision of ecosystem services. Since consumer expenditure is made-up entirely of human and human-made inputs, the value of the ecosystem service would effectively be zero. Second, interpreting consumer expenditure as a proxy for the exchange value, reflecting the amount people would be willing to pay additional to the actual consumer expenditure incurred, would imply the total value was double the observed consumer expenditure. Removing the input costs (captured in the SNA) would therefore not result in double counting given this assumption, as the "additional" consumer expenditure is interpreted as the exchange value of the ecosystem contributions. This interpretation assumes implicitly that if the ecosystem did not exist, these expenses would not be made. Economic theory would suggest, however, that – under this scenario – the quantity of trips demanded would be lower than those currently observed since the 'access fees' are additional to the travel cost – the exchange price is higher since it includes double the consumer expenditure. Therefore, even if consumer expenditure was an appropriate proxy for the exchange value of recreational benefits (over and above the travel expenses visitors must pay to get there and access the environmental asset), then not adjusting trip numbers to reflect the higher cost will overestimate the total exchange value to an unknown extent. The extent of this bias would depend on the price elasticity of demand.

Ecosystem economic accounts are primarily a tool to monitor changes in ecological and economic benefits produced by the ecosystem. In this regard, they are not as useful in terms of decision support as the other approaches identified above, which can readily be incorporated into a simulation environment. However, they can indicate the need to make changes if values decline and assess the success (or otherwise) of interventions through the change in ecosystem services generated.

Wellbeing frameworks and indexes

Well-being frameworks (WBF) and indexes have been developed to support the inclusion of social outcomes in decision-making (Coulthard 2012; United Nations 2012; Huppert 2014; Fudge *et al.* 2021). Importantly, WBF and indexes introduce a narrative of fairness, delivering conditions in which people and communities flourish and thrive as well as delivering sustainability for future generations. Key features of WBF and indexes are that they include objective and measurable indicators (e.g.,

education, income, housing status etc.) as well as individual stakeholders' subjective perceptions of wellbeing.

Several wellbeing frameworks and indexes have been developed which support social outcomes in decision-making processes. The thriving place index is a UK-based measure of liveability in a local government area using Office of National Statistics (ONS) data. The SEED (social, economic, environmental and democratic) wellness model (Crisp *et al.* 2023) is similarly UK based, considers several dimensions of wellbeing, including objective wellbeing (what people need) and subjective wellbeing (how people feel). The doughnut economic model (Raworth 2018) is a framework in which economic activity is constrained by ecological sustainability, while aiming to achieve a series of social foundations (deriving from the sustainable development goals reflecting minimum standards of living). The ONS wellbeing dashboard provides a number of indicators of wellbeing in the UK on an individual, community and national level, while the OECD Better Life Index measures 11 areas identified as essential across material living conditions and quality of life. Zeidler *et al.* (2022) identified a set of 29 individual indicators that reflect each of the key themes across all of these frameworks. Each framework was considered to have a different emphasis as to what constitutes the components of wellbeing, as represented by the number of indicators associated with each key theme (Figure 9) (Zeidler *et al.* 2022).

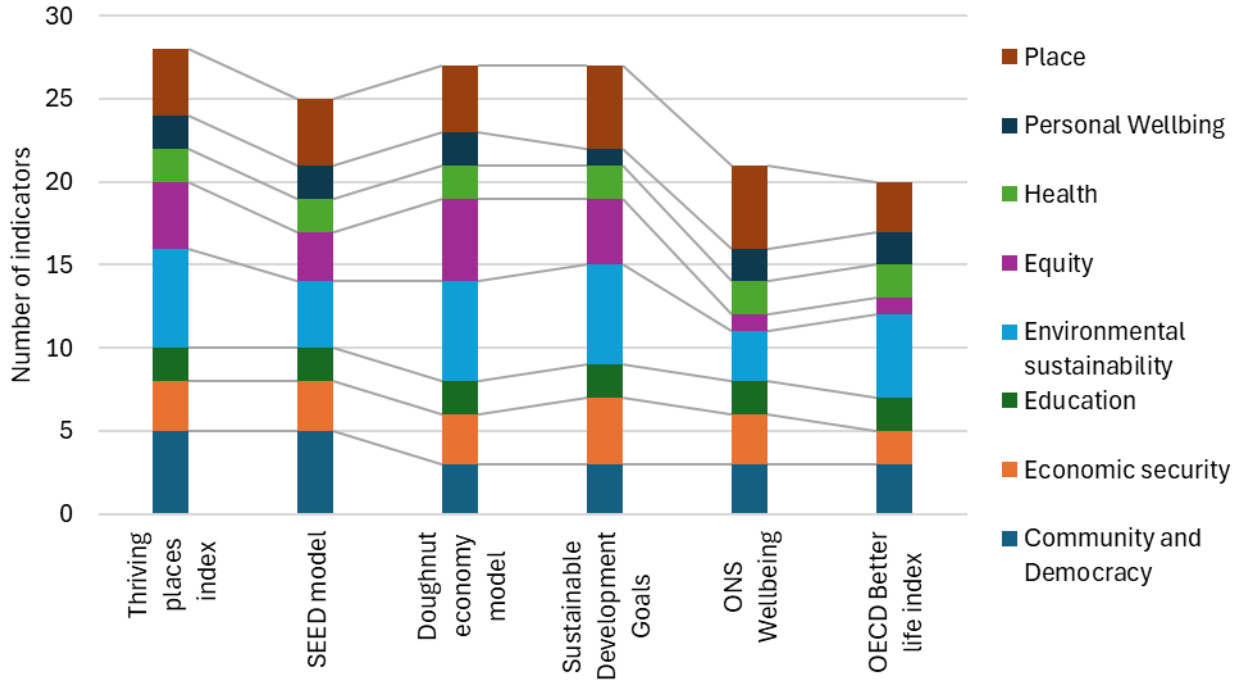


Figure 9. Components of key wellbeing frameworks

A fisheries-specific framework has also been developed to assess performance of fisheries using a range of indicators aligned with the triple bottom line (Anderson *et al.* 2015). These have been applied to a wide range of fisheries internationally to enable outcomes to be compared across different management systems (Asche *et al.* 2018). The framework, however, has not been applied in the context of MPAs or PPAs.

There are also well other established global wellbeing indexes. For example, the UN Human Development index, is derived from objective measures of wellbeing, including life expectancy, schooling and gross national income (a proxy for increase in living standards). In contrast, the World happiness Rankings are derived from the Cantril Ladder, a subjective measure of an individual's wellbeing (perceptions). Both these wellbeing measures enable countries to track their wellbeing over

time, as well as providing longitudinal benchmarking against other nations. National WBF or indexes are also being developed. For example, the Australian Government (Treasury and Australian Bureau of Statistics) have developed the Measuring What Matters Framework (MWMF) (Treasury 2023; ABS n.d.). The MWMF tracks wellbeing outcomes over time against five wellbeing dimensions, drawing on 50 indicators including a self-reported wellbeing indicator. In the same way as the government national economic accounts and macroeconomic metrics (e.g., GDP) provide a data picture of a nation's economic performance and economic prosperity, the MWNF extends the "data" picture to include health, security, sustainability and cohesion (Treasury 2023) to inform future policy decision making.

Coastal and marine environments impact human wellbeing through the provision of recreational opportunities, aesthetics and other factors (Russell *et al*. 2013; Spalding and Biedenweg 2017). For First Nations People, wellbeing is also derived through enduring cultural connections (Head *et al*. 2004). The impact of MPAs on wellbeing has been examined across domains of health, social and environment.

Impacts of MPAs and PPAs: review of empirical international studies

Ecological impacts associated with MPAs and PPAs

While the focus of this study is on the social and economic impacts of PPAs and MPAs, these are usually conditional on ecological objectives also being achieved. Several global meta-analyses have been undertaken exploring the ecological effects of MPA establishment, most of these on fully protected areas, including Gill et al. (2017), Molloy et al. (2009), Zhao et al. (2020) and Ban et al. (2017). Impacts or effectiveness of MPAs are traditionally measured using specific biological objectives or indicators, namely increased species richness (biodiversity proxy) or increased total biomass (Yates et al., 2019). Therefore, ecological impact is most often studied in terms of measurable impact on the density and recovery of commercial fish species. Less frequently ecological impacts are measured in terms of rare species (Pryor et al., 2020), habitat restoration/conservation (Selig & Bruno, 2010; Strain et al., 2019) and climate resilience (Bates et al., 2019).

In terms of MPA attributes that produce positive ecological impacts, it is widely accepted that older and larger MPAs generally have more of an impact on biodiversity and total fish biomass (Ban et al., 2017; Edgar et al., 2014). Babcock et al (2010) found that the time to detect direct effects on target species in an MPA was 5.13 ± 1.9 years, while indirect effects through trophic interactions were detectable only after 13.1 ± 2.0 years (Babcock et al., 2010). This was supported by results from Edgar and Barrett (2012) who, contrary to expectations, found that only two out of 11 exploited fish species in Australian MPAs showed signs of recovery over a three-year period. The same was true for invertebrate species which were expected to recover quicker compared to higher trophic level species (Edgar and Barrett 2012). The same study also posits that insignificant recoveries of exploited species may also have been due to low fishing pressure to begin with, highlighting a potential flaw in ecological success potentially leading to underestimation of the positive effect of MPAs if species recoveries are deemed 'insignificant'.

In a global meta-analysis on variables expected to impact fisheries and ecosystem health of an MPA, Ban et al. (2017) found that age of MPA, high environmental monitoring by users and high compliance were all drivers of increased ecosystem health. Enforcement and compliance strongly affect the success of ecological objectives. Gill et al (2017) determined that staff and budget capacities were the strongest predictors of conservation impact (measured in terms of fish populations)(Gill et al., 2017). Where MPA jurisdictions reported adequate staff capacity, the fish biomass response ratios were over 2.9 times greater. Strain et al. (2019) found evidence for indirect ecological benefits of old, well-enforced MPAs, showing that these management attributes, subsequent increases in herbivorous fish biomass and a decrease in algal cover all led to greater recovery of coral reefs (Strain et al., 2019).

The level of protection level has varying effects on the ecological impacts of a marine protected area, although generally the greater the level of protection the greater the ecological benefits. The level of these effects may depend on the type of ecosystem, the reserve design and other drivers and stressors on the marine environment (Clemente *et al.* 2011; Emslie *et al.* 2015; Pryor *et al.* 2020; Pereira *et al.* 2022). As expected, partially protected areas offer some conservation benefits but at a significantly lower magnitude than 'no-take' areas (Sciberras *et al.* 2013; Sala and Giakoumi 2018). Where fully-protected areas are infeasible, exclusion of certain activities may provide positive ecological impacts (Turnbull et al., 2018). A study in the Solitary Islands Marine Park, on the Eastern coast of Australia found that the exclusion of trawling had the most significant impact on increasing the abundance of fish taxa and increasing fish size (Pryor *et al.* 2020). A 2022 South American meta-analysis found that biodiversity increased by 45% in fully protected areas compared to open access areas, while partially-protected areas did not increase biodiversity compared to open access areas (Ferreira et al., 2022). The same study reported that partially protected areas required long times of protection (>15 years) and a high level of connectivity to produce significant impacts on biodiversity (Ferreira et al., 2022).

With the exception of connectivity, there has been little empirical work as to how the environmental attributes of an area contribute to positive ecological impacts in MPAs. Habitat connectivity is an ecological attribute widely accepted to contribute to the conservation success of MPAs (Balbar and Metaxas 2019). Connectivity refers to the extent to which spatially distinct ecosystems, communities, populations and habitats are linked. In their review of the inclusion of connectivity in MPA design, Balbar and Metaxas (2019) found that majority of the supporting literature for connectivity and connectivity in MPA design was from Australia.

There remains substantial debate over the merit of MPAs for ecological conservation. Beyond the practical challenges of achieving ecological objectives (namely budget and enforcement), a key limitation of MPAs to deliver ecological benefits is that MPAs do not address some of the biggest threats to marine ecosystems (Pendleton *et al.* 2018). A prime example is the repeated mass coral bleaching events in the Great Barrier Reef Marine Park due to climate change and ocean acidification. Bates *et al.* (2019) described a 'Protection Paradox' whereby MPA establishment protects from certain pressures (like fishing) but can select for species that are sensitive to other pressures such as warming (Bates *et al.* 2019). Increasing competition for coastal resources will have significant repercussions on MPAs due to, for example the destruction of associated ecosystems like mangroves and nursery grounds (Boersma and Parrish 1999; Meynecke *et al.* 2008). This is not to say that MPAs are not beneficial, only that MPAs alone cannot protect marine ecosystems from the effects of all human activity (Agardy *et al.* 2011; Hilborn 2018).

Economic impacts associated with MPAs and PPAs

For the purposes of assessing economic impacts of MPAs and PPAs, we have adopted an ecosystem - services approach to the definition of values and their impacts. We consider both MPAs and PPAs as part of a continuum of approaches to achieve conservation, economic and social benefits from the marine environment. Consistent with the TEV framework, we consider both use and non-use values. Most of the benefits and costs associated with commercial fisheries are captured by market prices and costs. Cultural services, including recreational activities such as recreational fishing, also generally requires non-market valuation approaches to elicit the non-market values associated with the activities.

Impacts on provisioning services

Commercial fishing

Fully protected areas have often shown long-term net economic benefits to fisheries as a result of enhances stock protection and the resultant spillover effects (Sala and Giakoumi 2017). However, fishers also often bear initial losses from the closure of fishing grounds (i.e., displaced fishing effort, forced gear diversification, increased travelling times, increased safety risk, and overcrowding) (van de Geer *et al.* 2013). PPAs are seen as a compromise solution that reduce the initial cost of environmental protection. The aim of a given PPA/MPA (e.g., enhancing specific target species or removing fishing pressure to reduce negative impacts on habitat) will determine the scale and nature of the benefits and costs (i.e., size of take/no-take zones, the section of the fishery industry affected, timescale for costs and benefits). For example, Pantzar *et al.* (2018) identified that in PPAs, fishermen using static gear to target low-mobility species were less likely to be considered incompatible with protected area objectives and therefore more likely to be allowed to continue fishing and benefit from on-site stock improvements. Conversely, fishers using mobile or bottom contact gear were more likely to be restricted within PPAs and their benefits expected to come from spill-over effects.

The economic value of fisheries is composed of the net benefit of producers and consumers (Davis *et al.* 2019). The effect of establishing a MPA on the benefits of producers is generally estimated as the difference in productivity and profits for the fishing industry with-and- without protection scenarios,

typically captured with data on CPUE (Goñi *et al.* 2011), and the net economic benefit (ABARES 2022). Similarly, the benefits for consumers (i.e., consumer surplus) will be represented by their willingness to pay (WTP) over the cost of a good or service and is estimated using a demand curve that plots the WTP for different amounts of a good. Market prices are used as an input to estimate the change in consumer surplus in response to changes in fish supply attributable to establishing a MPA (Davis *et al.* 2019).

Some limitations of using CPUE and market prices to estimate the benefits for consumers and producers include the potentially high levels of unreported or misreported catches, the difficulty to capture opportunity costs and depreciation, and the limited ability to reflect adjustments in behaviour and prices as a response to changes in fish supply and regulations (van de Geer *et al.* 2013; Davis *et al.* 2019).

The effects of MPAs on fisheries extend across the supply chain and related industries, such as seafood processors, equipment suppliers, retailers, consumers, and the community (often represented by the government) that benefits from job creation, taxes, and payment of fishing licenses. Although quantifying these flow-on effects may capture the interconnection between industries and prevent double counting, it requires multimarket models for the whole economy, which adds to analytical difficulties.

Bycatch and fishery waste

An often-unvalued disutility (from a non-market economic perspective) of commercial fishing is the non-market costs and option value of the incidental catch (i.e., bycatch) and discard of edible and commercially valuable fish due to their small size, quality, lower comparative market value, or regulations (Pascoe 1997; Pascoe *et al.* 2018). Some of the economic impacts of discarding include the forgone income associated with the future yield, environmental cost of discarding non-commercial species, and the costs of bycatch monitoring programs (e.g., observers, research vessels, logbook recording, electronic monitoring) and bycatch reduction engineering programs. In the USA, Patrick and Benaka (2013) estimated that bycatch discards represented a loss of approximately $4.2 billion in potential sales for commercial fisheries. During the last decade, Australia's commercial fisheries are estimated to have annually discarded approximately 37.6% of total captures (92,368 t) (Kennelly 2020). The Australian Government has recently announced the allocation of $3.9 million dollars to the Threatened and Migratory Species Fisheries Bycatch Mitigation Program (DCCEEW 2022). At the producer's level, part of the discarding costs is expected to be captured by fishing effort data and landed fish prices (as time spent at sorting and discarding is likely partially factored into fisher's costs); however, the real option value reflecting the possibility of more valuable uses in the future is rarely incorporated into fisheries policymaking.

The impact of MPAs and PPAs on discarding is mixed. Suuronen *et al.* (2010) found that effort displaced from an MPA to other areas resulted in greater catches of small fish that were subsequently discarded. Vilas *et al.* (2020) found both catch and discards increased in PPAs due to the spillover from adjacent FPAs. Di Lorenzo *et al.* (2022) found bycatches of elasmobranchs were higher in PPAs that unprotected areas, although this may reflect the differences in habitats within the different areas. These studies, however, pre-suppose that an objective of the MPA and PPA was to reduce bycatch and discards, which it may not have been. Fisheries related closures (discussed in subsequent sections) address this issue directly.

Non-use value of commercial species

Several studies have estimated the existence and option value attributed by the general population to the protection of commercial species over and above their provisioning service (Lew 2015). For example, Ojea and Loureiro (2010) used Contingent Valuation to quantify the increase in local welfare due to stock recovery levels for two commercial species. They estimated a median WTP for recovery

action of US$22.96 and US$35.63 for Norwegian lobster and European hake, respectively. A similar methodology was used by Bell *et al.* (2003) in Oregon to estimate the WTP of coastal residents for local coho salmon enhancement programs. Using a more general approach to assess the importance of the fisheries provisioning service for tourists and residents in two PPA, Christie *et al.* (2015) determined an annual willingness to pay for sustainable catch of desirable large fish in the Caribbean between US$0.23 and US$8.8 per household, with visitors willing to pay more in more degraded areas.

Subsistence fishing

Declining fish catches due to environmental degradation disproportionately affect coastal subsistence and small-scale fishing communities, with around 94% of the world's fishers belonging to those groups. Internationally, these fisheries not only produce about half of the fish used for human consumption, but also play a key food-security role in coastal communities, in addition to having a high representation of low socioeconomic and vulnerable groups (Busilacchi *et al.* 2013; Arthur *et al.* 2022).

In Australia, subsistence fishing is thought to play in the hybrid economy of Australian indigenous communities (Busilacchi *et al.* 2013). Several studies have applied a "replacement cost" approach using market prices to estimate the value of subsistence fisheries to indigenous communities (Gray and Altman 2006; Buchanan *et al.* 2009; Sangha *et al.* 2019). These approaches are controversial, and the distinction between subsistence fishing (a provisioning service) and traditional/cultural fishing (a cultural service) is often unclear. For example, Stoeckl *et al.* (2021) suggest that increased "connectedness" with the land through fishing on-country provides greater value to first-nations people than when fishing elsewhere.

Subsistence fishing related to the recreational sector is also complex, and the distinction between the activities can be fuzzy (Nyboer *et al.* 2022; Nguyen *et al.* 2025). "Subsistence" implies that the fish are necessary as a food source, whereas in recreational fishing the retained catch can usually be considered an intermediate service related to the recreational fishing activity (a cultural ecosystem service). In Australia, it is unlikely that consumed recreational catch constitutes subsistence, although the case may differ in other countries. For example, Nyboer *et al.* (2022) suggests that recreationally harvested fish species can be a hidden contributor to food security in some populations, and can contribute subsistence benefits to these communities. Recreational and subsistence fisheries, however, differ by the dependence on the activity to secure personal essential nutritional benefits through fishing products (Arlinghaus *et al.* 2021).

Although subsistence fisheries are one of the sectors that benefit most from the implementation of MPAs in the long term (FAO, 2021), they are also one of the most affected in the short term due to their limited capacity to displace or adapt fishing effort and resources. However, as traditional low-impact subsistence fishing is less likely to be in conflict with conservation objectives, this activity is not only often permitted within the boundaries of PPAs but can also benefit from reduction in competition as a result of restrictions to other fisheries (Kenchington *et al.* 2003).

Regulating services

Pest and disease management

MPAs can reduce the effects of invasive species by preventing the introduction of non-native marine plants and animals due to less maritime traffic and human use and by restoring/protecting top-down natural regulation processes (Ardura *et al.* 2016). Multiple examples worldwide have demonstrated the damaging ecological impacts that invasive species can impose on a system and their consequent economic cost to the fishing and tourism industry. Although the impact of marine invasive species in Australia has not been quantified, the cost of lost production, management, and clean-up is likely to be in the multi-million dollar range (Robinson and Cully 2022). Recognizing this, the Australian

government has designated $5 million to review and improve the national biosecurity system as part of its Marine Master Plan 2018–2023 (Department of Agriculture and Water Resources 2018).

Approaches to estimate the value of reducing the effects of invasive species can be taken from costs of prevention for private (e.g., measures taken by ship owners, recreational vessels owners with the aim of reducing the risk of transporting invasive species both in ballast water and as antifouling) and public initiatives (i.e., government expenditure in prevention, control/eradication, inspection programs, communication campaigns, and research). For example, Robinson and Cully (2022) determined that the cost of controlling an outbreak of black-striped mussels in Darwin was $2 million.

Climate mitigation (Blue carbon)

Marine habitats such as mangroves, kelp forests, tidal marshes, and seagrass beds sequester "blue carbon" in their vegetative structure and in sediments. By protecting these habitats, MPAs prevent the release of carbon into the atmosphere and ocean and its subsequent contribution to global climate change (Howard *et al.* 2017; Marcos *et al.* 2021). Also, given the direct relationship between carbon storage and biomass, MPAs contribute to additional sequestration through their positive effect on plankton and fish biomass (Roberts *et al.* 2017b; Hutto *et al.* 2021; Jacquemont *et al.* 2022). Pendleton *et al.* (2012) estimated that the carbon dioxide released due to the annual loss of marshes, mangroves and seagrasses worldwide might have a cost of US$6-42 billion.

Estimating the value of the additional sequestered carbon due to the establishment of MPAs has been based on the capacity of habitats/organisms to capture carbon multiplied by scale (e.g., additional hectares/kg of mangroves, increments in fish biomass) and the current market price of carbon (Murray *et al.* 2011; Zarate-Barrera and Maldonado 2015; Hutto *et al.* 2021). For example, Zarate-Barrera and Maldonado (2015) estimated that the annual economic benefits from carbon sequestration in mangroves and seagrasses after establishing a new network of marine protected areas in Colombia would range between 16 to 33 million Euros.

Although the use of the market price of carbon has been the preferred option to value the contribution of MPAs to carbon sequestration, some authors have opted for non-market approaches. For example, (Tyllianakis 2022) applied a choice experiment to estimate a WTP between 11.4 to 17.3 Euros/person/year of regular users in Malta for the increase in carbon sequestration due to marine ecosystems protection.

Climate adaptation (Erosion and coastal areas integrity)

The benefits of MPAs for coastal integrity include the preservation of sediment profiles and benthic communities. MPAs can be a management tool for protecting the seafloor from human disturbance (e.g., trawling). By limiting homogenising processes, MPAs can prevent the negative effects of disturbance such as the reduction of interspecies interaction, reduced habitat structural complexity, changes in biochemical processes, and reduced post-disturbance resilience (Jacquemont *et al.* 2022). Additionally, coral, rocky reefs, kelp forest, mangroves, tidal marshes and seagrass have the ability to protect the coastline and coastal inhabitants from waves and sea level rise through sediment accretion and/or wave attenuation (Marine Protected Areas Federal Advisory Committee 2009; Fairchild *et al.* 2021; Morris *et al.* 2021; Jacquemont *et al.* 2022; Strain *et al.* 2022).

Valuations of this benefit have relied on estimating the avoided replacement/ repairing cost of properties and infrastructure potentially affected by sea level rise and coastal erosion or the avoided cost of building and maintaining artificial coastal infrastructure (e.g., seawalls, rock revetments, groynes) (Angulo-Valdés and Hatcher 2010; Strain *et al.* 2022).

Despite the available literature on the coastal protection derived from specific marine ecosystems, values vary significantly across locations and organisms' biomass or density (Barbier 2015), and therefore, the use of benefit transfer has been discouraged unless site/region-specific data is available (Gaylard *et al.* 2020). In Australia, recent literature has identified approaches to estimate the contribution of marine ecosystems to coastal protection specific to Australian systems. For example, Morris *et al.* (2021) produced a guideline to assess nature-based methods for reducing risk from coastal hazards, and Gaylard *et al.* (2020) provided a summary of published coastal wetland ecosystem service values in temperate Australia, showing considerable potential for saltmarsh and mangrove protection value.

Although the use of the cost of repairing or replacing infrastructure has been the preferred option to value coastal protection, some authors have opted for non-market approaches. For example, using contingent valuation, Christie *et al.* (2015), estimated the WTP of tourists and residents in two PPAs in the Caribbean to improve coastal protection (reduce risk of flooding) between US$5 and US$10.26, with visitors' willingness to pay inversely related to the level of preservation of ecosystems (pristine vs degraded).

Cultural services

Cultural services from marine ecosystems have been extensively researched in the literature, particularly relating to recreational use and tourism. The economic value of these services consists of the market value—from expenditure—and the non-market value—namely the satisfaction or utility above the expenditure. While the former is commonly used as a measure of the value of the service to the economy, the latter tends to provide a more accurate reflection of the value to wellbeing. This is particularly critical for cultural services where values (e.g., providing entertainment and fulfillment to individuals) is not readily represented in market good and services.

Recreational fishing

Recreational fishing is a key ecosystem service that is impacted by MPA zoning. These activities, like commercial fishing, involve extractive use of the ecosystem. Thus, varying levels of restriction tend to be employed to limit access for these activities in MPAs.

The contribution of recreational fishing to regional economies is often considered a key indicator of its value. This contribution is largely based on expenditure by fishers. The expenditure value for recreational fishing is reflected in the purchase of inputs for recreational fishing include:

- trip-related costs such as: fuel, bait, tackle, ice, accommodation, entry/guide fees, boat/equipment hire
- off-trip or capital costs such as: purchase of fishing vessel, equipment (e.g., rods, reels), clothing and safety gear, insurance and registration, fishing club membership.

The benefits from recreational fishing expenditure will directly accrue to the economy, particularly for local businesses who depend on the sector. Recreational fishing surveys have been regularly administered by corresponding departments, with the most recent estimates available provided in Table 4.

The expenditure on recreational fishing also creates indirect and flow-on benefits for local communities and economies through provision of goods and services such as equipment and bait shops, transport and boating services, and expenditure on recreational groups and clubs. Consequently, the recreational fishing expenditure and indirect values for existing MPAs have been largely a function of the proximity to major population centres.

Table 4. Recreational fishing expenditure by State

State	Annual contribution	Source
Queensland	$638 million	BDO (2021)
New South Wales	$1.93 billion	McIlgorm and Pepperell (2013)
Victoria	$3.10 billion	Ernst & Young (2020)
South Australia	$175 million	Deloitte Access Economics (2017b)
Northern Territory	$51.5 million	West *et al.* (2012)
Western Australia	$2.54 billion	McLeod and Lindner (2018)

The impact of MPA on the expenditure component of recreational fishing is highly dependent on the restrictions associated with the policy. For example, if the MPA is purposed to restrict all extractive and direct use (i.e., a no-take zone), the recreational fishing effort and expenditure—including flow-on values—will expectedly decrease. However, MPAs that permit recreational fishing can attract tourism and recreation, preserving and potentially increasing these monetary values (Gray *et al.* 2010). Martin *et al.* (2019) found that the introduction of net-free zones in Queensland resulted in improved catches, with fishers travelling greater distances to visit the area than before the zones were implemented.

Although commonly undertaken, contribution studies provide little information of relevance to marine resource managers (Scheufele and Pascoe 2022). Expenditure is influenced by a wide range of factors not related to the resource being managed, such as fuel prices, food prices and the types of accommodation available. Expenditure is also influenced by fisher incomes, particularly with respect to boat purchase or hire. As a key potential benefit of MPAs and PPAs is to increase fish stock size and hence catch rates. Recreational expenditure is largely independent of catch rate, while the value of the activity to fishers is likely to increase if catch rates increase.

The value of recreational fishing to the fishers is a non-market value, reflecting the benefits they receive over and above their actual costs of access (i.e., consumer surplus). The non-market values of recreational fishing have been extensively studied in the literature, with many case studies in Australia (Coglan *et al.* 2021). The valuation approaches have generally either been travel cost methods or stated preference approaches. While the former would yield average non-market values for direct use, the latter allows elicitation of non-use values in the form of willingness to pay (WTP) for improved access, catch rates or other attributes associated with the activity.

In Australia, travel cost studies for recreational fishing have been conducted in key MPA sites including the Great Barrier Reef and Moreton Bay Marine Parks. Prayaga *et al.* (2010) found recreational fishers have a significant value for access to fish in the Capricorn Coast of $167 per trip for an angler, which was aggregated to an annual value of $1.6 million. Pascoe *et al.* (2014) estimated an economic value for recreational fishing at the Moreton Bay Marine Park to shed light on the impact of rezoning—specifically an increase in protected area from 0.5% to 16%. They estimated values between $58–108 per trip for an individual, depending on assumptions for cost inputs; this would amount to an approximate total value of around $20 million annually. Critically, they suggest that improved catch rates from the rezoning would increase economic values by up to $2.5 million annually.

Outside of MPAs, recent recreational fishing studies estimate values between under $10 per trip (Raguragavan *et al.* 2013) and up to $572 per individual's trip (Rolfe and Dyack 2011) depending on location, target species and assumptions for estimation. The positive impacts of increase in catch rates were also investigated in studies outside of MPAs in Australia. For example, Raguragavan *et al.* (2013) utilised a variant of a travel cost model in a state-wide study of recreational fishing site choices in Western Australia; they found a doubling of catch rates would result in an increase in economic values between $23–31 per trip depending on target fish species.

Tourism and other recreation

The increase in marine life biodiversity and the reputation of an area (i.e., designation effect) resulting from the protection of ecosystems after the implementation of MPAs is a powerful attraction for a series of recreational activities such as diving, snorkelling, whale watching, boating (e.g., Roncin *et al.* 2008). In fact, despite the low proportion of protected marine ecosystems in the world, they often attract more visitors compared to non-protected areas. For example, in Central America, 50% of diving trips occur in protected areas, although these represent a smaller proportion (about 20%) of the reefs within the region (Burke 2004). Likewise, Lynch *et al.* (2013)reported an increase in seasonal dive visitation rates after the re-establishment of shark aggregations because of the implementation of a Network of MPAs in eastern Australia. In the GBR, Deloitte Access Economics (2017a) estimated that tourism is the most significant direct use in the region, contributing an average of $5.7 billion to the Australian economy per year.

Given the commercial nature of most tourism activities, the estimation of the benefits of implementing MPAs is mainly based on market prices of tourism services, number of visitors, cost of provision of services, and constraints imposed by governments. The benefit for consumers is usually quantified in terms of changes in consumer surplus, determined from demand curves based on the ratio of price and quantity purchased or, conversely, the marginal willingness to pay for different quantities of the service (Davis *et al.* 2019). Lastly, the benefits of implementing MPAs can also reach the community in general (in the form of the government) through revenues (e.g., taxes paid by tourists and operators, visitor fees, operating licenses) and avoided public expenditures (e.g., employment generation) compared to the counterfactual scenario of what would have happened if the marine reserve was not established (Gravestock *et al.* 2008; Roncin *et al.* 2008; Sala and Giakoumi 2017).

While benefits associated with commercial tourism services are generally based on market data, the use of non-market valuation methods (i.e., revealed and stated preferences) dominate the estimation of the benefits of ecosystem protection for non-extractive non-commercial recreational activities. The most commonly used revealed preference methods to quantify the benefits of MPAs include travel cost and hedonic pricing (Spanou *et al.* 2020; Banarsyadhimi *et al.* 2022). Although not an analysis technique per se, benefit transfer has also been widely used (Rolfe & Gregg 2012; ref). For example, Deloitte Access Economics (2017a) estimated that recreational activities in the GBR provide $170 million in annual direct benefits to recreational visitors using a benefit transfer approach. The same study also calculated the annual benefits to domestic tourists at $1.5 billion using a travel cost method. Spanou *et al.* (2020) used a hedonic pricing approach to identify a relationship between proximity to MPAs and higher property values in Scotland's west coast.

The value of biodiversity and species protection for tourism

Studies of benefits to tourism associated with the protection of species or ecosystems have shown the economic relevance of establishing MPAs. For example, annual expenditure estimates have determined that the benefits associated with shark-diving tourism amount to more than $25 million in Australia (Huveneers *et al.* 2017). As also noted above, increased protection of grey nurse sharks through MPAs along eastern Australia led to an increase in the number of shark-related dive visits (Lynch *et al.* 2013) Similar approaches have highlighted the business revenue associated to the protection of shark populations in Malaysia (Vianna *et al.* 2018), Indonesia (Mustika *et al.* 2020) and Maldives (Zimmerhackel *et al.* 2018; Zimmerhackel *et al.* 2019).

The estimation of biodiversity-related values associated with MPAs has been based mainly on estimating the value of charismatic species such as turtles, sharks, marine mammals or iconic ecosystems such as coral reefs (Davis *et al.* 2019). For example, Wattage et al. (2011) determined the willingness of the Irish public to pay a personal tax of €1 to €10 to protect deep-sea corals by banning trawling in coral areas. Stated preference techniques, specifically contingent behaviour, have been

used to identify divers' willingness to pay for fees to support the creation and enforcement of shark sanctuaries in Malaysia, which would generate over US$2 million per year (Vianna *et al.* 2018). In Australia, Kragt *et al.* (2009) found that a decrease in coral and fish biodiversity could imply as much as 80% fall in reef trips in the northern section of the GBR. Using a choice modelling approach, Rogers (2012) estimated the values associated with protecting iconic marine fauna (e.g., turtles, sharks and habitats (e.g., seagrasses, coral reefs) in Ningaloo and Capes Park, Western Australia. A similar approach was used by Rolfe and Windle (2010) in the GBR to value diversity of fish and habitat health (seagrass, coral reef).

When linked to specific management policies, stated preferences have also been used to assess preferences for how an MPA should be managed (Rogers 2013). Using Contingent Valuation, Hall *et al.* (2002) estimated the WTP US$6 per family visit to support policies that increase the effectiveness and enforcement of MPAs in Southern California.

First Nation People and fishing

Marine ecosystems in Australia provide significant values to Indigenous People. These cultural values may be derived from knowledge sharing, spiritual connection to biophysical attributes of the ecosystems and connected catchments, and the passing on of customs, tradition and stories (Sangha *et al.* 2017). Smyth *et al.* (2018) outlines a comprehensive list of social, cultural and economic values of Indigenous cultural fishing practices for communities in NSW, South Australia and Northern Territory.

Regardless of the improved understanding of Indigenous values for marine ecosystems and resources, there is understandably limited attempts at quantifying these values. Some argue that these values should not be reflected in monetary values (Daw *et al.* 2015), particularly where valuation indicators are entirely based on Western sciences and not in collaboration with respective communities. This is because typical frameworks, such as ecosystem service frameworks, may not fully capture the value of traditional practices. For example, the values attributed from cultural fishing practices extend beyond the provisioning services of subsistence fishing that would otherwise be used to associate these values (Plagányi *et al.* 2013; Smyth *et al.* 2018). Similarly, Altman and Branchut (2008) found that the cultural and economic values Aboriginal communities have for water could not be disentangled in a manner consistent with prevailing frameworks. However, not considering these values in developing management tool risks omission of a significant socio-cultural service in resource allocation and access.

Despite this, there have been some attempt to capture the value of Indigenous fishing using non-market valuation globally. Zander and Straton (2010) utilised a choice model to estimate the values Aboriginal fishers had for the quality of fishing sites. Similarly, Oleson *et al.* (2015) also employed a choice modelling approach to estimate the cultural values for Indigenous fishers in Madagascar. Buchanan *et al.* (2009), Jackson et al (2011), and Sangha *et al.* (2019) all utilised variants of a replacement cost approach to estimate the subsistence fishing values. Manero *et al.* (2022) provides best practice advice on how and when to apply non-market valuation techniques to indigenous values relating to cultural services and activities.

Knowledge of marine ecosystems

MPAs have been classified as observatories or reference areas for research (Baxter et al., 2016), as they provide a counterfactual against which the effects of fishing-related pressures can be assessed. The knowledge generated from their establishment can contribute to management design (i.e., MPAs can be designed more profitable based on previous information) and improvement in the understanding of fishing dynamics (Hartmann et al., 2007) which in turn can help optimize fisheries management. Proxy indicators to estimate the value of these services include research expenditure including the

costs (e.g., permits, scholarships, salaries) incurred by research institutions to conduct studies in MPAs and the flow on effects of this expenditure to the local economy. Expenditure provides a lower threshold estimate of the value of research, as the assumption is that the benefits of the knowledge created would exceed the costs of its production (Bronzes *et al.* 2025).

Fishers' satisfaction

The motivation to participate in commercial fisheries can go beyond providing a source of income (Anderson 1980). There are a series of multi-dimensional attributes associated with the satisfaction of fishers, including psychological (i.e., outdoor experience, challenge, adventure) and social factors (i.e., identity, status, pride, tradition, sense of belonging) (Holland *et al.* 2020). Moreover, fisher's satisfaction has been considered a measure of the social performance of fisheries management and an indicator of the likelihood of compliance with regulations (e.g., Brooks *et al.* 2015; Pascoe *et al.* 2015; Pita *et al.* 2020) and its inclusion in management assessments is highly regarded. However, although methodologically possible to derive a monetary value for satisfaction (i.e., travel cost, indirect utility function), this is rarely undertaken as its monetary contribution is usually assumed to be captured by other measures such as fishing income (Coglan *et al.* 2021).

Economic and ecological impacts of fisheries management specific closures

The previous section focused on values associated with fully or partially protected marine areas. These are largely implemented with conservation as a key objective, but are recognised as potentially generating wider economic and social values.

Spatial fishery closures are often implemented specifically to improve commercial or recreational fisheries through ensuring stock rebuilding or allowing the species to grow to a more profitable size, or to correct or partially compensate for negative externalities generated by fishing activities, such as bycatch of species with high conservation value.

For the purposes of this section, we differentiate between seasonal closures and area/time closures with the former involving closure of the whole fishery for parts of the year, with the latter involving closure only of certain areas within the year. We further differentiate time/area closures from spatial (area) closures as the former are often temporary (i.e., seasonal) while the latter are longer term (operating over several years or permanently).

Seasonal and area/time closures

Seasonal and time/area closures are common fisheries management mechanisms, well suited to address known ecological and biological characteristics of some fisheries (e.g., species-habitat relationships, juvenile/nursery area or spawning seasons). These characteristics also make them attractive as a method to reduce bycatch (Grantham *et al.* 2008; Dunn *et al.* 2011) as well as increase the level of biomass through protecting species at critical times of their life cycle, or allowing the species to grow to a larger size before first capture. Consequently, there are potentially clear ecological benefits in their use as a fisheries management technique.

Given that the economic and ecological impacts of seasonal closures are largely intertwined, these are considered together in the following section.

Economic and ecological impacts

With regard to bycatch, there is a trade-off between protection of these species and the reduction in the economic opportunity to the fishing fleet. The latter is largely dependent on the potential for the fleet to re-allocate its fishing effort, the costs involved in this reallocation and the differences in revenues obtained (O'Keefe *et al.* 2013). The impact on fishing profits are potentially highly variable

between fisheries and also timing of the closures. For example, Smith *et al.* (2020) found that the lost economic opportunity due to a turtle conservation time-area closure was small, ranging from 0-6% in terms of a reduction in average vessel profits. However, when the time-area closure coincided with fishing for highly migratory pelagic species, vessel profits were found to be reduced by 30-40% (Smith *et al.* 2020). As there is often interannual variation where and when different groups of bycatch species are caught, Grantham *et al.* (2008) found that temporary area/time closures were more effective than seasonal closures in terms of both protecting the bycatch species and minimising the cost to fishers.

The potential benefits of seasonal closures to reduce bycatch is just one option available. Gilman *et al.* (2006) found that encouraging better communication between fishers may result in similar levels of bycatch reduction without the need of seasonal closures (and any subsequent loss in revenue). Similarly, O'Keefe *et al.* (2013) found that improving communication and/or the use of bycatch catch limits (e.g., quotas) produced better results in terms of bycatch reduction at less cost to the industry than seasonal or area/time closures.

Dunn *et al.* (2011) note that the use of seasonal closures may, in fact, prevent greater economic hardship, particularly if they protect juveniles or spawning stocks and lead to higher biomass levels and higher future catches/profits. Similarly, if they prevent overexploitation of iconic bycatch species that would otherwise results in longer periods of full fishery closure, then any short-term losses may be more than offset than the longer-term benefits.

Where the closures are implemented as a means of improving future catches for the commercial fishery (i.e. protect spawning aggregations or nursery areas), their effectiveness may also vary. Samy-Kamal *et al.* (2015) suggested that the main benefit of the seasonal closure was the reduction in total effort and consequent reduction in fishing costs. Karras and Agar (2009), Agar *et al.* (2019) and Owusu and Andriesse (2020) all found that seasonal closures in small-scale coastal fisheries may result in a decline in catch and income without subsequent (apparent) economic benefits. Weninger and Waters (2003) found that seasonal closures resulted in periods of substantial gluts on the market (following the re-opening of the fishery), with adverse impacts on prices and revenues received, while the race to fish resulted in a substantial increase in costs.

Seasonal closures have also been applied in other fisheries to allow the size at first capture to increase for economic rather than stock related reasons. For example, in several prawn/shrimp fisheries, seasonal closures have been introduced to allow prawn biomass to grow to an optimal size (allowing for trade-offs between growth rate and natural mortality) as a means of increasing economic returns from the fishery and preventing growth overfishing (Õnal *et al.* 1991; Somers and Wang 1997; Caillouet *et al.* 2008). Similarly, fisher revenues doubled in the Madagascar octopus fishery following the introduction of a seasonal closure to allow greater growth before capture (Oliver *et al.* 2015).

The potential benefits generated by seasonal closures aimed at increasing the current biomass through growth also depend on other management measures. For example, Cheng and Townsend (1993) estimated that seasonal closures in the US Lobster Fishery had the potential to increase revenues by as much as 18%, but the race to fish following the opening of the (open access) fishery dissipated all the potential profits (rents) that this might also generate. Griffin *et al.* (1992) and Smith *et al.* (2008) found similar results for the seasonal closures introduced in the Gulf of Mexico shrimp and gag fishery, with the seasonal closure resulting in short term increases in economic profits, but a subsequent increase in fishing effort reducing these to zero. Weninger and Waters (2003) concluded that much of the problems associated with seasonal closures identified above might be avoided through the use of a rights-based system of management.

The effectiveness of seasonal closures in multispecies fisheries is more problematic. For example, China imposes a summer moratorium on fishing in several sea areas with the aim of protecting the stock of the key commercial species during their spawning period (Shen and Heino 2014). While this

was expected to result in higher future stocks, the moratorium has been criticised as i) the key times required to protect the different species while spawning varies, so that the general moratorium does not achieve the desired outcomes and ii) a race to fish occurs once the moratorium is finished, rapidly reducing the remaining stock and also increasing the costs of capture, reducing the potential rent that could be generated in both the short and long term (Shen and Heino 2014). Samy-Kamal *et al.* (2015) found that time-based closures had varying impacts at the species level in multi-species fisheries, mainly due to different seasonal patterns of availability and growth. For fast growing species, catch rates (kg/day) increased as a result of a larger proportion of larger animals. For slow growing species, there was little impact on catch rates. For the faster growing species, the higher catch rates resulted in a lower price (as more was provided to the market), offsetting some of the benefits of the higher catch rates. Birkenbach *et al.* (2020) suggest that multispecies fisheries with well-designed property rights may avoid the need to have seasonal closures entirely, as fishers could choose when to optimally harvest their quota of each species.

While the focus of this section has been on commercial fisheries, recreational fisheries also often are subject to seasonal closures. Chagaris *et al.* (2019) suggest that the use of seasonal closures applied to specific recreational species rather than an area may be counterproductive, as anglers may re-allocate their effort (fishing harder elsewhere) or continue fishing for other species and discard the species for which the closures is imposed. In such case, the behavioural response of the anglers may offset the desired conservation outcome. Similarly, Abbott *et al.* (2022) suggested that seasonal closures were ineffective in managing recreational fishers if fishers could reallocate their effort to other areas or species, or if effort is increased (concentrated) during the open season as individuals know they are unable to fish at other times.

Haase *et al.* (2022) found that recreational fishers supported the use of seasonal closures in the Baltic to protect spawning stocks, as they were proposed in months when fishing activity was generally low. In terms of reducing actual total catch levels, seasonal closures were found to be not as effective as bag limits.

Spatial (area) closures

Economic and ecological impacts

We differentiate between fishery-related impacts from areas with a primary conservation objective (i.e., Marine Protected Areas, or MPAs) and those areas that are closed with a fishery related objective (i.e., to reduce a negative externality or enhance future production).

Marine protected areas not aimed primarily at providing fishery benefits have often been promoted as having potential benefits through the "spill over" effect into the areas open to fishing (Angulo-Valdés and Hatcher 2010; Di Lorenzo *et al.* 2016). In some cases, they are also seen as a complementary fishery management tool (Sumaila and Charles 2002; Mangi and Austen 2008; Weigel *et al.* 2014). However, evidence of the magnitude of the "spill over" effect is mixed (Di Lorenzo *et al.* 2020), and Armstrong (2007) suggests that most economic analyses of MPA are substantially more pessimistic than ecological analyses. Lenihan *et al.* (2021) found a substantial spillover effect in the case of a commercial lobster fishery adjacent to a MPA. Vandeperre *et al.* (2011) found a gradual increase in catch rates of key fish species following the introduction of MPAs, but the net benefit to commercial fisheries decreased as the size of the MPA increased (i.e., smaller MPAs produced the greatest net benefit). Kaiser (2005) and Hannesson (2002) found no benefit of MPAs to the fisheries sector or to stock conservation in the case of highly migratory fish species.

Area closures (other than MPAs) are often implemented with the aim of enhancing productivity of the fishery in the future. For example, spatial (area) closures have been shown to result in increased biomass as a result of protecting juveniles (Die and Watson 1992), with subsequent economic benefits

in terms of higher catch rates and revenues in the longer term. While this has been found to occur in some fisheries (e.g., Somers 1990; Taylor and Johnson 2020), others (e.g., Dalton and Ralston 2004) have found that effort increases outside the closed areas more than offset the benefits of effort reductions in the closed areas, with consequent stock decline.

Area rotations have been applied as a fisheries-management measure in several scallop fisheries (Valderrama *et al.* 2007; Hart *et al.* 2020) and other fisheries where the target species is relatively sedentary such as sea cucumber (Purcell *et al.* 2013; Plagányi *et al.* 2015) or trochus (Purcell 2004). The key perceived advantage of rotational closures is that it enables the local stock to recover and build while fishers still have access to other parts of the fishery. The area is re-opened (and another closed) when the stock and individual animals are at an optimal size (Valderrama *et al.* 2007). Hart *et al.* (2020) found that the protected stock may also help support the stocks outside the closed area through larval spill over. Little *et al.* (2010) found that protecting part of the stock each year may also help a fishery sporadically subject to catastrophic events.

In the absence of other controls, however, rotational fisheries may become rapidly depleted once opened. The high relative abundance and sedentary nature of the animals results in low costs of harvest and the potential for overexploitation and local depletion (e.g., O'Neill *et al.* 2020).

Opening closed areas to some fishing activity after a period of time may also provide fishery benefits to some sectors of the fishery. For example, Whitmarsh *et al.* (2003) found opening a closed area to artisanal fishers resulted in an improvement in their catch rates and economic performance compared to when they were competing with larger vessels in the open regions.

Area closures have also been implemented to limit bycatch by avoiding areas and habitats where particular bycatch species are prevalent. Hazen *et al.* (2018) notes however that, as a mechanism to control bycatch, the use of fixed (static) area closures is inefficient as environmental factors, including climate change, may result in a shift in the habitat and hence distribution of the bycatch species. Instead, they suggest the use of more dynamic closures that can change with changing environmental conditions to support long-term economic sustainability in concert with conservation of marine biodiversity (Hazen *et al.* 2018). Similarly, Beare *et al.* (2013) found that static area closures were not successful in protecting specific trawl bycatch species due to environmental changes which resulted in a higher proportion of the stock occurring outside the closed area than expected.

Closed areas in themselves, however, will not necessarily protect bycatch species in general. For example, Stobutzki *et al.* (2003) found no difference in the bycatch species composition or abundance inside and outside closed areas in the Northern Prawn Fishery. While these areas were not specifically set up to protect bycatch species, an implicit assumption was that they would benefit these species also.

A key issue with the use of fixed area closures is the displacement of effort, and the impact that has on both economic and ecological outcomes. Smith and Wilen (2003) found that changes in economic incentives as a result of an area closure impacts both participation levels and location choices of fishers, and ignoring these in closed area design may result in overly optimistic expectations about their conservation value. Dichmont *et al.* (2013a) modelled the ecological and economic impact of different closed area scenarios with the multiple objectives of protecting threatened species and minimising impact on fisheries. They found that MPA style closures – focused on key habitat areas – were less effective from both an ecological and economic perspective in terms of species protection than closing a larger area of less-intensive fishing grounds. They found that effort displacement from the species/habitat focused closure resulted in lower catches of target commercial species but not lower catches of the target conservation species. Closing areas of lower fishing intensity instead resulted in a larger overall closed area but also less effort displacement, resulting in less catch of the target conservation species with a lower impact on the catch of commercial species. Similarly, Smith and

Wilen (2004) showed that the reallocation of effort following the introduction of area closures in the California Sea Urchin Fishery resulted in increased pressure in more inshore areas with adverse stock and economic consequences.

The displaced effort may also have unintended consequences due to changes in fisher behaviour. For example, Abbott and Haynie (2012) found that closing some areas to protect one species caught as bycatch may result in changes in fisher behaviour (in addition to effort displacement) resulting in increased pressures on another bycatch species. This change in fisher behaviour is in response to the change in economic incentives generated by the closure, which resulted in a different fishing strategy being more profitable in the spatially restricted fishery. Increased crowding in the available areas may also result in changes in behaviour, changing the spatial and temporal distribution of fishing effort within the fishery (Dalton and Ralston 2004).

The magnitude of any benefits derived from area closures are largely dependent on other management arrangements in place in the fishery (Smith *et al.* 2008). In Australia, marine area closures for conservation purposes are often accompanied by effort reduction programs such as buybacks in order to offset the potential adverse effects of effort reallocation (see e.g., Fletcher *et al.* 2015). Determining the contribution of the closure to fishery productivity, then is obfuscated by the effects of the total effort reduction.

Social impacts of PPAs and fishery closures

Studies on the impacts of PPAs and MPAs were dominated by the assessment of ecological and economic impacts, although some consideration to social impacts was also provided in some cases. In most cases, social and cultural services impacted by PPAs and MPAs were assumed to be explicitly capture as economic impacts, with the main focus was on deriving values relating to recreational use and tourism (Ban et al 2019), mostly using non-market valuation approaches. In ecological-focused studies, social implications of MPAs were often acknowledged but not articulated further or directly measured. These studies generally assumed that positive ecological or conservation impacts were correlated to, and resulted in, positive societal social outcomes (Rasheed 2020). That is, improving the environment leads to improved social outcomes. In contrast, other studies assumed the counter position. That is, MPA and PPA reduce social outcomes (Turnbull *et al.* 2021). In this section, we review the available studies that explicitly consider social impacts separately (or in conjunction with) ecological and economic impacts.

Whilst there is an establish literature on the categorisation of social impacts from conservation initiatives (Suich and Dawson 2023), the application to PPAs and MPAs in developed countries was not consistent. A number of systematic reviews of the research literature have attempted to identify social impacts of MPAs (e.g., Mascia *et al.* 2010; Ban *et al.* 2017; Ban *et al.* 2019; Rasheed 2020; Phillips *et al.* 2025). These studies generally do not differentiate between MPAs in developed and developing countries, instead aggregating results across both, with studies from developing counties, for the most part, dominating the analysis. For example, Mascia *et al.* (2010) reported that more than 73% of the studies for their systematic review were from developing countries. Similarly, approximately 70% of the reviewed studies reported in Ban *et al.* (2019) and Rasheed (2020) that were less than 10 years old (i.e., post 2015) were from developing countries. Whilst this does bias the results of these reviews towards the assessments of social impacts in developing countries, the consensus of all the systematic review studies was that a narrow subset of social impacts was investigated namely livelihood (income), leaving room for improvement in assessing the broader social impacts around wellbeing (including social, health and cultural). A second common theme across the studies was that the negative social impacts of MPAs were more keenly felt by individuals, in contrast to positive social impacts that were realised at a community level such as benefits to the region's economy or environment.

Ban *et al.* (2017) synthesised studies which analysed impacts across 12 large MPAs (i.e., >10,000 km^2). Of the MPAs that satisfied their inclusion criteria, over half were in developed countries, of which four were Australian (Great Barrier Reef Maine Park, Great Australian Blight Marine Park, Macquarie Island Marien Park and Heard Island and McDonald Island Marine Parks). They conclude that the previous studies were biased towards ecological aspects of MPAs, with lower levels of documentation for social data, consequently limiting analysis.

Only one systematic review focused on PPAs in Australia, considering the published results available for nine PPAs across ecological, economic and social benefits (Phillips *et al.* 2025). Consistent with the above results, it was concluded that measurement of social benefits across PPAs studied lacked consistency, making it difficult to draw any meaningful conclusions. They noted that there was no reported measurement of impacts for First Nation Australians and cultural goals.

A common theme across the studies reviewed was that positive social impacts are frequently felt at a community level involving benefits to the region's economy or the environment with the negative impacts being carried by individuals.

Livelihoods and wellbeing: commercial fishers

Livelihoods and wellbeing measures of social impacts that have been assessed within the literature draw on both objective measures (e.g., income) and a range of subjective measures. These subjective measures range from perceptions of quality of the environment, access to resources by the community as well as other composite measures or indexes. However, in the case of both objective and subjective measures (i) terminology is fluid, (ii) conclusions are drawn about impacts even when they are not explicitly measured and, (iii) even when measured, approaches focus on aspects of wellbeing of a subset of stakeholders (Rasheed 2020). This limits conclusions to more thematic illustrations of impacts.

The empirical research that has been undertaken does underline the complexity of estimating social impacts for sub-stakeholder measured against different social dimensions. For example, a longitudinal study (covering 12 years) of the Lyme Bay MPA, UK, collected wellbeing data using a survey of commercial fishers to rank wellbeing indicators on a scale from 0-10. Indicators were derived from a learning history approach. Mobile gear commercial fishers experienced lower subjective wellbeing and higher material losses (objective wellbeing) compared to static gear fishers (Rees *et al.* 2021). These results are consistent with other studies of MPAs in the UK. For example Blampied *et al.* (2023) also examined the subjective welling of commercial fishers in two MPAs in Jersey, UK, across the domains of (i) job and income satisfaction; and (ii) perception of conflict and stress. Data were collected using a similar approach to Rees *et al.* (2021). Both studies found that mobile gear fishers reported lower levels of wellbeing across both domains (i.e., less job and income satisfaction and increased perceptions of conflict and stress) compared with static gear fishers. However, the Jersey MPAs assessment was for a new MPA with only one year's data, and despite the similar result in mobile fisheries subjective wellbeing, the study also found that profits had increased, implying an increase in objective wellbeing (proxying fishers' profit). A further a caveat relates to the fisher conflict outside of the MPAs (notably with the French static net fleet).

Similar subjective wellbeing results were also reported for commercial fishers in the Jurien Bay Marine Park (JBMP) in Western Australia (McNeill *et al.* 2018). In this study, data were collected against the same wellbeing domains from semi structured interviews, and a thematic approach was used to categorised positive and negative experiences. Again, the social impacts associated with subjective wellbeing were not distributed uniformly amongst commercial fishers. In this case, commercial fisheries who owned multiple fish licences reported lower subjective wellbeing, in contrast to those that held a single licence for rock lobsters. Interestingly, this study did report that the MPA had limited

impact on fishing activities or use of local waters, but it did not explicitly measure the social impacts associated with objectives measures on livelihood (e.g., profit or incomes).

Other studies have also suggested that subjective wellbeing can erode or dissipate over time even if objective wellbeing (livelihood income etc proxied by catch) improves as a result of the MPA (Pita *et al.* 2020).

Each of the above illustrates the unique (and non-comparable) approaches adopted to accessing social impacts for a given MPA. While there are similarities in the results, caution should be exercised in drawing robust conclusions across case studies.

A further dominant theme in the wellbeing studies was that inclusion in decision making and a better understanding of the sustainability impacts of the fish stock would increase commercial fishers subject wellbeing over time (Gill *et al.* 2017). This assumption is often generally applied in fisheries management (e.g., Micha and Kelling 2025) and reflects a perception by industry and recreational fishers that increased participation in decision making will result in better outcomes for them and correspondingly higher satisfaction (Crandall *et al.* 2019). Empirical evidence, however, suggests that the causality between satisfaction and participation may be more complex. In a fisheries management context, Msomphora (2015) found that satisfaction with the management process was a pre-condition for effective participation rather than participation increasing satisfaction per se. Pita *et al.* (2010) found that, even when participation processes were established, satisfaction depended on whether or not the fishers agreed with the outcomes; those who felt that they benefited from the management decisions were satisfied with the processes whereas those who felt they did not benefit felt that they were not being adequately consulted. Similar outcomes in an multi-user MPA context were also found by Perea-Muñoz *et al.* (2022); even with a formalised and established participation process, fishers were satisfied if the outcomes favoured them, but unsatisfied if the management outcomes favoured another user group at a cost to them.

Cultural values

There are well recognised cultural values from engaging with the marine environment, these are extensive and no universal terminology. For example, cultural values identified by Bryce *et al.* (2016) included engagement and interaction with nature, place identity, therapeutic value, social bonding, spiritual value, and memory/transformative value cultural benefits arise from a complex interaction between the characteristics of environmental spaces, practices, and transcendental cultural values. Whilst there is overlap, cultural values identified by Voyer *et al.* (2015) included peacefulness or naturalness, appreciation of beauty freedom, hedonism (joy, pleasure), spiritual wellbeing (religion or spirituality), personal connections (relationships with friends and family),knowledge and learning (personal learning and understanding or teaching others), health (self-esteem, fitness and mental or emotional wellbeing), guardianship (e.g., protector of the environment), benevolence (e.g., volunteer work, service to community), tradition or heritage (links with the past), social cohesion (feelings of community connection and unity), attachment to place (connection with specific location on the coast), acceptable practice (conformity), and unmediated nature (ability to connect directly with the natural world). This makes comparison between studies challenging.

Cultural values at the level off the community are not common to an interest group. For example a study of users of a NSW MPAs including surfers, recreational fishers, professional fishers, spear fishers, walkers, divers, snorkellers, kayakers and other community members, concluded that these groups were not homogeneous in their value systems, and within each included a diverse and complex mix of identities and values that were not associated with membership of a particular stakeholder group (Voyer *et al.* 2015). In contrast, a study in the UK across 151 marine areas of recreational anglers and divers was able to identify distinct regional patterns. For example, in Scotland, subjective cultural wellbeing was associated with connection to nature and beauty, whilst marine sites in southern

England were more associated with feeling more healthy and a sense of freedom and belonging (Bryce *et al.* 2016).

Realising positive social impacts for Indigenous and First Nation People

Indigenous owners are inextricably linked to ecological interactions, and MPAs have the potential to derive positive social benefits for this stakeholder group. This can occur through providing protection of the marine environment that underpins their cultural-ecological knowledge and local and tribal identity, and improving wellbeing through poverty alleviation as a result of improvement in traditional fish stocks and offering opportunities to exploit other income avenues e.g., tourism. (Davies *et al.* 2018).

These benefits can be influenced by where MPA and PPAs are located. Sites of MPA and PPA have traditionally been located based on scientific advice without explicit involvement of First nation People in decision making the significance of e.g., sacred sites or species, taboos and social norms is not captured (Schneider 2018). Whilst is it generally understood that realization of these social impacts required the engagement of Indigenous owners in PPA and MPA planning, establishment, and management, research on Indigenous governance and management is limited (Ban and Frid 2018).

Social impacts of fisheries-specific closures

Consideration of the social impacts of fisheries-specific closures has been relatively limited, and most studies have considered the impacts on local communities as a result of lost or relocated fishing activities rather than on fishers per se. Stevenson *et al.* (2013) found that the displacement of effort as a result of a network of MPAs in Hawaii resulted in centres of activity shifting, with consequent impacts on the local industries supporting the fishing sector (those in the areas where effort relocated to benefiting, while those in the areas where effort relocated from did not). Similarly, Owusu and Andriesse (2020) found that in small scale fishery dependent communities, the loss of income during the closure period may have substantial adverse impacts on the fisher families and also the local communities, particularly if alternative income opportunities were limited. In both cases, the impacts on fisher welfare were not recorded.

While not directly related to fisheries management, Ritzman *et al.* (2018) found that the temporary closure of a fishery in response to an algal bloom resulted in income and job losses not only in the fishing industry but also in fishing-dependent industries such as processing and hospitality industries, with further flow-on effects to the regional economy. These impacts on employment and income also resulted in increased stress and reduced emotional wellbeing of fishers and workers in related industries.

Case study 1: Moreton Bay Marine Park

Introduction

The Moreton Bay Marine Park (MBMP) (Figure 10) is located at a transition zone of tropical and temperate waters, is relatively shallow, and is affected by a large input of freshwater and sediment loads by river runoff. The four main rivers that flow into the Bay cause a steep gradient in water quality across the Bay: the western and southern parts exhibit turbid, nutrient-rich waters, whilst the east and north exhibit clearer water with lower nutrients levels due to connection with the ocean (McEwan *et al.* 1998). These characteristics makes it a highly productive yet fragile system. The seascape of the MBMP is diverse and complex, comprising a mosaic of habitats, including fringing coral reefs, seagrass meadows, estuaries, mangroves, intertidal flats, unconsolidated and unvegetated soft sediments, deep reefs, and offshore rocky reefs (Dennison and Abal 1999; Abal *et al.* 2005; Baltais *et al.* 2014). This diversity of habitats, in combination with its climate, promotes the Bay's high faunal species' diversity (including those of conservation concern) and abundance.

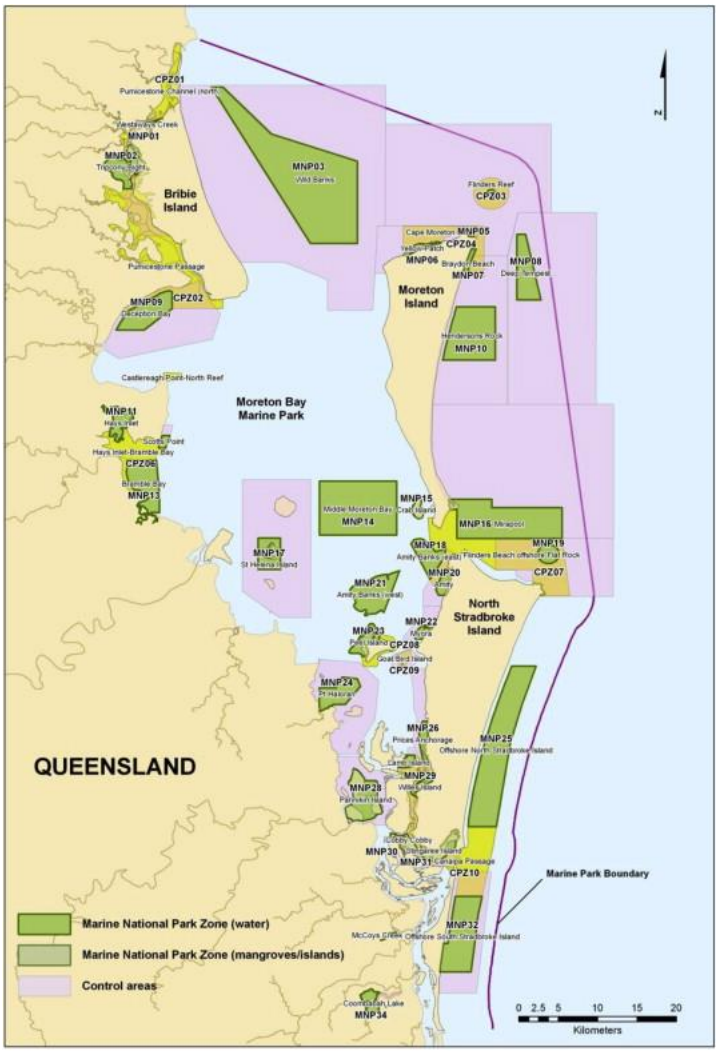


Figure 10. Moreton Bay Marine Park

The proximity of the Bay to a major metropolitan area (Brisbane) means that MBMP also has a high recreational use value as well as a significant conservation value. Activities such as recreational fishing, boating and scuba diving are undertaken by local residents and visitors to the region (Martin

2023). In addition, the Bay has an active commercial fishing fleet, providing fresh seafood (prawns, crabs and fish) to the Brisbane region and beyond.

In 2009, the area of the MBMP that is designated as no-take zones (where no form of fishing is permitted) was increased from less than 1% (representing the total of 'old' no-take zones established in 1993) to 16% (including 'new' no-take zones established in 2009). The major goal of this increase in protection of critical habitats and species was to enhance species abundance, biodiversity, and food webs, and, ultimately, the Bay's overall resilience to environmental variability (Bengtsson *et al.* 2003; Queensland Government 2007). It was anticipated that positive outcomes within the no-take zones would trigger cascading influences on connected and surrounding fished areas through so-called spill-over effects (Mumby *et al.* 2006; Babcock *et al.* 2010).

Objectives of the 2009 rezoning

The rezoning of MBMP was undertaken in consultation with an independent Expert Advisory Group, who developed a series of scientific guiding principles for developing the initial draft zoning plan. The overarching governing principle (i.e. objective) was "*to conserve the unique values (environmental, social, cultural and economic) of the marine park and to ensure its sustainable use for the enjoyment and benefit of present and future generations*" (Queensland Government 2007).

The guiding principles were separated into two sets of objectives: Biophysical Guiding Principles (Table 5) and Socio-economic Guiding Principles (Table 6). Both were based on an "efficient" approach to reserve system planning which aimed at meeting conservation goals while minimising the impact on other users; and also to facilitate efficient management and enforcement.

The socio-economic guiding principles (Table 6) provide a set of objectives aimed at achieving a balance between conservation goals and the need for continued sustainable use. These included a mix of means objectives (e.g., objective 2.2) and process objectives (e.g., objectives 2.1 and 2.3). As part of objective 2.3 in particular, the Expert Advisory Group also notes the importance of continuing off-reserve management (e.g., fisheries management and water quality strategies) to protect marine habitats, while objective 2.2 also notes the need for ongoing stakeholder engagement.

The zoning plan was reviewed in 2019 *(Marine Parks (Moreton Bay) Zoning Plan 2019),* although no zoning changes were made to the previous plan at this stage. Additional restrictions (e.g., go-slow zones) were imposed where activities may impact on natural areas or species (e.g., turtles and dugongs).

Table 5. Biophysical Guiding Principles

Derived from Queensland Government (2007)

Criteria	Principle	Explanation
Habitat representation	1.1. Represent a minimum amount of each 'habitat type' in no-take areas (NTAs)	Protect examples of each habitat type to ensure habitat maintenance and associated biodiversity within the marine park. The Expert Advisory Panel recommends that the CBD 10% minimum target be adopted in the Moreton Bay Marine Park Zoning Plan review.
Size and replication	1.2. Include adequate size and replication of 'habitat types' in NTAs	Each habitat type should be protected in more than one NTA in a reserve network to protect the full range of habitat types as a precaution against major localised damage. Larger reserves are preferred to smaller reserves to minimise edge effects.
Connectivity	1.3. Provide connectivity within the network of NTAs	Reserves in a network should be adequately spaced to ensure the movement of species, providing sufficient connectivity to support populations in reserves.
Vulnerable habitats	1.4. Protect in NTAs an adequate amount of vulnerable habitats	Vulnerable marine and coastal habitats and associated animals and plants need to be effectively protected in NTAs as they typically are easily disturbed or transformed by human actions and recovery is slow (e.g., coral reefs and seagrass beds).
Vulnerable life stages	1.5. Adequately protect species' vulnerable life-stages in NTAs	Vulnerable life stages of species need to be effectively protected in NTAs e.g., critical nursery areas, spawning or nesting site.
Species and areas of special interest	1.6. Include species, populations & areas of special interest in NTAs	Species and populations of conservation concern (e.g. threatened, rare, endangered or restricted-range species), and areas of special interest (e.g. areas with particular geomorphologic features; naturalness, amenity or cultural values; or areas of conservation concern) need to be effectively protected in NTAs.
Ecosystem linkages	1.7. Include consideration of ecosystem links among habitats and of sea and adjacent land uses in determining NTAs	Areas that support other habitats (ecosystem links), or are dependent on other habitats, need to be protected. Past and present uses may have influenced the integrity of biological communities and need to be considered when choosing NTAs. E.g. existing NTAs adjacent to terrestrial protected areas are likely to have greater biological integrity than areas that have been used for resource exploitation.
Resilience	1.8. Provide for future resilience against natural or human-induced changes or threatening processes	Areas that are less likely to be subject to impacts and have a high degree of naturalness (i.e. less exploited) ensure greater resilience against future change or threats.
Adaptive management	1.9. Design a reserve network to provide for scientific assessment of zoning effectiveness	Queensland legislation provides for the review of the zoning plan every 10 years. Zoning revision should be based on scientific evidence of the effectiveness with which they serve the governing principle of the marine park. Design of zoning should be informed by scientific best practice in experimental design and monitoring.

Table 6. Socio-Economic Guiding Principles

Derived from Queensland Government (2007)

Criteria	Principle	Explanation
Balancing conservation and sustainable use	2.1. Ensure the final selection of zones recognises social, economic, cultural and environmental costs and benefits	The final zoning selection needs to be made recognising the costs and benefits to the community.
Minimise impacts	2.2. Minimise the impact of zoning on human interactions with the Marine Park including access, activities, values and aspirations	Any proposed zoning should minimise impacts on users of the marine park. For example, fishing and boating should remain a significant and integral activity within the marine park and Traditional Owners' aspirations for their sea country and the importance of the marine park should be recognised. Engagement of stakeholders and the community in a participatory process that is open and transparent should be ongoing throughout the review process.
Management complementarity	2.3. Complement, where possible, other management mechanisms and arrangements that affect the Marine Park	In considering zoning options, other arrangements that may protect and/or manage the marine environment should be taken into account to minimise conflict and provide greater operational clarity. For example, policies and strategies dealing with marine pollution, international wetlands, national parks, fisheries management, water quality and coastal development.
Balancing conservation and sustainable use	2.1. Ensure the final selection of zones recognises social, economic, cultural and environmental costs and benefits	The final zoning selection needs to be made recognising the costs and benefits to the community.
Efficient and practical	2.4. Maximise the understanding of the Marine Park and the manageability of zones	The final zoning plan should consider operational and implementation issues to help provide for efficient management and enforcement. Uses in the marine park should be consistent, where practicable, with other State marine parks to help the community understand and appreciate conservation and use of the marine environment. An awareness campaign to maximise the understanding of the marine park should also be conducted.

Process of zoning decision making

The process of determining the final zones for the Marine Park were based on a combination of modelling and expert opinion. No weightings were associated with each of the objectives identified in the guiding principles, with trade-offs (where necessary) assessed qualitatively by the Expert Advisory Group supported by geo-spatial modelling (i.e., Marxan (Watts *et al.* 2009)) which aims at identifying areas where conservation and use (primarily fisheries) conflict.

The draft plan was also subject to extensive consultation, including a stakeholder reference group, face-to-face meetings with key stakeholders and stakeholder workshops. The release of the draft plan resulted in over 8000 submissions, with over 6000 of these supporting the plan (DERM 2008).

Ecological outcomes of Moreton Bay no-take zones

Marine ecosystems represent spatially heterogeneous seascapes. Many marine animals are mobile and move between habitats for recruitment, foraging, and reproduction, connecting important processes across ecosystems (Polis *et al.* 1997; Boström *et al.* 2011; Olds *et al.* 2012a). Meanwhile, they have different ecological functions, including browsing or grazing (by herbivores), predation (by piscivores and carnivores), and carrion consumption (by scavengers). The combination of the connectivity of habitats across seascapes and ecological functions of organisms defines the abundance and biodiversity (e.g., Mumby *et al.* 2004; Pittman *et al.* 2007; Nagelkerken *et al.* 2008; Boström *et al.* 2011), trophic interactions (e.g., Valentine and Heck 2005; Olds *et al.* 2012a; Davis *et al.* 2014), functioning (e.g., Mumby and Hastings 2008), and, eventually, the resilience to disturbance of ecosystems (Mumby and Hastings 2008; Olds *et al.* 2014). Removing fishing pressure through the establishment of reserves (i.e., no-take zones) may increase the abundance of harvested species and, consequently, top-down control effects that affect an ecosystem's species diversity, community assemblages, and habitat quality (Lubchenco *et al.* 2003; Lester *et al.* 2009; Sala and Giakoumi 2018). The ecological effects of 'old' (since 1993) and 'new' (since 2009) no-take zones within the MBMP have been studied since 2005 (Pillans *et al.* 2005). Here, we summarise published results on the effects of the removal of fishing pressure by the MBMP no-take zones on species in relation to different habitats and key ecological functions.

Coral reefs

Coral reefs provide habitat for primary producers and habitat, food, and shelter for a large variety of invertebrates, fish, and megafauna species (e.g., sharks and turtles). Due to its geographical location at relatively high latitudes, representing the limit of hard coral growth (Johannes *et al.* 1983), the MBMP comprises marginal coral reefs which exhibit considerably reduced growth and reproduction rates compared to coral at lower latitudes. Due to increased nutrient levels in combination with other favourable conditions, macroalgae can bloom and smother coral reefs (Gilby *et al.* 2015). The removal of turf algae and macroalgae by, respectively, grazing and browsing herbivores is a key ecological function on coral reefs as it decreases competition of mature corals with algae and creates space for coral larvae to settle and, therefore, coral recruitment and growth (Box and Mumby 2007; Mumby and Steneck 2008; Hughes *et al.* 2010; Igulu *et al.* 2014; Steneck *et al.* 2014). In the MBMP, the Black Rabbitfish (*Siganus fuscescens*) is an important browser with a preference for red and brown macroalgae (Pillans *et al.* 2004). It has been the only species observed to consume *Sargassum* (a genus of brown macroalgae) within the MBMP, making its functional redundancy low (Gilby *et al.* 2016b). Consuming turf algae, Bluebarred parrotfish (*Scarus ghobban*), Australian Sawtail (*Prionurus microlepidotus*), and Pencil Surgeonfish (*Acanthurus dussumieri*) are important grazers (Hughes *et al.* 2007; Olds *et al.* 2014). These herbivore species are harvested within the MBMP (Tibbetts and Townsend 2010; Olds *et al.* 2012a; Olds *et al.* 2012c), reducing their top-down control on lower trophic level species, potentially leading to trophic cascades and, ultimately, jeopardising the health and resilience of reef systems (e.g., Mumby *et al.* 2006).

No-take zones increase biomass of grazing fishes (Pencil Surgeonfish and Australian Sawtail) and their grazing rate on turf algae on reefs, particularly in correlation with biomass of surgeonfish (Yabsley *et al.* 2016). Macroalgal browsing rate does not seem to be correlated with any fish species (Yabsley *et al.* 2016), yet protected reefs within no-take zones in proximity of mangroves show increased macroalgae herbivory compared to unprotected reefs or protected reefs without nearby mangroves (Olds *et al.* 2012a; Yabsley *et al.* 2016). This is due to protection from fishing in combination with connectivity to mangroves increasing both the biomass and species richness of harvested roving herbivorous fish (Olds *et al.* 2012a). An interaction of protection and habitat connectivity does not enhance turf algae herbivory (Yabsley *et al.* 2016). This is likely because, whereas Black Rabbitfish (and Bluebarred parrotfish) migrate from coral reefs to mangroves and seagrass for foraging (Mellin *et al.* 2007; Olds *et al.* 2012b; Olds *et al.* 2012c; Davis *et al.* 2014), Pencil Surgeonfish and Australian Sawtail do not (Olds *et al.* 2013; Igulu *et al.* 2014). Increased herbivory rate reduces algae cover on reefs increasing reef growth and recruitment and, finally, resilience to environmental changes (Olds *et al.* 2012c). After a major flooding event in 2011 impacting the MBMP, protected reefs within no-take zones resisted the impact by extreme water quality degradation, whilst unprotected reefs in fished areas did not (Olds *et al.* 2014). Meanwhile, macroalgal cover on coral reefs that are consistently impacted by river runoff fluctuates more with highly variable bottom-up drivers for algal growth (i.e., water quality metrics) than increased browsing rates due to the removal of fishing, suggesting a limited impact of no-take zones in inshore waters (Gilby *et al.* 2015).

Piscivores are vital for shaping food webs within ecosystems and maintaining energy flow between habitats (Valentine and Heck 2005; Reiss *et al.* 2014; Stier and White 2014). In the MBMP, connectivity between coral reefs, mangroves, and seagrass meadows greatly enhances performance of no-take zones in that protected reefs in proximity of other protected habitats show an increased abundance of harvested piscivores (particularly Yellowfin Bream (*Acanthopagrus australis*) and Moses' Snapper (*Lutjanus russellii*) compared to similar fished areas (Olds *et al.* 2012a). Meanwhile, they contain a lower abundance of prey fish, indicating the cascading effects of removing fishing on other trophic levels in an ecosystem's food web and the potential of restoring trophic interactions (Olds *et al.* 2012a). The composition of protected reefs' fish communities is mostly related to the abundance of piscivorous fish and the distance of the protected reefs to nearest seagrass and mangrove habitats (Gilby *et al.* 2016a). The effectiveness of no-take zones on piscivores have been found to be insignificant when the spatial context of the protected habitats is ignored (Gilby *et al.* 2016a).

Seagrass meadows

Seagrass meadows provide nursery areas for fish and food for fish, shellfish, dugong (*Dugong dugon*), and turtles. As algae can harm seagrass by reducing light availability, grazing of algae by herbivorous species is a key ecological function to maintain the health of seagrass meadows (Heck and Valentine 2006; Olds *et al.* 2012c; Poore *et al.* 2012; Whalen *et al.* 2013; Duffy *et al.* 2015). The Black Rabbitfish, which consumes macroalgae on coral reefs (Gilby *et al.* 2016b), feeds on filamentous algae that grow in seagrass meadows (Capper *et al.* 2006). Additionally, it consumes epiphytic algae through feeding on seagrass blades (Chelsky Budarf *et al.* 2011). In abundance, this species dominates herbivorous fish assemblages in seagrass meadows within the MBMP (Ebrahim *et al.* 2014; Maxwell *et al.* 2014). The composition of harvested fish assemblages differs between seagrass meadows that are protected from fishing and those open to fishing, with protected seagrass meadows exhibiting an increased abundance of exploited Black Rabbitfish compared to fished meadows (Henderson *et al.* 2017). Although, generally, the abundance of Black Rabbitfish, Spangled Emperor (*Lethrinus nebulosus*), and Grass Emperor (*Lethrinus laticaudis*) on seagrass meadows increases with proximity to the open ocean, the effectiveness of protection of seagrass meadows was not influenced by seascape context (i.e., habitat connectivity). This contrasts with findings on the effect of habitat connectivity on the performance of protected coral reefs (Olds *et al.* 2012a). Increased grazing of algae by fishes within the MBMP no-take zones significantly enhances seagrass physiology, with the abundance of Dusky

Rabbitfish best predicting algal herbivory rate (Henderson *et al.* 2019). However, as only a small percentage of protected seagrass meadows exhibit a high abundance of Dusky Rabbitfish, the relationship between no-take zone effects and herbivory rate in seagrass meadows remains undetermined (Henderson *et al.* 2019).

Relatively small (< 6 km^2) and new (5 years) no-take zones that are dominated by seagrass meadows and mud flats and fringed by mangroves, have been found to be effective in protecting various exploited piscivorous finfish species of variable mobility, including Yellowfin Bream and Dusky Flathead (*Platycephalus fuscus*) (Pillans 2006). These species' relative density, mean size, and biomass are higher within no-take zones than in fished areas. Recapture data of both species outside no-take zones have indicated at least some spill-over effects, yet more research on its magnitude and significance of this effect is needed (Pillans 2006). The no-take zones were not found to be effective for Barred Javelin (*Pomadasys kaakan*) (Pillans 2006). No-take zones seem to positively affect community structure, species evenness (without decreasing species richness), and mean size of nekton species both within the no-take zones and adjacent fished areas (Pillans *et al.* 2007). This latter effect suggests again spill-over effects by the no-takes zones onto proximate fished areas.

Carrion consumption by scavengers limits the accumulation of decomposing animal material, which can otherwise cause disease and reduce water quality (Porter and Scanes 2015; Olds *et al.* 2018a). The Giant Mud Crab (*Scylla serrata*) is an important scavenging species that consumes carrion (Webley 2008). No-take zones dominated by mud flats and seagrass meadows have shown to positively affect Giant Mud Crab abundance and population structure as measured through increased catch rates, catch per unit effort (CPUE), and mean size of adult specimens compared to fished sites (Pillans *et al.* 2005). Within no-take zones, males greatly dominate catches and male catches are about five times higher than in fished areas (Pillans *et al.* 2005). In contrast to the fished areas, more than half of the males caught within protected zones are equal or greater than their legal size (Pillans *et al.* 2005). Sex ratio of mature mud crabs within protected zones tends to be largely skewed towards males (Pillans *et al.* 2005). Although male Giant Mud Crabs within the protected zones are larger, this effect has not been apparent in adjacent fished areas, suggesting there is no spill-over effect. This is likely due to the relatively small home range of the Giant Mud Crab and other invertebrate species (Pillans *et al.* 2005).

Estuaries

No-take zones in the MBMP dominated by estuarine habitat have been found to be ineffective in enhancing the abundance of harvested fish species (Gilby *et al.* 2017). This is likely because the protected estuarine habits are of low habitat value to fish including narrow estuaries with limited connectivity to mangrove habitats and the open ocean, given that the abundance of harvested fish species is known to be positively correlated to estuary size and connection to the ocean at low tide (Gilby *et al.* 2017). Carrion consumption is considered a key ecological function in estuaries (Olds *et al.* 2018a). Estuaries protected by no-take zones show a significantly different scavenger assemblage and increased scavenging potential compared to outside these protected zones (e.g., mud crab (*Scylla serrata*) more dominant outside NTZ, Yellowfin bream (*Acanthopagrus australis*) more dominant inside NTZ) (Webley 2008). Meanwhile, scavenging potential tends to increase with proximity of the protected estuary to the open ocean (Webley 2008). Scavenging within protected estuaries is dominated by the mud crab, whilst outside the no-take zones its scavenging level is decreased and equal to that of the Blue Swimmer Crab (*Portunus armatus*) (Webley 2008).

Mangrove forests

Mangrove forests provide shelter, nursery habitat, and food for invertebrates, fish, and marine megafauna (Sheaves *et al.* 2015; Sievers *et al.* 2019). In mangrove forests, herbivorous fish and invertebrates perform a key ecology function by removing algae from mangrove prop roots and pneumatophores. This kind of herbivory enables oxygen exchange by the roots, improving mangrove

growth (Cannicci *et al.* 2008). Health and growth of mangroves are also enhanced by scavengers that consume carrion which can be carried into the forests by the tide and waves and cover the mangrove roots (Olds *et al.* 2018a; Henderson *et al.* 2019). No-take zones protecting mangrove forests positively affect fish diversity and the abundance of harvested species, particularly when the protection zone is in proximity of reasonably sized seagrass meadows and the open ocean (likely in relation to water quality) yet far from coral reefs (Goodridge Gaines *et al.* 2020). Particularly the abundance of Yellowfin Bream, a species targeted by commercial and recreational fisheries and a dominant scavenger, is affected, yet the no-take zones do not seem to increase herbivory or scavenging rates (Goodridge Gaines *et al.* 2020).

Offshore rocky reefs and surf zones

Within two years after establishment, offshore no-take zones protecting rocky reefs were found to include an increased biomass of fish species, particularly harvested fish (including Snapper (*Chrysophrys auratus*), Spangled Emperor, Redthroat Emperor (*Lethrinus miniatus*), Blackspot Tuskfish (*Choerodon schoenleinii*), Maori Rockcod (*Epinephelus undulatostriatus*), and Goldspot Pigfish (*Bodianus perditio*)) (Haywood *et al.* 2019). However, increased abundance of snapper has been found to be more related to habitat type (kelp and rock habitat) and depth than ('old' or 'new') no-take zones (Terres *et al.* 2015). Given that most fish species mentioned above are long-lived, the two-year time-window was likely too early for the removal of fishing to affect their abundance (Haywood *et al.* 2019). Several of the same 'new' offshore no-take zones were found to not only increase the average size of mud crabs and Blue Swimmer Crabs (*Portunus* armatus) but also catch rates of mud crabs (Haywood *et al.* 2019).

Surf zones provide foraging, spawning, and nursery habitats for many fish (Olds *et al.* 2018b). Within no-take surf zones in the offshore part of the MBMP, fish diversity and abundance of harvested fish species tend to be highest when the zones stretch over more than 1.5 km, are located within 100 m of rocky headlands, and are located on smaller-size beaches with a heterogeneous seascape (Ortodossi *et al.* 2018). The current no-take zones protecting surf zones have not been shown to significantly increase fish assemblages compared to fished beaches (Ortodossi *et al.* 2018). This is likely because due to the limited zone size, fish readily move outside the boundaries of the no-take zones into adjacent waters without protection (Lester *et al.* 2009; Huijbers *et al.* 2015), whilst a lack of habitat heterogeneity at extended beaches represents isolation and limited feeding habitat (Vargas-Fonseca *et al.* 2016; Olds *et al.* 2018b).

Highly mobile megafauna

The MBMP contains a diverse community of highly mobile and protected megafauna species, including dolphins, dugongs, migratory whales, sea turtles, elasmobranchs, and resident and migratory wader birds (Lanyon *et al.* 2019). In the 'Go-slow' zones overlapping important feeding grounds of turtles and dugongs in mainly the eastern part of the Bay, vessels must reduce speed to mitigate the risk and impact of collisions. In most of the MBMP waters where bottlenose dolphins (*Tursiops aduncus*) occur, covering the eastern and southern Bay (Chilvers and Corkeron 2001; Ansmann *et al.* 2012), trawling is now prohibited. The reduction of trawling in the MBMP in the late 1990s seems to have changed the bottlenose dolphins' society from two discrete social communities to a more interconnected social network (Ansmann *et al.* 2012). This may increase information sharing and gene flow across the Bay's population and, ultimately, the population's resilience (Ansmann *et al.* 2012). It remains uncertain, however, whether this change in social network in relation to reduced trawling activity is due to decreased foraging by dolphins on trawling by-catch or increased fish stocks in the Bay (Ansmann *et al.* 2012). Other information on how the removal of fishing due to the implementation of the no-take zones within the MBMP affect megafauna species remains extremely scarce. This is mainly because most of

these species are long-lived and they exhibit vast home ranges that reach outside the MBMP (e.g., dugongs, whales, turtles, and birds).

Fisheries economic outcomes

Review of previous studies

Studies of the economic impact of the rezoning of the Moreton Bay Marine Park are limited, with most activity undertaken prior to the rezoning and only limited retrospective assessment of actual outcomes.

Studies contemporary with the rezoning in 2009 suggested that the economic impact of the rezoning on fisheries was likely to be substantial. For example, van de Geer *et al.* (2013) suggested that expectations of income reductions as a result of loss of fishing grounds prior to the 2009 rezoning averaged around 22%. A post-implementation survey provided an updated stated impact of around 23% reduction in income, consistent with expectations. Fishers attributed this reduction in income to loss of fishing ground, requiring higher costs to access new fishing grounds, and increased crowding on the remaining (open grounds) (van de Geer *et al.* 2013).

Around 25% of the fleet (mostly net and line boats) were actively fishing in areas that were to be closed with the rezoning, with this fishing effort moving to other (open) areas of the bay following the rezoning (Kenyon *et al.* 2018). A $15 million structural adjustment package was implemented to offset the loss of income and access to some extent (DERM 2008), with 119 licences being bought out (Sen 2010). While a portion of these were currently inactive licences, they were believed to represent roughly a quarter of the gross value of commercial take in the Bay (Sen 2010).

Evidence of an increase in crab and fish abundance within the new green zones was found within the first three years post-rezoning, although changes in abundance outside the zones (i.e. the spill-over effect) was not observed over this same period (Haywood *et al.* 2019). The potential for spill-over effects to increase catch rates in the commercial fishers longer term was uncertain. However, recapture data in an earlier study in Moreton Bay found at least some spill-over effects outside no-take zones for crab species (Pillans 2006).

Increased activity by recreational fishers along the borders of the no-take zones, however, suggested that some localised spill-over impacts may have occurred (Kenyon *et al.* 2018). Pascoe *et al.* (2014) estimated that even limited spill-over effects may increase non-market economic benefits to recreational fishers, with an estimated 5-12% increase in benefits expected (valued at between $1.3m and $2.5 million a year).

Changes in commercial fishing activity over time

Total vessel number operating in Moreton Bay decreased prior to the implementation over much of the decade prior to the rezoning of the marine park. As part of the rezoning, 119 licence were bought out in 2008 (Sen 2010) to offset the potential loss of fishing area. However, as 132 fewer vessels were operating in the fishery between 2000 and 2009, many of these licences were latent (i.e. unused). Only 32 fewer vessels operated in 2009 than 2008, representing 13% of the active licences. The removal of latent licences, however, was still beneficial as it prevented fishers with active licences that were removed through the structural adjustment program from re-entering the fishery by purchasing a latent licence. Since 2009, the number of line, net and pot licences active in Moreton Bay (QFISH fishing area X37) has continued to decline, although the number of otter trawlers has remained at around the 2009 level (Figure 11).

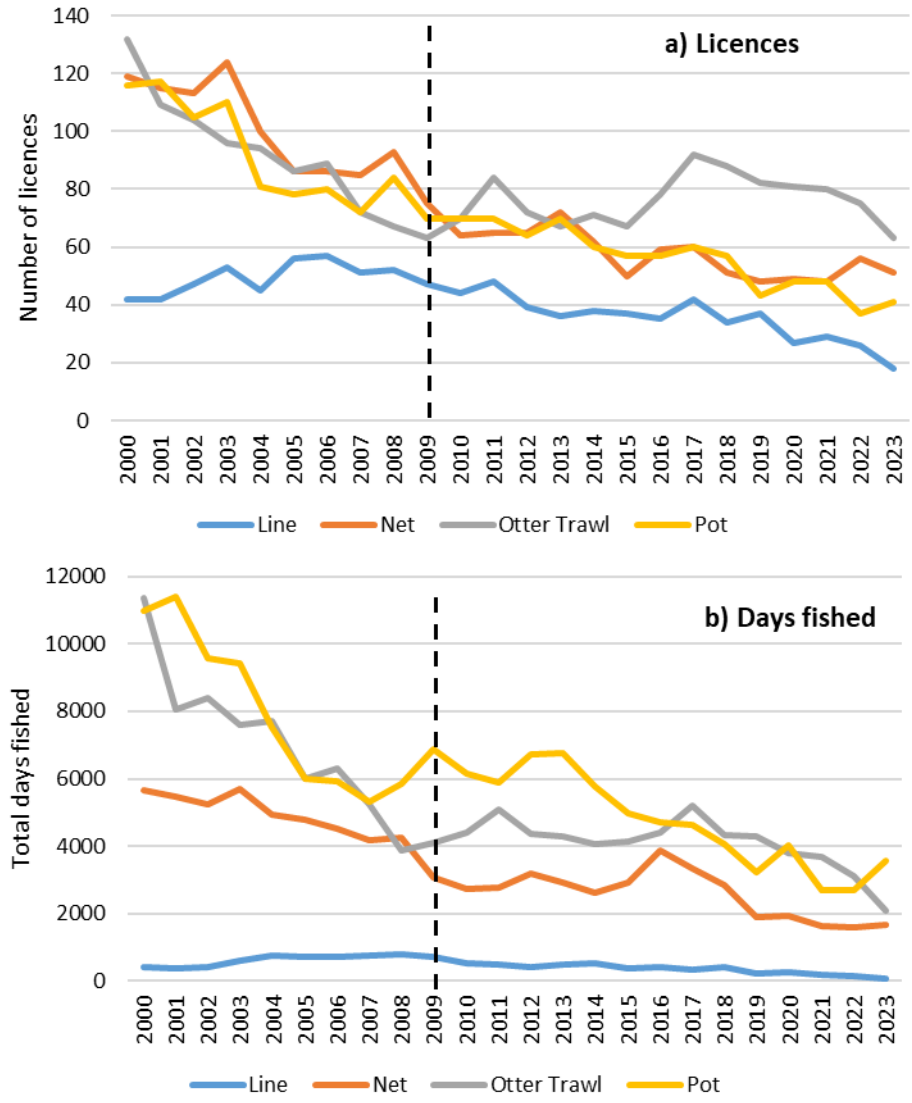


Figure 11. Number of a) active fishing licences and b) days fished in Moreton Bay

Fishing effort in terms of number of days fished has also declined since 2009 for all fleets (Figure 11), with net and pot boats having a similar rate of decline before and after the rezoning of the marine park.

The fishing industry is subject to a range of other pressures that affect prices and costs and hence the level of activity. Comparing otter trawl activity inside and outside Moreton Bay provides an indication of the potential impact of the marine park on this activity. Non-Moreton Bay trawl licence activity also declined over the same period of the data but continued to decline post 2009 (Figure 12). A simple linear regression model (Table 7) suggests that an additional 1.6 trawlers operate in Moreton Bay for each year since the rezoning than might otherwise have operated (based on the trend in non-Moreton Bay licences). That is, in 2023, an additional 24 licences are operating in Moreton Bay that might otherwise not be operating based on the trends in the broader fishery.

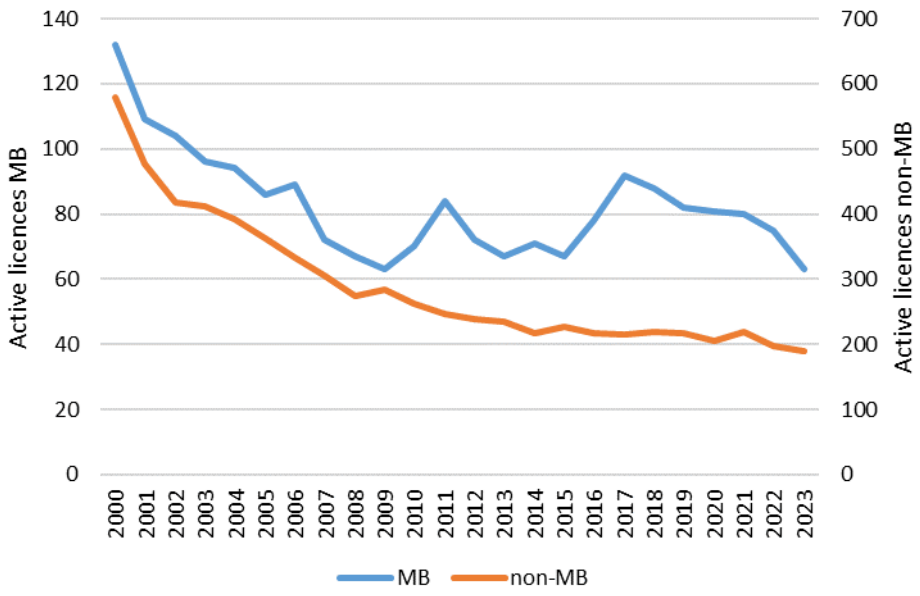


Figure 12. Number of active fishing licences otter trawl licences

Table 7. Linear regression of Moreton Bay trawl licences against non-Moreton Bay licences

	Coefficients	Standard Error	t Stat	P-value
Intercept	18.54	8.67	2.14	4.4%
Non-Moreton Bay licences	0.19	0.02	8.29	0.0%
Years of park operation	1.62	0.45	3.62	0.2%
R Square	0.79			
Adjusted R Square	0.77			

A similar comparison is more difficult for the other fleet types. Otter trawlers target the same main species (prawns) inside and outside Moreton Bay (with many having the ability to move in and out of the Bay based on relative economic conditions). Other fleet types tend to catch different species outside the Bay, particularly net and line fleets, and hence would be subject to different environmental and economic conditions. Potting boats, while also targeting different species outside the Bay (e.g., Spanner crabs), are likely to be the next closest to otter trawlers in having similar environmental and economic conditions. A comparison of active licence numbers inside and outside the Bay suggest that the change in licence numbers was not impacted by the marine park (Figure 13,Table 8).

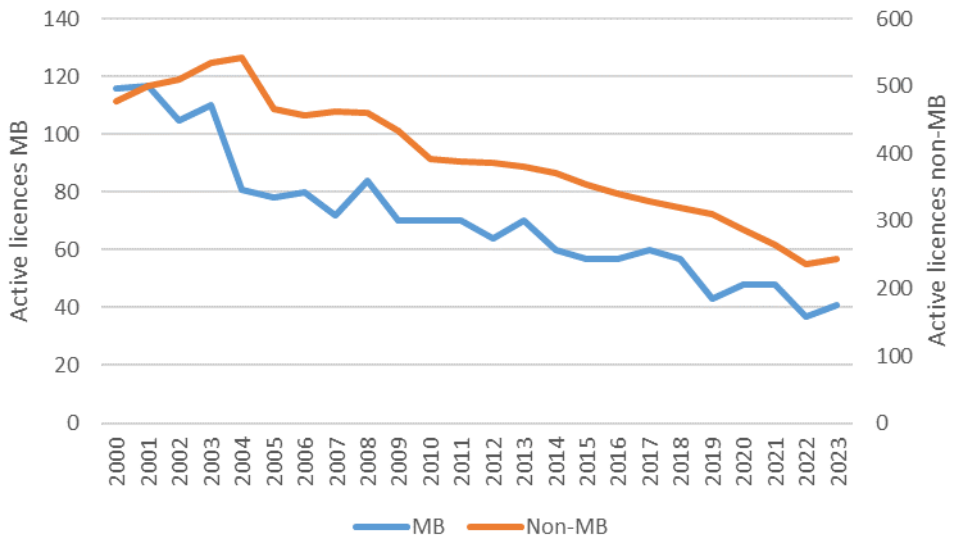


Figure 13. Number of active fishing licences: potting licences

Table 8. Linear regression of Moreton Bay potting licences against non-Moreton Bay licences

	Coefficients	Standard Error	t Stat	P-value
Intercept	-35.22	39.94	-0.88	39%
Non-Moreton Bay licences	0.26	0.08	3.11	1%
Years of park operation	0.67	1.44	0.46	65%
R Square	0.79			
Adjusted R Square	0.78			

Changes in commercial fishing performance over time

As noted previously, expectations of income reductions of around 22% as a result of loss of fishing grounds prior to the 2009 rezoning in-part instigated the implementation of a structural adjustment to offset this loss (van de Geer *et al.* 2013). Fishers reported a 23% reduction in income in a post-implementation survey, consistent with expectations (van de Geer *et al.* 2013).

Fisheries economic performance is a function of catch, effort, prices received and costs of fishing. Prices and costs are impacted by a wide range of factors. For examples, prices for product sold in the domestic market is a function not just of landings from Moreton Bay, but also landings of the same or substitute species elsewhere, as well as imported fish and crustaceans. Costs, particularly fuel costs, are also influenced by global economic conditions. Hence, changes in economic outcomes may be due to a range of factors unrelated to the rezoning of the marine park. In contrast, changes in catch rates are potentially directly impacted by the rezoning, as fishers displaced from new green zones are required to relocate their fishing effort to potentially less productive areas, or areas where competition for the resource is greater. As a consequence, changes in catch rates provide an indication of the effects of the rezoning on changes in economic performance.

Catch and effort information from the QFISH database (https://qfish.fisheries.qld.gov.au/) were used to estimate how catch per unit of effort changed as a result of the marine park rezoning. Catch per day and per boat fluctuated substantially over the period 2000 to 2022 (Figure 14). Average catch per

vessel did decrease for crab (pot) and prawn trawl vessels in 2010, consistent at least in terms of direction with the findings of van de Geer *et al.* (2013). In contrast, catch per vessel increased slightly for line and net boats targeting fish species.

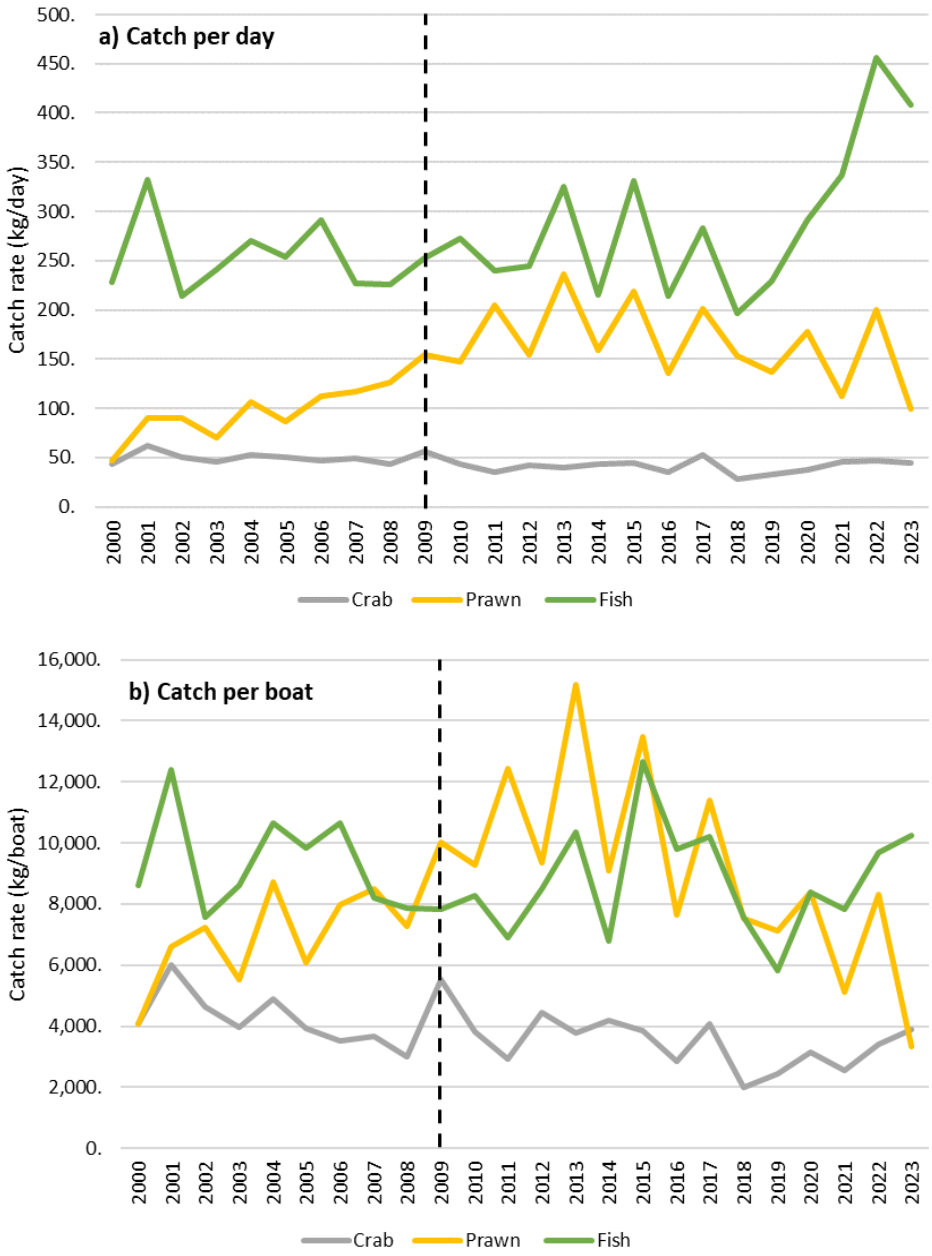


Figure 14. a) average catch per day and b) average catch per vessel in Moreton Bay, 2000-2023

However, catch rates – both on a per day basis and per vessel basis – fluctuated substantially over the period of the data, reflecting potential environmental influences or, on a per boat basis, incentives created by changes in prices and costs. Comparing the catch rates over the periods 2000-2008 and 2009-2022 (Table 9), catch rates for crabs declined following the rezoning on a per day basis, but were not significantly different on a per boat basis (i.e. catch per boat before and after the rezoning were not significantly different when the natural fluctuations in these rates were taken into account). Similarly, fish catch rates were not significantly different before or after the rezoning compared by both per day and per boat (Table 9). Apparent differences between periods in this case (e.g., between 2018 and

2023 in the case of fish catch per day) may be more reflective of factors other than the effect of the marine park rezoning.

In contrast, catch rates of prawns increased significantly on both the per day and the per boat basis after the no-take zones were implemented in 2009 (Table 9). The extent to which this is a result of a reduction in excess capacity in the fleet due to the structural adjustment package, or a spill-over effect due to the increase in protection areas is unclear, although it is likely that it is mostly a result of the former than the latter.

Table 9. Catch rates pre and post 2009.

	2000–2008		2009–2023			
	Mean	St Dev	Mean	St Dev	t-stat	Prob
Catch per day (kg/day)						
Crabs	49.51	1.92	41.84	1.96	2.80	1%
Prawns	93.97	8.23	166.17	10.16	-5.52	0%
Fish	253.81	12.84	286.41	19.08	-1.42	17%
Catch per boat (kg/boat)						
Crabs	4183.47	295.85	3530.52	233.90	1.73	10%
Prawns	6895.29	499.70	9179.79	791.95	-2.44	2%
Fish	9377.50	537.56	8721.46	454.83	0.93	36%

At the species level, prawn catches fluctuated from year to year (Figure 15). Banana Prawn catches peaked in 2011 and 2017 due to flooding conditions in the Brisbane River, contributing to the higher average catch rates post-2009. Catches of Endeavour, Eastern K and Tiger Prawns fluctuated around the pre-2009 average, with a slight decrease in Eastern King Prawn catch in recent years (i.e. post 2019). Catches of "other" prawns (e.g., Bay Prawns and others that that are largely used as bait) initially increased after 2009, but have decreased consistently since around 2013. The extent to which this decrease can be attributed to the marine park is uncertain given the catches remained above pre-2009 levels up to 2017.

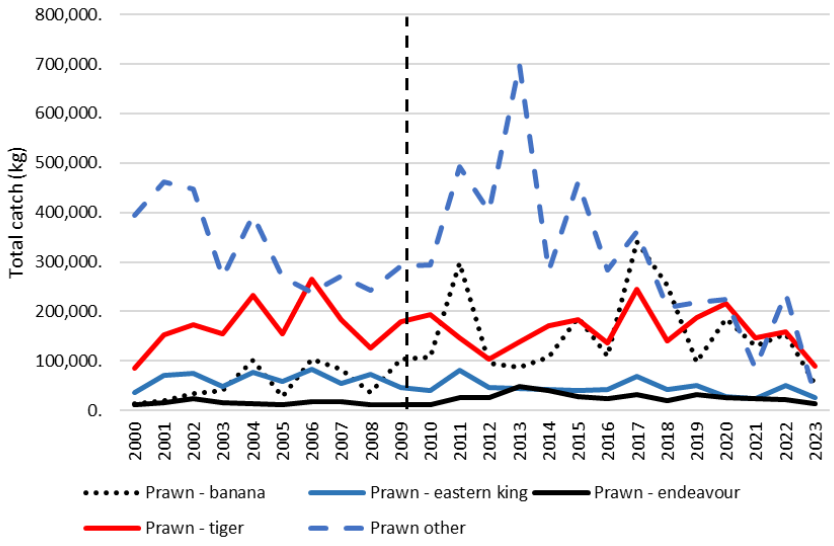


Figure 15. Total prawn catch by species in Moreton Bay, 2000-2023

Changes in recreational fishing activity over time

Estimates of total recreational fishing participation are undertaken on an irregular basis through a state-wide recreational fishing survey (e.g., Teixeira *et al.* 2021; QDAF 2024b). Since the re-zoning of Moreton Bay Marine Park, three surveys have been undertaken: 2010, 2013 and 2019; and they can be compared to just one survey prior to re-zoning in 2000.

Deriving the impact of the marine park on recreational fishing is uncertain as each survey represents only one point in time. The estimated number of days fished in Moreton Bay declined between 2000 and 2010, but similar declines were experienced elsewhere (Figure 16). Between the 2010 and 2013 surveys, days fished in Moreton Bay declined slightly, as did fishing effort in other parts of Queensland, but less so than in the adjacent regions (Gold Coast and Sunshine Coast) (Figure 16). However, between 2013 and 2019, the estimated number of days fished in Moreton Bay declined by 65% while days fished in the adjacent regions increased by over 200%.

What has driven these relative changes in uncertain. Catch rates in Morten Bay remained relatively higher (on a number of fish caught per day basis) than adjacent and other regions between 2000 and 2010, suggesting no significant impact on catch rates immediately following the designation of the Marine Park (Figure 17, (QDAF, 2024b)). However, catch rates decline between 2013 to 2019 to comparable levels with other regions. By 2019, catch rates in all regions were relatively similar, suggesting that catch rates were not responsible for the shift in the number of days fished in Moreton Bay between 2013 and 2019.

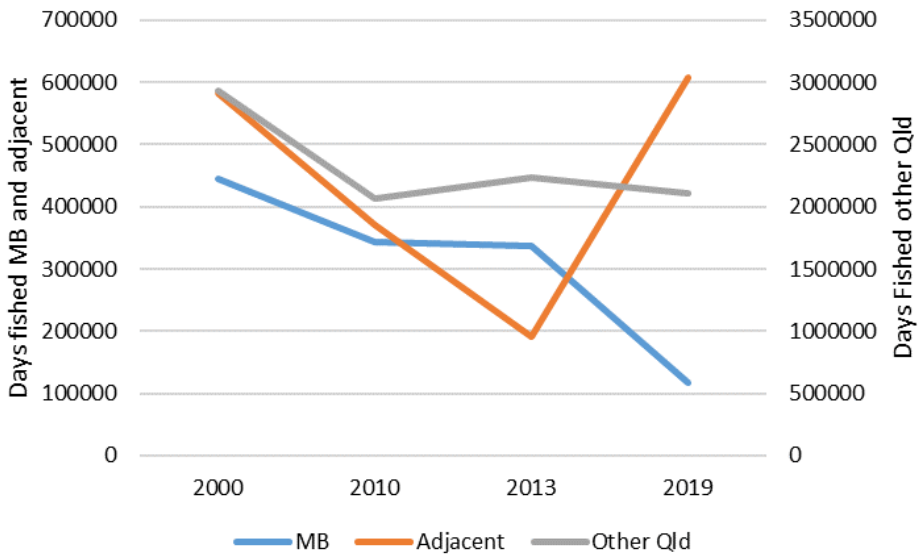


Figure 16. Recreational days fished in Moreton Bay and elsewhere, 2000-2019

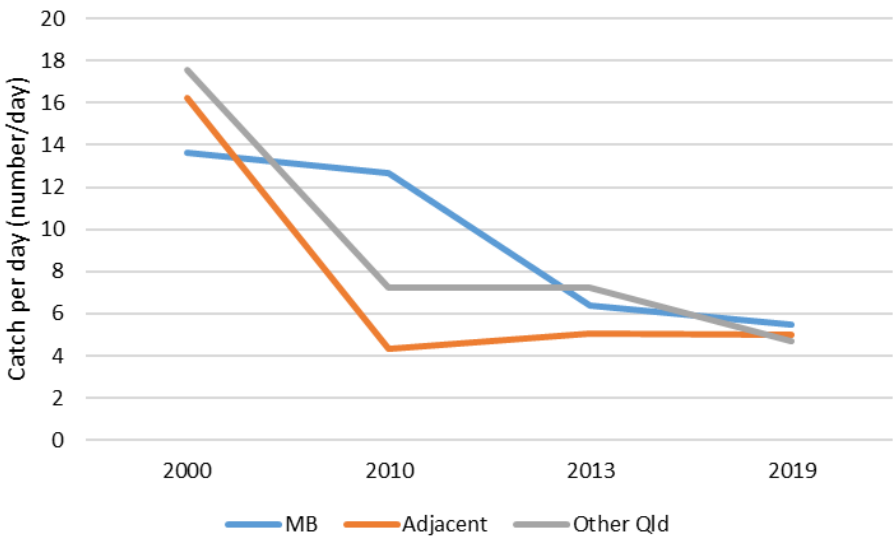


Figure 17. Catch rates in Moreton Bay and elsewhere, 2000-2019

More recent trends in fishing activity in Moreton Bay are available through the boat ramp survey program (QDAF 2024a). This series does not capture activity before the rezoning of the Marine Park but provides an indication of more recent trends in activity. The key indicator is the average number of boat-trailers in the car park at each boat ramp. This is assumed to reflect recreational fishing activity using boats but does not capture shore-based activity.

The average number of trailers observed during each survey has declined in Moreton Bay by close to one boat-trailer a year since 2016. A similar, but slower, decline in trailer numbers has also been seen at other Queensland boat ramps (Figure 18 (QDAF, 2024a)). The extent to which this greater rate of decline can be attributed to the Marine Park is uncertain and is more likely to reflect a range of other factors related to the broader Brisbane population and also the range of other recreational activities available. Also, not all trailers are used for recreational fishing as boating is also a popular activity in Moreton Bay, with only two thirds of vessel operators in Moreton Bay engaging in recreational fishing (Martin 2023).

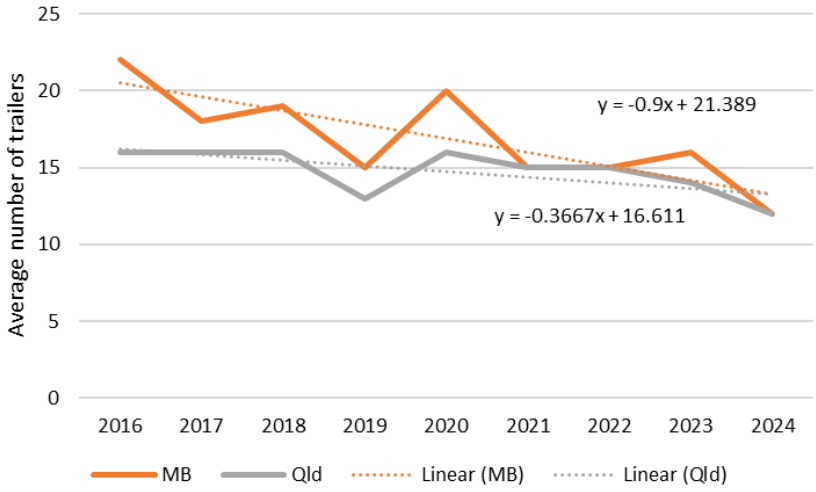


Figure 18. Average trailer numbers at boat ramps, Moreton Bay, 2016-2024

Social and cultural outcomes of Moreton Bay Marine Park

Moreton Bay not only provides significant recreational fishing opportunities (already described in the previous section), but it also provides other significant social and cultural outcomes. Recreational activities include swimming, snorkelling, sailing, wildlife watching, picnicking, walking and running, cycling, diving, and more (Kenyon *et al.* 2019). These activities often include a boating element. In addition to the value provided by these activities, there are more general appreciation values (in a human-centred manner) related to experiencing nature, places of tranquillity and natural beauty for wellbeing reasons and sharing the experience with others (Nasplezes *et al.* 2019; Martin 2023). These experiences have been linked to stewardship of Moreton Bay (Nasplezes *et al.* 2019). Combined with strong programs of information, environmental education and safety e.g., water quality suitable for swimming, these experiences also serve to maintain its social significance.

In this section, we outline how MBMP is providing social and cultural outcomes, focussed on the social enjoyment (of recreational activities and aesthetics), natural heritage and bequest values, Traditional Owners cultural heritage, and local stewardship and community cohesion.

Social enjoyment (aesthetic and recreational benefits)

Previous studies exploring how people value Moreton Bay and its tributaries found a strong desire throughout the region to protect and look after ecological integrity so as to ensure people can continue to derive social and cultural benefits, especially recreational outcomes (Ross *et al.* 2019). The no-take areas of Moreton Bay are often available for at least transiting through (i.e., recreational boating use), even if the area is closed to extractive commercial fishing.

Recreational boating and its associated recreational activities provide important social benefits. A recent study on the uses and values related to recreational boating found around 60% of survey respondents went recreational boating in Moreton Bay at least once a month and for some several times each month (seasonally dependent) (Martin 2023). Nearly all (97%) stated that they received substantial benefits from appreciating the natural beauty of the Bay, while over 90% stated that they gained benefits from experiencing nature (e.g., seeing dolphins, sea birds etc) (Figure 19 (Martin, 2023)).

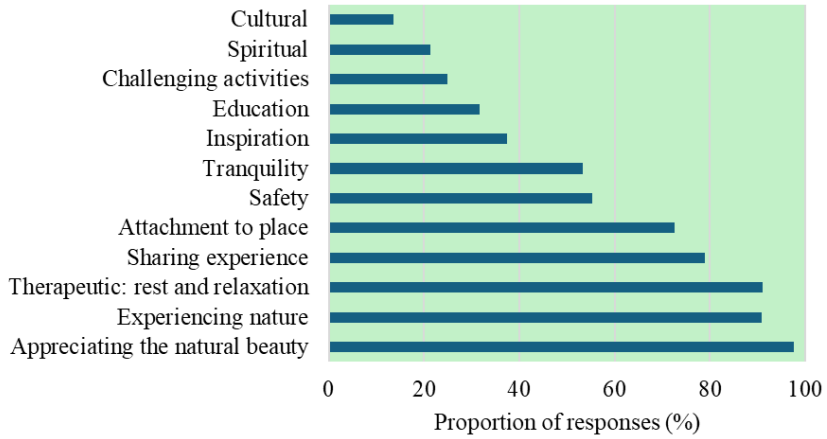


Figure 19: Importance of different benefit types for recreational boating

Source: Martin, 2023

Windle and Rolfe (2013) estimated the non-market value of beach-based recreation in Moreton Bay to be on average $40/day in 2012 values, roughly equivalent to $57/day in 2024 values. Activities undertaken on the beach were not identified, so may include a combination of recreational fishing and

boating activities defined in the previous studies. It is unlikely, however, that the change in the zoning of the marine reserve influenced these values as similar values were also identified for Gold Coast beaches (Windle and Rolfe 2013).

Changes in Traditional owner cultural heritage

The MBMP is of significant social and cultural importance to local communities, including Indigenous, Aboriginal Australian and Torres Strait Islander peoples. Traditional hunting is important to the social and spiritual wellbeing of the Quandamooka people, Traditional Owners of the Moreton Bay area. The new MBMP zoning has meant that the Quandamooka people have been able to progress their traditional hunting guidelines into a Traditional Use of Marine Resources Agreement (TUMRA) (SEQTOLSMA 2008).

Beyond conserving traditional places of significance and traditional uses, protected areas within the MBMP take account of the landscape within which they are situated (Low Choy *et al.* 2010). A key guiding principle of the Quandamooka people's sustainability strategic plan is that the environment, society and the economy are all in good condition and all work together as everything is connected *(Corporation 2021)*. This speaks to the importance of integrated management that enhances outcomes across the triple bottom line (Stephenson *et al.* 2019; Stephenson *et al.* 2023). It highlights the "holistic and social–ecological perspective over managing their land and sea country that contrasts sharply with the sectoralism of government" (Ross *et al.* 2019). As such, including landscape values within the social and cultural outcomes of the MPA is of importance for Traditional Owner cultural heritage.

The Healthy Land and Water (HLW) report card has started to measure the cultural values for Moreton Bay and SEQ waterways using four indicators (Healthy Land & Water 2023):

- recognition of rights and interests, reporting the proportion of areas across the SEQ region in which First Nations rights and interests are recognised so that their ability to lead the stewardship of Country and culture through Native Title is enabled
- cultural referrals, reporting the identification and recognition of First Nations peoples leading decisions related to Aboriginal cultural heritage.
- cultural surveys, reporting how much of the region has formally documented the archaeological surveys to maximise the protection of Aboriginal cultural heritage.
- cultural protection and management, reporting what proportion of the documented cultural significant Traditional Owner sites have cultural heritage management plans in place.

These cultural indicators do not currently investigate and report which indicators are supported specifically by protected areas within MBMP.

Changes in local stewardship and community cohesion

To some extent the changes in local stewardship and the associated community cohesion that comes from a sense of place is difficult to measure. (Roberts *et al.* 2017a) recommend taking bottom-up approaches to design with local communities large partially protected MPAs that remove damaging activities so that local and global benefits for climate change resilience are merged.

While not directly reflecting differences between the MBMP generally and the specific no-take (green) zones and preservation (pink) zones, the Healthy Land and Water (HLW) group have been measuring and reporting on several social benefits: satisfaction with experience, usability and accessibility, personal benefits, connection with nature and waterways as well as water-based recreation (classified as economic benefits under their framework) (Healthy Land & Water 2023). However, these are reported based on the south-east Queensland Local Councils that the survey participants reside in, rather than being reported based on which part of Moreton Bay (or SEQ waterways) the social values

(and activities) originate from. Similarly, there is no differentiation on whether the social outcomes arise specifically from green (no take) or partially protected areas within Moreton Bay.

In future, iterations of the HLW report card and stewardship reports could include questions such as these that could in turn provide meaningful monitoring of the social and cultural values arising from the protected areas. Ideally this would also be linked back to the socioeconomic guiding principles that were adopted for the marine park rezoning (Queensland Government 2017).

Finally, there is considerable literature calling for marine park protected areas in order to provide extra resilience from pressures driven by climate change (see section in the literature review), and this was also identified as a key objective by the marine protected area managers (see section on objectives). Jacquemont *et al.* (2022) suggest that social indicators could be crafted to measure with climate change adaptation and propose the following indicators: social organisation (conflict & cohesion), learning (environmental awareness and education), Agency (user rights & participation), and food security (CPUE and local nutrition).

Case Study 2: Commonwealth marine reserves (south-east marine reserve network)

Introduction

The South-east Commonwealth marine reserve network (Figure 20, (Director of National Parks 2013)) was established in 2007, comprising 14 separate marine reserves off the coast of Victoria, South Australia and Tasmania covering 388 464 km2 (Director of National Parks 2013). The south-east network is part of a larger national series of marine reserves established around the country, each with the goal of capturing a representative collection of marine habitats and key ecological features. Australia currently has 58 marine parks under its jurisdiction, formerly called Commonwealth Marine Reserves, covering approximately 2.8 million km2. After its proclamation in 2007, the Management Plan for the South-East Network 2013-2023 came into effect (Director of National Parks 2013). More recently, following the review of the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act), the succeeding management plan for the south east network has committed to an adaptive management framework in order to better map out the objectives for these MPAs and how to achieve these over the next decade (Hayes et al. 2021).

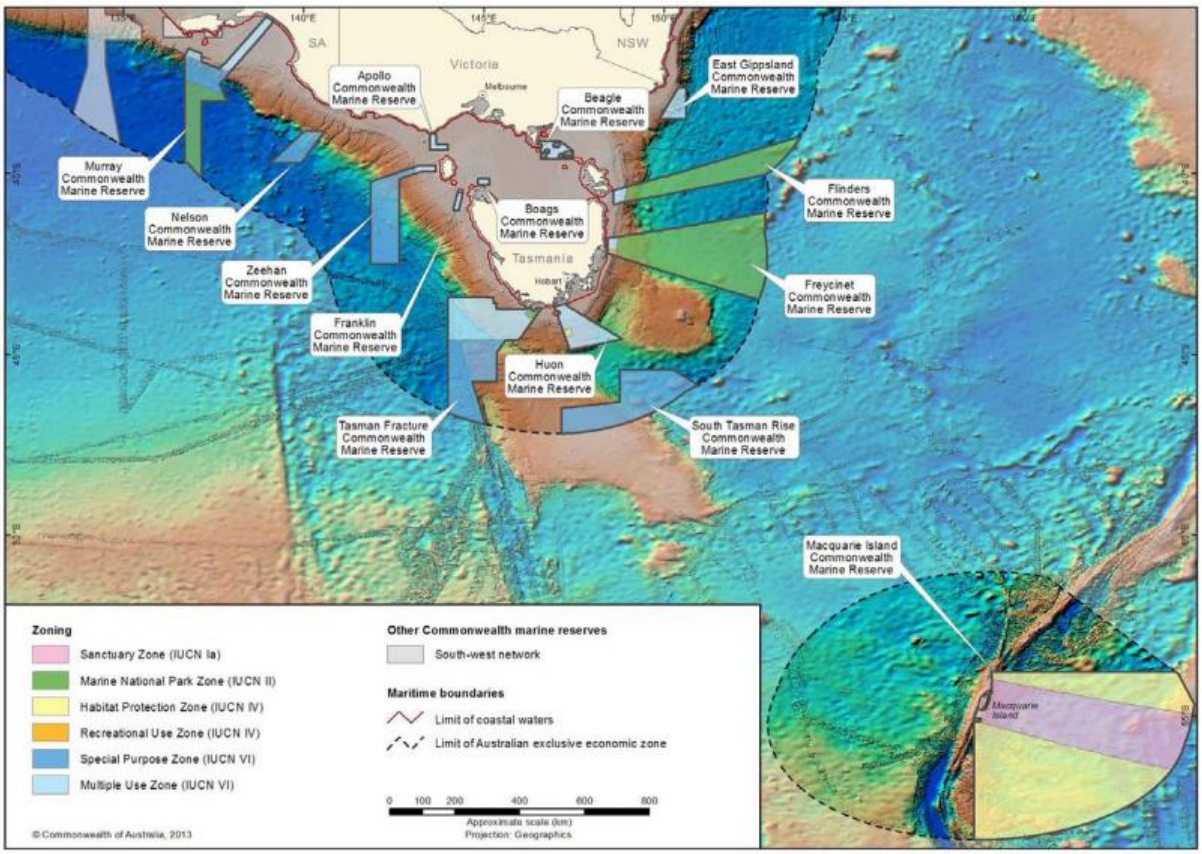


Figure 20: South-east Commonwealth Marine Reserves Network

Source: Director of National Parks (2013)

A key goal of each of the marine reserves is to capture examples of benthic and demersal biological features, such as habitats, communities, and sub-regional ecosystems, particularly those with high biodiversity value, and species richness at scale. The networks were also designed to accommodate climate change as far as practicable, including anticipated latitudinal or longitudinal movement in ecosystem or species distributions and changes in oceanographic features and currents, in response to climate change (Parks Australia 2024).

As with the Moreton Bay Marine Park, a number of guiding principles were developed to assist in determining the location, selection (when more than one option to meet the goals is available), design and zoning of suitable areas. These guiding principles are applicable to all Commonwealth Marine Reserve Networks and are detailed in Table 10.

Zoning within the marine reserves was based on the World Conservation Union (IUCN) categories of protection (Day *et al*. 2012). These include both highly protected zones (I and II) as well as multiple use zones (IV and VI).

The marine reserve network is also subject to a management plan (Director of National Parks 2013), the objectives of which are to:

1. provide for the protection and conservation of biodiversity and other natural and cultural values of the South-east Marine Reserves Network; and
2. provide for ecologically sustainable use of the natural resources within the South-east Marine Reserves Network where this is consistent with objective 1.

In this regard, the marine reserve network is managed to provide conservation, economic and social benefits, with the latter two being contingent on not compromising the conservation objectives of the reserve.

Table 10. Guiding Principles

Derived from Parks Australia (2024)

Consideration	Principle	Explanation
1. Location		
	1.1 Build on existing protected areas and sectoral measures	Take into account the occurrence and location of existing spatial management arrangements
	1.2 Minimise the number of separate marine reserves to maximise conservation outcomes	A smaller number of larger marine reserves is preferred to many small marine reserves
2. Selection (should take account of)		
	2.1 The capacity of a marine reserve to mitigate identified threats to conservation values	
	2.2 The occurrence of - spatially defined habitats for threatened and/or migratory species. - ecologically important pelagic features - known small-scale (tens of kilometres) ecosystems - listed heritage sites	
	2.3 Relevant available information about small-scale distribution of sediment types and sizes and other geo-oceanographic variables	
	2.4 Socio-economic costs should be minimised	
3. Design		
	3.1 Individual areas should include continuous depth transects	E.g., from the shelf to the abyss, as far as practicable
	3.2 Whole seafloor features should be included	E.g., geomorphic features should be included in full
	3.3 Features should be replicated within the system of marine reserves	That is, key features should be included more than once wherever possible
	3.4 Maximise connectivity	Size and shape should be orientated to account for inclusion of connectivity corridors and biological dispersal patterns within and across marine reserves.
	3.5 Boundary lines should be simple, and easily identifiable	As much as possible following straight latitudinal/longitudinal lines and where possible, coinciding with existing regulatory boundaries
	3.6 Size and shape of each area should be set to minimise socio-economic costs	

Process of zoning decision making

From the workshop, determining the most appropriate areas for the designation of the marine reserves is based on previously developed guidelines for establishing a national representative system of marine protected areas (ANZECC Task Force on Marine Protected Areas 1998). The guidelines used in the development of the south-east marine reserve network are largely derived from these broader national guidelines.

Zoning decisions are largely based on expert opinion (of key DCCEEW staff and external consultants) supported by geospatial information on the biophysical characteristics of each area and the level and type of existing activities. As with Moreton Bay marine park, the criteria for inclusion as a protected area or PPA are based on the guiding principles which are unweighted (i.e. each treated as equally important).

The process also involved the development of multiple alternatives, with the ecological benefits and potential economic costs of each estimated. These were developed through the use of an expert panel (i.e., Buxton *et al.* 2006).

The final decision as to which option was chosen was made at the Ministerial level.

Ecological impacts

The South-east Marine Reserves Network contains several key sea-floor features. The continental shelf component of the reserve includes rocky reefs and soft sediments, which foster diverse species. The shelf break intensifies currents and upwellings, boosting biodiversity in the key conservation zones as well as supporting the resource in commercially and recreationally fished areas. A series of sea-floor canyons provide habitats for sessile invertebrates such as corals and enhance local productivity and biodiversity through intensified currents and nutrient concentrations. Finally, a set of seamounts, located south and east of Tasmania, restrict and intensify deep ocean currents, preventing sediment build-up and creating habitats for unique deep-water coral and fish species (Director of National Parks 2013).

Biodiversity in the Network exhibits high diversity, with the South-east Marine Region noted for its extensive marine species diversity, including the world's most diverse marine floral assemblage. The region is believed to include over 600 different fish species (Director of National Parks 2013). A large number of the species in the Network are found nowhere else, including 85% of the fish fauna, 95% of the molluscs, 90% of the echinoderms, and 62% of macroalgae (seaweed) (Director of National Parks 2013). The region's stable climate, geological history, and historical changes in sea level (e.g., Bass Strait's submergence and emergence) have shaped its unique species composition and distribution.

Assessing the impact of the marine reserve on these key features is difficult, with the effects of management potentially taking more than a decade to manifest. Babcock *et al.* (2010) found that the time to detect direct effects on target species in an MPA was 5.13 ± 1.9 years, while indirect effects through trophic interactions were detectable only after 13.1 ± 2.0 years. While established in 2007, the management plan has only been in operation since 2013, with the effectiveness of this too early to assess.

A review evaluating the effectiveness of the current management plan (May *et al.* 2022) also noted that the key strategy objective of “minimising the impacts of activities through effective assessment of proposals, decision making and management of reserve-specific issues” was limited due to the current processes to monitor compliance with the obligations attached to licenses and permits. In response to this, as well as to develop sufficient information to assess network performance, Parks Australia has established a Monitoring, Evaluation, Reporting and Improvement (MERI) framework (Hayes *et al.*

2021) in a bid to produce sufficient data to assess the effectiveness of the marine reserve network. This approach was considered a "significant achievement" by an independent review of Parks Australia (May *et al.* 2022). As part of this, and to support evidence-based management, the South-East Marine Parks Network is the focus of a pilot study. The objective of this pilot is to explicitly determine monitoring priorities and risks to marine conservation values for the South East Marine Parks Network, which can then be expanded further to other MPAs and networks (Hayes *et al.* 2021).

Management priorities were the underlying basis of the monitoring program for natural values in the south-east marine reserve network, driven by the need for Parks Australia to understand what management is needed in order to achieve its conservation goals (Hayes *et al.* 2021). Monitoring questions and indicators were subdivided into natural value indicators (including but not limited to fish diversity, biomass, invertebrate diversity and habitat forming species), and anthropogenic pressure indicators (including but not limited to ocean acidity, turbidity, habitat modification, invasive species and marine pollution). Publicly accessible geospatial information systems (GIS) datasets were the primary measurement tool used to develop ecosystem maps and risk profiles for different parks in the South-East Marine Parks Network. The south-east network was selected for the pilot because of the relative abundance of knowledge and information on this network, despite this critical data and measurement gaps were still encountered (Hayes *et al.* 2021). Limitations in the availability of habitat mapping, recreational fishing efforts and the extent of different reef types were identified as the most critical gaps, which affected the confidence of ecosystem vulnerability assessments and impact modelling. These data limitations highlight the need for ongoing research and measurement on ecological attributes and human interactions with marine environments in order to support the Monitoring, Evaluation, Reporting and Improvement framework.

From the workshop, the aims of the monitoring program are threefold:

- to observe long term changes that are occurring in the Australian marine parks, including changes due to climate effects as a result of climate change;
- to assess the effects of different zoning arrangements in the marine park and how these influence conservation outcomes; and
- estimate cause and effect between the types of management actions, e.g., mitigation of key pressures, on the maintenance or improvement in natural values.

While the monitoring program is currently focused on ecological impacts, there is consideration of expanding the program to include also cultural and heritage values and other benefits in the future.

The brief literature review earlier in this report has shown that Marine Protected Areas can deliver positive ecological impacts on biodiversity and fish biomass. The designation of an MPA itself does not guarantee ecological outcomes. Proper management especially for higher protection levels (where possible), over extended periods of time and planning for ecological attributes including habitat connectivity will maximise the likelihood of positive ecological impacts. For Australia's South East Marine Parks Network, it may be too soon to tell what the specific effects on the areas' marine ecosystems will be.

Economic impacts

Estimates of the economic impacts of the south-east marine reserve network have mostly focused on the (potential) loss of fishing opportunities due to closure of some areas. Early assessments of the economic impact of the south-east marine park network estimated that the value of the loss of catch taken from within the network would be in the order of $11.6 million a year (across both State and Commonwealth fisheries) (Buxton *et al.* 2006). However, this ignored the potential to re-allocate effort to outside the closed areas. Most fishers surveyed indicated that they would relocate their fishing activities at a higher cost (estimated to be around $0.75 million a year) with an expect lower catch and

revenue, estimated to be around 60% of the pre-closure level (i.e. a net loss of revenue of $4.6 million a year) (Buxton *et al.* 2006).

Most previous studies on the economic impact of Commonwealth marine reserves have tended not to focus on the south-east network. For example, Larcombe and Marton (2016) estimated that the displaced fishing activity from the proposed expansion of the Commonwealth marine reserve network (after the south-east network) was in the order of $4.51 million to $8.21 million a year based on the final network design. The regional flow on effect of this was estimated to exceed $7.9 million a year (based on the least-impact design). This assumes, however, that the displaced fishing effort does not move to other parts of the fishery, in which case the final impact on fishing revenues (and profitability) and regional economic activity may be less.

More recently, Nichols *et al.* (2021) found that the designation of the south-east marine reserve network did not negatively impact the economic performance of adjacent fisheries. Nichols *et al.* (2021) found that effort displacement was not as substantial as estimated in the previous studies, as the network was designed with minimising these impacts in mind (as indicated in the guiding principles). What effort was displaced was not sufficient to negatively impact the performance of other vessels operating in these areas (Nichols *et al.* 2021).

The potential economic impact of the south-east marine reserve network may have been obfuscated by the fishing industry buyback under the "Securing our Fishing Future" package, which reduced the fleet size that previously had access to the area of the marine reserves by about 50% (The Auditor-General 2009). While this was not directly related to the marine reserve designation, the effort reduction would have mitigated to a large extent any reallocation of fishing effort from the closed areas.

As with the Moreton Bay marine park, the economic impacts of the south-east marine network appear to be limited. Critics, however, suggest that the fishing industry has been unfairly targeted by the zoning within the marine park, as fishing activity was not a major threat to the ecosystems or biodiversity within these zones (Kearney *et al.* 2012; Boag 2016). Despite the lack of evidence that the designation of the marine reserve had a direct substantial economic impact on the fishery, Boag (2016) suggests that the displacement of fishing activity may have had an indirect impact through increasing investment risk in the industry. That is, the property rights implicit in the statutory fishing right are lessened as access can be withdrawn to certain areas.

Studies on impacts of the marine reserve network on recreational fishers are also limited. Navarro *et al.* (2018) suggests that most recreational fishers are generally supportive of marine reserves in Australia, with little perceived impact on their fishing experiences. Navarro *et al.* (2021) undertook a choice survey of recreational fishers to elicit willingness to pay estimates for the network of Australian marine parks (AMPs) as a whole. They estimated that the net present value (i.e. cumulative discounted value over time) of the (non-market) value to recreational fishers of the multiple use zones in the AMPs was between $208 million and $1.1 billion (2021 values), depending on the assumptions about fishing activity levels within the marine parks. At the same time, loss of access to the closed zones within the marine parks was estimated to range from $85,000 to $168,000 a year.

Several studies have attempted to estimate the non-market value of marine reserve protection to the general public through the use of choice experiments. Burton *et al.* (2015) estimated the willingness to pay for improved management measures (such as additional codes of practice and monitoring and scientific research) for the south-east marine reserve network that would lead to better conservation outcomes. They found that individuals were, on average, willing to pay $11.95 (2015 values) for improved management of bioregions in general, and $13.30 for improved management of key ecological features.

The choice survey by Navarro *et al.* (2021) also considered the value (willingness to pay) to the Australian (non-fishing) public for the network of AMPs as a whole. They estimated that net present value of the AMPs to the general public was between $6.2 billion and $8.7 billion (2021 values), with the protected zones valued more highly ($7.52 for a 1% increase in zone area) than the multiple use zones ($1.92 for a 1% increase in zone area).

Other studies have also found that the general public generally has a high level of support for Commonwealth and other marine reserves, with Raabe *et al.* (2024) suggesting that most Australians would like to see a further expansion of the marine reserve network, irrespective of the potential impacts this would have on commercial and recreational fishing.

The evaluation report by May *et al.* (2022) noted that less work had been undertaken in the marine reserve to assess changes in social and economic values. The review also noted, however, that progress was being made in this area, with the report by Navarro *et al.* (2021) being a starting point in the development of appropriate social and economic baseline metrics.

Social impacts

Information on social impacts of the south-east marine reserve network is limited, but some potential impacts can be inferred from studies in other Australian Marine Parks. The Australian Marine Parks provide significant recreational fishing opportunities but more and more it is also recognised that marine parks provide significant social and cultural outcomes. For instance, there is also now much more understanding of the importance of marine sourced food especially protein is now linked clearly to human health and wellbeing—as a society—then this can be considered as a social outcome also even though it is primarily an economic one.

Some social outcomes may have monetary values (as identified in the previous section) but as for high-quality food security and many other social outcomes like aesthetic values (natural beauty, time in natural landscapes, etc) and existence many cannot be assigned monetary values. There are also heritage and cultural values in both the above- and underwater environments. The literature review reported in the earlier parts of the report go into more detail on these, but in summary the lack of monetary values remains the main reason that social outcomes have not had as much research focus as the economic and ecological outcomes.

At the forefront of the social outcomes is the notion of pride experienced by a community, region or even the nation when their marine parks are well-managed such that they preserve the intrinsic existence values. An example is when tourism facilities, normally thought of as an economic outcome, mean that the local community can also enjoy the local attractions even if they are offshore. The community's sense of place is known to be affected positively by the existence of marine reserves e.g., Cairns and Townsville are very proud of the outer reef experiences their tourism businesses have on offer.

Social and cultural outcomes often hinge on the reserve remaining intact, healthy and creating resilient ecosystems. Involvement of stakeholders and rightsholders, and Traditional Owners is vital to ensure the outcomes also provide appropriate stewardship of cultural heritage. These deepen and broaden the way that we consider outcomes that the reserves produce in terms of providing equity for future generations by conserving the bequest values such these marine ecosystems continue to provide ecosystem services for now and reliably into the future. Taking a landscape view of how reserves and protected areas contribute to aesthetics, traditions and wellbeing sees the concepts of local stewardship as inherently important to achieving adaptative management that responds to emerging threats in an effective manner (e.g., the initial incursions of invasive species are more likely to be detected when local people are actively engaged as stewards) (Nasplezes *et al.* 2019; Cannard 2023).

In the following subsections we outline some of the key social and cultural outcomes that Australian Marine Parks current aims to achieve and where these outcomes could be strengthened. The following discussion illustrates how a focus on the social enjoyment of recreational activities and aesthetics can align with cultural heritage protection, and is consistent with natural intrinsic and bequest values in an economic framework.

Previous Reviews and key literature

While the south-east marine reserve network was established in 2007, it took two decades of policy development and planning to establish Australia's first national MPA system in 2018 in Commonwealth waters, following the first proclamation in 2012 and another revision in 2015. However protection levels of the established MPA system were less when compared to the earlier iterations due to a substantial loss of the highly protected zones (IUCN I – II) in 2018 (Cockerell *et al.* 2020).

The Commonwealth Marine Reserves (CMR) review noted that "The primary goal of the NRSMPA was to establish a Comprehensive, Adequate and Representative (CAR) system of Marine Protected Areas (MPAs) to contribute to the long term ecological viability of marine and estuarine systems, to maintain ecological processes and systems, and to protect Australia's biological diversity at all levels" (Buxton and Cochrane 2016). This primary goal implicitly ensures the maintenance of existence/intrinsic values if it is thought of from a social perspective given, for example, the bequest values associated with marine and estuarine systems.

The CMR review terms of reference were to "consider how socio-economic issues could be better included in future decision-making". Striking the right balance between socially and economically important activities and conservation is vital for garnering stakeholder support related to the siting of PPAs within the MPA (Watson *et al.* 2021). Social and cultural issues are not necessarily fully represented when looking through a socio-economic lens (Cannard 2023). During the CMR review's significant stakeholder engagement and consultation two questions were used to determine whether issues raised were contentious in a socio-economic sense, namely: was there an opportunity to deliver either: a) a higher conservation outcome at no significant additional socio-economic cost (including for ongoing management) or, b) a lower socio-economic cost with no significant change to the conservation outcome? This approach aligned with recommendations to identify trade-offs between achieving biodiversity conservation priorities and the social costs, benefits and economic constraints (Seguin *et al.* 2023).

Changes in social enjoyment (aesthetic and recreational benefits)

Assessing the additional aesthetic and recreational benefits of MPAs and PPAs requires determining the degree to which recreation and exercise activities for human health (e.g., include swimming, running, diving, snorkelling, sailing, wildlife watching, walking and cycling, picnicking) in the marine areas and terrestrial areas adjacent to the marine parks have changed as a result of the marine park siting.

While economic measures of recreational uses have been well established (e.g., consumer surplus estimates associated with recreational fishing), not all social outcomes are easily measurable. Some of these insights from Moreton Bay reported that for instance more than half of the boaters surveyed appreciate the natural beauty of the area, experiencing nature and undertake their boating for therapeutic reasons such as for relaxation and for improved mental health and general wellbeing (Martin 2023). While some of these values may be implicitly captured in economic use-value measures, disentangling these is complex.

Non-use values have also been captured in an economic framework. The ecosystem services approach encapsulates the concepts of total economic valuation (TEV) which incorporates non-use

values such as existence values (of ecosystems), bequest values (to future generations), and aesthetic values, with established methods available to measure these values (but in practice rarely employed). These values represent more general appreciation values in an anthropocentric manner, and relate to experiencing nature, places of tranquillity and natural beauty for wellbeing reasons and sharing the experience with others (Nasplezes *et al.* 2019; Martin 2023). .

These experiences have been linked to stewardship of marine areas. Governance involvement through consultation and engagement processes with stakeholders and rightsholders also leads to improved social and cultural outcomes especially if the options and trade-offs were co-designed in the first place. To date the CMR has adopted the NRSMPA guidelines principles relating to social and cultural and heritage.

Changes in Traditional owner cultural heritage

As traditional ecological knowledge has been increasingly acknowledged and shared by Traditional Owners, we are seeing the move towards co-management of natural resources including MPA management and monitoring (Rist *et al.* 2019). Steps towards co-management enable safe sharing of both places of cultural significance and traditional uses. This leads to further understanding of key social outcomes such as sense of place, and leads to social learning (Berkes 2009). Low Choy *et al.* (2010)suggests that natural resource planners and managers should incorporate Indigenous landscape values within existing regional planning processes. This, in turn, enables the Traditional Owners to have a true decision-making role and to incorporate their *"holistic and social–ecological perspective over managing their land and sea country that contrasts sharply with the sectoralism of government"* (Ross *et al.* 2019). Landscape values then may also be a key social and cultural outcome that larger offshore MPAs can provide. As such, including landscape values within the social and cultural outcomes of MPA is of importance for both Traditional owner cultural heritage as well as general community social and cultural outcomes (Wescott and Fitzsimons 2016).

Traditional hunting is of significant social and cultural importance to local Traditional Owners for their social, community and spiritual wellbeing (Delisle *et al.* 2018). Therefore, traditional hunting should be provided for within all MPA but which areas are selected for PPA to allow that to occur should be determined by co-designing with Traditional Owners. It then makes sense to include Indigenous rangers in monitoring for compliance and/or species-specific stock assessment to ensuring ongoing sustainability of traditionally hunted species. The incorporation of Indigenous rangers is another way in which socio-economic outcomes can be secured for local communities (Buxton *et al.* 2006; Buxton and Cochrane 2016)

Changes in local stewardship and community cohesion

Co-management can and should be extended beyond Traditional Owners to other right holders and stakeholders so that there is increased local stewardship and community cohesion even though the belonging values that come from a sense of place are difficult to measure. Roberts *et al.* (2017a) recommend taking bottom-up approaches and designing large partially protected MPAs that remove damaging activities with local communities.

Objectives relating to PPAs and their preference weights

As noted in the methods section, a series of workshops were held with key Queensland and Commonwealth managers of marine protected areas to identify a range of objectives, often described as "values", that managers aimed to achieve when designing a marine protected area. Combined with a review of the literature, a set of objectives were derived. Given these objectives, a survey was implemented to elicit their relative importance weights.

A priori, the relative importance of each of these objectives was expected to differ between stakeholders. For example, objectives were expected to differ between conservation managers in different jurisdictions given the different ecological and policy environments under which they operate and between conservation focused groups and user groups (e.g., fisheries managers and commercial and recreational fishers) who are affected by the management decisions.

The relative importance stakeholder groups place on objectives impacts the overall net benefits (benefits minus costs) associated with the marine parks and associated partially protected areas. Increasing the transparency of these differences has the potential to guide future decision making and reduce the potential for conflict between different stakeholder groups when considering marine reserves and partially protected areas and are a critical part of informing the assessment of a marine park's societal impact.

Identification of objectives

The identified objectives were classified into different typologies using a standard objectives-classification system (Keeney 2007)

- The *fundamental objective* is the ultimate (overarching) aim of the marine protected area.
- Under this is a series of *means objectives*: sub-objectives focusing on key different components of marine park management with the aim of achieving the overarching fundamental objective. These means objectives also underpin the trade-offs between uses that may be permitted in any particular area. Hence, identifying the relative importance of each of these multiple means objectives to decision makers is essential to help guide optimal decision making.
- We also identified a number of *process objectives*, which prescribe how the objectives should be achieved (not necessarily what is to be achieved) when pursuing the means and fundamental objective(s).

The fundamental (overarching) objective identified through these meetings and review was to:

"Ensure functional marine environments that provide a range of demonstrable services and benefits, not just to users but local communities, potential future visitors, and even broader benefits nationally and internationally".

This fundamental objective captures three distinct components relating to the environment, the local and national economy (benefits), and the local community. In order to achieve this fundamental objective, a number of ecological, economic and social means objectives were identified (Table 11).

Ten process objectives were also identified (Table 12). These involve other considerations that do not directly impact the fundamental objective but guide how the fundamental objective is to be achieved. The distinction between several of these objectives and social objectives is relatively minor (e.g., the process objectives relating to equity and stewardship).

Table 11. Ecological, Economic and Social objectives of marine protected areas.

1.	Ecological	
	1.1	Maximise protection of unique habitats
	1.2	Maximise habitat representation
	1.3	Maximise connectivity of habitats
	1.4	Maximise effective biodiversity protection
	1.5	Maximise climate change resilience
	1.6	Maximise Threatened, Endangered and Protected (TEP) species protection
	1.7	Maximise keystone species protection
	1.8	Maximise protection of other species (e.g., keep "common" species common)
	1.9	Maximise resilience to regional growth pressures
2.	Economic	
	2.1	Minimise negative impacts on commercial fishing
	2.2	Improve resource condition for commercial fishing (e.g., fish stocks)
	2.3	Minimise negative impacts on other commercial operations (e.g., shipping, transport ferries, sand extraction)
	2.4	Improve opportunities for nature-based tourism businesses (e.g., scuba diving)
	2.5	Improve opportunities for other economic uses compatible with environmental conservation
	2.6	Minimise impacts on support industries (e.g., tackle shops, chandleries)
	2.7	Maximise economic benefits to Traditional Owners and First Nations Groups (e.g., jobs, employment, businesses, etc.)
3.	Social	
	3.1	Minimise negative impacts on recreational fishing
	3.2	Improve resource condition for recreational fishing (e.g., fish stocks)
	3.3	Reduce conflicts between different user groups (e.g., commercial and recreational fishers)
	3.4	Improve quality of other recreational experiences (e.g., scuba diving, boating)
	3.5	Maximise lifestyle opportunities
	3.6	Minimise loss of fresh fish/seafood to local markets and consumers
	3.7	Minimise negative impacts on local communities
	3.8	Maximise social benefits to Traditional Owners and First Nations Groups (e.g., rights to exercise traditional practices, passing down of traditional knowledge, etc.).

Table 12. Process objectives of marine protected areas.

P.1	Ensure all users have equitable access
P.2	Ensure First Nations objectives, uses and needs are considered
P.3	Increase nature and conservation appreciation through education during planning
P.4	Maximise community stewardship and buy-in through engagement with stakeholders
P.5	Coordination with other Government agencies (e.g., fisheries managers, tourism and transport agencies, local councils)
P.6	Ensure a viable mix between fully and partially protected zones
P.7	Ensure management and compliance resources are adequate
P.8	Minimise regulatory burden and encourage behavioural change as a management tool
P.9	Facilitate effective management of sea country by Traditional Owners and First Nations Groups within protected areas (ranging from full management, to co-management, to effective partnerships and through to appropriate consultation on decisions)
P.10	Ensure adequate programs are in place to monitor and evaluate management effectiveness

Elicitation of objective importance

Response rates

Survey invitations were sent to named individuals and generic organisational email addresses. For the purposes of this study, the response rate was estimated as the proportion of complete responses relative to the number of direct invites, less any automatic rejections received (e.g., no longer working there; the email address is invalid; and in some cases, rejected by the server as suspected spam).

In total, 82 individuals responded to the survey. Six respondents did not provide consent to participate in the survey and a further 17 respondents partially completed the survey. These surveys were excluded from the analysis (a total of 23 surveys). The effective sample used in the analysis was 59 responses, with an effective response rate, on average, of 32% (Table 13).

Table 13. Response rates for the online survey

Group	Invitations		Responses (complete)	Rejections (automatic)	Response rate (%)[a]
	Direct	Organisation			
Fisheries managers	61	5	18	8	33%
Commercial fisher organisations	30	10	5	10	25%
Marine conservation	8	3	0		0%
MPA managers	44	11	23	2	54%
Recreational fishing organisations	74	6	7	9	11%
Other			6		-
Total	217	35	59	29	32%

a) response rate is based on direct invitations less rejections

The response rate were highest for fisheries and marine park managers, 35% and 58% respectively. across all groups. No responses were received from conservation groups.

Sample characteristics

Of those who consented and completed the survey, 65% were male. Only 2% of respondents identified as Indigenous Australian. The sample was dominated by fisheries and marine protected area managers (37% and 30% respectively), with fisher organisations (commercial and recreational) making up around 12% of the sample each. The "other" group included charter and commercial fishers directly, as well as researchers supporting marine resource management (Figure 21).

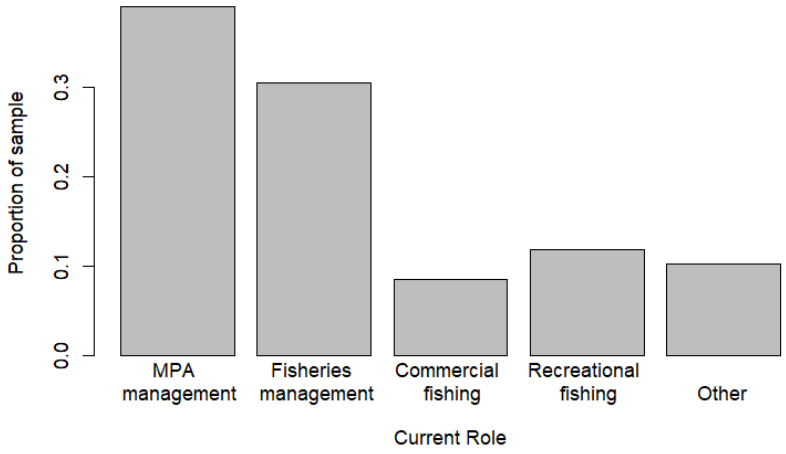


Figure 21: Current role in marine resource management

Roughly half of the sample (49%) had more than 20 years total experience in activities associated with marine resources. Around two thirds of the sample indicated that they had greater than five years' experience in their current role (Figure 22).

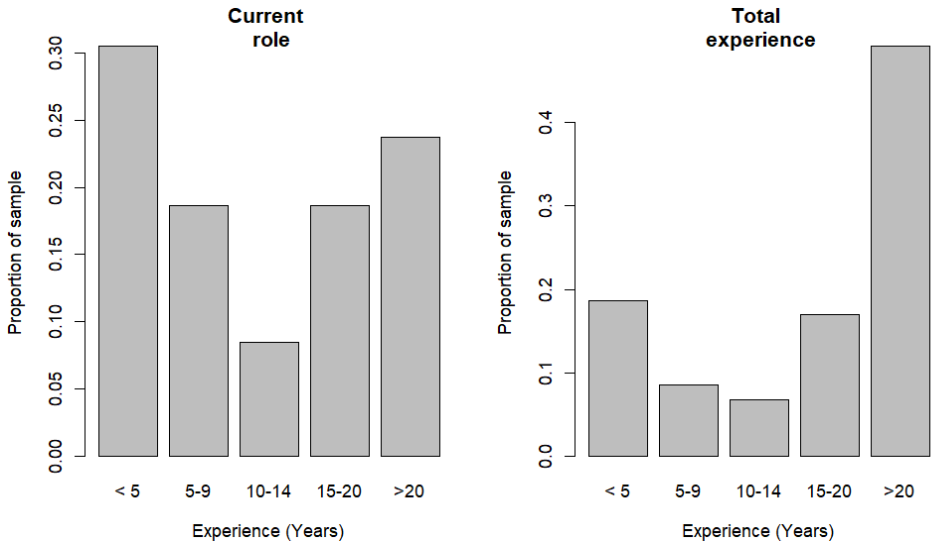


Figure 22: Experience in a) current role and b) total experience

High-level objectives

The relative importance of the high-level ecological, economic and social objectives varied both within and between stakeholder groups (Figure 23, Table 14). Both fisheries and MPA manager respondents tended, on average, to place higher weight on ecological outcomes. In contrast, respondents from commercial and recreational fisher organisations tended to place higher weight on economic

outcomes. These weightings align with the goals or objectives of these stakeholder groups. Social objectives were rated as the least important, with the exception of MPA managers respondents and "other" stakeholders respondents. This is consistent with the findings of previous studies on Australian fisheries objectives (Pascoe et al., 2019).

Table 14. Higher level objective means and standard deviations

Group	Ecological		Economic		Social	
	Mean	Std	Mean	Std	Mean	Std
MPA Manager	0.521	0.153	0.216	0.089	0.263	0.092
Fisheries Manager	0.519	0.139	0.265	0.124	0.217	0.114
Commercial fisher organisations	0.371	0.198	0.380	0.125	0.248	0.129
Recreational fisher organisations	0.317	0.131	0.384	0.149	0.299	0.129
Others	0.637	0.158	0.161	0.095	0.203	0.069

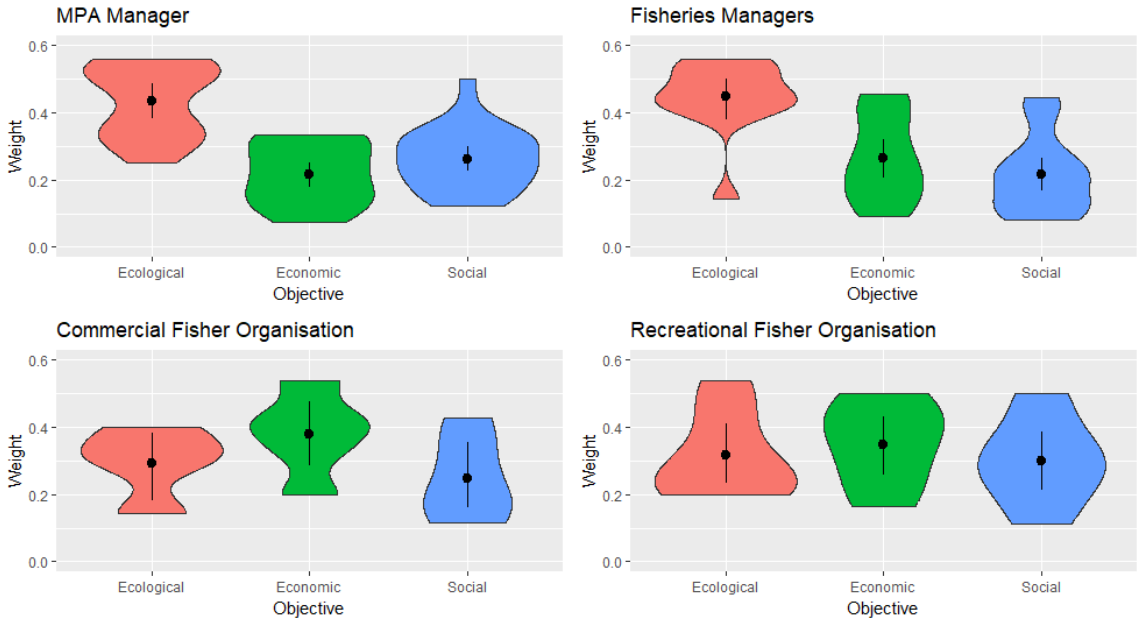


Figure 23: Distributions of importance scores for ecological, economic and social objectives by key stakeholder group.

Sub-objectives

Ecological

Ecological sub-objectives were mostly evenly distributed, with their weights generally similar across respondents from each of the stakeholder groups (Table 15). Respondents from commercial fisher organisations, had a higher preference for protecting unique habitats (Obj 1.1), while both MPA managers and recreational fisher organisations had a slightly higher preference for maximising habitat representation (Obj 1.2) (Figure 23).

Biodiversity (Obj 1.4) was rated quite highly across all groups. Contrary to a priori expectations, Protection of TEP, keystone and other species (Obj 1.6, 1.7, and 1.8, respectively) were given lower prioritisation by MPA managers (Figure 23).

Climate Change Resilience (Obj 1.5) was of greater importance to commercial fisher organisation compared to all other respondent groups. Maximising resilience to regional growth pressures (Obj 1.9) was also weighted relatively highly by commercial fisher organisation, with respondents from Queensland commercial fishing organisations (rating this objective as 9/9 importance) (Figure 23).

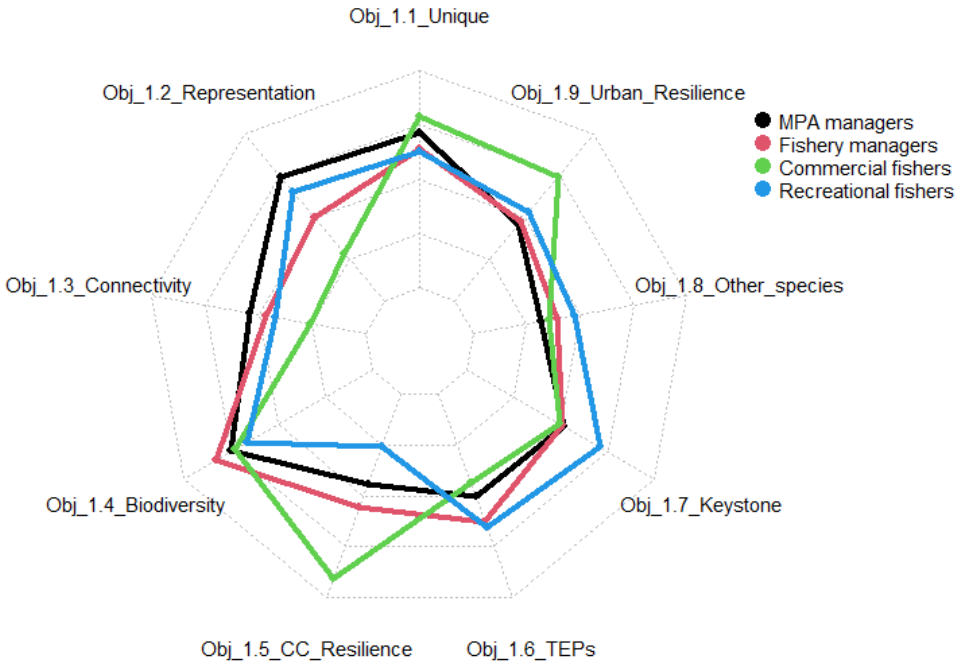


Figure 24: Relative importance of each ecological sub-objective.

Economic

Weightings of economic sub-objectives by survey respondents more closely reflected the interests and perceived responsibilities of their respective stakeholder group and is reflected in the differences in preferences in the individual economic sub-objectives (Figure 24**Error! Reference source not found.**).

MPA managers weighted the maximisation of economic benefits to Traditional Owners as the most important economic sub-objective (Obj 2.7). In general, MPA managers tended to have a higher preference for ensuring benefits to other users (Obj 2.4, 2.5 and 2.7), reflecting their broader management responsibilities.

Commercial fisher organisations and fisheries managers had a strong preference for minimising impact on their fisheries (Obj 2.1). This aligns with the perception of commercial fisher's that MPAs potentially impeded upon or negatively affect fishing activities and cause regional economic impacts (McNeill et al. 2018).

Objective 2.2 (improve resource condition for commercial fishing) was considered of higher importance compared by fisheries managers respondents, followed by recreational fisher organisations and commercial fisher organisations and least important by MPA managers.

Recreational fisher organisations had a strong preference for minimising flow-on impacts on supporting industries (Obj 2.6). Other users (e.g., tourism operators) were not considered in the survey, but were considered important by MPA managers. This explains the more "balanced" view of MPA managers who should account for the impacts on a broader range of stakeholders in their decision making.

Other economic objectives that survey respondents considered important in protected area planning (Survey Question 33) included minimising government red tape to encourage recreational fishing activities and incentivising more locally caught seafood trade.

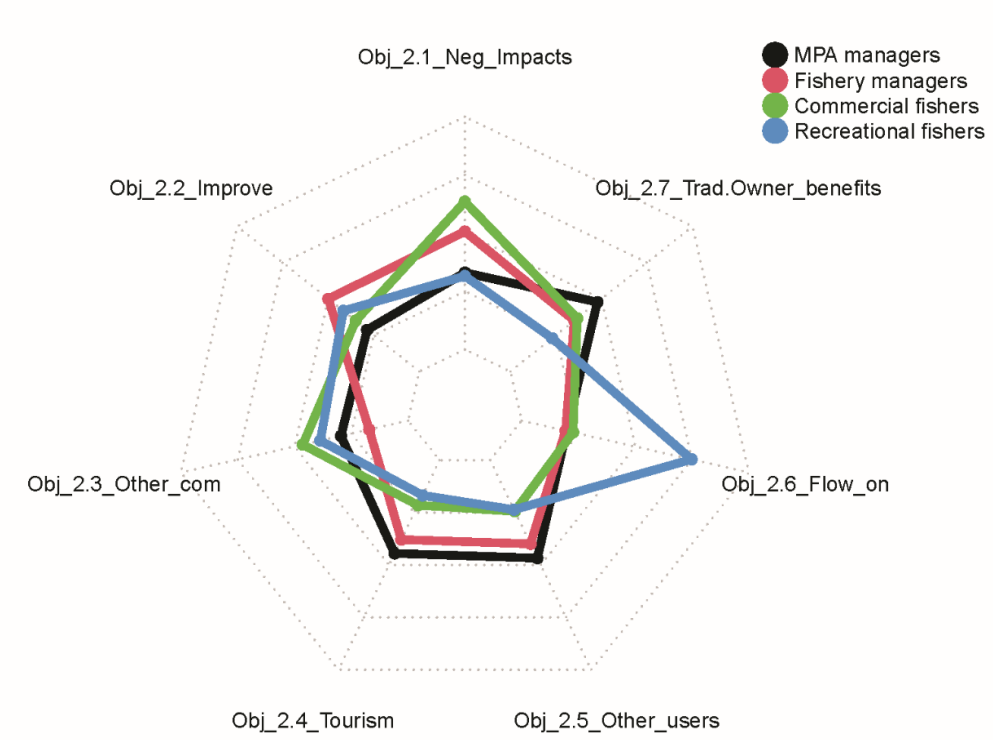


Figure 25: Relative importance of each economic sub-objective.

Social

Compared to ecological and economic sub-objectives, the relative importance of different social sub-objectives was substantially more varied across stakeholder groups (Figure 26).

Maximising social benefits to Traditional Owners (Obj 3.8) was weighted highest by MPA managers and other stakeholder respondents. This is consistent with the result reported earlier, for these stakeholder respondents who prioritise the maximisation of economic benefits to Traditional Owners (Obj 2.7, **Error! Reference source not found.**).

Both commercial and recreational fisher organisations placed higher weight on minimising impacts on local communities (Obj 3.7), reducing conflicts (Obj 3.3) and ensuring seafood supplies to local communities (Obj 3.6). This is consistent with the economic priorities of these stakeholder groups, including the protection of supporting industries discussed earlier. Both fisheries managers and recreational fisher organisations gave a higher weight to improving recreational fishing resources (Obj 3.2).

Unsurprisingly, commercial fisher organisation respondents placed low weighting on objectives related to improving recreational fishing experiences (3.2 and 3.4). Recreational fisher organisation respondents also weighted objectives allied to a broader range of recreational users low (Obj 3.4), whilst weighing objectives that directly impacted recreational fishing highly (Obj. 3.1 and 3.2). Respondents in both these groups ranked maximising lifestyle improvements low (Obj. 3.5) whilst ranking minimising negative impacts of the local community (Obj. 3.7) reasonable high. This appears to be contradictory result and may reflect an interpretation of negative impacts as economic (monetary) rather than social.

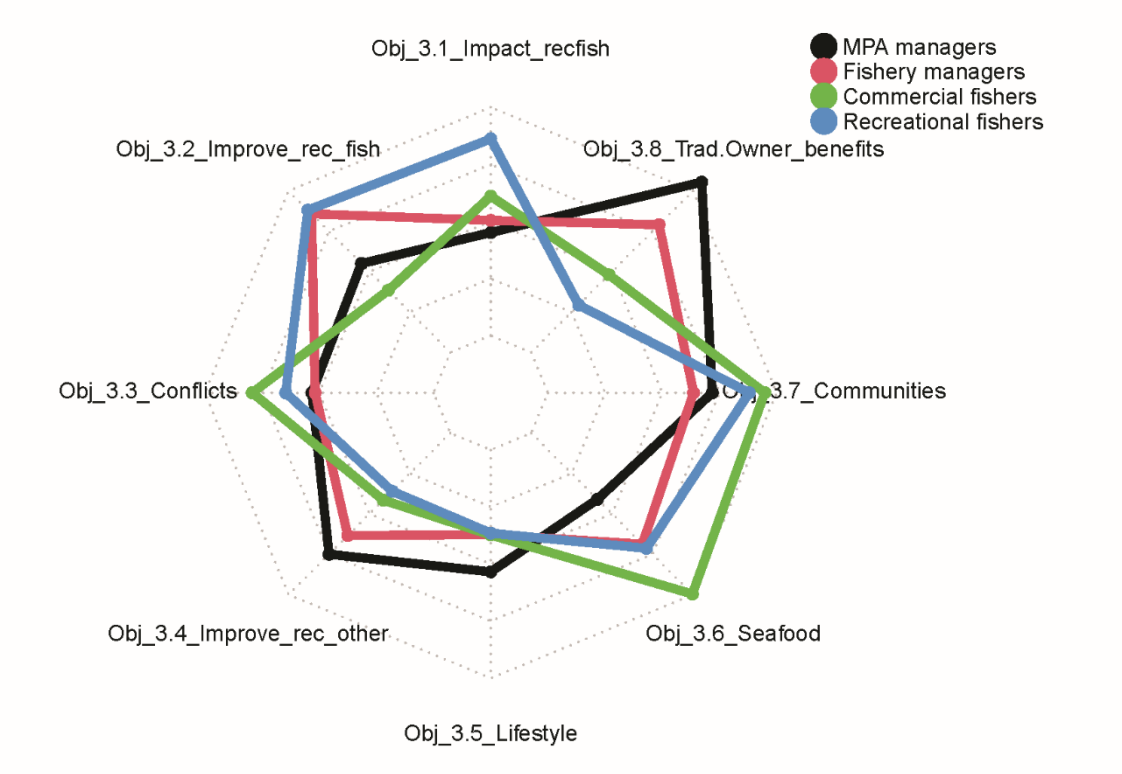


Figure 26: Relative importance of each social sub-objective.

Table 15. Importance weights for each ecological sub-objective

		MPA managers		Fisheries managers		Commercial fishers		Recreational fishers		Other	
Obj		Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
1.1	Maximise protection of unique habitats	0.149	0.088	0.134	0.062	0.235	0.092	0.131	0.077	0.100	0.126
1.2	Maximise habitat representation	0.159	0.128	0.092	0.056	0.067	0.011	0.140	0.098	0.218	0.068
1.3	Maximise connectivity of habitats	0.087	0.029	0.086	0.048	0.020	0.004	0.087	0.037	0.112	0.039
1.4	Maximise effective biodiversity protection	0.158	0.052	0.173	0.095	0.170	0.119	0.142	0.074	0.190	0.079
1.5	Maximise climate change resilience	0.091	0.045	0.123	0.087	0.134	0.048	0.060	0.052	0.062	0.050
1.6	Maximise TEP species protection	0.094	0.052	0.121	0.075	0.117	0.074	0.133	0.061	0.104	0.078
1.7	Maximise keystone species protection	0.105	0.050	0.095	0.035	0.092	0.058	0.126	0.095	0.098	0.067
1.8	Maximise protection of other species	0.068	0.038	0.062	0.034	0.084	0.046	0.088	0.038	0.070	0.073
1.9	Maximise resilience to regional growth pressures	0.091	0.053	0.115	0.074	0.081	0.053	0.092	0.059	0.047	0.028

Table 16. Importance weights for each economic sub-objective

		MPA managers		Fisheries managers		Commercial fishers		Recreational fishers		Other	
Obj		Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
2.1	Minimise negative impacts on commercial fishing	0.132	0.080	0.203	0.092	0.254	0.176	0.126	0.092	0.119	0.058
2.2	Improve resource condition for commercial fishing	0.115	0.053	0.200	0.124	0.140	0.128	0.167	0.076	0.165	0.055
2.3	Minimise negative impacts on other commercial operations	0.118	0.086	0.069	0.050	0.185	0.140	0.154	0.100	0.094	0.057
2.4	Improve opportunities for nature-based tourism businesses	0.178	0.069	0.152	0.114	0.086	0.040	0.067	0.028	0.185	0.105
2.5	Improve opportunities for other economic uses	0.187	0.088	0.160	0.100	0.098	0.051	0.095	0.046	0.183	0.068
2.6	Minimise impacts on support industries	0.079	0.039	0.077	0.054	0.091	0.047	0.299	0.133	0.067	0.043
2.7	Maximise economic benefits to Traditional Owners	0.191	0.074	0.140	0.087	0.147	0.071	0.092	0.054	0.188	0.076

Table 17. Importance weights for each social sub-objective

		MPA managers		Fisheries managers		Commercial fishers		Recreational fishers		Other	
Obj		Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
3.1	Minimise negative impacts on recreational fishing	0.090	0.042	0.101	0.052	0.122	0.100	0.172	0.077	0.103	0.043
3.2	Improve resource condition for recreational fishing	0.110	0.058	0.172	0.116	0.077	0.043	0.176	0.081	0.145	0.048
3.3	Reduce conflicts between different user groups	0.106	0.054	0.104	0.082	0.159	0.095	0.129	0.073	0.118	0.048
3.4	Improve quality of other recreational experiences	0.150	0.072	0.127	0.098	0.083	0.050	0.072	0.060	0.138	0.078
3.5	Maximise lifestyle opportunities	0.107	0.059	0.074	0.054	0.075	0.060	0.073	0.040	0.079	0.040
3.6	Minimise loss of fresh fish/seafood to local markets and consumers	0.082	0.044	0.138	0.095	0.199	0.070	0.143	0.101	0.093	0.047
3.7	Minimise negative impacts on local communities	0.144	0.053	0.127	0.063	0.190	0.082	0.176	0.077	0.102	0.043
3.8	Maximise social benefits to Traditional Owners	0.211	0.079	0.158	0.073	0.096	0.056	0.059	0.038	0.224	0.123

Table 18. Final relative importance weights for each sub-objective

Obj		MPA managers	Fisheries managers	Commercial fishers	Recreational fishers
Ecological		0.521	0.519	0.371	0.317
1.1	Maximise protection of unique habitats	0.077	0.069	0.087	0.041
1.2	Maximise habitat representation	* 0.083	0.048	0.025	0.044
1.3	Maximise connectivity of habitats	0.045	0.045	0.007	0.028
1.4	Maximise effective biodiversity protection	0.082	* 0.090	0.063	0.045
1.5	Maximise climate change resilience	0.047	0.064	0.05	0.019
1.6	Maximise TEP species protection	0.049	0.063	0.043	0.042
1.7	Maximise keystone species protection	0.055	0.049	0.034	0.040
1.8	Maximise protection of other species	0.035	0.032	0.031	0.028
1.9	Maximise resilience to regional growth pressures	0.047	0.06	0.03	0.029
Economic		0.216	0.265	0.38	0.384
2.1	Minimise negative impacts on commercial fishing	0.029	0.054	* 0.097	0.048
2.2	Improve resource condition for commercial fishing	0.025	0.053	0.053	0.064
2.3	Minimise negative impacts on other commercial operations	0.026	0.018	0.07	0.059
2.4	Improve opportunities for nature-based tourism businesses	0.038	0.04	0.033	0.026
2.5	Improve opportunities for other economic uses	0.040	0.042	0.037	0.036
2.6	Minimise impacts on support industries	0.017	0.021	0.034	* 0.115
2.7	Maximise economic benefits to Traditional Owners	0.041	0.037	0.056	0.035
Social		0.263	0.217	0.248	0.299
3.1	Minimise negative impacts on recreational fishing	0.024	0.022	0.03	0.052
3.2	Improve resource condition for recreational fishing	0.029	0.037	0.019	0.053
3.3	Reduce conflicts between different user groups	0.028	0.022	0.039	0.039
3.4	Improve quality of other recreational experiences	0.039	0.028	0.021	0.022
3.5	Maximise lifestyle opportunities	0.028	0.016	0.019	0.022
3.6	Minimise loss of fresh fish/seafood to local markets and consumers	0.022	0.03	0.049	0.043
3.7	Minimise negative impacts on local communities	0.038	0.028	0.047	0.053
3.8	Maximise social benefits to Traditional Owners	0.055	0.034	0.024	0.018

* Shows most highly weighted overall sub-objective for each stakeholder group

Table 19: Top five most highly ranked sub-objectives by each stakeholder group based on final relative importance weights

Rank	MPA managers	Fisheries managers	Commercial fishers	Recreational fishers
1	Maximise habitat representation	Maximise effective biodiversity protection	Minimise negative impacts on commercial fishing	Minimise impacts on support industries
2	Maximise effective biodiversity protection	Maximise protection of unique habitats	Maximise protection of unique habitats	Improve resource condition for commercial fishing
3	Maximise protection of unique habitats	Maximise climate change resilience	Minimise negative impacts on other commercial operations	Minimise negative impacts on other commercial operations
4	Maximise keystone species protection	Maximise TEP species protection	Maximise effective biodiversity protection	Minimise negative impacts on local communities
5	Maximise social benefits to Traditional Owners	Maximise resilience to regional growth pressures	Maximise economic benefits to Traditional Owners	Improve resource condition for recreational fishing

Weighted means

The sub-objective weights can be weighted by the overall mean for each of the higher-level objectives in order to produce cross-objective comparisons.

Ecological objectives dominated the top 5 most important objectives overall for MPA managers and fisheries managers respondents (Table 18). Sub-objectives of greatest importance to commercial fisher organisations respondents were mixed economic and ecological. Overall, recreational fisher organisations respondents have the most divergent sub-objective priorities compared to other stakeholder group respondents. Maximising benefits to Traditional Owners and First Nations Groups (both economic and social) remains one of the most contested objectives; ranking very highly for MPA manager respondents and commercial fisher organisations. Despite high-level social objectives being ranked the lowest overall by recreational fisher organisations respondents (Table 14, Figure 22, Economic > Ecological > Social), social sub-objectives appear in the top 5 most important sub-objectives (Table 19). This again demonstrates the value of using scaled importance weights in stakeholder studies.

Inshore versus offshore PPAs

Respondents in the MPA manager stakeholder group (n=23) were split into inshore and offshore managers (13:10, inshore:offshore participant split) to explore potential differences in higher level- and sub-objectives (Figure 26). A priori expectations were that inshore MPA managers would give greater weight to economic and social objectives as there would be more human activity in these MPAs than offshore MPAs. This seems to be the case, with inshore MPA manager respondents giving greater weight to social and economic objectives than their offshore counterparts. Overall, there was little difference in the weights of each of the sub-objectives between inshore versus offshore manager respondents.

Inshore MPA manager respondents weighted biodiversity (Obj 1.4) and protection of keystone species (Obj 1.7) higher than offshore MPA managers respondents. Offshore manager respondents favoured high-level ecological objectives, prioritising habitat representation (Obj 1.2) and protection of unique habitats (Obj 1.1). Climate Change and Urban Resilience (Obj 1.5 and Obj 1.9, respectively) were both weighted relatively poorly by both inshore and offshore groups. Minimising flow-on effects to support industries (Obj 2.6) was not prioritised highly by either group, while respondents in both groups prioritised benefits to Traditional Owners (Obj 2.7). Maximisation of social benefits to Traditional Owners (Obj 3.8) was also the most highly weighted social sub-objective for both inshore and offshore, reflecting the high importance placed on this by MPA manager respondents overall (Table 14). Minimising loss of seafood (Obj 3.6) was the most poorly weighted by respondents in both groups (Figure 27).

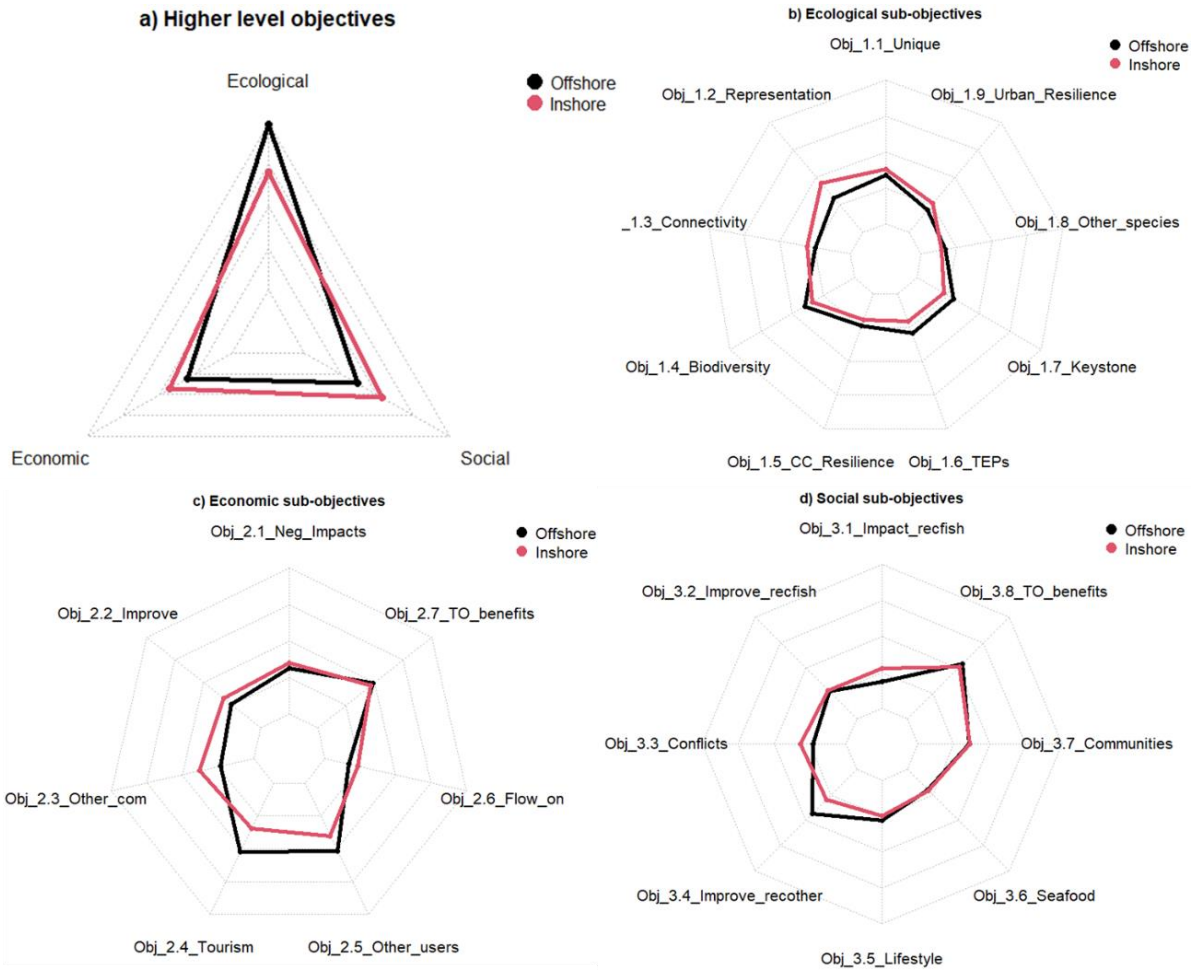


Figure 27: Inshore and Offshore MPA objective importance

Process objectives

Process objectives refer to how the management outcomes are to be achieved as well as complementary activities. These are often considered as institutional or governance related objectives that may sit beside the standard triple bottom line objectives (Stephenson *et al.* 2017). As with individual sub-objectives, process objective weights reflected what each group perceived to be within their control and their realm of responsibility. Compliance (P.07), monitoring (P.10) and coordination (P.05) were generally the highest weighted process objectives across all groups (Table 20). Equity was of greater importance to fisheries-manager respondents than MPA manager respondents. Coordination was slightly more important to fisheries manager and commercial fisher organisations respondents compared with MPA manager respondents. Recreational fisher organisations respondents weighted community stewardship (P.04) very highly, but increasing nature appreciation through education (P.03) was weighted very poorly. Social and economic benefits for Traditional Owners were consistently weighted highly by MPA manager respondents, but the process objective of ensuring First Nations objectives, uses and needs are considered (P.02) was weighted quite poorly here by MPA managers respondents. Minimising regulatory burdens (P.08) was considered relatively unimportant by all group respondents, including MPA and fisheries managers group respondents (Figure 28).

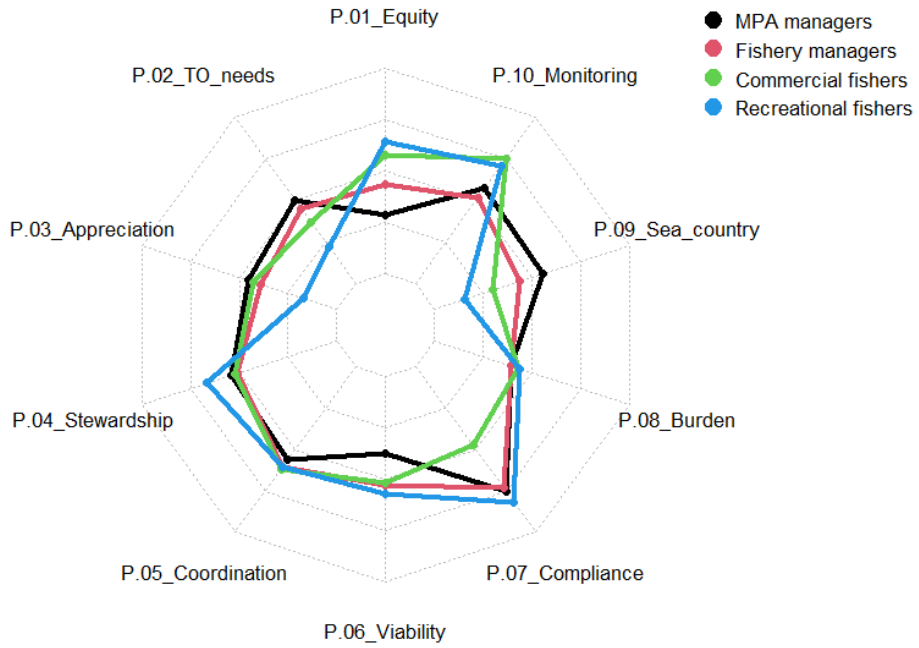


Figure 28: Relative importance of process objectives

Additional comments

At the end of the survey participants were asked to share any final comments they may have had, addressing points of importance that may not have been covered in the survey. Here, the most frequently made comment was objection to partially protected areas (38% of all responses). One respondent commented:

"One out, all out. If you truly want to have green zones [and] no fishing zones, then all human activity should be banned from that area… a true sanctuary zone from human impact" (*Recreational fishing organiser from NSW jurisdiction)*

Another frequently shared concern was the need for clearly communication and markings of protected area boundaries (15% of responses).

"I have worked for an extended period in compliance prosecuting offenders for activities undertaken in protected areas. Areas need to be easily identifiable and where possible, be square or marked by straight lines. Areas identified by depth at LAT are not easy for the public to understand and it creates confusion. Areas should be simple square or rectangular shapes marked on all 4 corners" *(Fisheries manager, Queensland jurisdiction)*.

Table 20. Importance weights for each process objective

		MPA managers		Fisheries managers		Commercial fisher organisations		Recreational fisher organisation		Other	
Obj		Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
P.01	Ensure all users have equitable access	0.057	0.037	0.087	0.076	0.116	0.092	0.129	0.084	0.065	0.057
P.02	Ensure First Nations objectives, uses and needs are considered	0.100	0.047	0.090	0.067	0.073	0.047	0.044	0.031	0.094	0.044
P.03	Increase nature and conservation appreciation through education during planning	0.091	0.041	0.078	0.052	0.086	0.097	0.035	0.012	0.115	0.067
P.04	Maximise community stewardship and buy-in through engagement with stakeholders	0.108	0.054	0.102	0.030	0.104	0.068	0.133	0.073	0.112	0.033
P.05	Coordination with other Government agencies (e.g., fisheries managers, tourism and transport agencies, local councils)	0.113	0.054	0.121	0.069	0.124	0.038	0.121	0.038	0.094	0.066
P.06	Ensure a viable mix between fully and partially protected zones	0.075	0.045	0.106	0.055	0.104	0.102	0.115	0.062	0.078	0.039
P.07	Ensure management and compliance resources are adequate	0.150	0.055	0.146	0.068	0.095	0.050	0.163	0.029	0.112	0.053
P.08	Minimise regulatory burden and encourage behavioural change as a management tool	0.079	0.029	0.078	0.047	0.087	0.061	0.088	0.055	0.080	0.046
P.09	Facilitate effective management of sea country by Traditional Owners and First Nations Groups within protected areas	0.112	0.048	0.088	0.057	0.060	0.048	0.031	0.017	0.125	0.069
P.10	Ensure adequate programs are in place to monitor and evaluate management effectiveness	0.115	0.039	0.104	0.048	0.151	0.081	0.142	0.031	0.126	0.070

Discussion

The purpose in the study was to establish the range of potential ecological, economic and social impacts of partially protected areas and fisheries-focused closures, as well as examine potential frameworks in which different management options could be assessed taking into consideration these impacts. We focused primarily on economic and social values, with the former also capturing many of the ecological benefits generated by the PPAs.

Decision making frameworks

We considered five frameworks for assessing the potential benefits of PPAs. The frameworks considered differed largely in how they applied economic values to ecosystem services, which underpinned the other four to varying degrees. Ecosystem services can be classified as provisioning, regulating or cultural services. These services overlap with the triple bottom line, with provisioning and regulating services largely relating to ecological and economic benefits, while cultural services overlap social and economic benefits.

The total economic value (TEV) framework considers these services as either use or non-use, and derives economic values for each service. A range of different valuation approaches have been developed to assess each ecosystem service in monetary terms. This allows costs and benefits of different management options to be compared.

Multicriteria decision analysis (MCDA) similarly allows different management options to be compared, with the outcomes against each criterion (e.g., management objective) expressed in a range of different units. We suggested the use of data envelopment analysis (DEA) as a means of comparing the different management options. DEA has many advantages over "traditional" MCDA approaches, particularly when the criteria/objectives are measured in different units. The approach can incorporate ecosystem services as well as changes in environmental assets and can be applied as a triple bottom line approach or ecosystem services approach (or some combination of the two).

Ecosystem accounting (EA) has attracted attention more recently as a means of allowing changes in environmental assets and the services they produce to be captured in a way analogous to the system of national accounts. EA is a useful tool to monitor how environmental assets and services are changing in both monetary and non-monetary terms. They are less useful as a decision-making framework, other than to identify areas of concern that may need management intervention.

Wellbeing frameworks conceptually provide a means of capturing many of the social outcomes that are less easy to enumerate in the other frameworks. However, as a consequence of this, comparing different outcomes of different management options is difficult as a result of this non-commensurability of key measures. Further, developing models or other approaches to estimate how these measures may change under different circumstances is also problematic.

Values associated with MPAs and PPAs

A considerable number of studies have been undertaken with respect to economic valuation of the use and non-use benefits of both MPAs and PPAs. These have largely focused on commercial fisheries, and the impacts of the PAs on fishing revenues and costs. Several studies were identified that aimed at considering the non-market values associated with protection of bycatch species. Cultural values including recreational fishing and tourism have also been extensively studied.

As objectives of PPAs vary, a general assessment of their benefits and costs is not possible. Studies have also been based on the main objectives that the PA aimed at achieving. For example, commercial and recreational fisheries benefits have only been estimated when this was an explicit objective of the PA. Costs to these, or other, sectors have not been systematically assessed when they were not considered a primary reason for the PA.

In contrast to PAs, both positive and negative impacts of fishery-specific closures have been estimated. Most studies have concluded that fisheries benefit directly from these closures either through larger stocks or more valuable size at first capture, or indirectly through avoiding potentially larger losses if closed due to bycatch issues.

Social aspects have also been given consideration with respect to fishery-specific closures. Area closures have been associated with shifting effort distributions with implications for onshore communities. Similarly seasonal closures have been associated with impacts on incomes during the closed season, with implications for fishing families, as well as processors and others dependent on a flow of product for their livelihood.

Economic impacts of MPAs, PPAs and fisheries closures

Evidence of substantial economic impact of MPAs and PPAs is limited. Most MPAs aim to minimise the impact on other users in their design, with the designation of PPAs aimed at achieving this mix of conservation and economic use of the marine resources. A key driver of economic impacts is the displacement of fishing effort. This has several possible consequences. Assuming profit-maximising behaviour of fishers, then the areas that they initially choose to fish are those that they expect to gain the most benefits from. Displacing this fishing effort is therefore expected to result in lower profits as a result of either lower catch rates or higher costs of access (e.g., the new fishing grounds are further from their home port). Displaced fishing effort also has consequences for fishers already operating in these areas, with the additional crowding reducing their catch rates or increasing their costs. Despite this, evidence of reduced profits as a result of MPAs is limited. In most cases, the designation of MPAs is accompanied by a buyback program to remove the otherwise displaced fishing effort, which may obfuscate the effects of the MPA per se on the economic performance of the fleets.

Conversely, theoretical models of MPAs suggest that the spill-over effect from marine reserves may actually enhance fisheries profits through increases in catch rates. While some evidence of these spillover effects exist, these tend to be relatively localised and close to the MPA borders (Di Lorenzo *et al.* 2016; Di Lorenzo *et al.* 2020). Goñi *et al.* (2010) suggest that these spillover effects may, in many cases, be sufficiently large to offset the effects of reduced fishing ground.

Fishery-related closures are generally designed to either improve stock conditions (e.g., through protecting juveniles, spawning stocks or small fish) or reduce bycatch. In the case of the stock-related closures, these (if successful) improve the sustainability of the resource with subsequent benefits to fishers in terms of higher catch rates (or catch of a more valuable size). Bycatch-related closures are aimed at reducing the extent of negative externalities associated with fishing. While these may negatively impact fishers' economic performance, they correct a market failure that would otherwise not be captured in the economics of the fishery.

Social impacts of MPAs, PPAs and fishery closures

Research on MPAs and PPAs has mainly focused on ecological and, to a lesser extent, economic impacts, often treating social and cultural impacts as secondary. Systematic reviews on social impacts of MPAs often do not differentiate between impacts in developed and developing countries. As a result, findings are biased toward developing countries where the majority of studies originate. These studies

have largely focused on livelihoods (mostly income), with limited attention to broader social impacts such as wellbeing, cultural identity, and health.

Where social implications of MPAs have been considered, these have been assumed linked to ecological outcome. That is, they are typically assumed to be positive if ecological outcomes are positive (and vice versa). Implicit in these studies is the assumption that communities benefit with improvements in ecological conditions, either directly through livelihoods or aesthetic values associated with a more natural environment. Positive impacts, such as improved environmental quality, regional tourism growth, or enhanced ecosystem services, tend to be realized collectively, at a community or regional level. Protecting the marine environment itself may also provide intrinsic benefits to communities; Raabe *et al.* (2024) found that most Australians would like to see substantially increased area of MPAs with a focus on protecting marine life rather than for economic or recreational reasons.

Conversely, where social impact of fishery closures (for fishery management purposes) have been considered, these have largely been negative. Reduced incomes and displaced effort imposes costs on the local communities as well as fishers themselves. Given that many MPAs also result in displaced fishing effort, improvements in community wellbeing at a community level may be at the expense of reduced wellbeing at the individual level (i.e. commercial fishers. Negative impacts, such as job losses, stress, reduced access to fishing grounds, are borne by individuals—especially those directly dependent on natural resources, like small-scale or commercial fishers. This indicates a disconnect between ecological goals and individual wellbeing, underscoring the need for more equitable planning and support for those who experience the brunt of the negative changes.

Generalising the social impacts of MPAs and PPAs has been made more difficult due to inconsistent measurement of social Impacts. Studies often use different methodologies, making it difficult to draw comparisons. For example, terminology varies, with “wellbeing,” “social benefit,” “livelihood,” and “cultural values” used interchangeably or inconsistently. Social outcomes are sometimes inferred rather than measured directly (e.g., ecological improvement implies social improvement), and when measured, they often focus on a limited subset of the population (e.g., commercial fishers or the local community), missing broader societal effects. The choice of perspective may distort the reported perception of the social impact. While wellbeing is the dominant framework applied (Fudge *et al.* 2021; Phillips *et al.* 2025), composite wellbeing indicators are missing or underutilized, and when used, they vary between studies, weakening cross-comparability. This reveals a gap in standardised social monitoring in MPA and PPA assessment frameworks.

Cultural benefits in particular are difficult to measure. The cultural benefits associated with engaging with marine environments are broad and diverse, including spiritual, recreational, educational, and aesthetic experiences. These values vary across stakeholders and regions. For instance, In Australia, users of MPAs formed a diverse mix of identities (from surfers to snorkellers) and value systems, challenging the assumption of homogenous interest groups. This diversity of cultural values makes it difficult to standardize impact assessment but also stresses the importance of inclusive stakeholder engagement.

Studies considering the inclusion of Indigenous and First Nations Perspectives are also limited. Although Indigenous peoples are recognised as key stakeholders in conservation areas, their participation in decision making is often overlooked. MPAs have the potential to deliver cultural and economic benefits to Indigenous communities—through protection of culturally important sites, enhanced resource availability, and ecotourism opportunities (Rist *et al.* 2019).

The wellbeing framework has been applied in many studies to capture social impacts. Multiple studies, particularly in the UK and Australia, have investigated how MPAs impact the subjective wellbeing of commercial fishers. These studies often go beyond just income (objective wellbeing) and explore

perceptions, satisfaction, stress, and conflict (subjective wellbeing). For example, in Lyme Bay (UK), trawlers experienced higher material losses and lower wellbeing scores as a result of displacement from the inshore MPA, pointing to long-term dissatisfaction (Rees *et al.* 2021). Similarly in Australia's fishers displaced by MPAs reported greater stress, loss of connection with others in the fishing community and subsequently lower subjective wellbeing (Gollan and Barclay 2021). In general, mobile gear fishers (i.e.., trawlers) report lower levels of subjective wellbeing compared to static gear fishers (e.g., pot fishers or line fishers), even in cases where income may have increased. This suggests that economic gains do not necessarily improve personal wellbeing, and that management decisions need to consider these nuanced impacts.

A standard method to assess potential trade-offs in outcomes within the wellbeing framework, however, is currently lacking. The wellbeing framework offers a means of capturing outcomes of past decisions, but do not provide a means to choose between alternative future management options. The remaining challenge then is to engage with stakeholders to develop social objectives that are linked to PPA outcomes, then developing management options to achieve these and maximise triple-bottom-line outcomes (Cannard, 2023). Potentially, aspects of multicriteria decision analysis could be incorporated into the wellbeing framework to provide a more quantitative assessment of different management options. For example, the Analytic Hierarchy Process (AHP) can be assess trade-offs, and linked with a Bayesian Belief Network (BBN), which have been successfully used in fisheries modelling to incorporate social drivers (Soma 2003; Little *et al.* 2004; van Putten *et al.* 2013; Pascoe *et al.* 2017).

Trade-offs are a common challenge in natural resources conservation and in the social sciences (Cheung and Sumaila 2008; Bennett *et al.* 2017; Bennett *et al.* 2018), ecosystem services and economics (Braat and de Groot 2012), and most recently, ecosystem accounting (Keith *et al.* 2017). In the MPA context, Seguin *et al.* (2023) recommend clearer assessments of trade-offs (and frameworks) can assist in achieving biodiversity conservation priorities while also factoring in the social and economic constraints.

Experiences: Moreton Bay Marine Park

The rezoning of Moreton Bay Marine Park had the multiple objectives of improving conservation and environmental values while also enhancing benefits from the use of Moreton Bay by the local community. To this end, minimising the impact on commercial and recreational fishing (if not enhancing these sectors) was a key consideration.

A review of ecological outcomes suggest that the biophysical objectives and ecosystem integrity have largely benefited from the rezoning. The removal of fishing by no-take zones in the MBMP has been shown to be effective in increasing abundance of key harvested fish and fish diversity in coral reefs (Olds *et al.* 2012a; Olds *et al.* 2012c; Yabsley *et al.* 2016), seagrass meadows (Pillans 2006; Pillans *et al.* 2007; Henderson *et al.* 2017), and mangroves (Goodridge Gaines *et al.* 2020), but not particularly in other habitats (e.g., estuaries (Gilby *et al.* 2017), rocky reefs (Terres *et al.* 2015; Haywood *et al.* 2019), and surf zones (Ortodossi *et al.* 2018). When effective, no-take zones can boost key ecological functions within these zones and contribute to the health of benthic ecosystems (Olds *et al.* 2012a; Olds *et al.* 2012c), improving the capacity of ecosystems to resist environmental disturbance (Olds *et al.* 2014).

The performance of no-take zones within the MBMP, depends on a variety of factors, including their connectivity with other (protected) habitats (Olds *et al.* 2012a; Gilby *et al.* 2016a; Henderson *et al.* 2017), position in relation to the water-quality gradient across the Bay (Gilby *et al.* 2016a; Henderson *et al.* 2017), the ecological function of the species within the protected zone (e.g., browsing versus grazing herbivore fish species) (Yabsley *et al.* 2016), resource type (e.g., turf algae versus macroalgae) (Yabsley *et al.* 2016), species' range of movement (across-boundary fishing pressure), habitat type,

and size of protected zone (Gilby *et al.* 2017; Ortodossi *et al.* 2018). One important limitation of previous studies has been the extent of observed effects by the current MBMP no-take zones in relation to the time of data collection, the year of establishment of the protection zones, and the life span of study species. In contrast to crab species, many targeted fish species (e.g., Snapper, and Spangled Emperor) have a long-life span and mature at relatively late in life and hence data documenting the effect of no-take zones on fish populations need to be collected over a greater time span (Grandcourt *et al.* 2006; Marriott *et al.* 2010; Andrews *et al.* 2011). For example, data collected between 2009 and 2011 revealed increased catch rates and mean size of Yellowfin Bream and Giant Mud Crabs within 'old' inshore no-take zones compared to 'new' no-take zones, and to 'fished' areas, including similar habitat types (Haywood *et al.* 2019).Similarly, catch rates and mean weight of snapper steeply increased in the 'new' inshore no-take zone at St Helena Island (Haywood *et al.* 2019). Sand crabs, however, did not seem to respond to the protection over several years by inshore no-take zones (Haywood *et al.* 2019).

Most research on the effect of no-take zones in the MBMP has focused on harvested herbivorous fish and their ecological function on coral reefs, followed by seagrass meadows, whilst the performance of no-take zones on other habitats (e.g., estuaries, mangroves, offshore deep reefs, surf zones, and unconsolidated soft sediments) and ecological functions (e.g., scavenging and predation) remain less well-studied. Although there exists some evidence for spill-over effects by no-take zones on targeted species within the Bay (Pillans *et al.* 2005; Pillans 2006; Pillans *et al.* 2007), this topic remains largely unstudied. Also, it remains uncertain how fishing pressure outside protected zones affects no-take zone performance; many species, particularly highly mobile species such as elasmobranchs, can move outside the boundaries of no-take zones during tidal, diel, or seasonal migrations, enduring risk from fishing (Kramer and Chapman 1999; Weeks *et al.* 2017). As such, an accurate evaluation of the full potential of no-take zones in the MBMP on species populations, species communities, and potential spill-over effects, including those of interest for fisheries, requires further future monitoring and research efforts.

The impact of rezoning to support the socio-economic objectives and change-and-benefit of extractive and non-extractive human use of Moreton Bay has been much less studied than the ecology of the Bay. The impact of the rezoning on the commercial fishing industry in Moreton Bay is also unclear. Unlike ecological outcomes, which have been extensively studied, only limited social and economic studies have been undertaken in the Bay, with most of these undertaken pre-zoning to provide estimates of the likely impacts of the rezoning. Expectations of income losses to commercial fishers resulted in a structural adjustment program accompanying the rezoning. The effect of this was to reduce the number of licences available to operate in Moreton Bay, although active licence numbers in Queensland in general were also declining due to economic pressures (e.g., prices and costs). The simple analysis in this study suggests that, post-rezoning, the number of prawn trawlers operating in Moreton Bay may have been higher than the counterfactual based on trends in the rest of Queensland, while the number of line and potting licences may have been similar with or without the marine park. Changes in catch rates of prawns was also statistically significantly higher post-rezoning, although this is more likely to be due to the two major flood events that occurred in 2011 and 2017 that had a positive impact on the banana prawn component of prawn catches within the Bay. Catches of fish and crabs were not significantly different post-restructuring than their previous average levels.

Assessing the economic impact of these changes in the fleets is not possible due to the lack of economic surveys undertaken in the region. van de Geer *et al.* (2013) suggested that fishing revenues were expected to decrease immediately following the re-zoning. Such a decrease is not supported by the catch and effort information provided in this review, although may still have occurred due to changes in prices and costs that are not related to the Marine Park. Separating out the effects of price change, cost change and productivity change on fleet economic performance requires a time series of economic information that is not available (i.e. through profit decomposition (e.g., Walden *et al.* 2022)).

Similarly, identifying the impacts of the Marine Park rezoning on recreational fisheries is also complex. Pascoe *et al.* (2014) estimated that spill-over effects from the no-take zone may increase non-market economic benefits to recreational fishers by around 5–12%. While catch rates were higher in Moreton Bay in 2010 compared to other Queensland regions, evidence of an increase is limited, and these perceived gains diminished by 2019. This suggests that recreational fisheries management and other drivers affecting recreational fishing participation may have played a larger role in changes in the fishery than the rezoning of the Marine Park. Determining an appropriate counterfactual is difficult with the limited data available.

While the economic analysis has focused on catch of recreational and commercial fisheries, users of the Marine Park often state other factors that affect their benefits. Over half of the boaters in the Marine Park do so to appreciate the natural beauty of the area, experience nature and for therapeutic reasons (e.g., relaxation and improved mental health) (Martin 2023). Similarly, Jones *et al.* (2016) found that over 93% of respondents to their survey valued experiences and interactions with nature as the key benefit of their use of Moreton Bay and the surrounding waterways, while 70% held deep appreciation for the natural beauty of the area. Hence, the Marine Park may have produced other benefits to other users.

In terms of achieving the original biophysical and socio-economic objectives, the rezoning of Moreton Bay appears to have improved the ecology of the Bay with minimal impact on recreational and commercial fishing. While there is no evidence that the Marine Park improved commercial and recreational fishing (i.e., due to spill-over effects), there is equally no evidence that the Marine Park resulted in an additional cost to these sectors.

Experiences: South-east Marine Reserve Network

A key difference between the south-east marine reserve network and Moreton Bay case studies was the volume of research activity that had taken place before and after the marine reserve designation and zoning. Moreton Bay has been extensively studies, mainly due to its coastal location (making access relatively easily), its geographically small area, its generally shallow water and proximity to a major metropolitan city, resulting in extensive (and intensive) use for both commercial and recreational reasons (and also availability of local research services through Universities and CSIRO). As a result, information on ecological, economic and social baselines were well established, and regular monitoring and research activities were frequent and relatively low cost.

In contrast, the south-east marine reserve network exists mostly offshore, with the different zones geographically dispersed and including areas of deep water. As a result, baselines were generally not as well established, and development of a monitoring program was required. While areas within the network were previously used for commercial fishing, the relatively small size of the protected areas compared with the range of the (highly mobile) key fisheries meant that assessing the impact of the closures on these activities was complex.

Despite these complexities, the available information suggests that the mix of fully and partially protected areas within the network have had little negative impact on the commercial fisheries sector, despite earlier predictions of potentially large impacts. This may have been helped by the buyback in 2007 as part of a broader fisheries policy initiative. As a result, the number of potentially impacted vessels was reduced, and the potential to reallocate fishing effort without substantial additional costs was improved. In this regard, the zoning could be considered successful in terms of minimising costs to existing users.

Whether ecological outcomes have been maximised is unclear. While a monitoring program has been established, the time lag between protection and observable benefits can be considerable, and it may be several more years before the ecological benefits of the protection are observable.

Objectives and objective importance

Balancing use and conservation in marine reserves is complex, as different stakeholder groups have different expectations about what is important in designing MPA and PPA. The range of potential MPA and PPA objectives were derived from the two workshops. While both groups worked under a different set of legislation and guiding principles, a common set of objectives could be identified that was consistent with the aims of both the Moreton Bay and Commonwealth marine park management. These covered the full range of triple bottom line objectives (i.e. ecological, economic and social objectives).

As seen in most other studies of objectives in marine resource management (e.g., Pascoe *et al.* 2019a), different stakeholder groups placed differing importance on the different objectives that were to be achieved. MPA and fisheries managers tended to place relatively similar importance on economic and social objectives, with ecological objectives on average being the most important (as might be expected). In contrast, commercial and recreational fishers both tended to place higher weight on economic and social objectives compared with the managers. Some conservation-focused stakeholder groups have argued that conservation benefits of no-take zones outweigh the benefits from PPA, arguing against, or limiting, their use (Hall et al. 2023). Others have argued that well-designed PPA may achieve economic and social benefits with little conservation loss (Bobiles and Nakamura 2019).

Opposition by fishers—both commercial and recreational—to MPAs usually focuses around potential negative impacts of restrictions, particularly regional economic and social losses (McNeill et al. 2018). This is reflected in the higher importance given by these stakeholders for these economic and social objectives. Other users (e.g., tourism operators) were not considered in the survey but were considered important stakeholder group by MPA managers. This, at least in part, may explain the more "balanced" view of MPA managers when including these stakeholders in their importance weightings.

Given this, the results of the survey were largely consistent with a priori expectations, with priorities for sub-objectives and process objectives closely reflecting the interests and perceived responsibilities of each stakeholder group. Within ecological objectives, Biodiversity (Obj 1.4) was rated quite highly across all group respondents. This may be attributed to biodiversity being a blanket concept often used to summarize multiple ecological objectives, allowing for a variety of interpretations by different individuals and stakeholder groups. For example, under the Guidelines for the Ecologically Sustainable Management of Fisheries (2007) biodiversity is defined as variability among living organisms and includes diversity within and between species, and diversity of ecosystems (Australian Government, 2007). Despite the high importance placed on biodiversity, specific biodiversity and conservation objectives (including Obj 1.6, 1.7, and 1.8) were given lower prioritisation by MPA manager respondents. This may again be attributed to biodiversity protection being a blanket concept that is perceived to include these other species-specific objectives.

Climate Change Resilience (Obj 1.5) was of greater importance to commercial fisher organisation respondents. As profit-seeking entities this is logical, as climate-related shocks will undoubtedly affect marine ecosystems and stocks, affecting commercial fishing activities. Therefore, encouraging resilient marine ecosystems is important in protecting the economic interests of this stakeholder group. Prioritisation of resilience may also be related to the rise of resilience as a new buzzword, similar to sustainability, because other ecological objectives like habitat connectivity (Obj 1.3) and habitat representation (Obj 1.2) are also important for ecological resilience, but were relatively poorly rated by the commercial fisher organisation respondents.

A 2009 study on stakeholder objective preferences in Australian Commonwealth-managed fisheries found that industry stakeholders prioritise economic benefits and sustainability of fish stocks (Pascoe et al., 2009). Results from commercial and recreational fisher organisations in this study still reflect

this, with greater awareness of the importance of reducing ecological impacts to achieve stock sustainability and resilience. Commercial fisher organisation respondents also placed greater importance on resilience to regional growth pressures (Obj 1.9), specifically commercial fisher organisations in the Queensland jurisdiction. Resilience to regional growth pressures is of great importance for fisheries areas like Moreton Bay in Southern Queensland; one of the most intensely used coastal areas in Australia (Thurstan et al., 2019). The Moreton Bay region makes up only 3% of the Queensland coastline but produces over 12% of Queensland's total fish catch (Thurstan et al., 2019), demonstrating the significant pressure placed on this marine ecosystem and the need for appropriate management to achieve 'resilience'. Despite the high importance placed on resilience by commercial fisher organisation respondents, process objectives pertaining to management and compliance (P.07) and conservation (P.03) were considered relatively unimportant by the respondents in this group (Table 10).

Recreational fisher organisation respondents had a strong preference for minimising flow-on impacts on supporting industries (Obj 2.6). The regional economic impact of recreational fishing through these flow-on effects is often used to promote the importance of the sector (Scheufele and Pascoe 2022), providing an incentive for recreational fishing to continue supporting local industries and businesses (a symbiotic relationship). The recent *National Social and Economic Survey of Recreational Fishers* found that recreational fishing in Australia contributes $11 billion to the Australian economy and supports 100,000 jobs (Moore et al., 2023). In 2021, flow-on effects from off-trip expenditure (to support local industries) constituted 83% of the total value of Queensland's recreational fishing industry (BDO Economics, 2021). Recreational fishers' aversion to supporting other recreational activities reflects their perception of these other activities being in conflict with or competing with recreational fishing.

Commercial fisher organisations and fisheries manager respondents had a strong preference for minimising impact on their fisheries (Obj 2.1). In the case of commercial fishers, this is consistent with the reported opposition to MPAs which are perceived to potentially impeded upon or negatively affect fishing activities and cause regional economic impacts (McNeill et al. 2018).

Interestingly fisheries and MPA manager respondents demonstrated little interest in minimising regulatory burdens and encourage behavioural change as a management tool (Figure 8; Process objective 8). Behavioural change is a relatively new concept in the management tool box and whilst potentially effective the weight of the evidence of traditional regulatory tool has in contrast an establish historical precedence

In terms of the inshore versus offshore marine protected area comparison, there was an a priori expectation that there would be lower prioritisation of economic and social objectives for offshore marine parks. MPA manager respondents responsible for offshore MPAs did prioritise higher level ecological objectives, but there was still consideration given to economic and social objectives. The importance of individual objective were very similar between the inshore and offshore managers, with only some minor differences. Most surprisingly was the relatively high weighting assigned to tourism by offshore manager respondents. This may reflect an increase in alternative, 'extreme' tourism events compared to traditional coastal tourism and recreational activities or be a framing effect of the word 'park' which may be intuitively linked to tourism activity.

The additional qualitative remarks revealed other priorities for stakeholder groups. These were around improving protection levels in marine parks and making boundaries and enforcement more tangible. These priorities reflect some of the dominant factors affecting the success of marine parks to provide ecological benefits, higher protection levels (Ferreira et al., 2022) and enforcement supported by budgeting and staff capabilities (Gill et al., 2017), which are globally considered to be the most influential factors affecting ecological impacts of MPAs.

This study has demonstrated the value of modified AHP and scaled importance weights for investigating divergent stakeholder objectives and confirmed the priority objectives for different stakeholder groups which can be further used to support policy decisions and assumptions. The results of this study can also serve as an aspirational guide for policymakers and influencers.

Consideration should be given to improving the practicality of process objectives in order to make meaningful progress towards specific sub-objectives, specifically addressing the gap between the importance of achieving benefits for Traditional Owners and First Nations Groups, and how to actually realise this to reduce the risk of "token objectives" and stagnated progress. Future collaborative efforts should also consider facilitating greater coordination and cooperation between groups. This was expressed by all stakeholder groups but results here clearly demonstrated that each group still only sticks to their perceived responsibilities which risks some objectives falling through the gaps between different stakeholder groups.

The scaled importance weights can potentially aid future MPA/PPA design through their inclusion in a multi-criteria decision analysis. This will allow an understanding of the potential attractiveness of different MPA/PPA designs from the perspectives of the different stakeholder groups and identification of the best compromise design for balancing use and conservation in marine reserves. This approach has been successfully applied in prior Australian fisheries contexts, including operationalising triple bottom line harvest strategies (Dichmont *et al.* 2013b; Pascoe *et al.* 2019a; Dichmont *et al.* 2020; Pascoe *et al.* 2023).

Conclusion

The purpose of this report was to provide an overview of Australian and international experiences with the application of PPAs to achieve social, economic and environmental (conservation) benefits. We also considered experiences with fishery-specific closures as a fisheries management tool. We examined a range of frameworks with which to assess the efficacy of multi-use marine reserves with respect to these key objectives. We also considered the relative importance different stakeholder groups place on these objectives.

We found that, designed effectively, partially protected areas within marine parks (i.e., multi-use zones) could result in conservation benefits being achieved with minimal impact on commercial and recreational fishing activities. This was demonstrated through the review of international experiences as well as in the two case studies considered. In both case studies, the impacts on the fishing sectors were substantially lower than originally anticipated. While evidence of fishery benefits through stock recovery and spill-over from protected areas was found in the international literature, there was little evidence of these benefits in the two case studies, with perhaps some inconclusive evidence in relation to recreational fishing. Most benefits, where observed, were generated through effort-reduction programs that either preceded the marine park zoning as part of a broader fisheries management initiative (i.e., the Securing Australia's Fishing Future program in Commonwealth fisheries) or as part of the marine park management plan itself.

A range of authors have proposed the use of MPAs as a fisheries management tool (e.g., Batista *et al.* 2011; Kriegl *et al.* 2021), based on the assumption of spillover benefits and as an aid to stock recovery. From our study, we conclude that MPAs are a marine conservation tool foremost and are not appropriate as fisheries management instruments. However, this is not to say that, designed effectively, MPAs may enable conservation objectives to be achieved with relatively little impact on fishing activities.

A range of fisheries-related closures were also examined, such as nursery area protection closures or seasonal closures to protect spawning stock and/or ensure fish grew to a more economically optimal size. Evidence of these generally providing benefits to the fishing industry was seen in most cases, although some issues (e.g., effort displacement) were also identified.

A recurring theme in the study, particularly in the components associated with social outcomes, was a lack of clear operational objectives against which MPA and PPAs can be assessed. Where objectives have been considered in marine protected area design, these have focused on ecological outcomes. A key outcome from our study is the first definition of a set of operational ecological, economic and social objectives for MPA/PPA management that are measurable and applicable to a range of different types of protected areas. These provide a means by which MPA/PPAs can be assessed, as well as provide a focus for future monitoring beyond just ecological monitoring.

Implications

The results of the study, primarily through the case studies and the objective weightings survey, suggest that marine reserve managers are cognisant of the impacts marine park zoning may have on commercial and recreational fishers, and design the reserves in order to minimise these impacts. Our results suggest that they have been largely successful in this, such that the integration of multi-use partially protected areas within marine reserves can contribute to conservation without necessarily imposing substantial costs on other users of the marine resources. The early collaboration with fisheries managers on marine park zoning as well as public consultation have also helped produce zoning plans which helped achieve conservation goals with minimal cost to fishers.

The further expansion of marine reserves, particularly as Australia approaches its national target of 30% of marine area under some form of protection, may place additional challenges for the fishing industry (both commercial and recreational). While impacts on these sectors have been limited to date, increased competition for the remaining marine estate with other users (e.g., offshore wind, oil and gas and potentially deep-sea mining) will place additional restrictions on the ability of the fishing industry to offset the effects of reducing fishing areas due to marine reserves.

Extension and Adoption

The workshops with the marine park managers included the project team providing a review of the international literature on the ecological, economic and social impacts of marine parks. Participants at the workshops involved marine park policy makers, planners and day-to-day managers. While the marine park planning was governed by a series of guiding principles, the process of objective development undertaken at the workshops also helped some managers think beyond just conservation. The evaluation approaches presented in the project also provide a framework for assessment of future marine reserve zoning changes.

Two conference presentations were delivered to international audiences, while two journal articles have also been produced (see next section).

Project materials developed

Journal papers

- Pascoe, S., Coglan, L., Roos, M., Cannard, T., Scheufele, G., Doshi, A. and Haro, I. (2025). Fisheries' Economic Impacts from the Rezoning of the Multi-Use Moreton Bay Marine Park, *Fishes* 10, 192. https://doi.org/10.3390/fishes10050192

Marine protected areas (MPAs) have been implemented globally to protect marine habitats and enhance biodiversity, often displacing commercial and recreational fishing activities previously occurring in the area. While the ecological impacts of MPAs have been the subject of considerable attention, the economic impacts on the displaced sectors have generally received less consideration. In this study, we examine the impacts of increasing the proportion of fully protected area within a coastal MPA—the Moreton Bay Marine Park in Queensland, Australia—on the economic performance of the fisheries operating in the area. This MPA is relatively unique as it is located adjacent to a major metropolitan area and, hence, heavily used for a range of activities. Analysis of commercial catch data suggests that the commercial fishery has been less impacted than expected by the loss of available area, although this result varies by species. Comparing fishing activity in the Bay with that in adjacent regions (assumed to reflect the counterfactual), we find evidence of strong improvements in fishery performance of the prawn trawl fleet, no significant change in fish net and line fisheries, and a small but significant decline in the performance of crab fishers. The impact on recreational fishing is uncertain as other external factors obfuscate the impacts of the rezoning.

- Sean Pascoe, Louisa Coglan, Ella Dewilde, Toni Cannard, Nipuni Abeysiriwardena, Amar Doshi, Isabel Haro, Gabriela Scheufele (*in review*) Perceptions of stakeholders regarding management objective importance in Australian marine protected areas

Most marine protected areas have a combination of unrestricted areas, partially protected areas (PPA) and fully closed (no-take) areas, with different activities allowed in each. Deciding which activities to allow in different areas requires balancing the often multiple and conflicting objectives confronting marine park managers. Disagreement between different stakeholder groups as to how park zoning is to be implemented may stall the process or lead to political lobbying and intervention in the process. An early understanding of the importance different groups give to these multiple objectives may help develop compromise solutions or policies that satisfice most groups. Increased transparency in terms of key motivators of each group also may help build trust and reduce conflicts between groups. In this study, we identify the key ecological, economic and social objectives faced by Australian marine park managers and planners and assess their relative priority from the perspective of marine park managers and other stakeholders. We find that commercial and recreational fishers place highest importance on objectives relating to minimising impacts on their activities, while resource managers place higher importance on achieving ecological objectives. While applied in an Australian context, the study provides an approach that can be applied elsewhere where trade-offs between use and conservation of the marine environment are required.

- Toni Cannard, Louisa Coglan and Sean Pascoe (*in preparation*) Social consideration for the implementation of marine fully and partially protected areas: a thematic review

Marine protected areas (MPAs) are generally implemented with a key aim of protecting or enhancing marine habitats and species, primarily from extractive activities such as commercial and recreational fishing. The removal of these activities, however, potentially result in economic and social impacts on fishers and the local community. Partially protected areas (PPA) allow some level of extraction to continue, but at a potential cost to conservation outcomes. Balancing these impacts is complex,

largely as social and economic objectives associated with MPA/PPA design are rarely specified and social impacts in particular rarely considered. In this paper we undertake a thematic review of previous studies considering social impacts of MPAs. We identify several common themes, namely that social objectives are rarely considered in MPA design; developing a social licence to operate is a key requirement to achieve compliance; a perception that social benefits are linked to conservation outcomes; and, the importance of stakeholder engagement to ensure that placement of MPAs and PPAs achieve an appropriate balance across all objectives.

Conference Presentations

- Coglan Louisa, Pascoe Sean, Cannard Toni, Scheufele Gabriela, Doshi Amar, Haro-Bilbao Isabel, Kenyon Rob, Dunstan Piers, Perry Michelle, Udy Nicola. *The multiple values of Australian marine partially protected areas: information to improve management outcomes.* Paper presented at the World Recreational Fishing Conference, 2023.

Most of the marine protected areas consist of partially protected areas (PPAs) that are open to some form of recreational or commercial fishing activity. For example, 75% of Australian marine protected areas are PPAs (Roberts et al., 2020). Resource managers must balance environmental, economic, and socio-cultural values of multiple users, which is challenging given that the values of some of these benefits, such as recreational fishing, are not readily quantified or easily understood. This study will discuss the findings from the first part of a wider FRDC-funded project on multiple values of Australian MPAs. This includes the results of a detailed review of environmental, economic, and socio-cultural values in Australian marine protected areas, including gaps in existing knowledge. Critically, it will outline a discussion of the prevailing frameworks utilised to encapsulate these values, with advantages and limitations to facilitating management. Finally, a roadmap will be presented of the next phase of the wider project, which involves extensive consultation with resource managers of two distinct case studies, to understand (1) the differences in key values, (2) how this impacts the establishment of PPAs, (3) evaluation of PPAs against multiple use objectives, and (4) how these may inform adaptive management strategies. The socialisation of this first phase will provide context and information to relevant stakeholders of the multiple values of marine PPAs and provide further avenues for discussion on management approaches that aim for fairness and flexible arrangements through the development of appropriate decision-making tools.

- Sean Pascoe, Louisa Coglan, Toni Cannard, Nipuni Abeysiriwardena, Amar Dosh, Isabel Haro, Gabriela Scheifele *Balancing use and conservation in marine reserves: what do managers aim to achieve?* Poster presented at IIFET 2024, July 2024, Penang.

Most marine parks have a combination of open areas, partially protected areas and fully closed (green) areas, with different activities allowed in each. Most restrictions relate to recreational and commercial fishing, with recreational fishing allowed in some areas and commercial fishing allowed in others to differing degrees. Deciding which activities to allow in different areas requires balancing the often multiple objectives confronting marine park managers. These generally have a solid foundation in marine conservation, but also need to take into account others who historically have used the area. In this study, we identify the key ecological, economic and social objectives faced by Australian marine park managers and planners, and assess their relative priority from the perspective of both managers and stakeholders. We do this for two types of marine parks – a coastal marine park adjacent to a large capital city and a marine reserve network in offshore federally managed waters. We find, as might be expected, that the relative importance of these objectives differs across stakeholder groups and also across context of the reserves (e.g., coastal or offshore).

- Toni Cannard, Louisa Coglan and Sean Pascoe (2024) *Finding the triple bottom line in partially protected area*s, ANZSEE Conference, Oct 2024, Stradbroke Island

While marine protected areas (MPA) have a primary conservation objective underlying this is an imperative to create triple bottom line ecological, economic, and social outcomes. Partially protected areas (PPA) are a type of MPA that could provide opportunities to achieve these outcomes especially the social aspects including heritage, cultural identity and human wellbeing. In preparation for conducting workshops for two case studies, the Moreton Bay marine park and the Commonwealth marine reserves, we conducted an extensive literature review on the social outcomes of PPA as these are still the most understudied. In doing so we analysed the social outcomes provided by the marine park zoning alongside the ecological and economic outcomes. Our review concluded that social legitimacy is required for social licence and in order to address social inequities which is often at the core of users' activity-based conflicts. Highlights from our two case studies will also be presented. This work has taken an interdisciplinary approach and has uncovered several integration opportunities including how conservation goals, social and economic issues at multiple levels can also consider sensible trade-offs while considering cumulative impacts. More importantly, it underscores the importance of participation such that bottom-up input from communities excludes the most damaging activities from partially protected areas that are designed with local communities to optimise both local and global benefits. Overall, the participatory approach will ensure that in the future closed area placement will also be based on social considerations.

Appendices

List of researchers and project staff

- Louisa Coglan, Ella Dewilde, Nipuni Abeysiriwardena (QUT)
- Isabel Haro, Amar Doshi (Natural Capital Economics)
- Sean Pascoe, Toni Cannard, Marjolene Roos, Gabriela Scheifele, Rob Kenyon, Piers Dunstan (CSIRO)
- Nicola Udy, Michele Perry (QDESI)

Online survey: Partially protected areas in Australia

Using multiple values to inform the management of Australian marine partially protected areas

Participant Information
CSIRO, in collaboration with QUT (Queensland University of Technology), NCE (Natural Capital Economics) and the Fisheries Research and Development Corporation (FRDC), is currently undertaking a study on the use of multiple values to inform management of marine partially protected areas within Australia. This project aims to consider the key environmental, social and economic values and their trade-offs in the use of partially protected areas, and identify how understanding these values can assist management decision making when implementing marine parks more broadly in Australia. This survey is part of this study and aims to understand how the relative importance of these different values and objectives differ between jurisdictions and different stakeholders impacted by the designation of partially protected areas.

What is involved?
You will be asked to assess the relative importance of different aspects of marine park management and the designation of partially protected areas. The survey has three sections:

- Section 1: Provides further information on the study and instructions as to how to complete each set of questions.
- Section 2: Asks you to rate (using a slider) the relative importance of different key ecological, economic and social considerations; and
- Section 3: Asks for information about you and your current role in marine resource management.

The survey is anonymous, and the information provided will be treated confidentially in accordance with the Privacy Act 1988. No information used in the study will be individually identifiable, and all information will be presented in an aggregated form.
The survey is expected to take less than 15 minutes to complete.

Participation and withdrawal
Participation in this study is completely voluntary and you are free to withdraw by stopping the survey at any time. As all responses are anonymous, it is not possible to withdraw your participation once submitted.

Risks and benefits
Aside from giving up some of your time, there are no foreseeable risks associated with participating in this study. If you have any concerns about any aspects of the study, please contact Dr Sean Pascoe (see below for contact details).
The outcome of the research will provide benefits to the future planning of multiple use areas within Marine Parks in Australia.

Confidentiality
All individual information collected in this study will remain confidential. The primary data will only be seen by members of our research team and will be stored in a secure area that is inaccessible to other individuals. Your information will only be used for research purposes.

Will I receive any payment for taking part in the study?
You will not receive any form of direct payment from CSIRO or the other project partners for taking part in this survey.

How will my information be used?
The information provided by you us will be combined with other respondent's answers and statistically analysed to better understand how preferences vary between jurisdictions and stakeholder groups. The results will not present any individual information. The information will also be used to prepare scientific reports and manuscripts for academic publication. Your personal information will not be identifiable at any stage in these publications.

How will my privacy be protected?
Your personal information is protected by the Privacy Act 1988 (Cth) (Privacy Act). CSIRO will handle your information in accordance with this Act and the National Health & Medical Research Council (NH&MRC) National Statement on Ethical Conduct in Human Research (2007, updated 2018), or as otherwise required by law. Results from the project will be de-identified and/or aggregated and published/presented or shared in a variety of forums and with third parties. This includes reports to the funding agency (FRDC), conference presentations and papers submitted to peer-reviewed journals. For further information on how CSIRO handles your personal information and our access, correction and complaints process please read our privacy policy available on our website or by contacting us at privacy@csiro.au.

How can I find out more about the study?
Please feel free to contact us at any time during the study. This study is being funded by the Fisheries Research and Development Corporation (FRDC), with in-kind contributions from the other partners.

Ethical clearance
This study has been approved by CSIRO's Social Science Human Research Ethics Committee (project 058/24) in accordance with the Australian National Statement on Ethical Conduct in Human Research. If you have any questions concerning your participation in the study, please feel free to contact the researchers involved. Alternatively, any concerns or complaints about the conduct of the study can be raised with the Manager of Social Responsibility and Ethics on 07 3833 5693 or by email at csshrec@csiro.au.

Key project contacts from each organisation Dr Sean Pascoe, CSIRO Environment Tel: 07 3833 5966 Email: Sean.Pascoe@csiro.au; Prof Louisa Coglan, QUT School of Economics and Finance Tel: 07 3138 5135 Email: l.coglan@qut.edu.au; Dr Amar Doshi, Natural Capital Economics Tel: 0422 882 180 Email: amar.doshi@nceconomics.com
Thank you for your help with this important research.

I consent to participating in this survey

- ○ No (1)
- ○ Yes (2)

Skip To: End of Survey If I consent to participating in this survey = No

Decision making in marine conservation and the use of partially protected areas

The designation and management of marine parks is important to ensure the ongoing protection of the marine environment. However, limiting or prohibiting access to areas of the marine environment also has impacts on other users of these areas. These may be both positive and negative. For example, excluding commercial fishing from an area may have a negative impact on those whose livelihoods were linked to that activity. However, improved fisheries resources resulting from this exclusion may benefit other commercial fishers in adjacent areas, or recreational fishers who may still have access to these areas. Similarly, it may provide opportunities for new economic activities to develop (e.g., scuba diving).

As a result of these costs and potential benefits, many marine reserves contain a number of different zones which allow different uses. Some zones are aimed purely at conservation ("green zones"), while others allow different levels of fishing or other activities (i.e., partially protected areas). Determining the appropriate trade-offs between conservation and use in different areas requires (1) an understanding of the objectives of management and (2) the relative importance of these objectives to the different groups responsible for managing the marine environment (e.g., marine park and fisheries managers).

The project team has undertaken a series of workshops and meetings with several State and Commonwealth marine park managers to identify a range of objectives that were considered when planning marine park zoning and the uses that may be undertaken in each zone. These objectives were classified into different typologies using a standard objectives-classification system. The *fundamental objective* is the ultimate (overarching) aim of the marine park. Under this is a series of *means objectives*: sub-objectives focusing on key different components of marine park management with the aim of achieving the overarching fundamental objective. These means objectives also underpin the trade-offs between uses that may be permitted in any particular area. Hence, identifying the relative importance of each of these multiple means objectives to decision makers is essential to help guide optimal decision making. We also identified a number of *process objectives*, which prescribe how the objectives should be achieved (not necessarily what is to be achieved) when pursuing the means and fundamental objective(s).

The fundamental (overarching) objective identified through these meetings was to *"ensure functional marine environments that provide a range of demonstrable services and benefits, not just to users but local communities, potential future visitors, and even broader benefits nationally and internationally"*. In order to achieve this fundamental objective, a number of ecological, economic and social means objectives were identified. The relative importance of each of these objectives is likely to differ between conservation managers in different jurisdictions given the different ecological and policy environments under which they operate, and also between conservation focused groups and user groups (e.g., fisheries managers) who are affected by the management decisions. Increasing the transparency of these differences will help guide future decision making and (hopefully) reduce the potential for conflict between different stakeholder groups when considering marine reserves and partially protected areas. Similarly, the importance of different processes may vary between jurisdictions and stakeholder groups, and understanding these differences is important when considering the overall outcomes (benefits and costs) associated with the marine parks and associated partially protected areas.

The following questions are intended to derive the implicit relative importance you attribute to each of the different means and process objectives. More detailed instructions are on the following page. There is no right or wrong answer, and all responses are anonymous.

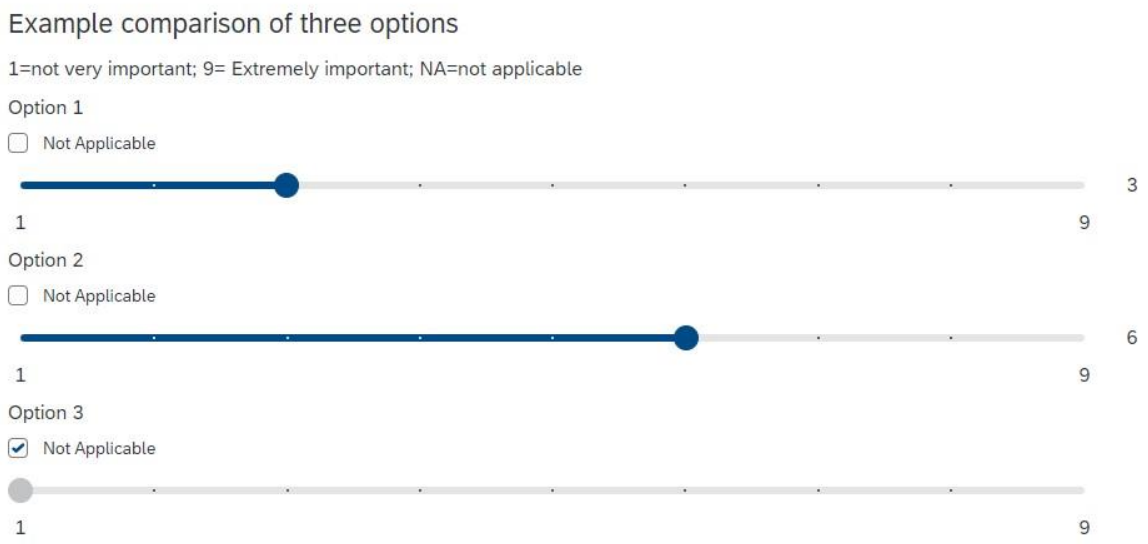
Instructions

A number of potential objectives have been identified through discussions and meetings with State and Commonwealth Marine Park Managers and policy makers. As noted in the introduction above, the aim of this survey is to assess the relative importance of these objectives to different stakeholder groups. Understanding these preferences increases transparency in decision making and allows overall benefits to be maximised and costs to be minimised where they can.

The means objectives are separated into environmental, economic and social objectives (i.e. the triple bottom line). A number of process objectives have also been identified.

The relative importance you give to each objective can be expressed through the use of a "slider", where a value of 1 indicates "Not very important" and a value of 9 indicates "Extremely important" (note: we use a 9-point rather than 10-point scale for reasons that are important for the subsequent analysis of the preferences, but otherwise similar in concept to a standard 10-point Likert scale). We also have an option for a "Not Applicable" if you believe the objectives is not relevant (rather than not important).

Your task is to place the sliders at the positions that best reflect your relative importance. An example of this is given below. In the example, option 1 is considered somewhat important, so is given a score of 3. Option 2 is considered more important than option 1 and is given a relative score of 6. Option 3 is not considered relevant at all (not just low importance) so the "Not Applicable" box is ticked. Note that the point selected is also indicated at the end of the slider.



The full range (1-9) does not need to be used. For example, the highest score you may give an objective may be a 7 if it is important but not highly important, and the lowest score may be 3 if it is of some importance (but not unimportant). Unlike ranking approaches (where a strict ordering needs to be imposed), you may consider some of the objectives to be equally important. In this case you move the slider to the same point for each. However, differentiation of the importance of these objectives is of greater use to help inform future decision making.

Section 2: Comparison of objectives

Ecological means objectives:
When considering ecological objectives when planning a marine reserve, what is the relative importance of each of the objectives below?

	Relative importance of the objective. 1= Not very important; 9= Extremely important									Not Applicable
	1	2	3	4	5	6	7	8	9	
Maximise protection of unique habitats ()										
Maximise habitat representation ()										
Maximise connectivity of habitats ()										
Maximise effective biodiversity protection ()										
Maximise climate change resilience ()										
Maximise Threatened, Endangered and Protected (TEP) species protection ()										
Maximise keystone species protection ()										
Maximise protection of other species (e.g., keep "common" species common) ()										
Maximise resilience to regional growth pressures ()										

Economic means objective:
When considering economic objectives when planning a marine reserve, what is the relative importance of each of the objectives below?

	Relative importance of the objective. 1= Not very important; 9= Extremely important	Not Applicable
	1 2 3 4 5 6 7 8 9	
Minimise negative impacts on commercial fishing ()		
Improve resource condition for commercial fishing (e.g., fish stocks) ()		
Minimise negative impacts on other commercial operations (e.g., shipping, transport ferries, sand extraction) ()		
Improve opportunities for nature-based tourism businesses (e.g., scuba diving) ()		
Improve opportunities for other economic uses compatible with environmental conservation ()		
Minimise impacts on support industries (e.g., tackle shops, chandleries) ()		
Maximise economic benefits to Traditional Owners and First Nations Groups (e.g., jobs, employment, businesses, etc.) ()		

Social means objectives:
When considering social objectives when planning a marine reserve, what is the relative importance of each of the objectives below

	Relative importance of the objective. 1= Not very important; 9= Extremely important	Not Applicable
	1 2 3 4 5 5 6 7 8 9	
Minimise negative impacts on recreational fishing ()		
Improve resource condition for recreational fishing (e.g., fish stocks) ()		
Reduce conflicts between different user groups (e.g., commercial and recreational fishers) ()		
Improve quality of other recreational experiences (e.g., scuba diving, boating) ()		
Maximise lifestyle opportunities ()		
Minimise loss of fresh fish/seafood to local markets and consumers ()		
Minimise negative impacts on local communities ()		
Maximise social benefits to Traditional Owners and First Nations Groups (e.g., rights to exercise traditional practices, passing down of traditional knowledge, etc.). ()		

Overall comparison of ecological, economic and means social objectives:
How important are the different broad considerations overall when planning a marine reserve?

	Relative importance of the objective. 1= Not very important; 9= Extremely important	Not Applicable
	1 2 3 4 5 5 6 7 8 9	
Ecological objectives ()		
Economic objectives ()		
Social objectives ()		

Process objectives (how the means objectives are to be achieved)
What process objectives are important when planning a marine reserve? By process, we mean **how** the reserve is to be planned, not what it is trying to achieve.

	Relative importance of the objective. 1= Not very important; 9= Extremely important 1 2 3 4 5 5 6 7 8 9	Not Applicable
Ensure all users have equitable access ()		
Ensure First Nations objectives, uses and needs are considered ()		
Increase nature and conservation appreciation through education during planning ()		
Maximise community stewardship and buy-in through engagement with stakeholders ()		
Coordination with other Government agencies (e.g., fisheries managers, tourism and transport agencies, local councils) ()		
Ensure a viable mix between fully and partially protected zones ()		
Ensure management and compliance resources are adequate, monitoring programs in place ()		
Minimise regulatory burden and encourage behavioural change as a management tool ()		
Facilitate effective management of sea country by Traditional Owners and First Nations Groups within protected areas (ranging from full management, to co-management, to effective partnerships and through to appropriate consultation on decisions) ()		

What other considerations should be important when planning a marine reserve? For example, are there key means or process objectives not covered above?

_____

_____

_____

_____

_____

Section 3: Some Information about you and your current role

What is your current role?

- ○ Marine park management/policy (1)
- ○ Fisheries management/policy (2)
- ○ Conservation organisation (3)
- ○ Fisher organisation (4)
- ○ Other (please specify) (5) _____

How long have you been in this role?

- ○ < 5 years (1)
- ○ 5- 9 years (2)
- ○ 10-14 years (3)
- ○ 15-20 years (4)
- ○ >20 years (5)

Have you had a different role related to marine resource management before your current role? (tick all that may apply)

- ☐ No (1)
- ☐ Yes: Marine park management/policy (2)
- ☐ Yes: Fisheries management/policy (3)
- ☐ Yes: Conservation organisation (4)
- ☐ Yes: Fisher organisation (5)
- ☐ Yes: Other (please specify) (6)

_____

Overall, how long have you been involved in marine resource management and/or conservation?

- ○ < 5 years (1)
- ○ 5- 9 years (2)
- ○ 10-14 years (3)
- ○ 15-20 years (4)
- ○ >20 years (5)

What jurisdiction are you currently working in?

- ○ Commonwealth (1)
- ○ Queensland (2)
- ○ NSW (3)
- ○ Victoria (4)
- ○ Tasmania (5)
- ○ South Australia (6)
- ○ Western Australia (7)
- ○ Northern Territory (8)

What other jurisdiction(s) have you worked in? (tick all that may apply)

- ☐ Commonwealth (1)
- ☐ Queensland (2)
- ☐ NSW (3)
- ☐ Victoria (4)
- ☐ Tasmania (5)
- ☐ South Australia (6)
- ☐ Western Australia (7)
- ☐ Northern Territory (8)

☐ International (please specify) (9) ____

What is your gender?

- ○ Male (1)
- ○ Female (2)
- ○ Non-binary / third gender (3)
- ○ Prefer not to say (4)

Do you identify as an Indigenous Australian?

- ○ No (1)
- ○ Yes (2)

Do you have any final comments you would like to add around multiple use of marine parks and partially protected areas?

_____

End of Block: Default Question Block

Balancing use and conservation in marine reserves: What do managers aim to achieve?

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1). CSIRO Environment; 2). QUT School of Economics and Finance; 3) Natural Capital Economics

Introduction

Most marine parks have a combination of unrestricted areas, partially protected areas (PPA) and fully closed (no-take) areas, with different activities allowed in each. Most PPA relate to recreational and commercial fishing, with recreational fishing allowed in some areas and commercial fishing allowed in others to differing degrees. Deciding which activities to allow in different areas requires balancing the often-multiple objectives confronting marine park managers. These generally have a solid foundation in marine conservation, but also need to consider others who have historically used the area. In this study, we identify the key ecological, economic and social objectives faced by Australian marine park managers and planners and assess their relative priority from the perspective of both managers and stakeholders.

Methods

Workshops were held with managers from two different marine protected areas (MPA) in Australia (one inshore and one offshore). From these workshops, a set of ecological, economic and social objectives were determined that captured the main considerations of the managers. These were subsequently validated by the managers in each workshop group.

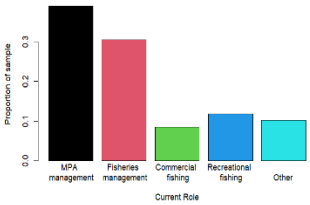
An online survey was conducted using the Modified Analytic Hierarchy Process (MAHP) approach (Pascoe et al. 2024) to elicit the importance weights associated with each objective. The sample included MPA managers, fisheries managers and policy makers, as well as commercial and recreational fishing organisations with an involvement in fisheries management.

Results

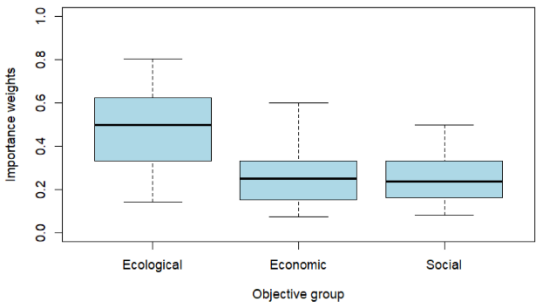
The objectives developed during workshops are presented in the below table:

1.	**Ecological**
1.1.	Maximise protection of unique habitats
1.2.	Maximise habitat representation
1.3.	Maximise connectivity of habitats
1.4.	Maximise effective biodiversity protection
1.5.	Maximise climate change resilience
1.6.	Maximise Threatened, Endangered and Protected (TEP) species protection
1.7.	Maximise keystone species protection
1.8.	Maximise protection of other species (e.g. keep "common" species common)
1.9.	Maximise resilience to regional growth pressures
2.	**Economic**
2.1.	Minimise negative impacts on commercial fishing
2.2.	Improve resource condition for commercial fishing (e.g. fish stocks)
2.3.	Minimise negative impacts on other commercial operations (e.g. shipping, transport ferries, sand extraction)
2.4.	Improve opportunities for nature-based tourism businesses (e.g., scuba diving)
2.5.	Improve opportunities for other economic uses compatible with environmental conservation
2.6.	Minimise impacts on support industries (e.g. tackle shops, chandleries)
2.7.	Maximise economic benefits to Traditional Owners and First Nations Groups (e.g. jobs, employment, businesses, etc.)
3.	**Social**
3.1.	Minimise negative impacts on recreational fishing
3.2.	Improve resource condition for recreational fishing (e.g. fish stocks)
3.3.	Reduce conflicts between different user groups (e.g. commercial and recreational fishers)
3.4.	Improve quality of other recreational experiences (e.g. scuba diving, boating)
3.5.	Maximise lifestyle opportunities
3.6.	Minimise loss of fresh fish/seafood to local markets and consumers
3.7.	Minimise negative impacts on local communities
3.8.	Maximise social benefits to Traditional Owners and First Nations Groups (e.g. rights to exercise traditional practices, passing down of traditional knowledge, etc.).

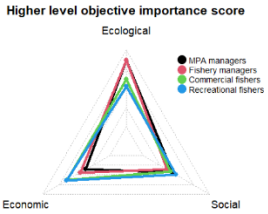
Eighty-two individuals responded to the survey invitation, although only 59 consented and answered all questions (an effective response rate of 32%). MPA and fisheries managers made up roughly 70% of the final sample. The "other" group consisted mostly of scientists associated with fisheries management.



Across the sample, Ecological objectives were generally considered more important, with Economic and Social objectives being secondary and roughly equal in importance.



Of interest in this study, however, is how these objectives, and the more detailed sub-objectives, differ in importance to the different stakeholder groups.



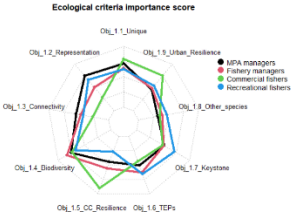
MPA and fisheries managers tended to place relatively similar importance on Economic and Social objectives, with **Ecological objectives on average being the most important** (as might be expected).

In contrast, commercial and recreational fishers both tended to place **higher weight on Economic and Social objectives** compared with the managers.

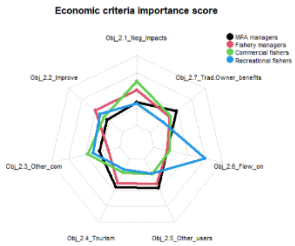
Within the Ecological objective, stakeholders indicated generally similar weights of importance across the associated criteria.

Commercial fishers, however, had a higher preference for **improving resilience** (Obj.1.5 and 1.9).

Both MPA managers and recreational fishers had a slightly higher preference for **maximising habitat representation** (Obj. 1.2)



Greater differences in preferences were seen in the individual criteria for Economic and Social objectives.



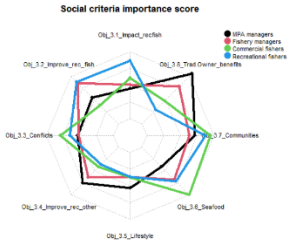
Commercial fishers and fisheries managers had a strong preference for **minimising impact on their fisheries** (Obj. 2.1).

Recreational fishers had a strong preference to **minimise flow-on impacts** (Obj. 2.6).

MPA managers tended to have a higher preference **for ensuring benefits to other users** (Obj. 2.4, 2.5 and 2.7), reflecting their broader management responsibilities.

Both commercial and recreational fishers placed higher weight on **minimising impacts on local communities** (Obj. 3.7), **reducing conflicts** (Obj. 3.3) and **ensuring seafood supplies to local communities** (Obj. 3.6) than the MPA managers.

Both Fisheries managers and Recreational fishers also gave higher weight to **improving recreational fishing resources** (Obj. 3.2).



Discussion and conclusions

Balancing use and conservation in marine reserves is complex, as different stakeholder groups have different expectations about what is important in designing MPA and PPA. Some conservation-focused stakeholder groups have argued that conservation benefits of no-take zones outweigh the benefits from PPA, arguing against, or limiting, their use (Hall et al. 2023). Others have argued that well designed PPA may achieve economic and social benefits with little conservation loss (Bobiles and Nakamura 2019).

Opposition by fishers – both commercial and recreational – to MPA usually focuses around potential negative impacts of restrictions, particularly regional economic and social losses (McNeill et al. 2018). This is reflected in the higher importance given by these stakeholders for these Economic and Social objectives.

Other users (e.g. tourism operators) were not considered in the survey but were considered important stakeholder group by MPA managers. This, at least in part, may explain the more "balanced" view of MPA managers when including these stakeholders in their importance weightings.

The importance weights can potentially aid future MPA/PPA design through their inclusion in a multi-criteria decision analysis. This will allow an understanding of the potential attractiveness of different MPA/PPA designs from the perspectives of the different stakeholder groups, and identification of the best compromise design for balancing use and conservation in marine reserves while optimising outcomes.

Acknowledgements

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