Non-Market Values to Inform Decision-Making and Reporting in Fisheries and Aquaculture

An Audit and Gap Analysis

Final report

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2020

FRDC Project No **2018-068**

ISBN 978-1-925553-26-0

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2018-068

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

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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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Acknowledgments

The authors would like to thank the survey and workshop participants for their valuable input into the project. We would also like to thank Professor John Rolfe for his participation in the workshop and advice given on the use of benefit transfer. Thank you to FRDC Human Dimensions Research Subprogram for their support and comments on earlier versions of this work and the anonymous reviewers for their careful reading of our manuscript and their comments and suggestions.

Executive Summary

What is this report about?

Effective management of marine resources requires the inclusion of both the negative and positive impacts of fishing, inclusive of commercial and recreational activities, and aquaculture activities on the wider ecosystem and community. Not all of these costs or benefits have an explicit market value, so non-market valuation is required to derive an appropriate equivalent monetary value.

The inclusion of the non-monetary costs and benefits of managing marine resources to capture this in decision making is not controversial. However, empirical estimation of critical non-market values can be costly (with respect to both time and resources). This presents fisheries and aquaculture managers with pragmatic constraints that often results in non-market values being excluded in decision-making.

Given the imperative for evidence-based decision-making in fisheries and aquaculture will intensify, this study examined the issues around non-market values requirements and identified potential sources of robust and defensible estimates of key values, including those generally viewed as difficult to measure.

Background

There is increasing need for fisheries and aquaculture management to take account of the cumulative environmental, economic and social impacts of all human uses of marine resources. At an operational level, this shifts decision-making beyond simple financial analysis, that accounts for the direct monetary effects of management decisions on industry, to consideration of an economic evaluation framework (e.g. cost-benefit analysis) that also captures the broader non-monetary impacts.

However, a previous study found that one of the main reasons that such values have not been widely adopted in fisheries and aquaculture management is due to a lack of understanding and trust around their validity (Marre *et al.* 2015). It is important to note that this finding does not mitigate the expertise that fisheries and aquaculture managers bring to the table re identifying non-market values. Fisheries and aquaculture managers have scientific and practical expert knowledge that can inform the identification of non-market values required to support decision making. But it does suggest that issues around trust and validity of non-market values in decision making needs to be addressed. The theory and methodology behind non-market values is strongly embedded within economics. As such, deriving non-market values requires specialist economic expertise. Benefit transfer methodology provides an alternative approach, as it uses existing empirical estimates of non-market costs and benefits. This approach also requires economic expertise and the pre-existence of robust non-market valuation studies.

Aims/objectives

The key project objectives were:

1. To support robust and defensible evidence-based decision-making in fisheries and aquaculture decision making that is understood and supported by key fisheries and aquaculture managers;
2. To provide managers with an understanding of the resources available to account for non-market values in fisheries and aquaculture decision making; and
3. To identify key research gaps and make recommendations related to the need for further empirical non-market valuation studies.

Methodology

A survey was undertaken of key management/policy analysts and decision-makers in fisheries and aquaculture at the State and Commonwealth levels to determine the current use of, needs for and barriers to using non-market values to support evidence-based decision making and reporting. A follow-up workshop of key management/policy analysts, decision makers and researchers sought to better inform managers and policymakers about the valuation techniques required to derive the desired values, the potential ways in which these values can be used and establish priorities for which values are most in need for fisheries and aquaculture management. A review of relevant non-market valuation studies and audit of existing databases (national and international) was undertaken to determine the availability of non-market valuation studies to support robust and defensible evidence-based decision making in fisheries and aquaculture based on the priorities identified. A key criterion of this audit was the feasibility of using these values to support benefit transfer.

The above elements informed a gap analysis using the "Importance–Performance Analysis" (IPA) approach. This identified key research gaps from which recommendations related to the need for further empirical non-market valuation studies are made.

The project identified a need for increased education around the types and uses of economics data in general for many fisheries and aquaculture managers. This was supported by the project output *Using non-market values in cost-benefit analysis in Australian fisheries and aquaculture*.

Results/key findings

The project identified thirteen types of non-market values that fisheries and aquaculture managers considered as potentially important to their decision making. Of these, the top four involved values related to users of the fisheries resources, including fisher satisfaction, values to Indigenous Australian fishers, and the value of fish and experience to recreational fishers. The next four involved impacts of fishing on others, including habitats, species, local communities and other users of the marine environment.

The gap analysis identified that recent values for most of the values of potential use to fisheries and aquaculture management were unavailable. This limits the role of benefit transfers and identifies a need for further primary studies of non-market values.

Implications for fisheries and aquaculture managers

The key implications of the study are:
1. there is a need to undertake further non-market valuation studies in the key priority areas identified by fisheries and aquaculture managers to address the key barrier of availability of relevant non-market values; and
2. there should be a specific program of education/information dissemination to fisheries and aquaculture managers and policy makers around the types and uses of economics data in general to ensure appropriate use of non-market values in decision making.

Recommendations

The project identified a need for increased education around the types and uses of economics data in general for many fisheries and aquaculture managers. A first step is provided in the guidance notes developed in this project (and also provided as an Appendix to this report). However, further training of fisheries and aquaculture managers in the use of non-market values would be beneficial.

Given the imperative for evidence-based decision-making in fisheries and aquaculture will intensify and the lack of existing studies, it is recommended that consideration is given to:
1. examining the extent to which existing recreational fishing survey data can be used and future data collection can be adapted.
2. estimation at a national level to deriving values for key priority values including bycatch (in general and of Threatened, Endangered and Protected species (TEPs) and habitat damage.

Keywords

Non-market valuation, economic costs, economic benefits, benefit transfer, gap analysis, fisheries and aquaculture management

Introduction

Background

The marine environment produces a wide range of ecosystem services, many of which may be affected by fisheries and aquaculture management, and need to be explicitly considered in fisheries and aquaculture management decision-making. These include recreational fishing, cultural services (e.g. aesthetics and heritage) as well as supporting services for a range of marine species that are harvested commercially, either through wild caught fisheries or aquaculture, or are not harvested but are of conservation value to society (e.g. seals, seabirds and dolphins).

Fisheries and aquaculture management in Australia has predominantly adopted an explicit overarching objective of maximising the net economic returns to the community (e.g. Department of Agriculture and Fisheries 2017; Department of Agriculture and Water Resources 2018). However, fisheries and aquaculture production can have impacts on the environment that are not explicitly captured in the estimation of net economic returns. For example, the "cost" to the community of bycatch, or the impacts of habitat damage on the generation of other ecosystem services (Pascoe *et al.* 2018). While policies are in place to minimise these effects, trade-offs between fisheries/aquaculture production and broader impacts on ecosystem services are not captured in an integrated framework, largely as information on the non-market value of these impacts is not well understood.

This project was developed in response to a need to acknowledge and where appropriate include these non-market values, with the aim of improving fisheries and aquaculture policy decision-making.

Ecologically Sustainable Development and the triple bottom line

The FRDC RD&E plan 2015-2020 recognised the shift in focus within natural resources management from a purely commercial perspective to a broader social perspective, reflecting the adoption of the Ecologically Sustainable Development (ESD) (Fletcher *et al.* 2005) and wellbeing framework (Voyer *et al.* 2017) for fisheries and aquaculture management. Most fisheries jurisdictions now require consideration of recreational and Indigenous Australian fishers in management decision-making, while some have additional social consideration requirements. The fundamental principle of both Commonwealth and State fisheries policy is to ensure that Australian marine resources are allocated across competing uses and users, such that the net benefits to current and future generations of society are sustainable and maximised.

This principle requires all benefits and costs associated with the 'use' of marine resources to be accounted for, with the term 'use' being defined broadly to include both use values (e.g. commercially and recreationally harvested fish) and non-use values (e.g. continued existence of an endangered species). Effective management of marine resources therefore needs to be cognisant of the cost associated with potential negative impacts of fishing and aquaculture activities on the wider ecosystem and community including (i) environmental and ecosystem impacts (for example, water and habitat quality, carbon footprint and visual amenity etc.) and (ii) recreational, cultural, and social impacts.

The triple bottom line (TBL) approach is the general framework currently used to assess policy outcomes against economic, social, and environmental dimensions. TBL requires articulation of impacts in each of these dimensions, but these may be qualitatively assessed. Significant progress has been made in incorporating some of them into fisheries management decisions, particularly prioritising different objectives of fishery management (e.g. Pascoe *et al.* 2013; Jennings *et al.* 2016). In some cases, development of semi-quantitative approaches have been used to assist in decision-making across these multiple dimensions (e.g. Dichmont *et al.* 2013), including in some cases Indigenous Australian values (e.g. Plagányi *et al.* 2013). Recent research has also extended this focus to develop a robust articulation of Indigenous customary fishing values to enable their inclusion when developing fisheries management policies (Australian Institute of Aboriginal and Torres Strait Islander Studies (AIATSIS) 2018).

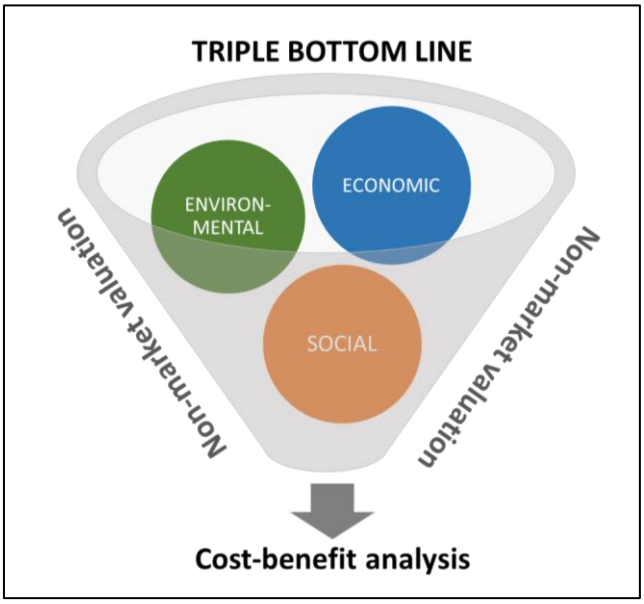
However, optimal decisions require the trade-off between costs and benefits to be considered. TBL approaches do not explicitly consider this trade-off, resulting in challenges in identifying optimal outcomes. As each component is assessed separately, there is also the potential for double counting. For example, employment may be considered a social outcome, while income from employment may be considered an economic outcome.

A number of different approaches have been developed to assess trade-offs between different components of the triple bottom line. These approaches generally fall into two "camps" – multi-criteria decision analysis (MCDA) and cost-benefit analysis (CBA).

MCDA approaches to assessing trade-offs between pillars of the triple bottom line have been applied in a range of Australian fisheries to assess management options (e.g. Dichmont, Pascoe *et al.* 2013; Pascoe *et al.* 2019; Dichmont *et al.* 2020). MCDA requires assessing the relative importance of each outcome and assigning it a weight, as well as assessing how each outcome changes as a result of the management option. With different preferences for different outcomes by different stakeholder groups, MCDA can identify which option is most preferred by which group, but does not necessarily identify which option is best overall.

While MCDA can play an important role in assessing management options, the focus of this study is the use of non-market values to provide an economic assessment of the costs and benefits of different management options. Economics provides an alternative framework for identifying management options that produce the highest net benefits (benefits less costs) to society. This requires, however, value estimates for each of the benefits and costs. Where these costs and benefits are estimated in monetary terms, assessing the trade-off requires deducting the expected costs from the expected benefits (commonly referred to as cost-benefit analysis (CBA)). If the analysis includes the costs and benefits associated with each of the TBL dimensions, CBA essentially captures the essence of the TBL approach, with the added advantage that the net benefit or cost of an action can be determined (Figure 1).

Figure 1. Economic value and the triple bottom line



In practice, CBA only requires estimating costs and benefits that change as the result of a management decision. For example, if employment is unchanged, then incomes do not need to be assessed. In this regard CBA, operates at the margin.

However, in fisheries (as in many environmental management areas), many costs and benefits do not have an explicit monetary value since they are not bought and sold in markets. Hence, decisions about the use and management of marine resources increasingly requires objective information on the non-market value of benefits (and costs). Some attention has been focused on the estimation of non-market values of recreational fishing (e.g. Prayaga *et al.* 2010; Pascoe *et al.* 2014), although only limited attempts to-date have been made to use these values in supporting management decision-making (e.g. Lindner *et al.* 2006). Many other values have not been quantified, and their use in fisheries and aquaculture management has not been fully explored.

This shifts decision-making beyond simple analysis, that accounts exclusively for market benefits and costs of management decisions on industry, to consideration of an economic evaluation framework (e.g. cost-benefit analysis) that also captures the non-market benefits and costs to all sectors of the community, inclusive of those associated with Indigenous and recreational fishing. Including these values in a cost-benefit framework may change what is considered an optimal outcome. For example, the recent FRDC project 2015-202 found that including a non-market cost associated with bycatch changes the optimal level of harvest in a fishery (Pascoe, Hutton *et al.* 2018; Schrobback *et al.* 2018).

CBA and its role and reach in decision making is often misunderstood. CBA is an economic decision-making tool to determine the total economic value of a decision. The theoretical underpinnings of CBA are neoclassical economic theory. It is principally a methodological approach that is grounded in utilitarian principles which assume that the value of each cost or benefit (as measured in dollar terms) counts equally regardless of who incurs or receives them. Hence, CBA does not explicitly account for the distribution of costs and benefits amongst members of the affected stakeholder groups (interpersonal distribution), since it is sufficient that total economic value is maximised given that net benefits of a decision (capturing both market and non-market costs and benefits) are positive. Whilst this may be considered a weakness of CBA, this does not preclude analysis of the distributional impacts in decision making. The distributional impacts of costs and benefits to different stakeholders can be derived from a CBA and provided to policy makers who have the responsibility to determine if the distributional impacts are acceptable. Additional steps can then be taken to address potential distribution inequities through, for example, compensation programs or redistributive income policies.

Whilst CBA does not explicitly consider distribution of costs and benefits between relevant stakeholders, it does account for intergenerational distributional impacts. These are captured through discounting of future net benefits using an appropriate social discount rate. For a thorough discussion of distributional impacts and CBA refer to Campbell and Brown (2015).

The need for non-market values in fisheries and aquaculture management

The estimation and use of values associated with non-market benefits and costs is a well-established process in environmental and resource management and as such the inclusion of the non-market costs and benefits of managing marine resources is itself not controversial. However, empirical estimation of critical non-market values is costly (with respect to both time and resources). This presents managers with pragmatic constraints that often results in non-market values being excluded in decision-making (Rolfe et al, 2015).

Fisheries and aquaculture managers have scientific and practical expert knowledge that can assist in the identification of the non-market values required to support decision-making. However, deriving relevant non-market values requires specialist economic expertise. In the absence of specialist non-market valuation expertise, knowledge and understanding of these values, focus can be distracted by "valuation" studies that meet the objectives of some groups but are not necessarily useful for fisheries and aquaculture management decision-making e.g. recreational fishing expenditure surveys (e.g. FRDC 2012-214, FRDC 1999-158); wellbeing studies that identify but not quantify benefits (e.g. FRDC

2011/217)). While some appropriate studies have been undertaken (e.g. FRDC 2010-536), these have not necessarily been specifically designed to support broader fisheries and aquaculture management.

To date, an appropriate research program to provide non-market values relevant for fisheries and aquaculture decision-making has not evolved due to a lack of information as to management needs. This project aims to address this gap.

Objectives

The key objective of this project is to determine what type of non-market costs and benefits values managers of Australian fisheries and aquaculture most need to ensure:

- decision-making also captures the broader non-monetary costs and benefits to all sectors of the community, and
- the extent to which these values have been estimated within Australia and elsewhere.

Given the existence of appropriate non-market values, the benefit transfer methodology may offer a potential means for managers to incorporate these values into their decision-making.

This project aims to enhance fisheries and aquaculture managers understanding of and use of non-market values to support robust and defensible evidence-based decision making that maximises the benefit to the community from use of the marine ecosystem. The project objectives were:

1. To support robust and defensible evidence-based decision-making in fisheries and aquaculture decision making that is understood and supported by key fisheries and aquaculture managers;
2. To provide managers with an understanding of the resources available to account for non-market values in fisheries and aquaculture decision making; and
3. To identify key research gaps and make recommendations related to the need for further empirical non-market valuation studies.

Method

Methods overview

A previous study has found that one of the main reasons that such values have not been widely adopted in fisheries management is due to a lack of understanding and trust around their validity (Marre, Thebaud *et al.* 2015; Marre *et al.* 2016). The key objective of this project was to provide managers with an understanding of and the resources available to account for non-market values in their decision-making.

The specific research questions addressed were:

1. What non-market values are required to support decision-making in fisheries and aquaculture (inclusive, as appropriate, of recreational, Indigenous cultural/customary, and commercial)?
2. What is the availability of empirical estimates of these identified values, and what are the key gaps that need to be addressed?

The research was undertaken in four stages.

The <u>first stage</u> aimed to identify what empirical estimates and approaches are currently available. This was conducted through desk-based research, involving:

(i) An audit of relevant non-market valuation studies and of existing databases (national and international). This included examining the feasibility of using these values to support benefit transfer;

(ii) Identification of current use of non-market values in Australian fisheries and aquaculture decision-making;

(iii) A review of the current methodological literature to determine best practice in how these values can be used to support decision-making (including limitations and alternative methods when explicit non-monetary values are either not available or considered not appropriate).

The <u>second stage</u> focused on what non-market values are required by managers to support decision-making, their level of understanding and current use of non-market values. This involved a survey of key management/policy analysts and decision-makers at the State and Commonwealth levels to determine the needs of decision-makers for non-market values to support evidence-based decision-making and reporting. The results of this survey were used to identify levels of understanding and key knowledge gaps, as well as plan a subsequent workshop with decision-makers.

The <u>third stage</u> involved a workshop of key management/policy analysts, decision-makers and researchers. Building on the results from the survey, the key aims of the workshop was to better inform managers and policy-makers about the valuation techniques required to derive the desired values (i.e. what is required, potential time frames to get results, key assumptions and limitations), and the potential ways in which these values can be used. Attendees were re-surveyed post-workshop, to evaluate the impact on their understanding of the role and application of non-market values in decision-making. A key outcome of the workshop was to establish priorities for which values are most in need for fisheries and aquaculture management.

The <u>fourth stage</u> of the project involved a synthesis of the first three project elements which informed an assessment of key research gaps and recommendations related to the need for further empirical non-market valuation studies. An audit of available databases was conducted to ascertain what information is currently available and accessible for fisheries and aquaculture managers. Next, a gap analysis was undertaken based on these findings to help identify key areas for future research.

This report reflects the synthesis of these findings.

Literature review and database audit

The literature review and database audit were undertaken simultaneously. The role of the database audit was to determine which current database, if any, might be useful for fisheries and aquaculture managers. Several databases were identified through an online (Google) search. Previous key studies, such as that undertaken by the Productivity Commission (Baker and Ruting 2014), also identified a number of databases that were considered in the study.

Separate online searches for potentially relevant papers were also undertaken. These were undertaken using Google Scholar (https://scholar.google.com.au/), ScienceDirect (https://www.sciencedirect.com/), Web of Science (https://apps.webofknowledge.com/) and Scopus (https://www.elsevier.com/en-au/solutions/scopus). The aim of this separate search was to identify studies not included in the databases that may be of relevance to fisheries and aquaculture managers.

The aim of the review and database audit were to provide an understanding of the potential availability of existing studies, which may be able to provide useful information for managers through benefit transfer. As noted previously, benefit transfer involves the use of non-market values taken from one study and applying these to a separate study site elsewhere. For example, benefit transfer may involve the use of recreational fishing values from a study in Northern Queensland to assess recreational fishing impacts in South Australia. The suitability of non-market values for benefit transfer can only be determined case-by-case. Ideally, the characteristics of the two case study sites should be identical (Boyle and Bergstrom 1992), although in some cases it may be possible to adjust the values depending on differences in local characteristics (e.g. species, population characteristics) provided information is known on these for both original study and target (policy) sites, and this information was used in the derivation of the original values. The application and development of such benefit transfer functions has been found to provide more reliable results than just direct benefit transfer (i.e. using the unmodified values), although even this requires a high degree of similarity between the environmental goods or services being valued, and that errors from applying benefit transfer can be large, even across seemingly similar amenities (Kirchhoff *et al.* 1997). Other studies have also suggested that the use of benefit transfer functions predominantly overstate benefits in the policy site, and that this error increases over time (i.e. time between the original study and application to the policy site) (Downing and Ozuna 1996). Similarly, Zandersenab *et al.* (2005) found that changes in preferences over time resulted in substantial changes in economic values for environmental assets, and that benefit transfer over time – even using transfer functions – could result in substantial errors.

While empirical studies suggest that the potential usefulness of value estimates diminishes as differences between the original site and target policy site increases, other studies have suggested that benefit transfer has been "successful" even internationally. In this case, "successful" indicates that errors encountered through international benefit transfer were no larger than those from transfer study sites within the same country as the policy site (Ready and Navrud 2006). Hence, it is difference in sites, not necessarily just geographical location that may affect their usefulness for benefit transfer.

Given these previous findings, the set of available databases were assessed primarily on their currency (i.e. how recent the reports included were). Attention was also given to the availability of Australian case studies, although the potential relevance of case site studies in other countries were also considered.

Examples of values drawn from these databases and the online searches were also considered more broadly. While many of these studies may not be suitable for benefit transfer purposes, they illustrate the range of values that have been derived elsewhere and demonstrate that such valuation is possible.

Manager computer assisted telephone interview (CATI) survey on the use of non-market values

Interviews with fisheries and aquaculture managers and policy makers were conducted in September 2018 to determine the extent to which key managers and policy makers were familiar with non-market values, the extent of their use of non-market values and the potential barriers to their use. The survey was developed as a Computer Assisted Telephone Interviewing (CATI) survey (Shangraw Jr 1986; D'Souza *et al.* 2010). That is, the interviewer was prompted for each question, with additional questions prompted based on the answers of the previous question.

CATI surveys have several advantages over both online web-based surveys and paper-based surveys (Pierzchala *et al.* 2004). A key advantage of CATI-type surveys is that complex skip-instructions can be included without affecting the flow of the telephone interview (Niemann 2003). While these can also be included in web-based surveys, CATI offers the potential for respondents to clarify if the question is not clear, and the interviewer can also identify if the respondent has missed the main point of the question and ask it again (Pierzchala, Wright *et al.* 2004). The interviewer was also able to enter the responses directly into the program (QUALTRICS) for data storage and subsequent analysis, saving time and reducing the potential for transcription errors associated with use of paper-based surveys. A key disadvantage of CATI surveys, however, is that they are more labour and time intensive than online surveys, and therefore cannot expect the same volume of responses. For a relatively small population, such as the target population in this study, this was not a major drawback.

The target population of the survey was managers employed in government and associate agencies involved in fisheries and aquaculture policy and management. These included all jurisdictions (Commonwealth, states and territories), and included individuals involved with both commercial and recreational fisheries as well as aquaculture. Industry organisations were initially considered for inclusion, but as they do not have a direct management or policy role, were subsequently excluded from the survey.

Forty-six potential interviewees were identified through online searches of management agency and government department websites. As many of these sites did not include individual manager or policy maker details, the online search also included recent FRDC reports, FISH magazine articles and conference papers. In some cases, referrals from early identified individuals were also sought. All identified individuals were contacted via email requesting their participation in the study.

A total of 14 individuals were interviewed, with each interview taking around 20 minutes to complete. The sample consisted of managers and policy makers involved with commercial fisheries (70%), recreational fisheries (10%) and aquaculture (20%).

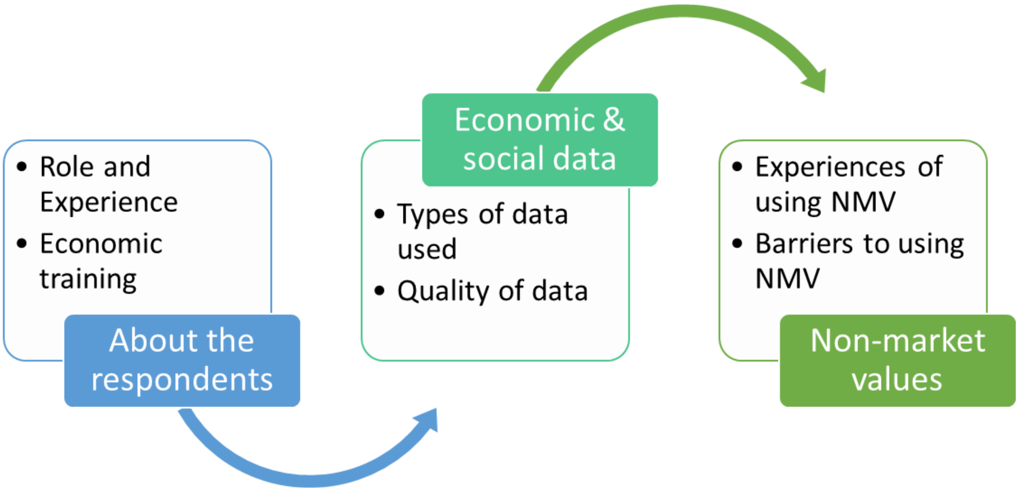
The interviews involved both closed and open-ended questions to gather deeper level of information (qualitative) as to the (i) need for non-market values, (ii) understanding of the use of non-market values and (iii) barriers to using non-market values in fisheries and aquaculture management decision. The survey questionnaire was informed by a previous survey of coastal resource managers by FRDC PhD student Jean-Baptise Marre on the use of ecosystems valuation in Australia coastal management (Marre, Thebaud *et al.* 2015). A copy of the questionnaire, along with the participant information sheet provided prior to interview, is provided in Appendix A. The survey was approved by the QUT University Human Research Ethics Committee (UHREC) (QUT Ethics Approval Number 1900000690).

The questionnaire involved three sections (Figure 2). The section "About the respondents" asked about the jurisdiction that the manager/policy maker worked in, and the area of key involvement (e.g. commercial fisheries, recreational fisheries or aquaculture). Respondents were also asked about the types of role(s) as a manager they had in final decision-making. Four types of roles were considered: (i) informative, where the main role of the manager/policy maker was collecting/analysing information and delivering it to others; (ii) consultative, where the manager/policy maker was involved in providing recommendation to others; (iii) contributive, where the manager/policy maker contributed to the final decision and/or management plan; and (iv) decisive, where the manager/policy maker made the final decision. Managers could (and usually would) participate in more than one of these roles, and the proportion of time involved in each role was elicited. Other information collected included the level of experience in their current and previous similar roles and if they had had any economic training and to what level.

In the section "Economic and social data" respondents were asked about the legislative requirement to consider social and economic considerations in fisheries and aquaculture management decision-making. Next, they were asked about the type of economic and social data they used in decision making. The final part of the interview focused explicitly on non-market values, including experiences of using non-market values in decision making contexts and identifying barriers to using non-market values in the past and future.

Individuals were also asked if they would be prepared to participate further in the project, including attending the project workshop.

Figure 2. Structure of telephone interview



Manager workshop

A workshop was held in Brisbane on 31 October 2019. Managers and policy makers who indicated an interest in attending the workshop during the survey were contacted and asked if they would be able to attend the Brisbane meeting. Not all were able to attend, so suggestions for a suitable replacement from their organisation were requested. In total, 13 managers/policy makers attended the workshop, representing (nearly) all jurisdictions,[1] and included representatives of commercial and recreational fisheries and aquaculture management and policy.

The first half of the workshop involved a series of presentations around the use of non-market values in fisheries and aquaculture, the conditions necessary for the use of benefit transfer, and the availability of information in existing databases. The second half of the workshop involved group discussions about the types of non-market values that managers/policy makers would find most useful and would most likely use. Prioritisation of these values was initially intended to be undertaken as part of the workshop, but due to time constraints was undertaken as part of a post-workshop evaluation survey.

[1] The South Australian delegate had to cancel last minute.

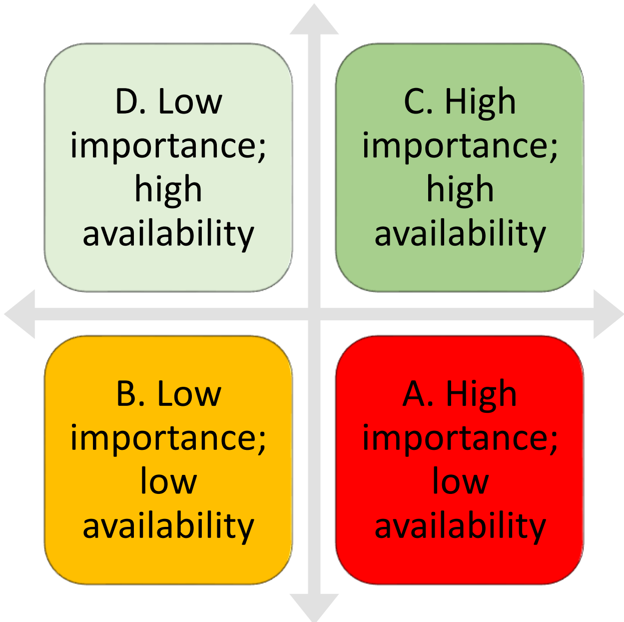
Gap analysis

The audit of the available databases provided an indication of what information is currently available and accessible for managers. A gap analysis was undertaken based on this to help identify key areas for future research.

The gap analysis adapted the "Importance–Performance Analysis" (IPA) approach (Martilla and James 1977; Bacon 2003). The approach has been applied widely in areas such as healthcare (e.g. Yavas and Shemwell 2001) and education (e.g. O'Neill and Palmer 2004), and has particularly gained widespread acceptance in the hospitality and tourism research (Azzopardi and Nash 2013; Lai and Hitchcock 2015), including marine based tourism (Tonge and Moore 2007).

The approach is conceptually simple: a matrix is constructed consisting of four quadrants, the axes being importance and performance (Figure 3). The importance of each of the different types of non-market values was determined in the workshop and post-workshop survey as noted above. In the above cited studies, performance is generally measured in terms of customer or visitor satisfaction. For this study, we have replaced performance with a measure representing the relative quantity of available studies, adjusted for their recency and relevance.

Figure 3. Importance-Availability matrix



For relevance, we consider Australian based studies only, as analyses of the potential for international benefit transfers suggest that these may involve high errors (Lindhjem and Navrud 2008). To include a recency consideration, we discount studies based on their age. While updating estimates of nominal values of non-market values produced by previous valuation studies through use of the consumer price index (CPI) is a common approach, changes in preferences (e.g. development of other opportunities in the case of recreation) and underlying economic conditions over time (e.g. incomes) may increase the distortion in the inflated values (Eiswerth and Shaw 1997). Changes in non-market values of ecosystem services over time can be driven by changes in the markets for related ecosystem services, changing preferences for landscapes or biodiversity, increased understanding of the level of scarcity of the ecosystem service, or changes in the level of resources available to manage an ecosystem. (Rosenberger and Johnston 2009; Scheufele 2012; Hein et al. 2016; Bryan et al. 2018). Skourtos et al. (2010) found that non-market values were relatively robust over 1-5 years (i.e. could be adjusted based on changes in income), but beyond this, preferences changed substantially, decreasing their relevance with time, such that simple adjustment was less appropriate. Since valuation methods have improved over time and preferences have changed, dependence on older valuation data can affect the accuracy and relevancy of values derived through benefit transfer (Pendleton et al. 2007; Scheufele 2012). Likewise, there is considerable uncertainty as to future preferences and how they will change, so older estimates of values are likely to be less relevant to future values. Even basing decisions on the most recent information may still be flawed if preferences change in the future, but may be reasonably appropriate for the short term, with these decisions needing re-assessment over time through adaptive management (Johnston and Rosenberger 2010).

Given this, the availability measure was developed on the basis that i) older studies are potentially less relevant than more recent studies, and ii) several studies provide more information than a single study. As there is no agreed method to discount studies based on their recency, studies conducted in the last five years were considered current (i.e. not discounted following Skourtos, Kontogianni et al. (2010)),[2] while studies older than five years were subject to a straight line "depreciation rate" of 20% a year. This effectively removed papers that were over 10 years old from the analysis.

The final availability score was based on the discounted age of the most recent published study in each State or Territory contiguous to the coast (i.e. excludes the Australian Capital Territory (ACT)). A State-based analysis was considered as most relevant for potential benefit transfer. This still remains a crude indicator, as noted with the most recent study, although relevant to the State, may be less relevant to different fisheries or aquaculture industries within the State. The final availability score was calculated as the sum of the discounted ages of the most recent study in each State/Territory divided by the total number of States/Territories contiguous to the coast (i.e. 7). This gives a value between 0 (no relevant or recent studies) and 1 (a current (i.e. 2020) study in every State and Territory). In this case, a value of 0.5 represents a current study (i.e. within the last 5 years) in at least four of the seven states, or a number of less recent papers over more States.

The combination of importance and availability helps identify which values are potentially in most need of further research. Values that fall into the top left quadrant of Figure 3 (low importance; high availability) require low additional attention. In contrast, values that fall into the bottom right quadrant (high importance; low availability) require additional attention.

[2] The five year period was taken as from 2015 as 2020 was incomplete at the time of the analysis. Studies identified in 2020, however, were also included in the analysis.

Results

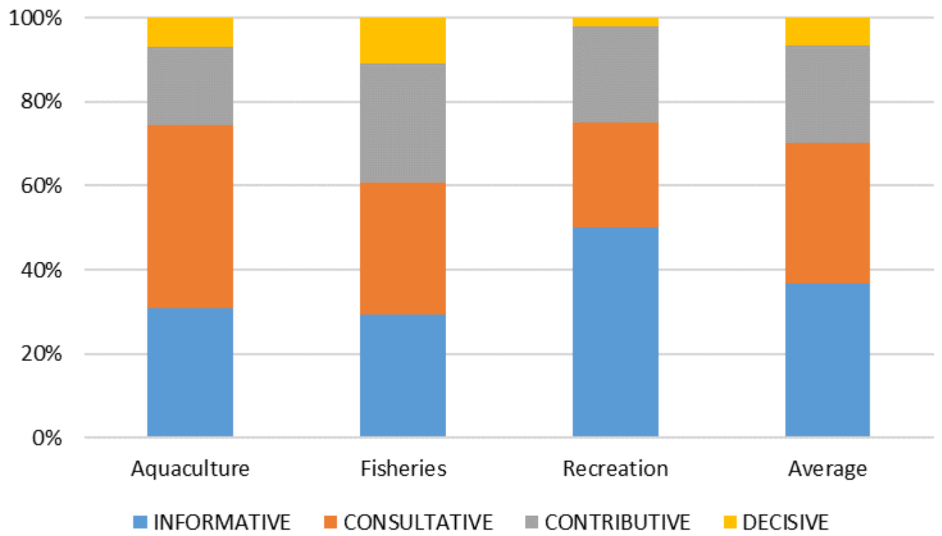
CATI survey of managers and policy makers

Characteristics of the sample

Forty-three fisheries and aquaculture managers and policy makers were approached for the interview, from which 14 were interviewed, a response rate of about 33%. Reasons for non-participation were largely due to non-availability at the time the interviews were being undertaken. The interviews took place over the telephone using a computer aided telephone interview (CATI) approach, with each interview taking around 20 minutes to complete. The sample consisted of managers and policy makers involved with commercial fisheries (70%), recreational fisheries (10%) and aquaculture (20%), and covered all jurisdictions (Commonwealth and State).

The sample of interviewed managers and policy makers were involved in all aspects of the decision-making process, from information collection and analysis through to the final decision-making (Figure 4). The proportion of involvement in each aspect varied by individual, with some having a greater role in the informative processes, and others having a greater role in the final decision.

Figure 4. Contribution to decision-making

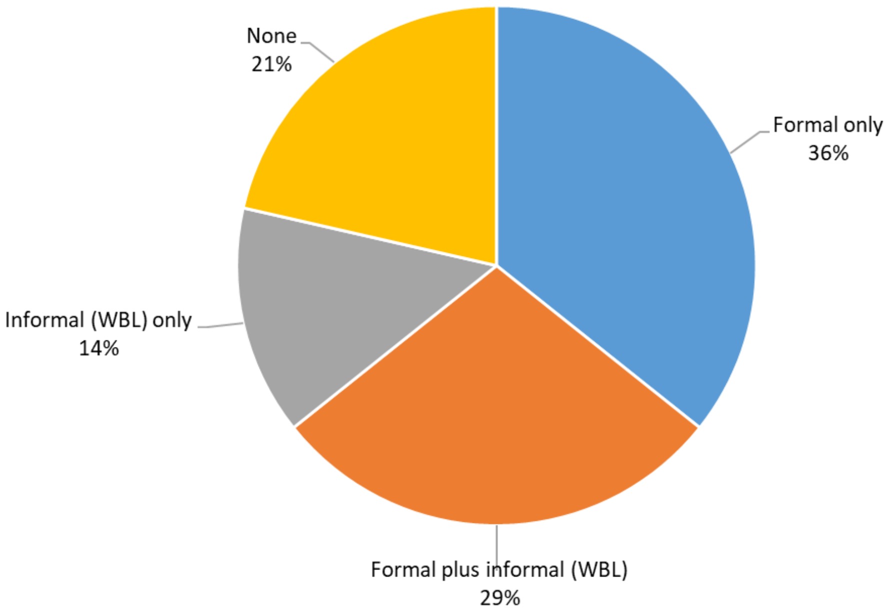


All had greater than 2 years of experience in their current role, with most (80%) having more than 5 years of experience in fisheries and aquaculture management or policy over their career. This experience was primarily Australian based (i.e. little international experience). Prior to working in fisheries and aquaculture management or policy making directly, most individuals were involved in a fisheries-related position, such as a fisheries officer, fisheries biologists, or training, but others had broader experience, including owning and running a business. Around 70% of the interviewed group were males. These results are largely comparable with respect to years of experience and gender identified in previous studies of the use of non-market values in coastal and resource management in Australia (Marre, Thebaud *et al.* 2015).

Most (79%) of the individuals in the sample had been exposed to some form of training in economics (Figure 5), with this ranging from formal to informal training. The latter was primarily work-based learning, where individuals worked with economists or economic information and learnt in the process.

Formal training varied considerably, ranging from full degree level education in economics or natural resource management with economic components to short courses. Short courses included three days courses in economics for fisheries managers as well as the FRDC Master Classes that were run as part of previous FRDC projects.

Figure 5. Exposure to economics training



A common key theme that emerged during the interviews was that there was no systematic formal economics training for the industry, with short courses provided on an ad-hoc basis only.

Use and availability of economic and social data

The theory of economic values and the related terminology are part of the epistemology of economics. Founded on the concept of opportunity costs (Green 1894), economic values capture both explicit costs and benefits (e.g. are priced by the market and have a monetary value) and implicit costs and benefits (e.g. costs and benefits that are incurred but for which no market exists). The concept of opportunity costs has been used to value ecosystem services through monetarised payments since the 1970s. (Gómez-Baggethun *et al.* 2010).

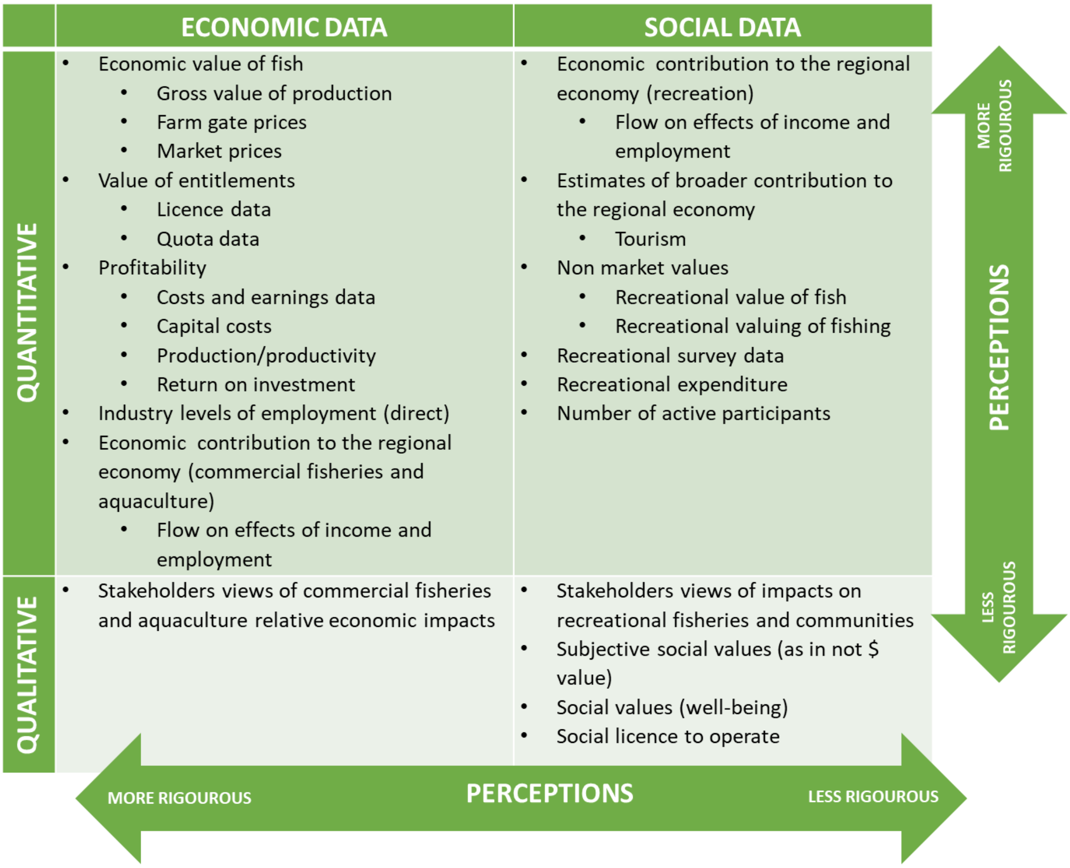
What has been termed as non-market values in this report is a valuation principle that captures the environmental, economic and social impacts of all human uses of marine resources. From a policy perspective the economics values (inclusive of both explicit and implicit costs and benefits) ensures formal consideration of non-priced environmental impacts (e.g. loss of ecosystem habitat) and benefits (e.g. gains from recreational fishing access by the community) in decision making. However, the broadening of social sciences has resulted in many interpretations of what constitutes economics and social values and the corresponding data that captures these values. The interviews were intentionally framed to better understand what fisheries and aquaculture managers and policy makers were interpreting as economic and social data.

All individuals noted that there was a legislative requirement to include economic and social considerations in management, however the objectives of their relevant Acts were quite broad, and were not specific as to how to include these considerations. As a consequence, the use of economic and social information varies considerably, largely based on the availability of data. For example, where there are quantitative data, formal consideration is usually applied; but where there are no data, or only qualitative data, considerations are informal, subjective, and less rigorous.

Across the sample, different types of data were available for use (Figure 6). These included both quantitative and qualitative data. The latter largely included stakeholder views, as well as social "values" such as wellbeing indicators. These qualitative data were generally considered less reliable than quantitative data.

By and large, individuals considered quantitative measures of commercial fisheries and aquaculture data as "economic", but recreational fisheries data and broader community impacts as "social", even if these related to economic measures. These included measures of economic contribution of the recreational sector to regional economies, while the equivalent commercial fisheries contribution was considered economic information. Similarly, where non-market economic values were identified (mostly relating to recreational fisheries), these were also considered "social" by the individuals.

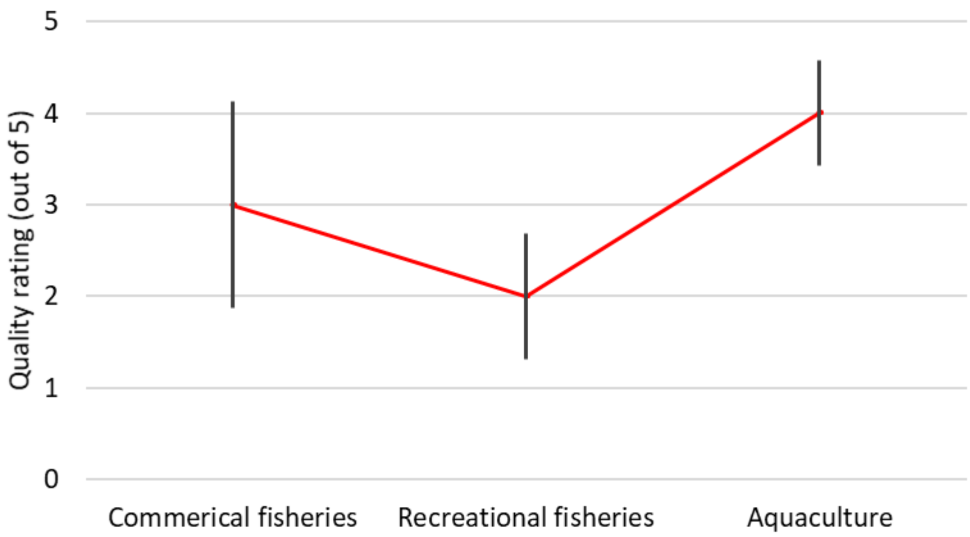
Figure 6. Fisheries and aquaculture managers perceptions of economic and social data



The perceptions of managers of what constitutes economic and social data demonstrates a divergence in understanding of the use of these terms by researchers and managers. For managers, quantitative (non-biological) data relating directly to the fishery is considered economic, while quantitative data relating to fishery impacts or other groups affected by the fishery is considered social, even if these were economic measures as defined by economists. For example, economic contributions to the regional economy from commercial fishing or aquaculture (Figure 6) are considered economic data, but economic contributions to the regional economy by the recreational sector are considered social data. Calls for increased social data by managers, therefore, may reflect a need for data on impacts on the broader community whether "economic" or "social" as defined by the respective researchers.

Perceptions of economic and social data quality varied by jurisdiction (Figure 7), with only aquaculture-related data considered to be generally of good quality. Many respondents suggested that most of their data were outdated, and that there was no mechanism for regular data collection. Where data were available, these were often collected and analysed by third parties, with no direct access to the data. Social data was considered more nebulous than economic data, with no clear frameworks to collect or analyse these data.

Figure 7. Quality of available economic and social data



Experiences with the use of non-market values

All but one respondent was familiar with the concept of non-market values, but their use in fisheries and aquaculture resource management was limited. Respondents suggested that non-market values are not explicitly and systematically used to make resource allocation decisions.

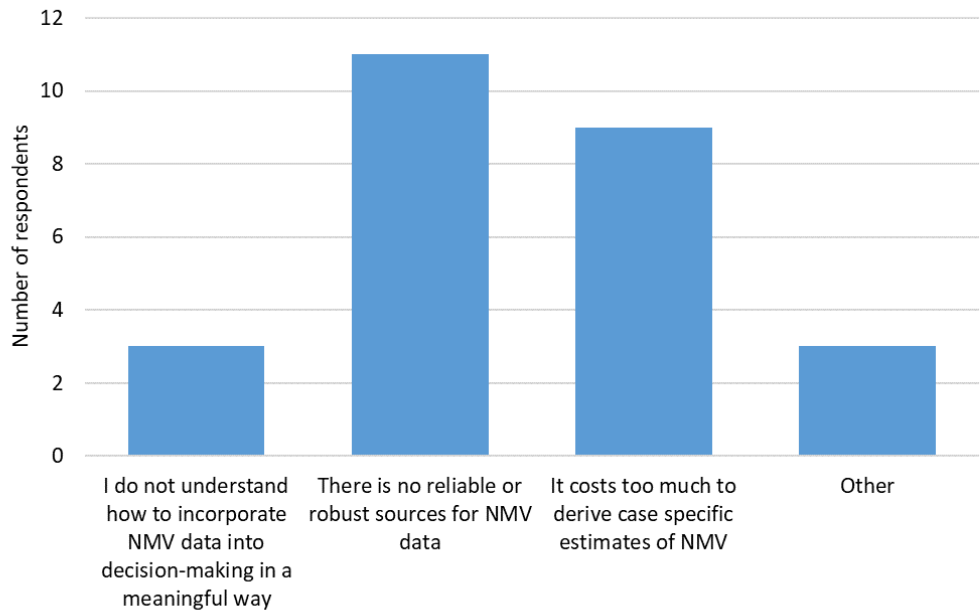
Most respondents believed that the use of non-market values would add credibility to the decision-making process, particularly when involving activities that enhance marine environmental resources. Examples of their potential use given by managers include their use in aquaculture for valuing water quality; to estimate the net benefit of reef restoration programs and to quantify any loss of habitat due to fishing or aquaculture. In practice, there was also a strong perception of some respondents that non-market valuation is primarily used for quantifying and "protecting" the benefit and/or impacts of the recreational fishing industry and/or measuring the social impact.

Only one respondent suggested that they had never been in a situation where non-market values would have assisted decision-making. In contrast, 36% suggested that non-market values would be often useful for their decisions, while the remainder suggested that non-market values would assist some decisions. When available, non-market values were used in the decision-making process by over two-thirds of the respondents.

Barriers to the use of non-market values

Respondents were asked what they thought the key barriers were to the use of non-market values in fisheries and aquaculture management. The key factor identified was their availability (Figure 8). Most respondents indicated that there were no reliable sources of non-market values available for their use, and that deriving the values was too expensive to justify. Relatively few respondents indicated that lack of expertise in the use of non-market values was a barrier. Other reasons given for their non-use included that it was an imprecise science (i.e. lack of trust in the values), and the lack of an established framework for their use.

Figure 8. Barriers to the use of non-market values



Values and non-market values

While most survey respondents indicated a familiarity with the concept of non-market values, discussions in the workshop suggested that the concept was not well understood more broadly. Even within the survey respondent group, there was some uncertainty over some key concepts. Given this, the review section of the project was re-designed to provide an initial overview of the key concepts underlying economic values – both market and non-market. This is aimed at providing managers less familiar with economic thought a basic grounding in the key concepts.

From the manager workshop, key non-market values were identified for which managers and policy makers saw a need. Managers at the workshop were presented with an overview of the types and potential uses of different non-market values that were identified in the literature review, so had a better understanding of these values than when interviewed prior to the workshop. These values were then prioritised in a follow up survey of workshop participants as part of the workshop evaluation. Given this, the review focuses on those values that were identified in order of their relative importance as identified by managers.

Economic values

Marine resources provide a stream of goods and services, some of which are directly beneficial to humans. The use of goods and services that are provided by marine resources generates social wellbeing, which can be measured in terms of economic value.

Economic value is an anthropocentric concept based on consumer choice. Only goods and services that are beneficial to people have economic value. Economic value is a measure of social wellbeing, which is generated through the supply and demand of goods and services. An increase in social wellbeing is defined as a benefit, whereas a loss of wellbeing represents a cost.

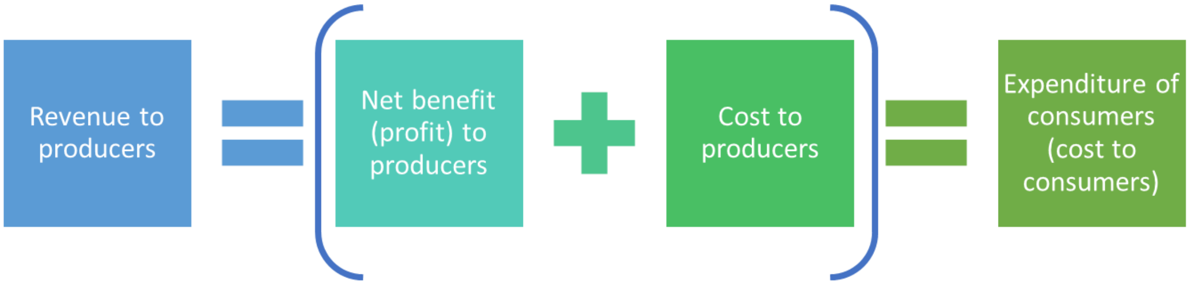
To some degree, economic value is reflected in the expenditure for a good. That is, a consumer will only purchase a product if they value that product at least as much as the market price. This concept underlies many expenditure-based metrices used to value recreational fishing (e.g. revenue from direct spending of recreational fishers), as well as commercial fishing and aquaculture at a basic level (e.g. gross value of product, GVP).

As a measure of economic value, however, expenditure is flawed. The higher the expenditure, the more inputs are also used. As a measure of economic contribution, resource 'waste' is rewarded through expenditure measures. For example, buying 5kg of bait, while increasing expenditure, may have no impact on recreational fishing catch if 0.5kg is sufficient. Expenditure also may reflect other benefits not related to the intended activity. For example, a restaurant meal during a fishing trip increases expenditure, but the value is attributable to the dining experience, not the fishing experience. Finally, the actual benefits to the consumers (e.g. the recreational fishers) over and above the level of expenditure are ignored.

Given this, we can consider economic value to have two components: the net benefit to the producer and the net benefit to the consumer. For the producer, the prospect of earning a profit (difference between revenue and costs) provides the incentive for firms to supply goods and services that people want. The extent of supply is measured by the amount firms are willing to accept as compensation, which reflects their costs of production.

Given this simple framework, we can link revenue of the producer (GVP) to the expenditure of the consumer for a market-based industry (e.g. commercially caught fish) (Figure 9). Within this, part of the revenue is "consumed" through the use of intermediate inputs (e.g. fuel, crew, vessel capital etc), leaving only part as the measure of the actual net economic value generated by the activity (i.e. the profit). These profits form the producer surplus, or the amount of revenue received over and above what is required to provide the good or service (i.e. the cost of its production or provision).

Figure 9. Link between expenditure and industry revenue



Similarly, consumers will only purchase a good if the cost of the purchase is less than the benefit they think they will receive. As a result, the total benefit to the consumer from the good or service will be greater than or equal to the price paid (Figure 10). The net benefit to the consumer is the amount they are willing to pay over and above the cost of an item or activity. This consumer surplus will differ from one individual to the next.

Within this framework, the total economic value is the sum of the net benefit to the consumers (consumer surplus) and producers (producer surplus) (Figure 11). This may differ substantially from the level of expenditure by consumers or the gross value product of the producers (i.e. revenue).

For goods and services traded in the market, the market price and amount of a good or service that is bought and sold in a competitive market is found where demand (determined by the willingness to pay for the goods or services) equals supply (determined by the marginal cost of provision of the good or service).

Figure 10. Consumer benefit, price and consumer surplus

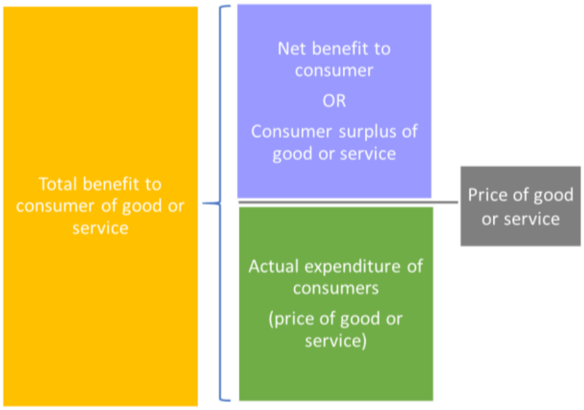
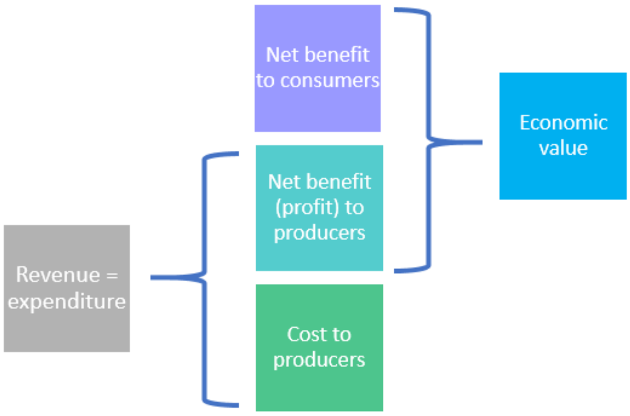
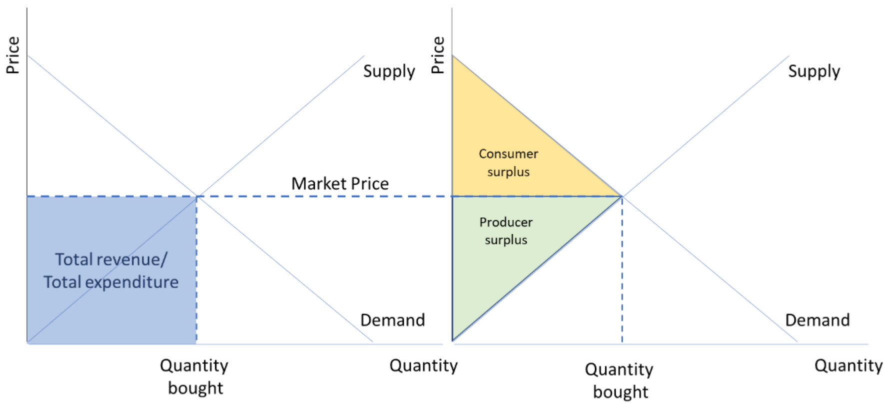


Figure 11. Deriving economic value from producer revenue and consumer surplus



At the market price, buyers purchase the amount that maximises their consumer surplus, whereas sellers supply the amount that maximises their producer surplus. Consumer surplus is the difference between what buyers are willing to pay and the price they actually have to pay. Producer surplus is the difference between revenue sellers earn and the cost they bear. As with Figure 11, the sum of consumer and producer surplus represents the economic value (net benefit) of the 'use' of a marine resource (Figure 12). This sum can vary considerably from the total revenue or expenditure that also occurs at the market equilibrium.

Figure 12. Traditional economic model for a competitive market

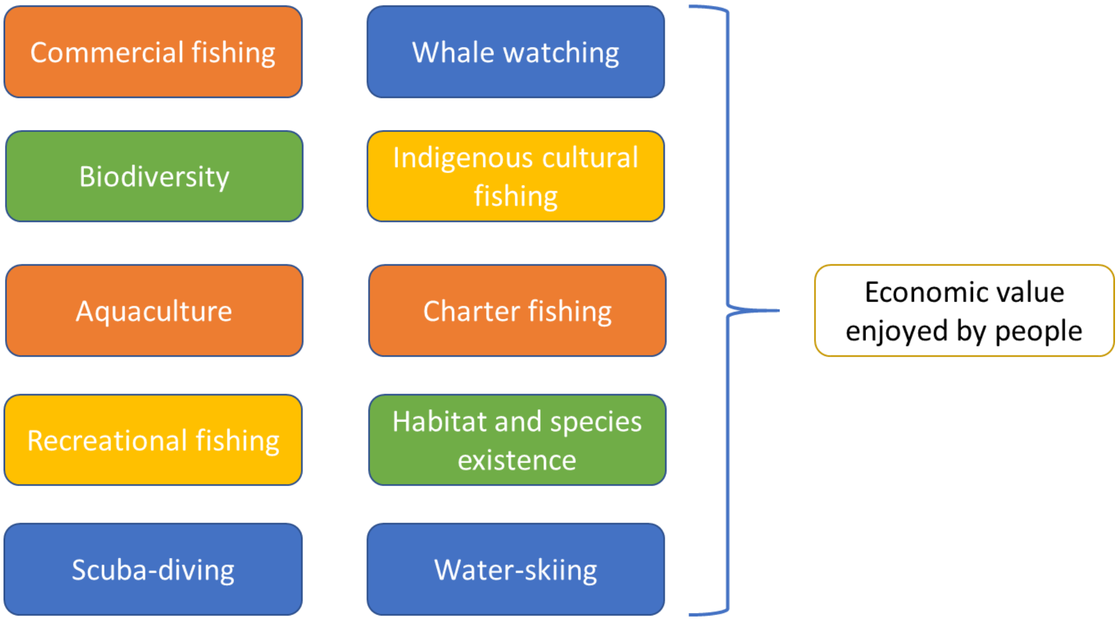


Non-market values

Many goods and services provided by marine resources are not bought and sold in markets. Economic value, however, is still generated through the demand and supply of goods and services, irrespective of the existence or absence of markets. These values are termed non-market values, and are usually measured as consumer surplus from their use.

Some examples of market values (see orange boxes) and non-market values (see gold boxes) of marine resource 'use' are shown in Figure 13. Some 'uses' (see blue boxes) can generate both market values (e.g. commercial whale watching tours, which generates economic profits as well as consumer surplus) and non-market values (privately organised whale watching trips, which just generates consumer surplus). Other values are also generated by the marine environment (see green boxes), such as biodiversity and individual habitats or species that have a value to humans even if not directly used.

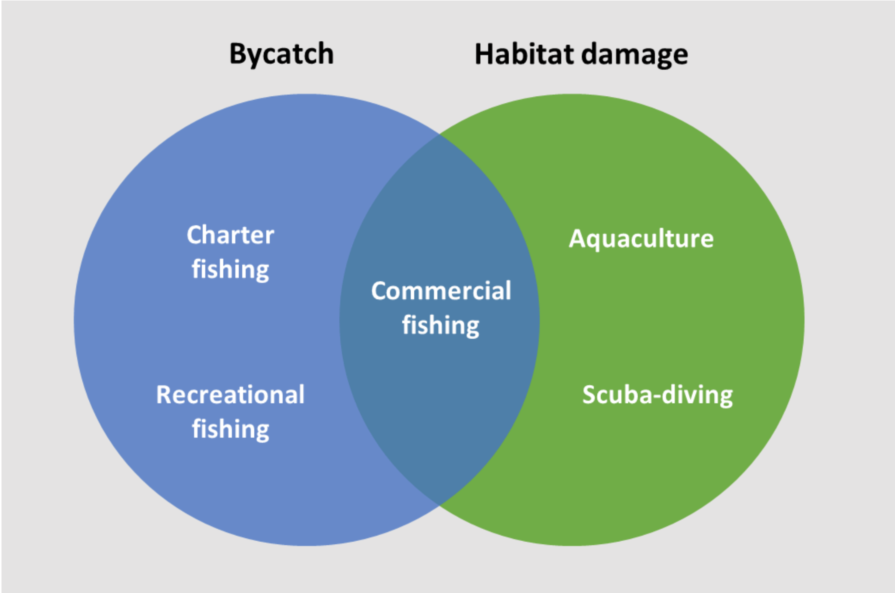
Figure 13. Examples of market and non-market values from marine resources



These values are often classified into use and non-use values (see Appendix B). Recreational fishing and scuba diving are examples of use values (where the benefit is derived through use of the environmental assets), while habitat and existence values may be considered non-use values.

Activities in the marine environment can also produce non-market costs. For example, in Figure 14, commercial fishing may also produce non-market costs through bycatch of some species or habitat damage. Similarly, some of the other activities may also result in some negative externalities. The costs of these unintended outcomes are not included in the decision making of those undertaking the activity. As a result, it is up to policy makers and managers to take these potential non-market costs and benefits into account. An example of how non-market values may aid decision-making is provided in Box 1.

Figure 14. Examples of potential negative non-market values that may be generated from some activities in the marine environment



Box 1. How might non-market values of bycatch of protected species aid decision-making? A hypothetical scenario

A hypothetical scenario is presented where managers are confronted with a problem of dolphin bycatch. The current management results in six dolphin deaths a year. In this example, we assume that there are only two alternative management options that could be applied to reduce this problem. Option 1 involves a series of area closures that modelling suggests would reduce the bycatch of dolphins from six to three, but would also result in the bycatch of an additional three turtles. The cost to the industry of Option 1 is zero. Option 2 involves the use of area closures and new bycatch reduction technology which reduces bycatch of all species to zero, but results in an additional $1m a year in costs to the industry.

		OPTIONS		
		Current management	1	2
BYCATCH	DOLPHINS	6	3	0
BYCATCH	TURTLES	0	3	0
	COSTS	0	0	$1m

The key questions facing managers are: is it better to catch three turtles and save three dolphins; or is the benefit of saving all of these species to society (i.e. reduce bycatch to zero) worth more than the $1m additional cost imposed on the industry? From a societal perspective, if the value of saving six dolphins was more than the $1m additional cost to the industry, then Option 2 would be preferable to current management. Similarly, if the value of a turtle was less than the value of a dolphin (irrespective of the value of a dolphin), then Option 1 would also be preferable to current management. It may also be the case that current management is the most economically efficient outcome, depending on the relative and absolute values of the different species. Without non-market values for the dolphins and turtles, the option that provides the most net benefits to society cannot be determined.

Estimating non-market values

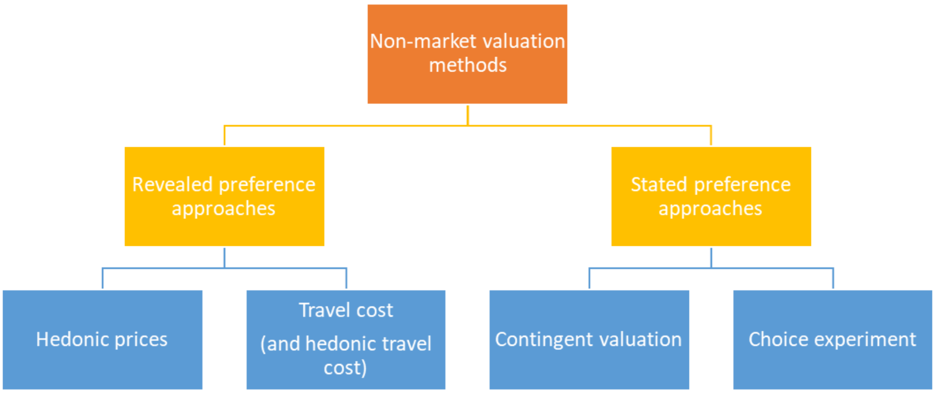
The main role of markets is to generate information on how much people are willing to buy and sell at different prices. This information can be used to measure economic value as noted above. In the absence of markets, non-market valuation methods are required to achieve this task.

There are essentially two types of non-market valuation methods (Figure 15). Further details on these methods are provided in Appendix C.

Revealed preference are based on observed behaviour and contain an element of market value in the derivation of the non-market values. These methods include the estimation of hedonic prices, where values are derived from the characteristics of goods that have a market price such as real estate. For example, differences in house prices can be separated into different architectural features (e.g. number of rooms), and their environmental setting (e.g. near a beach or forest). Travel cost approaches derive consumer surplus values from observed visitation rates and the costs of each trip. These can also be combined with characteristics of different sites (e.g. catch rates, aesthetic values) to derive values for these separate characteristics.

Stated preference approaches are based on individual estimates of willingness to pay for an ecosystem service, either directly in the case of contingent valuation, or implicitly through the choice of a bundle of services using a choice experiment approach.

Figure 15. Non-market valuation methods



The two stated preference approaches and hedonic pricing generally are used to estimate marginal values. The travel cost method can be used to estimate either average values, marginal values or both within the same estimation. Both types of values have a role in fisheries and aquaculture decision making.

Marginal values represent the value of an *additional* unit of the environmental good and are generally smaller than average values. Using recreational fishing as an example, we may estimate the marginal value of catching an additional fish using any of the above methods. This can potentially be used, for example, to assess the potential benefits of a change in bag limits or assessing the potential benefits of a new artificial reef to enhance recreational catch. Marginal values are not necessarily constant. For example, it would be expected that the marginal value of a fourth fish would be less than the marginal value of a third fish, so use of marginal values to value large changes is not appropriate. In the case of aquaculture, when considering changing regulations aimed at improving water quality, the marginal value of improved water quality to the broader community can be compared to the additional costs of infrastructure and potential production losses imposed on the farmers.

In contrast, average values represent the proportion of the total value divided equally amongst all units of a good. For example, in the case of recreational fishing, the average value of a trip can be calculated using the travel cost approach (as described later). From this, the total value of recreational fishing can be estimated from multiplying the average trip value by the total number of trips. This can be useful when considering management options that may change the total number of recreational fishing trips to an area. For example, installing a new boat ramp.

Related approaches

A number of related approaches have also been developed to capture non-market benefits and costs (but in a non-monetary way). For example, utility-based approaches, based on relative strength of preferences of those impacted for one outcome over another, provides an alternative measure of outcomes under different management options. These do not provide an economic value per se, but instead can be used to produce a weighted outcome measure that can be used in cost-utility analysis or some other form of multi-criteria decision analysis. Such an approach may be a useful alternative when there are a large number of environmental or social outcomes that may need to be combined into a single value for determining optimal outcomes. Utility-based measures can also be used to link the derived weights to monetary values (Pascoe *et al.* 2019). These are also further discussed in Appendix C.

Benefit transfer is also often applied as a means to incorporate non-market values into decision making. This is technically not a non-market valuation approach in the same sense that the revealed and stated preference approaches are. Benefit transfer, however, is used to derive appropriate estimates of value using the results of these other non-market valuation studies and can provide a cost-effective approach to deriving costs and benefits of a management decision without requiring direct value estimation (provided appropriate valuation studies have been conducted elsewhere that can be used).

What type of non-market values do managers see as important?

At the workshop, managers and policy makers were asked to identify what non-market values they saw as important to aid in management in capturing the non-market impacts across three areas, commercial, recreational and aquaculture activities, on the wider ecosystem and community. In a follow up online survey, workshop participants were asked to prioritise these values on a 10-point scale.[3] Participants were asked about their likelihood of using the non-market values if they were available, from not likely (a score of 1) to extremely likely (a score of 10). Over half of the workshop participants responded to the survey.

The key values and their average priorities, as identified by the workshop participants, are shown in Figure 16, sorted in order of average importance. The diamond in Figure 16 represents the geometric mean score given by the respondents, while the line represents the range of responses. As there was considerable variability in some of the scores, a geometric mean was used rather than an arithmetic mean to reduce the influence of extreme values.

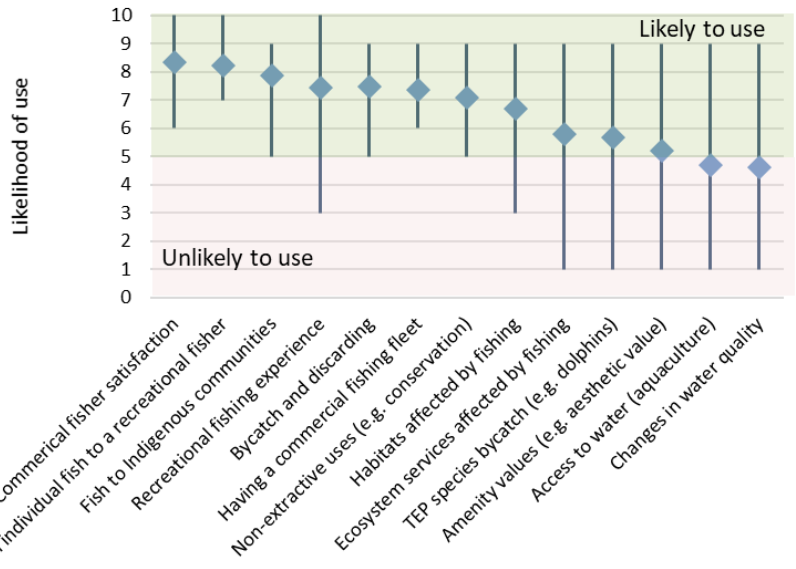
A score of 8 or higher in Figure 16 indicates that managers are fairly to extremely likely to use the values for decision-making. In contrast, a value of 5 or less indicates that managers are unlikely to use the values if available. Most respondents suggested that they were likely to use non-market values related to the use of the resource by commercial, recreational and Indigenous fishers, but likelihood of use was highly variable for values around ecosystem services affected by fishing and aquaculture.

The results in Figure 15 should be interpreted with care. The lower scores attached to likelihood of using non-market values related to aquaculture activities does not reflect their lesser importance and is potentially an artefact of a small sample size. It may also be the case that the inherent link between aquaculture and ecosystem impacts relative to wild fisheries (both commercial and recreational) are considered less problematic by managers in general. An explanation for this could be that aquaculture operations are subject to additional local government planning and health regulations re pollution and

[3] Originally, the prioritisation process was planned for the workshop itself, but time did not permit this. The ranking to derive the relative importance of non-market values presented in Figure 13 is based on a sample that consisted of managers and policy makers involved with commercial fisheries (70%), recreational fisheries (10%) and aquaculture (20%). This distribution may not be representative of Australian managers and policy makers in these three sectors.

noise (Haward 2016). Alternatively, the sample included managers whose primary responsibility was for fisheries, and hence would not be expected use values more relevant to aquaculture. These managers would have rated their likelihood of use of aquaculture values low. The wide variance in responses for these values (ranging from 1 through to 9) suggests that, for some managers – those likely to have responsibility for aquaculture, these values were highly important. The anonymous nature of the data collection does not enable these subtleties to be extracted.

Figure 16. Relative likelihood of use of non-market values identified by workshop participants



What type of non-market values have been estimated?

The following section provides examples of studies from the literature that estimated NMVs across a range of marine resources to inform policy and management decisions. They are presented roughly in order of relative importance as identified by the workshop participants, although some values are grouped where appropriate.

Fisher satisfaction

Having roots in anthropology and latterly economics, fisher satisfaction is now often interpreted as an indicator of the social performance of fisheries management (e.g. Pascoe *et al.* 2014; Brooks *et al.* 2015; Heck *et al.* 2016; Carlin and Morison 2018; Pouso *et al.* 2019). Such measures are routinely collected as part of the economic and social monitoring of key fisheries (e.g., Carlin and Morison 2018). For example, in Australia, qualitative satisfaction measures are a common feature of many recreational fishery surveys (e.g. Tate *et al.* 2020). Qualitative measures of fisher satisfaction in of themselves are not and cannot be interpreted as non-market values. But quantitative measures of satisfaction can be measured using non-market valuation approaches.

Whilst commercial fisher satisfaction was identified as the non-market value of main interest to workshop participants for completeness, the rationale/approach to how fisher satisfaction is captured in non-market and market values for both recreational and commercial fishers is discussed below. Consideration of valuation of fisher satisfaction for both commercial and recreational fishers is appropriate if outcomes under both sectors are to be compared or traded-off.

Fisher satisfaction as an economic concept is not without controversy. In economics, it is normally assumed that wages reflect both the tangible and intangible costs of employment. In a well-functioning market, should a worker be able to be better compensated for their services, then rational decision-making means that they would switch employment. The concept of a non-market value that is attributed to fisher satisfaction is grounded in the reasoning that fishing is more than a livelihood. Effectively it is based on an assumption that fishers are not seeking alterative employment options and will remain in the fishery even when incomes decline (McCay 1988). Within the economics literature, Anderson (2011) developed a conceptual model of the components of economic surplus which included a worker satisfaction bonus, but assumed that it applied in other occupations as well.

Fisher satisfaction has generally not been measured in monetary terms, but usually is measured on a relative Likert or similar scale (more often expressed as a score out of 10). Satisfaction is a multi-dimensional indicator that captures performance of fisheries management but also other external and internal drivers of production (Arlinghaus *et al.* 2017; Brinson and Wallmo 2017). The relative importance of these different drivers of satisfaction may vary between time periods (Arlinghaus, Alós *et al.* 2017) and between individuals, and disentangling these different drivers is complex.

There has been a range of studies that assessed satisfaction of commercial and recreational fishers. The level of satisfaction with fishing and with being part of the commercial fishing industry has been associated with a willingness to remain in the fishery in periods of lower incomes (Holland *et al.* 2019) as well as the desire to exit the industry (Pascoe *et al.* 2015). Satisfaction levels have also been linked to achieving the planned outcomes of management (as well as being affected by management). For example; dissatisfied fishers were less likely to comply with management regulations (Crona *et al.* 2017). Similar links between compliance and satisfaction have been noted for recreational fisheries (Ward *et al.* 2016; Arlinghaus, Alós *et al.* 2017).

The above studies are based on a qualitative rather than a monetary value. However, deriving monetary values from satisfaction-based measures is not without precedent. For example, in other areas, satisfaction based measures have been used to derive non-market values for particular environmental externalities or attributes (e.g. air pollution, scenic amenity, biodiversity conservation) through the derivation of indirect utility functions based on satisfaction measures (Ambrey and Fleming 2011; Ambrey and Fleming 2012; Ambrey and Fleming 2014; Ambrey *et al.* 2014).

While fisher satisfaction is a potentially important metric to support fisheries decision making, the question remains: does satisfaction need to be explicitly monetarised, or are values associated with fisher satisfaction better captured in estimates of other market and non-market values?

To a large extent, the value of "satisfaction" is included in other values, both market and non-market. In recreational fishing, satisfaction is linked to such factors as catch rates. Graefe and Fedler (1986) found that a significant portion of the variance in satisfaction between recreational fishers could be explained by a set of identifiable components of the total experience, with the quantity of fish caught being the main factor driving satisfaction. Arlinghaus (2006) found that absolute catch was not the key determinant of satisfaction, but catch based on expectations was. That is, a fisher expecting a high catch will be satisfied if they catch what they expect, but a fisher with low or no expectations will be satisfied if they catch anything (Arlinghaus 2006). Beardmore *et al.* (2015) found that satisfaction was not just determined by quantity of catch reaching expectations, but also composition of catch, with low catches of trophy fish potentially more important than total catches overall. These factors can all be included in measures of consumer surplus through travel cost models (see Appendixes C and D), and can potentially affect both consumer surplus per trip and the total number of trips taken.

For commercial fishers, the existence of a non-monetary job satisfaction bonus has been established in several fisheries (Anderson 1980; Holland, Abbott *et al.* 2019). Holland, Abbott *et al.* (2019) found that most of this (80%) was attributable to being outdoors, working on the water, maintaining family history in the industry and the identity of being a commercial fisher. These factors are largely not affected by fisheries management. In contrast, livelihood factors such as income and job safety accounted for only 15% of job satisfaction. However, changes in management may affect incomes, and may disproportionately impact satisfaction scores. In a broader (non-fisheries) study, Boes and Winkelmann (2010) found an asymmetry between income and satisfaction, with income playing a relatively small role in overall life satisfaction, but a reduction in income having a disproportionate negative impact on satisfaction scores. In such a case, the impacts of reduced satisfaction can be captured through changes in fisheries incomes, which is captured through market values.

Hence, while measures of satisfaction provide a useful indicator as to how management is being perceived by fishers (both recreational and commercial), for the purposes of assessing changes in costs and benefits the impacts of changes in fisher satisfaction are largely included in other measures. As a result, attempting to value satisfaction and incorporate these changes into a benefit-cost framework may result in double counting if the other changes in values are already captured. Values associated with catches of individual species and the fishing experience which also link to fisher satisfaction were identified as separate priorities for valuation by managers (Figure 16). These values are further discussed in the following sections.

Recreational fishing experience and the value of a recreational fish

While non-market valuation of recreational fishing was considered less important that other measures by the workshop participants, the value of a recreational fish was considered highly important. The estimation of the latter is usually derived as part of the estimation process of the former, so the two components are discussed together in this section. While a large number of studies have been undertaken on recreational fishing values (i.e. the values of the recreational fishing experience), relatively few studies have been undertaken on the value an additional fish per se.

Both types of values are potentially important for fisheries management decision making. An example is presented in Box 2, based on an example presented at the workshop.

A wide range of methods have been applied to estimate the experiential value of fishing as well as the value of a recreational fish.

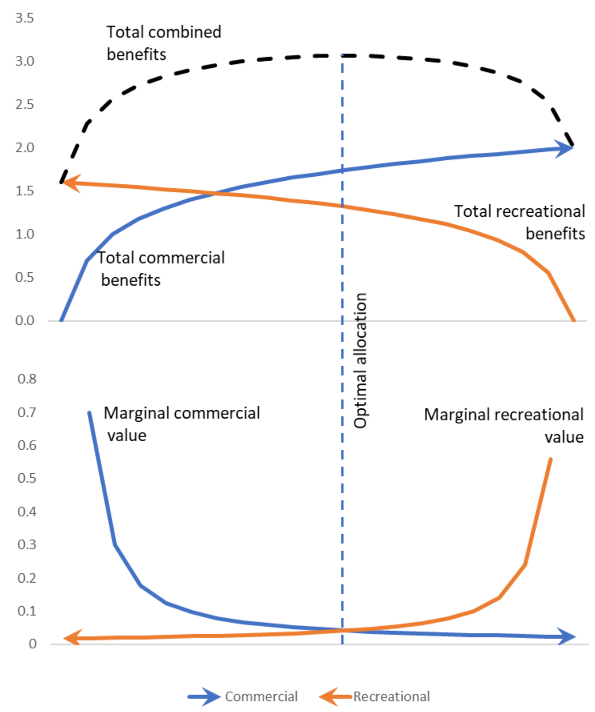
In the coastal and marine environment, the travel cost method has been used extensively to derive values for recreational fishing (Bockstael *et al.* 1989; Grogger and Carson 1991; Shrestha *et al.* 2002; Alberini *et al.* 2007; Rolfe and Prayaga 2007; Prayaga, Rolfe *et al.* 2010; Properjohn and Tisdell 2010; Carlén *et al.* 2019; Deely *et al.* 2019; Pouso *et al.* 2019). These studies have largely estimated average values of recreational use.

A range of values derived from Australian travel cost studies are given in Table 1. From these, recreational fishing values vary substantially, largely due to different assumptions as to what costs are included (e.g. accommodation costs, travel time). These previous studies primarily targeted recreational fishers who were actively fishing at the time of the study (i.e. site-based surveys, often boat ramp based, rather than broader community surveys). Consequently, the likelihood of contacting more serious recreational fishers was higher (the endogenous stratification problem), and it is reasonable to expect that these fishers would place a higher value on the activity. Although corrections can be made to allow for the effects of endogenous stratification, a high proportion of high frequency visitors (higher than the general population) in an on-site survey can still potentially lead to biased estimates of consumer surplus.

Box 2. How might recreational fishing non-market values aid decision-making?

The need to develop specific allocations of fisheries resources between commercial and recreational fishers is increasingly becoming an issue in many Australian fisheries. From a societal perspective, any policy change that involves a reallocation of marine resources across competing uses should achieve an improvement in net economic benefits. Following this argument, the question then becomes: will a reallocation generate additional benefits to the recreational sector that are larger than the additional costs to the commercial sector? Or vice-versa?

To ensure this is the case, the amount of fish catch quota to be reallocated should be determined at the 'margin'. In the adjacent figure, the marginal value of additional recreational fish decreases as the total catch (allocation) increases (i.e. right to left). Similarly, the marginal value of additional commercial catch decreases as it increases (i.e. left to right).



Reallocation will increase net economic benefits (the total combined benefits in the top panel) when the marginal benefit of the two uses are equal (Edwards 1990).

Without the non-market value of recreational fish catch, the allocation that provides the greatest net economic benefits to society cannot be determined.[4]

In contrast, Pascoe (2019) estimated the value of recreational fishing as a primary (i.e. main purpose of the trip) or secondary activity (i.e. undertaken while down the beach rather than the main reason for going to the beach). In this case, the value of fishing as a secondary activity was found to be not significantly different to zero. This value may be more appropriate for the "average" recreational fisher in the broader population, who may participate in fishing but not as a primary activity. Given that 70-80% of recreational fishers only undertake between one and five trips a year, a higher proportion in the sample may have also undertaken some recreational fishing but not identified it as a primary or secondary reason for visiting the beach.

Applying an estimate of the consumer surplus per trip for fishing as a primary reason for the trip to all people identified as participating in fishing at some point over the year will substantially overestimate the total consumer surplus generated by recreational fishing. Based on the results of Pascoe (2019), the consumer surplus generated by the recreational fishing activity itself for almost three quarters of recreational fishers may be zero, with the benefits they received attributable only to the enjoyment of the natural environment or the other activities undertaken.

[4] Abbott (2015) notes that while this approach is theoretically appropriate, the establishment of informal sectoral property or access rights based on existing nominal "allocations" (i.e. current catch shares) would make any reallocation difficult in practice.

Table 1. Willingness to pay estimates (value per trip) from Australian recreational fishing travel-cost studies (since 2010)

Region	Year of data collection	Value in 2018 dollars	Type	Reference
NSW				
Non-Sydney	2018	$30.07	Primary activity	Pascoe (2019)
	2018	$0.00	Secondary activity	
Sydney	2018	$23.75	Primary activity	
	2018	$0.00	Secondary activity	
Queensland				
North Queensland	2007	$479.11	Site access	Prayaga, Rolfe *et al.* (2010)
Gladstone	2016	$148.94	Site access	Windle et al. (2017)
Moreton Bay	2010	$185.38	Site access	Pascoe, Doshi *et al.* (2014)
South Australia				
Coorong	2006	$1,208.97	Site access	Rolfe and Dyack (2011)
Western Australia				
Whole coast	2000	$3-$10	Site access	Raguragavan *et al.* (2013)
Victoria				
Southern bluefin tuna	2010	$39-$155	Fishery access	Ezzy *et al.* (2012)
Salmon/trout	2013	$92-$220	Site access	Hunt *et al.* (2017)

Marginal values have also been derived from travel cost models (Table 2). Pascoe, Doshi *et al.* (2014) included the level of catch in the travel cost model to determine the marginal value of catch, and from that extrapolate to the potential benefits of a marine reserve given various assumptions as to how this affects catch. Raguragavan, Hailu *et al.* (2013) developed a multiregional random utility travel cost model which included catch of different types of species in each region, allowing their marginal values to be determined.

Table 2. Marginal value of a fish derived from Australian recreational fishing travel-cost studies

Region	Year of data collection	Value in study year	Type	Reference
Queensland				
Moreton Bay	2010	$7.50	Lower estimate	Pascoe, Doshi *et al.* (2014)
		$12.90	Upper estimate	
Western Australia				
	2000	$2.30	Butter fish	Raguragavan, Hailu *et al.* (2013)
		$4.65	Table fish	
		$15.94	Prize fish	

Hedonic pricing has also been applied in the context of recreational fishing as a means of estimating the marginal value of sportfishing harvest. Carter and Liese (2010) used data from the market for offshore charter fishing in the Gulf of Mexico, with the marginal value of sportfishing harvest identified based on differences in charter fees charged and spatial variation in harvest rates and fish sizes (taking account also of other factors such as vessel characteristics and distance travelled). They estimated values ranging from $9.36 to $13.25 for an extra fish per angler per trip and from $1.41 to $3.64 for an extra pound on each fish (2003 US Dollars).

Contingent valuation has been applied internationally in several fisheries studies, primarily with regard to recreational fishing (e.g. Sorg and Loomis 1986; Toivonen *et al.* 2004; Deely, Hynes *et al.* 2019). Several Australian studies have also used contingent valuation. The FRDC project "Golden fish: evaluating and optimising the biological, social and economic returns of small-scale fisheries" (FRDC 2016-034) is using contingent valuation approaches to value recreational fishers' willingness to pay for additional catch. The aim of the study is to estimate the potential benefits of a stock enhancement program for the recreational fishing sector. Yamazaki *et al.* (2013) estimated the value of a day's recreational fishing using contingent valuation for two Tasmanian fisheries. The mean WTP for the last day of fishing in the inshore saltwater fishery was found to be $78.18, while the mean WTP for lobster fishing was $87.43 per day.

Discrete choice experiments have been applied in a number of recreational fishing studies (Chen and Cosslett 1998; Train 1998; Breffle and Morey 2000; Provencher *et al.* 2002; Provencher and Bishop 2004; Huang *et al.* 2020). These have also been used to estimate values of ecosystem service enhancement (e.g. through habitat rehabilitation) to recreational fishing (e.g. Huang, Young *et al.* 2020). Whitehead and Lew (2020) combined both travel cost and choice experiments to estimate the values of three key recreational fish species in the US.

Several studies have used meta-analysis to derive values of recreational fish that can be used for benefit transfer purposes. For example, Johnston *et al.* (2006) estimated that the real WTP per fish over the sample ranged from US$0.048 to US$612.79, with a mean of US$16.82 (June 2003 values). The meta-data were drawn from 391 observations from 48 non-market valuation studies that estimate the marginal value (or WTP) that anglers place on catching an additional fish or allow such a value to be calculated. The model results suggest that WTP is systematically influenced by both methodological variables and by resource, context, and angler characteristics. Similarly, Mazzotta *et al.* (2015) used benefit transfer approaches to estimate the costs to recreational fishing from mining operations that affect stock abundance. They estimated values per species using meta-analysis, and found that the values for freshwater bass species ranges from around $15 to around $22 (2012 US dollars) per additional fish caught, and the value of other species range from around $3.80 to around $4.50 per additional fish caught. Sturtevant *et al.* (1995) used meta-analysis as benefit transfer approach for deriving values of freshwater recreational fishing.

Value of fish and fishing to Indigenous Australian communities

Ecosystems in which commercial fisheries operate in Australia also provide values to Indigenous Australian People that are not considered in market interactions. These non-market values are considered as cultural or customary values and may be derived from teaching and learning activities, cultural importance of the regions or species in the ecosystem, maintaining customs and connecting with history (Sangha *et al.* 2017). Consideration of these non-market values is an important issue for fisheries and aquaculture management.

In many Indigenous societies it is considered unethical to value some types of cultural services, particularly spiritual (Daw *et al.* 2015), but to omit them altogether is to risk them being given an implicit value of zero (Sangha *et al.* 2019). By establishing an economic value, decision-making can be improved, providing more sensitive and realistic considerations of relevant costs and benefits. Increasing the understanding of the values that Indigenous Australian People place on customary fishing for example, will ensure that fisheries and aquaculture managers and policy-makers understand the importance of accounting for these values and lead to more efficient catch allocations and water access. There is a need for Indigenous Australian customary values to be includable, thus clearly defined to be appropriately considered in the decision-making process.

The standard methods of economic valuation as a method for determining the relevant values may not be suitable to addressing these cultural and customary values, due to their diverse nature. Adamowicz *et al.* (1998) noted several problems of traditional valuation approaches when applied to Indigenous communities. These included challenges eliciting values for individuals as well as aggregating individual values into measures of social welfare. The influence of sacred or taboo goods, and variations in property rights are factors to address in assessing individual values, while differences in political and property rights systems in Indigenous communities are seen as limits to aggregating values for randomly selected individuals (Adamowicz, Beckley *et al.* 1998). Further, use of these values to compare welfare across culturally different groups (i.e. Indigenous and non-indigenous resource users) was also problematic. Since valuation is endogenous to specific social environments, aggregations of Indigenous and non-indigenous measures of social welfare may be inappropriate (Adamowicz, Beckley *et al.* 1998).

Despite these challenges, some attempts have been made to try and capture appropriate values for Indigenous fisheries. For example, Oleson *et al.* (2015) used a choice modelling approach to elicit cultural and bequest values from Indigenous fishers in Madagascar. They also applied multicriteria decision-making approaches to derive relative preferences as a form of validation of the economic values.

Indigenous values have been captured in other sectors, and some of these approaches may also be applicable for fisheries. For example, McDaniels and Trousdale (2005) ran a series of workshops with Aboriginal Canadian People to understand the cultural significance of resource losses to their communities as a result of petroleum exploration and development. Using a multi-attribute value assessment, the total non-market losses to the Metis community were estimated to exceed $2.6 million per year in 2000 Canadian dollars, whereas the annual compensation received by the community was only CAD$325 000. Hence, ignoring the cultural significance resulted in a substantial underestimate of the loss of benefits to the community. While this example is not related specifically to fisheries, the concepts are transferrable.

Others have identified challenges in deriving “traditional” economic values for Indigenous communities. Most resources used by Indigenous communities provide multiple services. For example, Altman and Branchut (2008) found an overarching Aboriginal view that water is a resource with inseparable cultural and economic values. That is, it has both significant religious and livelihood values. Similarly, Toussaint (2010) noted that fish and fishing in Indigenous societies has multiple values, and that what constitutes value cannot be understood by adopting a singular or fixed form of comparison. In this regard, standard valuation approaches may only capture one element of the true value of the resource, and Toussaint (2010) opted for the use of multicriteria approaches to derive relative values (rather than monetary values).

Sangha, Stoeckl *et al.* (2019) estimated Indigenous Australian values associated with Northern Territory fisheries through two approaches. The first approach used market prices to determine equivalent replacement costs for catch by Indigenous fishers (i.e. what they would have had to pay for the fish if they were bought). The second approach was aimed at estimating the well-being benefits that people obtain from their coastal/marine systems. This was assumed to equate to 25% of the total government welfare expenditure on healthy lives, early childhood learning and development, secure environment, and economic development that directly link to the services people accrue from coastal and marine resources, following an approach used previously by Sangha, Russell-Smith *et al.* (2017).

Jackson *et al.* (2011) recognised the multiple service function of resources to Indigenous communities, and also recognised that, in some cases, capturing only part of these values may be possible. In their study, they considered only replacement values (i.e. equivalent market values) to represent the harvest and consumption of subsistence food items. The use of gross replacement values was in part to minimise the use of a survey, responses from which they considered variable at best, and to also allow for a straightforward calculation of these values, preserving the easily understandable nature of the replacement value method (Buchanan *et al.* 2009). Replacement values (based on market prices) were also used by Gray and Altman (2006) to value subsistence fisheries harvest by an Indigenous Australian community in NSW.

Zander and Straton (2010) applied a choice experiment approach to estimate the value of fishing quality to Aboriginal residents in three river catchments in Queensland, the Northern Territory and Western Australia. They found that the WTP of Aboriginal respondents to access “good waterholes” was $522/person/year compared to “ok waterholes” with a value of $280/person/year.

Value of having a commercial fishing fleet to tourism and local communities

Fisheries can provide other benefits to local communities, particularly in the area of tourism and provision of local seafood. There is growing evidence that the existence of a fishing fleet in a region can also provide intrinsic benefits, with the ability to see the fishing industry and sample local seafood can enhancing the tourism coastal experience. For example, Voyer *et al.* (2016) found that 76% of New South Wales (NSW) residents were of the view that eating local seafood was considered an important component of the holiday experience while visiting the coast. Furthermore, 64% of NSW residents expressed interest in observing fishers at work during their holiday (Voyer, Barclay *et al.* 2016).

Developing monetary values for these non-market benefits has had little previous attention, although there have been some recent studies in the Australian context. Research on the value of fisheries for tourism and the local coastal communities has recently been undertaken as part of a PhD research project (Paredes 2020). The study used a contingent travel cost method to estimate the contribution of supplies of local seafood and the option of seeing the local fishing fleet to the value of a trip to a coastal town in Southeast Queensland. The results of the study indicate that the fishing and seafood industries are of value to day trippers and holiday makers, such that the loss of these industries would reduce the value per trip (on average) (Paredes 2020). The study also estimated the value of consuming locally caught fish by the local community, building on earlier analysis of Pascoe *et al.* (2016). Using a choice experiment, the study estimated that the community valued locally caught fish 15% higher than non-local fish of the same species.

Examples of these recent Australian studies are given in Table 3. Again, these studies provide an indication of the positive externalities created by the existence of a fishing fleet in an area, which may affect decisions in some circumstances that result in fleet restructuring.

Table 3. Willingness to pay estimates relating to the value of a commercial fishing fleet to tourism and local communities

Region	Year of data collection	Value in 2018 dollars	Type	Reference
Qld	2018-2019	$10.51/trip	Value of local seafood to tourists	Paredes (2020)
		$7.36/trip	Value of local fishing industry to tourists	
	2016	$5.90/kg	Value of local seafood to local residents	Paredes (2020), Pascoe, Innes *et al.* (2016)

Further examples of fisheries related tourism include seafood festivals (Claesson *et al.* 2005; Kim *et al.* 2017), as well as through "pescatourism" – where tours involving working fishing boats are undertaken (though the latter has primarily been undertaken in Europe) (Piasecki *et al.* 2016). The economic benefits generated by these activities can usually be assessed using market-based methods, such as contribution studies. These are not non-market values per se, but do reflect positive externalities generated by the fishing industry to other sectors that should be considered in any management that changes the level of fishing activity.

Value of non-extractive uses

In the coastal and marine environment, the travel cost method has been used to value recreational use in marine parks (Pendleton 1995; Park *et al.* 2002; Bhat 2003; Chae *et al.* 2012); whale watching (Loomis *et al.* 2000); scuba diving (Tapsuwan and Asafu-Adjaye 2008; Du Preez *et al.* 2012; Pascoe *et* *al.* 2014); and beach visitation (Bin *et al.* 2005; Pendleton *et al.* 2006; Yeh *et al.* 2006; Blackwell 2007; Rolfe and Gregg 2012; Zhang *et al.* 2015; Windle *et al.* 2017; Pascoe 2019).

Examples of some recent Australian studies are given in Table 4. Most of these studies are of little direct use to fisheries or aquaculture management but are of potential use for broader marine multiple use and environmental management. Examples where such values may be of benefit are for determining potential benefits of different zonal allocations in multiple use marine reserves.

Table 4. Willingness to pay estimates relating to the value of coastal non-extractive uses

Region	Year of data collection	Value in 2018 dollars	Type	Reference
NSW				
Sydney residents	2016-2017	$24.87	Beach use values	Pascoe (2019)
Other residents		$12.78	Beach use values	
Qld				
	Not stated	$41.18	Beach visit	Rolfe and Gregg (2012)
	2011	$22.40	Beach visit	Zhang, Wang *et al.* (2015)
	2014	$37.17	Beach visit	Windle, Rolfe *et al.* (2017)
	Not stated	$9.55-$14.37	Beach use per person (local residents)	Prayaga (2017)

Note: when year of data collection is not stated, values are inflated based on their year of publication. Otherwise based on the year of the data collection.

Value (non-market cost) of protected species, habitats, commercial bycatch and fishery waste

To our best knowledge, no non-market valuation study exists to date focusing on commercial bycatch and fishery waste. Commercial bycatch and waste (often considered synonymous) in this case refers to commercial species that are caught and subsequently discarded. Reducing this waste is a key priority for fisheries managers and has been linked to the social licence to operate in commercial fisheries (Murphy-Gregory 2018; van Putten *et al.* 2018).

In terms of management, the social cost of commercial discarding has similar implications as that for bycatch of protected species or habitat damage (with the caveat that the social cost associated with these may vary) on the optimal levels of catch/effort. These implications are illustrated in Box 3.

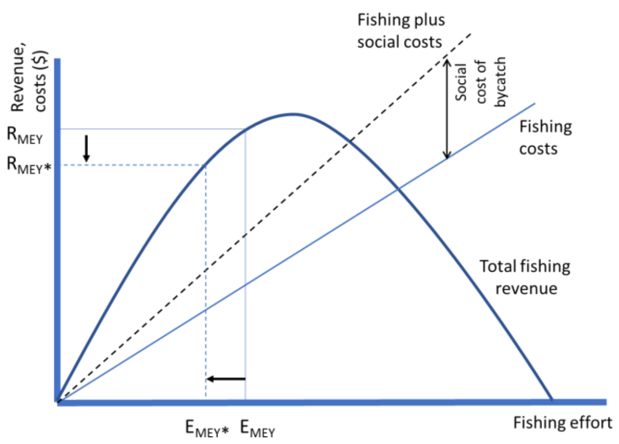
Choice experiments and contingent valuation have been used to value protected or endangered species and habitats that interact with fisheries. For example, Johnston *et al.* (2015) and Wallmo and Lew (2016) estimated values of several species and habitats including turtles, whales, seagrass and coral in the US using choice experiments. Lew (2015) reviewed such studies internationally and found that most were undertaken in the US, with European studies comprising most of the remainder. Further, these studies mostly estimated the value of protecting or recovering species, and most did not estimate the value of an individual species. Examples of these values ranged from US$22 for an individual loggerhead turtle to US$100 per household to improve the conservation status of Hawksbill turtles in general. In Australia, Farr *et al.* (2014) used CVM to estimate the value of seeing a range of different iconic species on reef trips in the Great Barrier Reef. While not directly related to fisheries impact, it provides a measure of value of these species (although this also includes an experiential component also).

Lew (2015) also identified a number of studies that valued recovery of commercial stocks (over and above the value of the provision ecosystem services they produced). For example, one contingent valuation study estimated a willingness to pay of US$22.96 and US$35.63 for programs to ensure stocks of Norwegian lobster and European hake, respectively, are recovered above their minimum viable population level. This is effectively a non-use value held by the general population for the protection of commercial species.

Box 3. Implications of the social cost of bycatch and habitat damage

The social cost associated with bycatch or habitat damage can be considered in the same way as fishing costs in the determination of economic target reference points (e.g. maximum economic yield). For simplicity, and assuming a linear relationship between the level of bycatch/damage and fishing effort, including the social cost increases total fishing costs, which results in a lower level of optimal fishing effort (E_{MEY} to E_{MEY*}) at a lower yield and revenue of the target commercial species (R_{MEY} to R_{MEY*}). Further details of this can be seen in (Pascoe, Hutton *et al.* 2018).

Understanding the value of TEPs and bycatch can also help resolve management issues such as those identified in Box 1.



In the coastal and marine environment, the travel cost method has been used to value some key habitats that can be affected by fisheries, such as coral reefs (Seenprachawong 2001; Carr and Mendelsohn 2003; Seenprachawong 2003; Ahmed *et al.* 2007; Andersson 2007; Andersson 2007; Tapsuwan and Asafu-Adjaye 2008). In Australia, Huang, Young *et al.* (2020) used a random utility based travel cost model to estimate the contribution of seagrass to recreational fishing benefits.

Choice experiments have also been used to a lesser extent. Börger *et al.* (2014) estimated conservation values of marine protected areas for comparison to lost fishing opportunities in these areas. In Australia, Rogers (2013) used a choice experiment approach to establish differences in preferences (rather than values) between scientists and the broader community for marine reserves. Pascoe, Doshi *et al.* (2019) used a combination of choice experiments and preference elicitation techniques to estimate the non-market value of a range of coastal and marine habitats that could potentially be affected by fisheries. Davis *et al.* (2019) also used a choice experiment to value habitat protection, and found that this increased linearly up to around 25% additional protection, after which the value of additional protection decreased.

Table 5. Willingness to pay estimates relating to the value of Australian habitats and species that may be affected by fishing and aquaculture

Region	Year of data collection	Value in 2018 dollars	Type	Reference
Habitats				
NSW				
Seagrass	2016-17	$109.45	$/ha/household	Pascoe, Doshi *et al.* (2019)
Rocky reefs		$102.55		
Sandy seabeds		$121.87		
Mangroves		$110.66		
Estuaries		$58.11		
Wetlands	Not stated	$5,175	$/ha	Taylor and Creighton (2018)
Victoria				
Seagrass	2014-16	$19.20	$/trip for 10% increase	Huang, Young *et al.* (2020)
Qld				
Habitat protection	2017	$1.80 - $4.70	per percentage point increase in habitat protection/household	Davis, Burton *et al.* (2019)
Species				
Qld				
Sharks and Rays	2007-2009	$41.38-$51.56	Value of seeing the species $/trip	Farr, Stoeckl *et al.* (2014)
Turtles		$28.95-$38.71		
Whales and dolphins		$53.50		
Large fish		$22.31		
Variety of species		$38.74		

Production function approaches have been used to estimate the contribution of habitats to fisheries (Kurniawan *et al.* 2020). In Australia, Taylor and Creighton (2018) used a production function approach to estimate the values of coastal wetlands to prawn production in order to justify restoration works in the area. This approach does not estimate additional benefits or costs per se, but identifies the marginal contribution of habitats and other environmental assets to producer surplus. As a result, their measurement can allow estimates of changes in these assets.

Amenity values of marine ecosystems

Amenity values capture the benefits derived from the appreciation of a particular area. Amenity values can be both tangible (e.g. visual amenity) and intangible (e.g. cultural heritage). Fishing and aquaculture activities have the potential to result in a deterioration of amenity values. For example, an aquaculture development may reduce the amenity value of the marine environment as a result of increase noise from operations and impact on the natural beauty of an area.

Only few studies have estimated amenity values for natural resources, and particularly, of marine resources. Blignaut *et al.* (2016) used contingent valuation to value the amenity value of the coastal and marine resources of Abu Dhabi to beach visitors. Others have used hedonic property pricing to estimate the amenity values of National Wildlife Refuges near urban areas on the eastern coast of the US (Xiangping *et al.* 2013) and of wetlands in Perth, Australia (Tapsuwan *et al.* 2009).

Changes in water quality in aquaculture production

Several studies have considered the negative impacts of aquaculture on water quality, availability and visual amenity through an examination on house prices in areas close to aquaculture facilities (Huang 1990; Evans *et al.* 2017). Chen *et al.* (2006) used such hedonic prices to estimate the optimal level of aquaculture production given the values of the externalities associated with groundwater use and water pollution.

Other Australian studies have estimated the value of changes in water quality. Schrobback, Pascoe *et al.* (2018) used a production function approach to estimate the economic impact of changes in water quality on oyster production. Alam *et al.* (2006) and Hatton MacDonald *et al.* (2015) used a choice experiment to estimate the value of changes in water quality to local communities. While not directly related to aquaculture, the results could be extrapolated to water quality impacts of aquaculture.

Other values

Non-market values have been applied in a number of other fisheries and aquaculture related contexts. For example, Bennear *et al.* (2005) used hedonic pricing to estimate the implicit value of fishing licences in the US, where the value of the licence was capitalised into the value of the vessel. These values provide an indication to managers of changes in economic performance (e.g. Arnason 1990; Pascoe *et al.* 2019).

Hedonic pricing has been applied in a wide range of fisheries and aquaculture contexts, predominately around the contribution of fish characteristics to the price consumers were willing to pay in the market place. Examples include studies of the influence of landed size on price (Asche *et al.* 2014; Lee 2014), where the fish was caught (reflecting the environmental conditions of the region) (Asche and Guillen 2012; Asche, Chen *et al.* 2014), and how the fish was caught (Asche and Guillen 2012). Other studies have considered the value of sustainability certification (Roheim *et al.* 2011). Such values are useful for managers when considering management options that may change the characteristics of the catch (e.g. size or environmental certification).

The potential use of benefit transfer[5]

Benefit transfer uses research results from pre-existing primary studies at one or more sites or policy contexts (often called study sites) to predict welfare estimates or related information for other, typically unstudied sites or policy contexts (often called policy sites). Values from a source study are transferred to other situations with appropriate adjustments.

Benefit transfer has an immediate intuitive and pragmatic appeal in its application within economic evaluation, particularly where time and funding are constrained or not justified given net benefits of the proposal are relatively small. Time and costs savings can be potentially realized through avoiding expensive (both in terms of costs and time) primary research including lengthy data collection exercises and employing expertise in relevant non-market valuation estimation techniques.

The use of benefit transfer is not without precedent. It is an indispensable component of virtually all large-scale cost-benefit analyses in the United States, European Union and elsewhere. Australia is no exception with values derived for use in policy decision contexts including health and the statistical value of life (Department of the Prime Minister and Cabinet 2018), as well as transport and the value of time saved (Australian Transport Assessment and Planning n.d.).

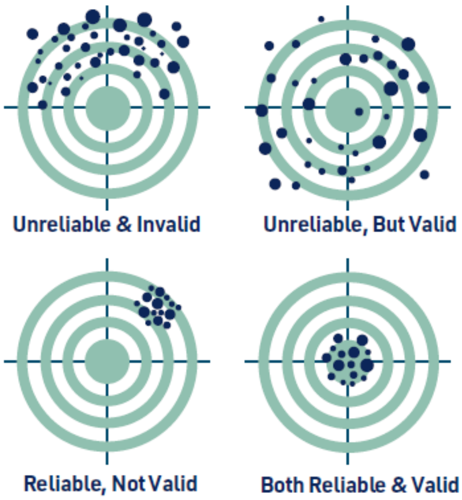
[5] This section was adapted from a presentation by John Rolfe at the FRDC Non-market Valuation Workshop, Brisbane, October 2019.

However, despite the fact that benefit transfer is well-embedded in economic evaluation, the critical prerequisite of the existence of relevant studies (source studies) that have estimated non-market values for similar benefits/costs presents a major road block in its application to deriving non-market values for use in Australian fisheries and aquaculture policy decision making.

Evaluation of candidate studies

To ensure validity and reliability of benefit transfers, source studies (also referred to as primary studies) should be similar in characteristics to that of the policy target site. This includes biophysical conditions, type and scale of the environmental change and population characteristics. In addition, the primary studies also need to be robust and technically competent. Studies that are unreliable and/or invalid may result in substantial transfer errors (Figure 17).

Figure 17. Reliability, validity and transfer error



Source: Dilmen (2012)

The potential for invalid and unreliable benefit transfers is large. Inevitability there will be variations between source sites and populations. Variability between the source study and target site will result in generalisation errors, and reduce the reliability and accuracy of the transfer values. Validity can be improved by controlling for measurement error through avoiding unreliable source studies and thereby avoiding the transfer of errors from the original primary studies.

Benefit transfers are not a panacea for estimation of non-market values and therefore does have to be approached with caution. There is no common reporting standard with respect to published nonmarket valuation studies (Menegaki *et al.* 2016) and evaluation of existing research is limited (see for example (Subroy *et al.* 2019; Gunawardena *et al.* 2020). In the case of fisheries and aquaculture this problem is compounded by a general lack of non-market valuation studies in general.

In addition to the above, there is also no consensus on protocol regarding the benefit transfer process. At a practical level, this has meant that practitioners often make informal, and sometimes uninformed judgments, about the applicability and importance of different types of similarity when identifying primary studies. Furthermore, the benefit transfer values are also sensitive to the value transfer method used (the method by which a value is transferred from the primary study to the policy site – which is further discussed below – are not always applied appropriately (Johnston *et al.* 2015)).

Value transfer methods

There are three main value transfer methods (i) unit value transfer (ii) value transfer function and (iii) meta-analytic function transfer.

Unit value transfer involves the transfer of a single value from the primary study to the policy site with adjustments to reflect differences in income and purchasing power. Whilst it has intuitive appeal, accuracy is highly dependent on there being close similarity between the primary study and target site.

Value transfer functions uses a value function to calibrate the value being transferred. The coefficients of the value function for key characterises of the primary study (e.g. ecosystem characteristics and population and demographic characteristics) denoted as $X_{primary}$ and $Z_{pirmary}$ (Equation 1) are transferred to the value function for the policy site, denoted as X_{policy} and Z_{policy} and the transfer value is estimated (Equation 2).

$$
\begin{array}{llll}
(1)\ \text{Primary study value} & = \alpha + \beta_{primary} & X_{primary} + \gamma_{primary} & Z_{primary} \\
 & & \downarrow & \downarrow \\
(2)\ \text{Transfer value} & = \alpha + \beta_{policy} & X_{policy} + \gamma_{policy} & Z_{policy}
\end{array}
$$

The benefit transfer literature commonly finds that transfers of benefit functions outperform transfers of fixed (unit) values when sites are not identical. There is also evidence that value transfer functions are reliable in transferring values between international sties (Andreopoulos and Damigos 2017).

The meta-analytic function transfer estimates the value function from the results of multiple primary studies. Value estimates of site attributes and methodological approaches are collected from primary studies. Parameters are estimated using a meta-regression model and corresponding benefit transfer values are then estimated using the characteristics of the policy site.

Whilst the approach has been applied in the estimation of non-market values in ecosystems, Johnston and Bauer (2020) identified several issues in the use of meta analytic function transfer in generating valid and reliable benefit transfer values. These included the use of appropriate econometrics and specification issues with respect to the sensitivity of values to scope, scale, and spatial dimensions. They also noted that the estimation of meta-analytic function transfer estimates for ecosystem service valuation requires robust primary study data, with sufficient primary study numbers necessary for meta-regression model. In fisheries and aquaculture, limited primary studies would rule out this approach.

Regardless of the methodical approach adopted, benefit transfer does have to be approached with care. Below are the common pitfalls to avoid.

- Identifying the total values for an entire environmental asset instead of the policy relevant question of marginal values for clearly specified changes in protection or condition. For example, valuing total marine habitats rather than change in the marine habitat. Similarly, average values in other studies may not reflect marginal values associated with the change in marine habitat.
- Inaccurately or imprecisely identifying the key elements that provide economic value. This can result in potential for double counting or omission of values when elements overlap or do not cover the full scenario of interest.
- Transferring values between very dissimilar source and target studies. The primary study or studies should be similar to the target policy site.
- Relying on source studies that are not robust and do not follow best practice standards.
- Transferring values without adjustment for variations in policy scale or geographic scope.
- Transferring values without adjustment for variations in other important factors, such as population differences.
- Extrapolating values to larger populations than is appropriate, or ignoring the fact that values may decline as one moves further from an affected resource or area.

- Ignoring the fact that iconic sites (e.g. Great Barrier Reef) or resources will often have greater value than non-iconic sites or resources.

Where can we find values for benefit transfer?

There are a number of online environmental valuation databases that provide access to historical environment valuation studies. Whilst these databases include valuation studies relevant to fisheries and aquaculture where they exist, there is no evaluation database that is specific to fisheries and aquaculture or any that are maintained or managed within Australia. A key objective of these databases is to broaden access to valuation studies and facilitate the use of existing non-market values in decision-making through benefit transfer. An overview and audit of the key non-market databases are presented in the following section.

Audit of non-market valuation databases

For this audit, four key databases were examined: the Environmental Valuation Reference Inventory (EVRI); the National Ocean Economics Program (NOEP) Non-Market Valuation Database; the Marine Ecosystem Services Partnership (MESP) Mapper & Valuation Library; and the Recreation Use Values Database (RUVD). The databases that were audited all drew on a wide range of sources including published peer review literature, government reports and other grey literature. All of the databases reviewed were free to access and provided summary level information from the valuation studies. Access to the full version of a specific study may incur an additional cost or be restricted depending on publisher access rights.

A summary of each of the four databases used in the review is provided in Table 6, while further details on the contents of each database are presented in the following sections. A number of other databases were identified. However, many of these have not been updated for several years. These are also briefly described below.

Each of the four main databases were searched for the relevant priorities and their sub-topics where applicable. The studies were screened based on their valuation method (non-market valuation methods), publication date (2010-2020) and direct relevance to the given priority. For example, with respect to the value of recreational fishing – studies which valued recreational fishing as the primary activity being valued were included in the study. Studies which captured recreational fishing as part of a broad range of recreational activities were only included if the study provided a separate value for recreational fishing.

Table 6. Summary of the key features of the databases examined

	EVRI[1]	NOEP[2]	MESP[3]	RUVD[4]
Free text search	Yes	Yes	Yes	Excel spreadsheet which can be searched
Guided search	Yes	Yes	Yes	Use codes to sort through studies
Abstract	Yes	Yes	Sometimes notes include abstract	
Summary of NMV	Yes	Yes	Yes	Yes
Provides year of data collection	Yes	Yes		Yes
Provides Method	Yes	Yes	Not always	Yes
Export results into excel doc	Yes			Yes (entire database in excel)
Australian studies	Yes	Yes	Yes	
Non-Australian studies	Yes	Yes	Yes	Yes
Updated in the last 10 years (2010-2020)	Yes	Yes	Yes	Yes
Most recent paper	2020	2019	2017	2016
Additional features	Detailed filter on search page (e.g. year, country, ecosystem etc). Can be downloaded as csv or xlxs format	Detailed filters, but must be applied prior to search.	Interactive map function. Access to bibliographic list of the database.	Entire database can be downloaded as excel spreadsheet. Stated Preference/ Revealed preference studies. Original and 2016 values reported.

Notes
[1] Environmental Valuation Reference Inventory (EVRI)
[2] National Ocean Economics Program (NOEP)
[3] Marine Ecosystem Services Partnership (MESP)
[4] Recreation Use Values Database (RUVD)

Environmental Valuation Reference Inventory (EVRI)

The Environmental Valuation Reference Inventory (EVRI) (Environmental Valuation Reference Inventory 2020) was developed by the Environment and Climate Change Canada in collaboration with valuation experts and organisations from across the globe. The database was launched in 1997 and has since grown to include 4000+ summaries of environmental and health valuation studies. The database can be accessed once a free account has been made with EVRI. The database is not only one of the largest databases that was reviewed, but is also one of the most recent, with the database being continually updated. The homepage of the database also provides users with notification of the latest studies that have been added into the database. Users are also encouraged to submit studies to the database.

The database allows for both free text search and a guided search. Once a search is completed, users can filter the search by: recency of publication (last 5 or 10 years), document type, region, environmental assets, type of value/ usage, study type, valuation techniques and economic measures. Each study summary generally includes: the reference details of the study, the abstract of the study, classification of study, identification of the focus of the study, methods used and the values produced by the study. A key feature of the database is the ability to save completed searches to the users online account or the results can be exported into xlsx or csv format. EVRI also provides users with access to a comprehensive user manual.

Table 7. Studies identified in the EVRI database 2010-2020

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
Fisher satisfaction	0		0	Stoll and Ditton (2006)
Value of recreational fishing				
• Freshwater	4	Hunt, Scarborough *et al.* (2017)	15	Grilli *et al.* (2018)
• Sea/coastal	3	Pascoe, Doshi *et al.* (2014)	12	Barrientos *et al.* (2017); Hynes *et al.* (2017)
• Mix of locations	1	Hailu *et al.* (2014)	1	Vesterinen *et al.* (2010)
Marginal value of a fish				
• Freshwater	0		4	Metcalfe *et al.* (2018)
• Sea/coastal	1	Pascoe, Doshi *et al.* (2014)	3	Carter and Liese (2010)
Value of fish to Indigenous communities	0		2	O'Garra (2012); Oleson, Barnes *et al.* (2015)
Intrinsic value of having a commercial fishery	0		1	Ropars-Collet *et al.* (2015)
Value of non-extractive use				
• Beach recreation/ protection	2	Prayaga (2017)	20	Glasgow and Train (2018); Leggett *et al.* (2018); Lyon *et al.* (2018); Mayer and Woltering (2018); Talpur *et al.* (2018); Zambrano-Monserrate *et al.* (2018)
• Surfing	0		1	Christie and Gibbons (2011)
• Kayaking	0		1	Bell *et al.* (2012)
• Wildlife watching/viewing	0	Wilson and Tisdell (2003)	3	Paltriguera *et al.* (2018)
• Diving	0	Kragt *et al.* (2009)	9	Roberts *et al.* (2017)
Value of habitat				
• Marine/coastal habitat protection	3	Pascoe, Doshi *et al.* (2019)	9	Guignet *et al.* (2017)
• Marine protected area	1	Gillespie and Bennett (2011)	29	McClenachan *et al.* (2018); Paltriguera, Ferrini *et al.* (2018); Yu *et al.* (2018); Zambrano-Monserrate, Silva-Zambrano *et al.* (2018)
Value of protected species	0	Carr and Mendelsohn (2003); Wilson and Tisdell (2003)	17	Cazabon-Mannette *et al.* (2017); Halkos *et al.* (2017); Lew and Wallmo (2017); Lim *et al.* (2017)
Value of bycatch/ fishery waste	0		0	
Amenity values of marine resources	1	Prayaga (2017)	8	Leggett, Scherer *et al.* (2018)
Value of water quality	2	Hatton MacDonald, Ardeshiri *et al.* (2015)	32	Alemu I *et al.* (2019); Shan *et al.* (2019)
Value/use of water (environment) affected by aquaculture	0		5	Aanesen *et al.* (2018)

Notes: The same study could have multiple entries; References in blue are more than 10 years old but are the most recent identified study

National Ocean Economics Program (NOEP) Non-Market Valuation Database

The National Ocean Economics Program (NOEP) non-market valuation database (National Ocean Economics Program 2020) provides users free access to summaries of studies relating to marine and coastal resources. Submissions to the database are encouraged and are subject to review. Although NOEP is primarily based in the USA, the database also includes studies from across the globe. International studies (i.e. studies from outside the USA) are identified with three red asterisks.

The database allows for both free text search (in both the title, keywords or authors) and a guided search – or combination of both. The search can be customised through a range of filters (publication type, year, location, recreational activities, assets valued, methodology, data source, non-use values, order, number of records to show), however must be applied before starting the search (i.e. the results cannot be further filtered on the results page – they can only be filtered when on the search page). The results can only be accessed while on the site.

For each entry on the page, the following details are displayed: Title, authors, publication year, source (bibliographic details of source), publication type, document type, assets valued, recreational activities and location. Each entry also has a 'See Details' link which provides more detailed information about the entry. In addition to the details shown on the results page, the 'see details' link also provides: the location description of the study, methodology, abstract and 2010 value estimates.

Table 8. Studies identified in the NOEP database 2010-2020

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
Fisher satisfaction	0		0	
Value of recreational fishing				
• Freshwater	1	Zander *et al.* (2010)	3	Deely, Hynes *et al.* (2019)
• Sea/coastal	6	Pascoe, Doshi *et al.* (2014)	22	Rhodes *et al.* (2018)
• Mix of locations	0		2	Wang *et al.* (2020)
Marginal value of a fish				
• Freshwater	0		2	Deely, Hynes *et al.* (2019)
• Sea/coastal	1	Pascoe, Doshi *et al.* (2014)	2	Lew and Larson (2011)
Value of fish to Indigenous communities	1	Zander and Straton (2010)	1	Oleson, Barnes *et al.* (2015)
Intrinsic value of having a commercial fishery	0		0	
Value of non-extractive use				
• Beach recreation/ protection/ improvement)	7	Prayaga (2017)	52	Dahal *et al.* (2019); Kipperberg *et al.* (2019); Lankia *et al.* (2019); Liu *et al.* (2019); Matthew *et al.* (2019); Parsons (2019); Rodella *et al.* (2019); Schuhmann *et al.* (2019); Shen *et al.* (2019)
• Surfing	0	0	2	Scorse *et al.* (2015)
• Wildlife viewing	1	Farr, Stoeckl *et al.* (2014)	10	Paltriguera, Ferrini *et al.* (2018)
• Scuba diving/ snorkelling	0	Kragt, Roebeling *et al.* (2009)	27	Zimmerhackel *et al.* (2018)
Marine Habitat				

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
• General	0	Kazmierczak (2001)	9	Nieminen *et al.* (2019)
• MPAs	2	Gillespie and Bennett (2011)	39	Pham *et al.* (2018); Yu, Cai *et al.* (2018)
Marine Protected Species	2	Farr, Stoeckl *et al.* (2014)	7	Anna and Saputra (2017); Lim, Jin *et al.* (2017)
Bycatch/ fishery waste	0		0	
Amenity values of marine resources	0	Anning *et al.* (2009)	14	Dahal, Grala *et al.* (2019)
Water Quality (general)	2	Rolfe and Gregg (2012)	33	Tan *et al.* (2018); Lankia, Neuvonen *et al.* (2019)
Value of water quality/habitats affected by aquaculture	0		1	Cruz-Trinidad *et al.* (2011)

Notes: The same study could have multiple entries; References in blue are more than 10 years old but are the most recent identified study

Marine Ecosystems Services Partnership (MESP) Mapper and Valuation Library

In an effort to increase knowledge with respect to global marine ecosystem services, the Marine Ecosystem Services Partnership (MESP) was proposed in 2010 (Marine Ecosystem Services Partnership 2020). The MESP is currently maintained by Duke University's Nicholas Institute of Environmental Policy Solutions. Members of the MESP are a community of marine ecosystem stakeholders, policy makers, researchers and economists. While providing an opportunity to contribute and seek information/news in relation to marine ecosystems, the MESP also has a database of valuation studies. The MESP database contains 1000+ entries of economic valuation data (Marine Ecosystem Services Partnership 2020). The data can be accessed through the MESP Mapper and Valuation Library.

Table 9. Studies identified in the MESP database 2010-2020

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
Fisher satisfaction	0		0	
Value of recreational fishing	3	Ezzy, Scarborough *et al.* (2012)	2	Gao and Hailu (2011)
Marginal value of a fish	0		0	Agnello (1989)
Value of fish to Indigenous communities	1	Zander and Straton (2010)	0	
Intrinsic value of having a commercial fishing fleet	0		0	
Value of non-extractive uses				
• Whale watching	0		0	Larson *et al.* (2002)
• Swimming	0		0	Grigalunas *et al.* (2004)
• beach recreation	1	Zhang, Wang *et al.* (2015)	14	Bann and Başak (2013); Bann and Başak (2013)
• scuba diving	0		0	Asafu-Adjaye and Tapsuwan (2008); Tapsuwan and Asafu-Adjaye (2008)
Value of habitat				

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
• MPA	1	Gillespie and Bennett (2011)	7	Castaño-Isaza *et al.* (2015)
• Marine general	0		3	Brander and McEvoy (2012)
Marine Protected Species	0		4	Wallmo and Lew (2012)
Value of bycatch/ fishery waste	0		0	
Amenity values of marine resources	0		2	Christie and Gibbons (2011); Schumann (2012)
Value of water quality (general)	1	Hatton MacDonald, Ardeshiri *et al.* (2015)	8	Becker *et al.* (2012); Kaffashi *et al.* (2012); Östberg *et al.* (2012)
Value of water quality affected by aquaculture	0		1	McDonough *et al.* (2014)

Notes: The same study could have multiple entries; References in blue are more than 10 years old but are the most recent identified study

The MESP Mapper provides an interactive way for users to access the valuation studies. By clicking on a particular coastal zone on the map, studies which are relevant to that country will appear in the table underneath the map. The studies can also be further refined or searched by using specific filters: Exclusive Economic Zones & Ecosystem Type. The table underneath the map can be customized to include the title of the study, country of origin for the study, subregion, ecosystem, valuation (USD by default) and Valuation details. When clicking on a row in the table, the study title and sample values appears on the map and also provides a link to the more detailed entry. The more detailed entry generally provides the title of the study, location of the study, type of ecosystem, sample value estimates, region, data source, publication information and additional notes (if any).

Users can also search the valuation library through the free text search function or by browsing through the full bibliographic list of studies. Access to these resources are freely provided by the MESP for public use.

Recreation Use Values Database (RUVD)

Recreation Use Values Database (RUVD) was last updated in 2016 (funded by the USDA Forest Service, Pacific Northwest Research Station and in collaboration with USGS) (Oregon State University - College of Forestry 2016) and contains 421 economic valuation studies from North America with a date range of 1958-2015 (Rosenberger, 2016). The studies are a mix of both land and water based recreational activities.

Table 10. Studies identified in the RUDV database 2010-2020

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
Fisher satisfaction			0	
Value of recreational fishing				
• Freshwater			69	McKean *et al.* (2014)
• Saltwater			1	Gornik *et al.* (2013)
Marginal value of a fish			1	Ng (2011)
Value of fish to Indigenous communities			0	

Priority	*Number of Australian records*	*Most recent Australian study/studies*	*Number of Non-Australian records*	*Most recent non-Australian study/studies*
Intrinsic value of having a commercial fishing fleet			0	
Value of non-extractive uses				
• Non-motorized boating			2	Loomis and McTernan (2014); Aanesen, Falk-Andersson *et al.* (2018)
• Motorized boating			1	Gornik, Lin *et al.* (2013)
• Beach recreation			6	Parsons *et al.* (2013)
• Sight seeing			0	Bhat *et al.* (1998)
• Swimming			0	McKean *et al.* (2005)
• Wildlife viewing			1	Gornik, Lin *et al.* (2013)
• General recreation			1	Landry *et al.* (2016)
Value of protected habitats			5	Landry, Lewis *et al.* (2016)
Value of protected species			11	Edwards *et al.* (2011)
Bycatch/ fishery discards			0	
Amenity values of marine resources			0	Lew and Larson (2005)
Value of water quality			2	Loomis and McTernan (2014)
Value of water quality affected by aquaculture			0	

Notes: The same study could have multiple entries; References in blue are more than 10 years old but are the most recent identified study

A total of 3192 estimates have been extracted from these studies. Marginal values with respect to site quality/condition are excluded from the database, however net willingness to pay and consumer surplus for recreational activities are provided. The original values from each study are reported in the database, however "estimates in per person per day units, adjusted to 2016 USD" are also reported (Rosenberger 2016).

The full database can be downloaded as an Excel workbook and is freely available to the public. The database provides the bibliographic details of the study, study location details, the recreational activity being valued, site characteristics, survey and sample characteristics, valuation method, the characteristics of the method used, benefit measure and standardized consumer surplus estimates. The studies contained in the database are either revealed preference, stated preference or a mixture of both methods. The majority of the search categories in the database are numerically coded, however the 'Recreational Use Values Database Coding Template' is provided in the first sheet of the excel workbook.

Other databases

Other databases such as the Ecosystem Services Valuation Database (ESVD) (Ecosystem Services Partnership 2020) and Envalue (NSW EPA n.d.) were considered for the review, however were excluded on the basis that these databases have not been updated in the last 10 years (i.e. between 2010 – 2020). In 2010, the Ecosystems Services Partnership developed a database for the Economics of Ecosystems and Biodiversity (TEEB) study (Ecosystem Services Partnership 2020). The 2010 ESVD TEEB database (Van der Ploeg and de Groot 2010), is currently being updated as of 2019. Once available, the new ESVD will contain over 600 studies (or 4000 value records) (Ecosystem Services Partnership 2020). The Envalue database is not totally excluded from the review, given that studies in Envalue were incorporated into the EVRI database (Dobes *et al.* 2016).

While other databases such as the Beneficial Use Values Database (Agricultural and Resource Economics - University of California n.d.) and Sportfishing Values Database (Industrial Economics Incorporated n.d.) may have been useful in this review, the links to these databases were not active during the time of the review (March 2020 – May 2020). According to Krishnan (2002), the Sportfishing Values Database contains studies from 1975-1996 – hence this would not have satisfied the publication year requirement for this audit. The ValueBase SWE database has previously been cited as another database that contains non-market valuation studies (Lantz and Slaney 2005), however this database could not be found during the time of the audit (i.e. previously cited link to the database did not work). The Review of Externalities Data (RED) database (Institute of Studies for the Integration of Systems n.d.) was an active project between 2002 and 2003 and therefore did not meet the 2010-2020 criteria. Similarly, the New Zealand Non-Market Valuation Database does not include studies past 2010 (as identified by the search filter function) (Lincoln University n.d.), hence this database was also excluded from the audit. The Greek Environmental Valuation Database was also excluded from the audit as the database was last updated in 2007 (The Laboratory of Mining & Environmental Technology (LMET) and the National Technical University of Athens (NTUA) 2007).

Gap analysis

The gap analysis was based on the Australian studies identified in the literature review above. As noted in the methods section, the gap analysis is based on the importance-performance analysis (IPA) approach. Importance of each of the types of non-market values identified by managers in the workshop was determined through a survey, and is represented by a score out of 10. We use the geometric mean value of these importance scores for the purposes of the gap analysis. For the performance measure, we use a measure of the availability of studies in each state or territory, with the most recent study in each state or territory (if any) discounted by its age.

The results of the analysis can be seen in Table 11 and Figure 18. All but three of the priorities fell into the high importance – low availability category. For the two priorities that fell into the low importance group based on the mean usage rating, these both related to aquaculture, and it is likely that the low average likelihood of use was an artefact of the small proportion of aquaculture managers compared with fisheries managers in the survey. For at least some managers – assumed linked to aquaculture – these values were identified as having a high likelihood of use. In contrast, the value of the recreational fishing experience was well represented, with studies undertaken in six of the seven states since 2010, with half of these undertaken in the last five years. For other priorities, some studies have been undertaken in the last 10 years, although these are relatively sporadic and still fall in the "in most need of further improvements" category (the bottom right quadrant of Figure 18).

Figure 18. Importance-availability analysis of non-market valuation studies in Australia by priority (numbered)

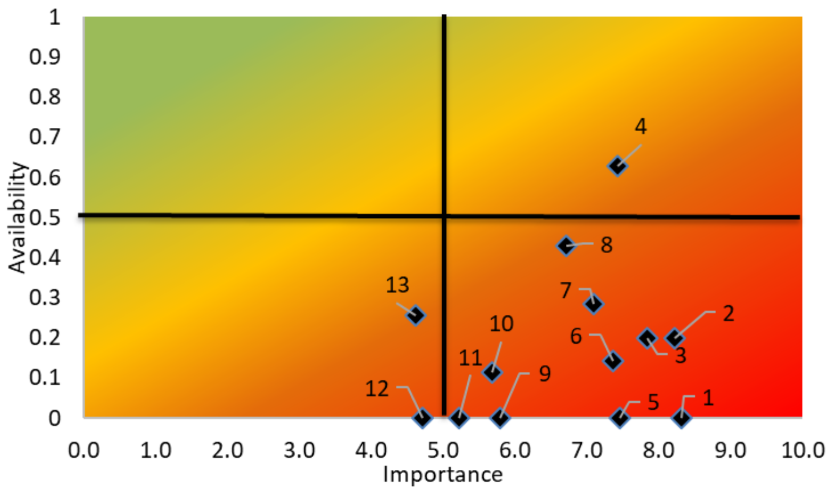


Table 11. Importance and availability of recent NMV studies in Australia (2010-2020)

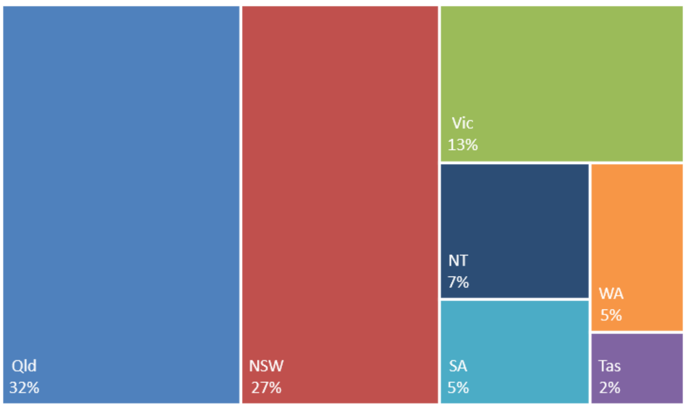
	Importance	Discounted age of most recent NMV study							Av. Score
		Qld	NSW	Vic	Tas	SA	WA	NT	
1. Commercial fisher satisfaction	8.32								0
2. An individual fish to a recreational fisher	8.22	0.8					0.6		0.200
3. Fish to Indigenous communities	7.85	0.2					0.2	1	0.200
4. Recreational fishing experience	7.43	1	1	1	0.6	0.2	0.6		0.629
5. Bycatch and discarding of commercial species	7.46								0
6. Having a commercial fishing fleet	7.37	1							0.143
7. Non-extractive uses	7.10	1	1						0.286
8. Habitats affected by fishing	6.71	1	1	1					0.429
9. Ecosystem services affected by fishing	5.80								0
10. TEP species bycatch	5.68	0.8							0.114
11. Amenity values	5.22								0
12. Access to water (aquaculture)	4.72								0
13. Changes in water quality	4.63		0.8			1			0.257

For five of the priorities, no study could be identified in the last 10 years. This includes non-market values for commercial fisher satisfaction, the highest priority identified by fisheries and aquaculture managers at the workshop. A number of qualitative measures of fisher satisfaction have been undertaken (e.g. Pascoe, Cannard *et al.* 2015; Carlin and Morison 2018), but these have not been developed to represent economic values that could potentially be used in a cost-benefit analysis. However, as noted earlier, market values such as fisher income may provide a reasonable estimate of the value of commercial fisher satisfaction, and greater value may accrue from gaining a better understanding of how management affects satisfaction in order to better design management options.

The value of bycatch and discarding were also given a relatively high priority, but no studies were identified in Australia in the last 10 years. Bycatch and discarding of both commercial and non-commercial species have been identified as a factor affecting social licence (Hobday *et al.* 2018). No Australian studies were identified focusing on the negative externality value associated with commercial bycatch. Only one study was identified that estimated values of non-commercial species (Farr, Stoeckl *et al.* 2014), although these were values to tourists and would be difficult to include in a fisheries cost-benefit analysis.

Coverage of these priorities at the State and Territory level (Figure 19) can be estimated by summing down the columns of Table 11 and dividing by the number of priorities (i.e. 13). Both Queensland (32%) and NSW (27%) have the greatest coverage, although this represents less than a third of the priorities being covered by recent studies.

Figure 19. Coverage of non-market values at the State and Territory level



Discussion

Estimate or borrow?

When done poorly, the inclusion of non-market values can undermine the scientific rigour of decisions and reporting, weakening trust in practitioners and users of non-market values and in decision-making. Conversely, excluding non-market values implicitly assumes that these impacts have a zero value, and can lead to decisions that do not maximise the net economic returns to the broader community.

Deriving non-market values to support decision making is a potentially costly and time-consuming activity. Benefit transfer provides a cost-effective means of including non-market values in decision making. However, benefit transfer is not without problems, and in some cases may lead to wrong decisions being made if the values are not valid and/or robust for the situations being considered.

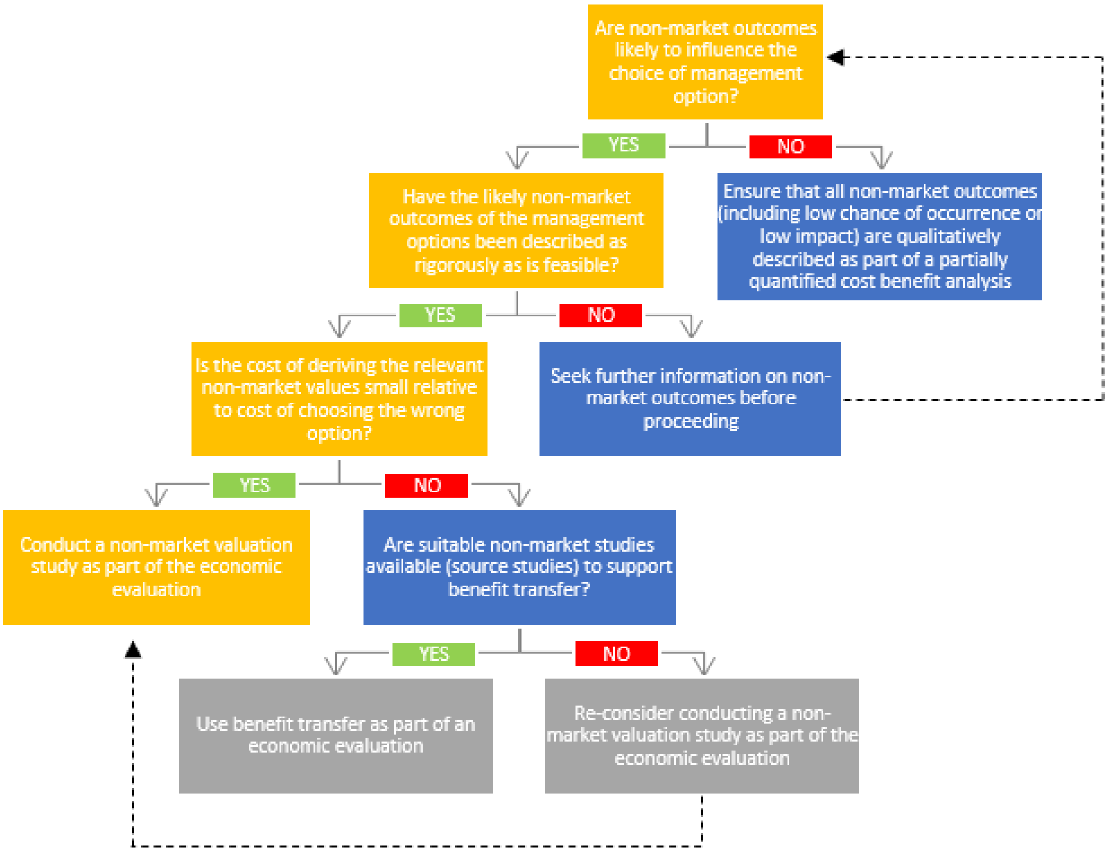
Baker and Ruting (2014) developed a flow-chart to determine when and how non-market values should be used for decision-making (Figure 20). The use of this flow chart is illustrated using the earlier example where managers of a fishery were confronted with a problem of high dolphin bycatch (Box 1). To recap, the current management strategy results in six dolphin deaths a year. There were two alternative management options to reduce this problem. Option 1 involved a series of area closures that modelling suggests would reduce the bycatch of dolphins from six to three, but would also result in the bycatch of an additional three turtles. Option 2 involved the use of spatial closures and new bycatch reduction technology which reduces bycatch of all species to zero, but adds an additional $1m a year in costs to the industry.

The first step in any decision-making framework is to ensure that all possible options have been considered. While in this hypothetical scenario we assume that all options have been considered, in other cases there may be alternative management options that do not have an external impact.

In this case, we assume that impacts on the TEP species is unavoidable and some form of non-market values associated with the bycatch of the different species are required to determine which option is best from a broader societal perspective. While Option 2 has no non-market impacts, in order to compare it to Option 1 and the current management strategy non-market values for these alternatives need to be estimated. The potential additional cost to the industry of option 2 is assumed known to be $1m, while the additional costs of the other options to the industry are zero. If the cost of deriving the non-market value for averting bycatch is substantially less than this then collecting and analysing these data should be considered. Conversely, if the cost of collecting the data are considered excessive and existing valuation studies exist that could be adapted, then benefit transfer should be considered.

There may be some circumstances where including the non-market values will not change the decision as to the optimal outcome. For example, in the case where the non-market impacts are the same for all management options. In such cases, the potential non-market impacts still need to be identified to ensure that they have been explicitly recognised. In cases where the non-market values of the impacts of the different management options are expected to influence the choice of the preferred management option, such as in the case above, estimation of these values is required. Primary data collection and analysis is always preferable to benefit transfer, and benefit transfer should always be seen as a second-best approach. However, it is not always pragmatic or feasible to calculate case-specific non-market values. For example, in some circumstances the costs of undertaking such analyses may not be warranted given the potential implications of the outcome. In Figure 19 (adapted from Baker and Ruting (2014)) the decision points to determine if non-market values should and can be estimated are presented and the conditions under which benefit transfer are appropriate are identified.

Figure 20. Flow chart for deciding if benefit transfer or a primary study is appropriate



Adapted from Baker and Ruting (2014)

Availability of appropriate values

From the gap analysis based on the literature review, there are only limited Australian estimates of values that could be potentially applied in benefit transfer.

The results of the literature review also found that where several non-market values were available (e.g. recreational fishing experiences), these values may vary substantially. This reflects not only differences in the underlying demographics of the different populations (e.g. Queensland versus Tasmanian fishers), but also in the estimation approach used and the assumptions involved. For example, some travel cost analyses assumed that travel time had an explicit cost (e.g. Rolfe and Dyack 2011) whereas in other cases travel time was assumed to have zero cost (e.g. Pascoe 2019).[6]

Meta-analysis is an approach that has been widely adopted for benefit transfer to allow for adjustments in different sample characteristics as well as methodological assumptions (Rosenberger and Loomis 2000; Bergstrom and Taylor 2006; Stapler and Johnston 2009). Meta-analysis involved the development of regression models that model the willingness to pay estimate as a function of the different sample and survey characteristics. A key feature of meta-analysis is that the transfer error can be estimated using the model, so the reliability of the estimates can be better evaluated. While examples in fisheries and aquaculture are limited, such an approach has been used in the marine environment to develop estimates of coral reef values for benefit transfer (Londoño and Johnston 2012) and (some) marine TEP species (Richardson and Loomis 2009).

[6] Further details on these assumptions are given in Appendix D.

While the audit of key non-market valuation databases also highlighted the sparsity of Australian values, it also highlighted the broader range of international studies. The potential usefulness of international studies in benefit transfer is mixed. Ready and Navrud (2006) and Shrestha and Loomis (2001) suggest that errors encountered through international benefit transfer were no larger than those from transfer study sites within the same country as the policy site. More important is the differences in sites, not necessary just geographical location that may affect their usefulness for benefit transfer (Ready and Navrud 2006). Similarly, Czajkowski *et al.* (2017) found that provided appropriate adjustments are made for differences in income levels and other demographic characteristics, then international benefit transfer can provide appropriate values. However, Ready *et al.* (2004) found that adjusting values for differences in demographic characteristics did not reduce benefit transfer errors. Other studies, based on meta-analysis, have suggested that errors introduced through international benefit transfer may be substantial, and recommend that international benefit transfer be avoided (Muthke and Holm-mueller 2004; Lindhjem and Navrud 2008).

Costs versus reliability of different valuation options

The approach adopted for incorporating non-market values into decision making involves a trade-off between robustness and cost. Estimation of non-market values requires collection of primary data as well as the specialist expertise required to derive the models. Benefit transfer, again, requires specialist expertise as well as estimates of relevant non-market values.

The trade-off between the different approaches is illustrated in Figure 21. In Figure 21, red indicates a problematic feature, green indicates a beneficial feature, and orange indicates somewhere in between. Details of what is required for each method is provided in the Appendix.

Basic unit value transfer, where values are just taken from studies with similar characteristics can be undertaken at low cost and require little expertise. However, this is likely to be the least robust approach as discussed previously. Given the size, complexity and relative disorganization of the literature on non-market valuation, there is often a divergence between transfer practices recommended as best practice and those commonly applied within policy analysis (Johnston and Rosenberger 2010). With many varying values to potentially choose from, there is also the potential for selection bias, with managers able to pick and choose the values that best support their *a priori* preference.

Modifying the values to allow for different demographic and other site-specific differences can improve the robustness of these results. This may involve the use of a simple value function transfer (i.e. based on the original model used to estimate the non-market value) or the results of a meta-analysis. This would involve greater expertise and cost (depending on the approach undertaken) than the basic approach. From the gap analysis, there are also insufficient Australian examples to develop a robust meta-analysis for many values, requiring reliance on international studies that may add additional complications in developing reliable estimates (Lindhjem and Navrud 2008; Czajkowski, Ahtiainen *et al.* 2017).

In terms of non-market valuation methods, cost and complexity can vary substantially depending on the method adopted. A simple travel cost model can produce results that are relatively robust, but are limited to values on a per trip or per person basis for access to a particular area or activity. While both the simple travel cost analysis and tailored benefit transfer approach are both labelled as moderate in terms of robustness, the site-specific travel cost analysis would be more robust than the benefit transfer approach.

Figure 21. Trade-offs between cost and robustness

	Robustness	Expertise required	Cost	Timeliness
Benefit transfer				
• Basic benefit transfer	Low	Low	Low	High
• Tailored benefit transfer	Moderate	Moderate	Moderate	High
Non-market valuation methods				
• Travel cost analysis (basic)	Moderate	Moderate	High	Moderate
• Travel cost analysis (hedonic or contingent)	High	High	High	Moderate
• Choice experiment	High	High	High	Moderate
• Deliberative approaches	Low	High	Low	High

Derived from Hester *et al.* (2019)

The usefulness of travel cost analyses can be enhanced through the use of hedonic or contingent behaviour travel cost models. These require greater expertise in terms of survey design and analysis than the basic travel cost approach, but can also provide greater information on the values of particular features of the trip (e.g. species caught, activity undertaken etc). Choice experiments also can provide robust value estimates, although the expertise required to design the surveys and analysis is higher than some of the simpler approaches.

The cost of data collection for the non-market valuation studies can vary depending on the attribute being examined. In some cases, where the potential beneficiaries are limited (e.g. visitation to a particular site), onsite surveys may be required. These are both expensive and time consuming. In contrast, for more "general" values (e.g. cost of bycatch), surveys may be distributed more broadly online which may lower the cost of the data collection and also reduce the time required to collect an appropriate sample.

Deliberative approaches for deriving non-market values have been advocated in some instances as an alternative to stated or revealed preference methods. These involve small groups deciding what an appropriate non-market value might be for a particular environmental asset or ecosystem service. Approaches range from the use of citizen juries, where a small group is presented with evidence provided by experts to increase their understanding (Kenyon *et al.* 2001; Robinson *et al.* 2008) to deliberation between citizen-stakeholders in more of a workshop environment (Wilson and Howarth 2002). In some cases, these can be combined with other stated preference approaches, with the stakeholder group using the derived values as a starting point for their deliberation (Macmillan *et al.* 2002; Álvarez-Farizo and Hanley 2006).

As the process of deliberation requires citizens to go beyond their own private self-interest, it is believed that the outcome will have greater social equity and political legitimacy (Wilson and Howarth 2002). However, the approach has many critics. Deliberative non-market valuation has been criticised for the lack of a theoretical base for the interpretation of the monetary values produced (Bunse *et al.* 2015). Others have questioned the lack of inclusiveness in the valuation process (Vargas *et al.* 2017), with some groups excluded from the process. In other cases, valuation outcomes are potentially affected by issues relating to group power dynamics (Liski *et al.* 2019). Others have suggested that there is a lack of clarity in the process, such that different groups given different institutional settings produce different value estimates (Spash 2007). In regard to this last criticism, however, attempts have been made to improve the robustness of the approach through developing a set of uniform guidelines based on best practice (Schaafsma *et al.* 2018). The extent to which such guidelines have been adopted is not known. The Productivity Commission concluded that deliberative valuation performs poorly relative to other non-market valuation approaches in providing value estimates that are reflective of community preferences and can be included in a cost–benefit analysis (Baker and Ruting 2014).

The collection of fisheries data in general is expensive. Compared to collection of other data and analyses (e.g. fishery independent survey, stock assessments), the costs of collecting non-market values may not be excessive, particularly if values can be applied across multiple decisions and to several fisheries or aquaculture industries. For example, estimates of the value of TEP species or other bycatch could potentially be applied to a number of fisheries.

Priorities for non-market value data collection and analyses

Several studies have noted a sparsity of non-market values relating to marine and coastal ecosystem services and environmental assets. In 2007, Pendleton, Atiyah *et al.* (2007) noted that there were insufficient non-market valuation studies in the coastal and marine environment to adequately support management. Beaumont *et al.* (2008) identified the sparsity of estimates available for coastal management in the UK, and subsequently estimated values for eight (out of thirteen) key ecosystem services identified as relevant to marine and coastal management, as their value had not previously been estimated. In a comprehensive international review of existing studies, Katsanevakis *et al.* (2011) found that, other than relating to recreation, few non-market valuation studies existed relating to the marine environment. In the US, Raheem *et al.* (2012) similarly found few non-market values relating to marine ecosystem services. Where studies were identified, the reported values relating to coastal ecosystems varied widely. Lipton *et al.* (2014), however, noted that the number of non-market valuation studies in the coastal and marine environment was on an upwards trajectory, corresponding to an increasing demand for their use in management.

A perception regarding the sparsity of valuation studies is that they are difficult to estimate as individuals are less familiar with the coastal and marine environments, and hence deriving valid values is more difficult than in a terrestrial environment. However, while substantially fewer studies of the marine environment exist than for the terrestrial environment, the database audit has demonstrated that non-market valuation studies have been successfully undertaken somewhere and at sometime with regard to all coastal and marine ecosystem services and environmental assets. This suggests that it is not the perceived additional complexity that is the main constraint to these studies, but the demand for their use (which subsequently leads to funding for their estimation).

Primary data collection and analyses

The gap analysis identified that recent values for most of the values of potential use to fisheries and aquaculture management were unavailable. Studies that had been undertaken were largely ad hoc in nature, addressing a particular issue of relevance to a particular case study. The studies have also been developed independently, and often use different assumptions around, for example, key issues such as treatment of multi-purpose trips, multi day trips, other related costs such as food and the cost of travel time.

Pascoe (2019) and Pascoe, Doshi *et al.* (2019) demonstrated that a wide range of non-market values can be estimated cost-effectively at a State-wide level using online surveys. For many values, such as habitat, bycatch and protected species values, State-wide or even National surveys may be appropriate.

In some cases, primary data may already exist but be underutilised. For recreational fisheries, there is considerable information already being collected in most states, particularly boat ramp surveys as well as state-wide expenditure surveys. A national survey is also currently underway (www.nationalrecsurvey.com.au). While not collected with the intention of undertaking travel cost analyses, the data collected often contains information that can be used in a travel cost model.

Raguragavan, Hailu *et al.* (2013) used such data in their estimation of the non-market value of recreational fishing and also the marginal value of key species in Western Australia. Given the importance of these values to managers, further integration of appropriate economic questions into these generally regular surveys may also be feasible. These surveys can also be used as vehicles for other valuation exercises.

Of foremost importance to fisheries and aquaculture managers was commercial and recreational fisher satisfaction. These have largely been assessed on a qualitative basis. Such analyses may be useful in modelling how different management options may change satisfaction levels, but precludes them from direct inclusion into cost-benefit analyses.

Intuitively, incorporating changes in satisfaction into a cost-benefit analysis framework has appeal. A management option that improves economic and sustainability outcomes but reduces the social benefits associated with fishing is not as desirable than another that achieves the same economic and sustainability outcomes without the loss of social benefits to fishers.

Internationally, there has been growing interest in measuring satisfaction in monetary terms, particularly in the field of health economics where improvements in health outcomes are often measured as quality adjusted life years (QALYs). QALYs are utility-based measures that are based on outcomes over a number of health-related factors, each of which is based on a subjective assessment by the patients. They are used in a cost-utility framework to compare different treatments, where the cost per QALY is used to compare outcomes against the costs of the treatment. Critical monetary values of QALYs to determine where a treatment is “acceptable” or not from a resource rationing perspective are usually administratively set.

Attempts to improve the estimates of the value of QALYs by linking them to willingness to pay have been mixed. Pinto-Prades *et al.* (2009) used a choice-experiment approach based on the different attributes underlying the estimation of a QALY and found substantial variation in the estimation of a monetary value for a QALY (MVQ), and that this decreases with the magnitude of any health gain (so is not invariant to the starting point). Similarly, Shiroiwa *et al.* (2013) used a choice experiment and found the MVQ was higher for worse health states than for better health states. In contrast, Robinson *et al.* (2013) used contingent valuation to estimate the value of each QALY component separately, from which an overall value could be determined based on the weights associated with each component. This resulted in more consistent values over the range of QALYs considered.

Given the subjective and qualitative basis underlying the measurement of QALYs, then suggesting a monetary value for fisher satisfaction measures is not unreasonable, particularly where these measures are compared against other economic outcomes in decision making. To some extent, this has been done in other areas. Huang *et al.* (2018) modelled life satisfaction as a function of income, health (measured in terms of QALYs) and a range of other factors. The model was used to estimate the value of QALYs based on the trade-off between income and health keeping satisfaction levels constant (i.e. the marginal rate of transformation). The study used data from the Household, Income and Labour Dynamics in Australia (HILDA) survey. While the focus of the study by Huang, Frijters *et al.* (2018), the approach could also be adopted to consider the marginal value of a unit of life satisfaction as a function of income. Linking fisher satisfaction to life satisfaction through additional analyses could result in an estimate of the monetary value of fisher satisfaction.

So, while it may be technically feasible to derive monetary values for fisher satisfaction, the question still remains as to whether this is appropriate. As noted in the earlier parts of the report, many aspects of fisher satisfaction are included already in monetary (e.g. income measures) and other non-market values (e.g. consumer surplus). However, satisfaction scores are likely to remain an ongoing indicator of management performance. Despite this, three has been little attention in fisheries as to how best to measure satisfaction. While this goes beyond the scope of this study, experiences in other sectors (e.g. tourism and other service sectors in particular) have demonstrated that how satisfaction is measured is crucial to improving service provision (Yuksel and Yuksel 2001; Yuksel and Yuksel 2001). Job satisfaction can be influenced by a range of factors independent of management (e.g. factors affecting general life satisfaction (Perrewe *et al.* 1999)) and hence a single satisfaction score may be misleading. Stradling *et al.* (2007) suggests that satisfaction measures should be decomposed into a number of different factors, and that an aggregate satisfaction measure derived using a set of importance weights associated with each. This allows both an overall consistent measure to be derived as well as providing information as to which factors are associated with dissatisfaction (similar to the development of the standard QALY measure in health), and hence provides guidance as to how managers should respond (if needed).

Spagnoli et al. (2012) suggests that gaining an understanding of the determinants of satisfaction is important to try and minimize any short-term impacts from management change. Other studies have suggested that improved communication and increased stakeholder participation in the change process can minimize reductions in job satisfaction during the change process (Kim 2002; Bull and Brown 2012).

In this regard, a better understanding of factors affecting satisfaction with fishing would be of benefit to managers as a separate area of research. While it may be feasible to develop non-market values of satisfaction change, minimizing short run impacts during any management change is likely to be a better option.

Development of meta-analysis models

A number of meta-analyses were identified that could potentially assist in benefit transfer for key non-market values, including values of key habitats (Johnston *et al.* 2005; Brander *et al.* 2006; Londoño and Johnston 2012), TEP species (Richardson and Loomis 2009), recreational fishing (Sturtevant, Johnson *et al.* 1995) and the value of recreational fish (Johnston, Ranson *et al.* 2006). A common feature of these studies, however, is that most are a decade old or older, and most include few Australian studies that would increase their relevance for local fisheries and aquaculture management.

From the literature review and audit of the key databases, there are relatively few recent Australian studies for most non-market values of interest, but a considerable number of international studies. It is recognised that there may be the potential issues with the use of international values in benefit transfer. Development of meta-analysis models that incorporate available Australian and more recent international valuation studies may be of use in decision making for managers, but as stated previously primary data collection and analysis is always preferable to benefit transfer.

What alternatives are there if non-market values are not acceptable?

A critical constraint affecting the use of non-market values is their acceptability by the key stakeholders who will be affected by their use. In some cases, individuals may hold ethical objections to valuing the environment in this way, as they argue that the environment has 'intrinsic value' that is unrelated to human preferences (Baker and Ruting 2014). Lack of understanding about the valuation approaches and their application may also make some managers and policy makers reluctant to incorporate these values in decision making (Rogers *et al.* 2015). In other cases, lack of trust in the values result in decision makers preferring to use other types of information (Marre, Thebaud *et al.* 2015; Marre, Thébaud *et al.* 2016).

Increasing emphasis on stakeholder participation in decision making may also reduce the acceptability of the use of non-market values, especially if this use had an impact on the outcomes that key stakeholders see as detrimental. For example, including the cost of bycatch into harvest limit estimates may result in lower total allowable catches (e.g. Pascoe, Hutton *et al.* 2018). In this case, it is in the fishers' interests not to use these values, or to discredit their estimation. This approach may be counterproductive. For example, when all costs and benefits are correctly accounted for (inclusive of non-market values) it would be possible to determine the switch point at which reducing the bycatch of

TEPs will be suboptimal. In the absence of this information, regulation will be imposed that may result in this suboptimal outcome.

Other, more qualitative, approaches have been applied to aid fisheries decision making in such circumstances. Multi-criteria decision-making approaches collect information on preferences and expected outcomes, and develop relative performance measures of different management options based on this information. Some of these approaches are described in the Appendix D. These approaches are not necessarily less expensive than non-market valuation approaches as they also rely on survey-based information as well as subsequent analysis. They also do not help determine if a given option has a positive or negative net benefit; just which of a set of options performs the best over a range of criteria.

Conclusion

The key project objectives were:

1. To support robust and defensible evidence-based decision-making in fisheries and aquaculture decision making that is understood and supported by key fisheries and aquaculture managers;
2. To provide managers with an understanding of the resources available to account for non-market values in fisheries and aquaculture decision making; and
3. To identify key research gaps and make recommendations related to the need for further empirical non-market valuation studies.

The project objectives were fully met.

Previous studies of managers' attitudes to the use of non-market values in Australia (Marre, Thebaud *et al.* 2015; Rogers, Kragt *et al.* 2015) suggested that most managers were interested in using non-market values, but their lack of availability and understanding of their use were major barriers. Rogers, Kragt *et al.* (2015) suggested that increased education of managers and policy makers is the key to increased acceptability of the use of non-market values in decision making. In our study, we found that most managers surveyed claimed they were familiar with the key concepts. This statement should be taken in context. A key theme that emerged during the interviews conducted as part of this study was that there was no systematic formal economics training and that, consequently, understanding of what constituted economic and social data amongst fisheries and aquaculture managers varied greatly. A key output of this project "*Using non-market values in cost-benefit analysis in Australian fisheries and aquaculture*" (Appendix E) seeks to address this issue directly. However, the availability and the cost of obtaining the information were seen by fisheries and aquaculture managers as a key constraint to including non-market values in decision making. Our study confirmed the lack of recent and relevant non-market valuations studies in Australia.

The study identified thirteen types of non-market values that fisheries and aquaculture managers identified as potentially important to decision making. Of these, the top four involved values related to users of the fisheries resources, including fisher satisfaction, values to Indigenous fishers, and the value of fish and the experience to recreational fishers. The next four involved impacts of fishing on others, including habitats, species, local communities and other users of the marine environment.

The review of the available studies indicated that few empirical estimates for most of these values currently exist. The value of recreational fishing experience was the best represented, although only a few recent studies were available. At the State and Territory level, Queensland and NSW were the best represented in terms of available non-market data, although not all values were available nor recent.

For some of these values, there is the opportunity to use existing data to develop estimates of non-market values. In particular, recreational fishing values (experience and value per fish) could potentially be estimated from existing boat ramp and/or expenditure surveys which are routinely collected by the States and Territories. Ideally, more primary studies would also be undertaken for other key values identified in the study. Meta-analyses could also be developed for some of the other key priorities. While Australian studies are limited, there are substantial international studies available that may be able to inform values for Australian fisheries and aquaculture management.

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Recommendations

While most fisheries and aquaculture managers stated that they were familiar with the key concepts underlying the use of economic data in general, there was still some confusion as to what economic data was, and also how non-market values could be used in a decision-making framework.

- Increased education around (i) the types and uses of economics data in general and (ii) use of non-market valuation in decision making is still required for many fisheries and aquaculture managers. The *"Using non-market values in cost-benefit analysis in Australian fisheries an aquaculture*" provide one example of an educational resource. However, an education program could be developed based on developing a range of flexible resources (e.g. short explanatory videos, podcasts etc.)

The project identified that relatively few recent estimates of non-market values relevant to Australian fisheries and aquaculture exist. Given this, a number of research priorities were identified:

(i) Many jurisdictions currently collect recreational fisheries data that could potentially be used to derive non-market values (e.g. boat ramp surveys, expenditure surveys etc). Given the importance of the value of recreational fishing experience and a recreational fish to mangers, the potential use of existing recreational fishing survey data should be examined to determine the extent to which it can be used to derive non-market values. This could be undertaken in one State in the first instance as a case study, or several States to compare and contrast the usefulness of the existing data and identify what additional data may be required to estimate non-market values. This is likely to be a relatively low-cost exercise (as the data have been collected) with a high return to managers.

(ii) Only very limited information is currently available on the value of externalities generated by commercial fishing such as bycatch in general (i.e. discarded fish species bycatch) and of TEPs, and habitat damage. Primary studies need to be conducted to estimate these values. Potentially these values can be estimated for a wide variety of key species and habitats at a national level, which would be applicable to a wide variety of fisheries.

(iii) Given the importance placed on fisher satisfaction by managers, gaining a greater understanding of how satisfaction levels are affected by management would be of benefit (even if not derived as a monetary value). Valuing satisfaction in monetary terms has not been undertaken in fisheries in Australia or elsewhere, but has been considered in other areas, particularly in health. Deriving a standardised method for assessing satisfaction and an equivalent monetary value would appear to be of highest priority to fisheries and aquaculture managers. However, while such a valuation could theoretically be undertaken, for example through a choice experiment, the dynamic nature of satisfaction changes in response to management change will make the use of any derived values difficult. Deriving a standardized method for assessing satisfaction, and understanding what drives these measures, rather than an equivalent monetary value may be of greater benefit to fisheries and aquaculture managers.

(iv) The application of meta-analysis to capture other key values should also be considered. In particular, value of changes in water quality will be of benefit to aquaculture managers.

Extension and Adoption

A key objective of the extension and adoption plan was to ensure that Australian fisheries and aquaculture managers are familiar with:

- the concept of non-market values;
- what type of values are appropriate for decision-making; and
- the range of values that are currently available (and how they could potentially be used).

The pathway adopted to improving fisheries and aquaculture managers understanding and trust of non-market values and improved transparency, consistency and reliability in the use of non-market values in fisheries and aquaculture in their decision-making has been addressed through:

(i) a workshop attended by fisheries and aquaculture managers and policy makers from all fisheries jurisdictions. The workshop sought to identify what non-market values they saw as important to aid in management in capturing the non-market impacts across three areas, commercial, recreational and aquaculture activities, on the wider ecosystem and community to support a well-managed ecosystem and the realisation of broader societal benefits, reflecting the adoption of the principles of Ecologically Sustainable Development (ESD) in fisheries and aquaculture management

(ii) the development of *"Using non-market values in cost-benefit analysis in Australian fisheries and aquaculture"* (Appendix E). These have been developed as a standalone output and will be available from the FRDC website.

This report provides fisheries and aquaculture managers and policy-makers an overview of the available non-market value studies in Australia; and is a key resource to potentially support the use of non-market valuation in fisheries and aquaculture decision making through benefit transfer.

Finally, this report provides an evidence base to assist the FRDC and the sub group Human Dimension Research with respect to the gaps in non-market values. This evidence can be used to direct research funding to support primary studies of non-market values estimation at a national level (e.g. bycatch in general and of TEPs, habitat damage, and impacts on other resource users) as indicated in the recommendations.

Project materials developed

Conference presentations

Coglan, L., Pascoe, S., Scheufele G. and Pickens, A. (2020). The barriers to fisheries managers in using key economic values in decision making: the case of the missing nonmarket value. Paper presented at Behavioural Economics, Society and Technology: Human Behaviour & Decision Making 2020 Conference (13-14 Feb), QUT, QLD.

Journal articles

Coglan, L., Pascoe, S., Scheufele G. (in submission). Non-market values to support decision making in Australian fisheries and aquaculture: an audit and gap analysis. Submitted to *Sustainability*.

Other materials

Coglan, L., Pascoe, S., Scheufele G., Paredes, S. and Pickens, A. (2020). *Using non-market values in cost-benefit analysis in Australian fisheries and aquaculture*.

Appendix A. Telephone interview questions and participation sheet

Participant information sheet

PARTICIPANT INFORMATION FOR QUT RESEARCH PROJECT – Telephone interview –
Non-market values to inform decision-making and reporting in fisheries and aquaculture – an audit and gap analysis **QUT Ethics Approval Number 1900000690**

RESEARCH TEAM

Principal Researcher:	Associate Prof Louisa Coglan **School of Economics & Finance, Queensland University of Technology (QUT)**
Co-researcher:	Dr Sean Pascoe **Commonwealth Scientific and Industrial Research Organisation (CSIRO)**
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Why is this study being conducted?

This project was funded as a priority need by Fisheries Research and Development Corporation (FRDC) to:

(i) enhance fisheries and aquaculture managers' understanding of the use of non-market values in decision-making, and

(ii) determine the key variables that managers require to capture the broader societal perspective within decision-making across all Commonwealth and State commercial (inclusive of wild caught and farmed) and recreational fisheries.

You are invited to participate in this project because you have a role as a fisheries or aquaculture manager or policy analyst or policy maker for a State or the Commonwealth.

Your participation in this this telephone interview will assist in informing a program of research to determine what non-market values are required to support the shift in focus within natural resources management from a purely commercial perspective to a broader social perspective, reflecting the adoption of the Ecologically Sustainable Development (ESD) framework for fisheries and aquaculture management.

What does participation involve?

Participation will involve a telephone interview by a member of the research team.

In this telephone interview, you will be asked about your understanding of the current use of non-market values in fisheries and aquaculture management decision-making. We will also ask you to provide examples of the type of non-market values you may have used, how they have informed the decision-making process and the problems you have experienced with using non-market values in fisheries and aquaculture management decision-making. Examples of the type of question you will be asked include:

- providing brief details about your professional and educational background; and
- your experience using non-market values to inform fisheries and aquaculture management.

The interview will take approximately 20 minutes of your time. The researcher will ask you a question and record your answer electronically on the questionnaire.

Your participation in this research project is voluntary.

If you agree to participate, you do not have to complete any question(s) you are uncomfortable answering.

If you do agree to participate, you can withdraw from the research project without comment or penalty. You can withdraw anytime during the interview.

If you withdraw within THREE DAYS after your interview, at your request, any information already obtained that can be linked to you, will be destroyed.

Your decision to participate or not participate will in no way impact upon your current or future relationship with QUT, CSIRO or FRDC who is funding this project.

You can request to review the record of your responses after the interview.

You will have the opportunity to indicate to us if you are agreeable to us contacting you for any follow up information.

What are the possible benefits for me if I take part?

A key expected benefit of this research is improved fisheries and aquaculture management practices at both Commonwealth and State level that better align to achieving a well-managed ecosystem and the realisation of broader societal benefits.

Your answers will be used to inform the development of resources targeted at enhancing fisheries and aquaculture managers' understanding of the use of non-market values in decision-making for the broader community involved in fisheries and aquaculture management decision-making at both Commonwealth and State level (inclusive of policy analysts).

What are the possible risks for me if I take part?

There are no risks beyond normal day-to-day living associated with your participation in this project.

Privacy and confidentiality

Only the direct project team, identified above, will have access to the collected raw data, which will include your personal data.

Every effort will be made to ensure that the personal data you provide cannot be traced back to you in reports, publications and other forms of presentation. We will not use your name or make reference to the specific details of examples you provide in the presentation of results. For example, if a respondent stated *"I have used non-market values in the management of the Northern Prawn Fishery"*, this will be generalised in the presentation of results to *"Examples were given of non-market values being used to inform decision-making in a highly valued commercial fishery"*.

This research project is partly funded by FRDC Research Program and they will have access to data in de-identifiable form obtained during this project. A report with main results will also be provided to the FRDC. Please note that non-identifiable data collected in this project may be used as comparative data in future projects.

Any data collected as part of this research project will be stored securely as per QUT's Management of research data policy. Data will be stored for a minimum of 5 years, and can be disclosed if it is to protect you or others from harm, if specifically required by law, or if a regulatory or monitoring body such as the ethics committee requests it.

Consent to participate

Responding to the invitation email and agreeing to undertake the telephone survey will be taken as an indication of your consent to participate in this research project.

Questions / further information about the research project

If you have any questions or require further information please contact one of the listed researchers:

Louisa Coglan l.coglan@qut.edu.au 07 3138 5135

What if I have a concern or complaints regarding the conduct of the research project

QUT is committed to research integrity and the ethical conduct of research projects. If you wish to discuss the study with someone not directly involved, particularly in relation to matters concerning policies, information or complaints about the conduct of the study or your rights as a participant, you may contact the QUT Research Ethics Advisory Team on +61 7 3138 5123 or email humanethics@qut.edu.au.

Citation

2018-068 Non-market values to inform decision-making and reporting in fisheries and aquaculture – an audit and gap analysis is supported by funding from the FRDC on behalf of the Australian Government.

THANK YOU FOR HELPING WITH THIS RESEARCH PROJECT.

PLEASE KEEP THIS SHEET FOR YOUR INFORMATION.

Telephone Questionnaire

Start of Block: WELCOME TO THE PROJECT

Q1 Interviewer to confirm the following before commencing the survey

- ▢ Thank the participant for agreeing to undertake the survey (1)
- ▢ Confirm that the participant has read the participation sheet (2)
- ▢ Check that they have no further questions re the participation sheet (3)
- ▢ Reiterate that the interview should take approximately 20 mins (4)
- ▢ Confirm if they are happy to continue now or wish to reschedule (5)
- ▢ Remind them that you will be noting down their answers (6)
- ▢ Remind the participant that they can choose not to answer a question (no justification is required) (7)

Q2 Record any comments/additional information shared by the respondent below

End of Block: WELCOME TO THE PROJECT

Q3 Are you involved in fisheries or aquaculture management decision-making?

- ◯ Fisheries (1)
- ◯ Aquaculture (2)
- ◯ Both (indicate % estimate) (3) ____

Q4 Inclusive of your current position, how long have you been involved in fisheries or aquaculture management decision-making?

- ▢ Less than 1 year (1)
- ▢ 2-3 years (2)
- ▢ 4-5 years (3)
- ▢ more than 5 years (record number of years) (4) ____

Q5 Record any comments/additional information shared by the respondent below.

Q6 Who do you currently work for?

- ▢ Government and associated agencies (policy & management) (1)
- ▢ Government and associated agencies (research) (2)
- ▢ Other (3) ____

Q7 Record any comments/additional information shared by the respondent below

Q8 Which of the following marine jurisdictions does your work relate to?

- ◯ Commonwealth (1)
- ◯ State (2)
- ◯ Both (indicate estimate of % (6) ____

Q9 Which state

- QLD (1)
- NSW (2)
- NT (3)
- VIC (4)
- TAS (5)
- WA (6)
- SA (7)

Q10 In your current role, what would you identify as your main areas of contribution to fisheries management?

(Rate each out of 100)	0 10 20 30 40 50 60 70 80 90 100
Informative (collating information and delivering it to others) (1)	
Consultative (providing advice and recommendations to others) (2)	
Contributive (contributing to the final decision and/or management plan) (3)	
Decisive (deciding whether or not a decision is implemented) (4)	
Other (5)	

Q11 Record any comments/additional information shared by the respondent below

Q12 How long have you been in your current position?

- less than 1 year (1)
- 2-3 years (2)
- 3-4 years (3)
- 5 years or more (4) ____

Q13 Record any comments/additional information shared by the respondent below

End of Block: YOUR EXPERIENCE IN FISHERIES AND AQUACULTURE MANAGEMENT DECISION-MAKING

Start of Block: YOUR PREVIOUS EXPERIENCE IN FISHERIES MANAGEMENT

Q14 In the last FIVE YEARS, have you been involved in fisheries management decision-making prior to your current position?

○ Yes (1)

○ No (2)

Skip To: End of Block If In the last FIVE YEARS, have you been involved in fisheries management decision-making prior to y... = No

Q15 Record any comments/additional information shared by the respondent below

____

Q16 Who did you work for in your LAST substantive fisheries management decision-making role?

▢ Government and associated agencies (policy & management) (1)

▢ Government and associated agencies (research) (2)

▢ Other (3) ____

Q17 Record any comments/additional information shared by the respondent below

____

Q18 In which country?

▢ Australia (1)

▢ Other (2) ____

Skip To: Q19 If In which country? = Australia

Q19 Which of the following marine jurisdictions does your work relate to?

○ Commonwealth (1)

○ State (2)

Display This Question:
If Which of the following marine jurisdictions does your work relate to? = State

Q20 Which state?

- ▢ QLD (1)
- ▢ NSW (2)
- ▢ NT (3)
- ▢ VIC (4)
- ▢ TAS (5)
- ▢ WA (6)
- ▢ SA (7)

Q21 Which of the following best describes your role in that position?

- ☐ Fisheries manager (1)
- ☐ Fisheries policy analyst (2)
- ☐ Policy-maker (3)
- ☐ Other (4) ____

Q22 Record any comments/additional information shared by the respondent below

Q23 In your last substantive fisheries management role, what would you identify as your main areas of contribution to fisheries management? (Rate each out of 100)

	0 10 20 30 40 50 60 70 80 90 100
Informative (collating information and delivering it to others) (1)	
Consultative (providing advice and recommendations to others) (2)	
Contributive (contributing to the final decision and/or management plan) (3)	
Decisive (deciding whether or not a decision is implemented) (4)	
Other (5)	

Q24 Record any comments/additional information shared by the respondent below

Q25 How long did you work in this position?

- ○ less than 1 year (1)
- ○ 2-3 years (2)
- ○ 3-4 years (3)
- ○ 5 years or more (4) ____

Q26 Record any comments/additional information shared by the respondent below

End of Block: YOUR PREVIOUS EXPERIENCE IN FISHERIES MANAGEMENT

Start of Block: OVER VIEW OF MAIN AREA OF CONTRIBUTION IN FISHERIES MANAGEMENT

Q27 Over your ENTIRE fisheries management career, what would you identify as your main areas of contribution to fisheries management decision-making? (Rate each out of 100)

	0 10 20 30 40 50 60 70 80 90 100
Informative (collating information and delivering it to others) (1)	
Consultative (providing advice and recommendations to others) (2)	
Contributive (contributing to the final decision and/or management plan) (3)	
Decisive (deciding whether or not a decision is implemented) (4)	
Other (5)	

Q28 Record any comments/additional information shared by the respondent below

End of Block: OVER VIEW OF MAIN AREA OF CONTRIBUTION IN FISHERIES MANAGEMENT

Start of Block: GENERAL CHARACTERISTIC QUESTIONS

Q29 Have you ever attended any economic training including in your current or previous roles?

- Yes (provide details) (1)
- No (2)

Q30 Record any comments/additional information shared by the respondent below

The following information will assist in identifying the representation of society in a policy decision-making context.

Q32 What is your gender?

- Man (1)
- Woman (2)
- Other (3)
- I do not wish to answer (4)

End of Block: GENERAL CHARACTERISTIC QUESTIONS

Start of Block: YOUR USE OF ECONOMIC AND SOCIAL DATA

Q33 In your current role, in your jurisdiction, is there any legislative requirement to consider social and economic considerations in fisheries management decision-making? For example are you required to do Social Impact Assessment, Economic Impact Assessment, Benefit-Cost Analysis, Resource allocation; etc.

◯ Yes (ask respondent to provide details and record below) (1)

◯ No (2)

Q34 Record any comments/additional information shared by the respondent below

Q35 In your current role, do you use economic and social data either qualitative or quantitative?

◯ Yes, in some form (1)

◯ No, none at all (2)

Skip To: Q39 If In your current role, do you use economic and social data either qualitative or quantitative? = Yes, in some form

Q36 Record any comments/additional information shared by the respondent below

Q37 If **NO** in the following management contexts, what type of economic and social data would you require? (Where possible, please give examples)

◯ Management of commercial fisheries (e.g. market prices costs and earnings, employment) (1) ________________

◯ Management of recreational fisheries (e.g. value of recreational fishing) (2) ________________

◯ Management of aquaculture (e.g. market prices, costs and earnings, employment) (3) ________________

Q38 Record any comments/additional information shared by the respondent below

Skip To: Q43 If Record any comments/additional information shared by the respondent below Is Displayed

Q39 If **YES** in the following management contexts, what type of economic and social data do you typically use? (Where possible, please give examples)

◯ Management of commercial fisheries (e.g. market prices costs and earnings, employment) (1) ________________

◯ Management of recreational fisheries (e.g. value of recreational fishing) (2) ________________

○ Management of aquaculture (e.g. market prices, costs and earnings, employment) (3) ____

Q40 Record any comments/additional information shared by the respondent below

Q41 How would you rate the quality of the economic and social data you have access to?

Management of commercial fisheries (1)	☆	☆	☆	☆	☆
Management of recreational fisheries (2)	☆	☆	☆	☆	☆
Management of aquaculture (3)	☆	☆	☆	☆	☆
Management of customary fisheries (4)	☆	☆	☆	☆	☆

Q42 Record any comments/additional information shared by the respondent below

Q43 In your current jurisdiction, what are the decision-making process for allocating catch between commercial, recreational and customary fishers?

▢ Ad hoc/no formal process (1)

▢ Based on relative current catches of each of the sectors (2)

▢ Based on relative values of the resource to each of the sectors (3)

▢ Exogenous to the management of the fishery (e.g. political expediency) (4)

▢ Mix of all of the above (5)

Q44 Record any comments/additional information shared by the respondent below

Q45 Non-market values (NMV) capture the broader non-monetary costs and benefits to all sectors of the community. In a fisheries context, NMV gives an economic value (in dollars) to such impacts as, for example, value of habit damage by trawling or the value derived from participation in recreational fishing.

Q46 Have you been involved in a decision-making process where NMV would have been of use (provide examples)?

○ Yes, often (1)

○ Yes, only a few times (2)

○ Not sure (3)

○ Never (4)

Skip To: Q52 If Have you been involved in a decision-making process where NMV would have been of use (provide exa... = Never
Skip To: Q52 If Have you been involved in a decision-making process where NMV would have been of use (provide exa... = Not sure

Q47 Record any comments/additional information shared by the respondent below

Q48 When it has been appropriate to use NMV, and data was available, was it used?

○ Yes. Record example below explaining context. (1)

○ No (2)

Skip To: Q52 If When it has been appropriate to use NMV, and data was available, was it used? = Yes. Record example below explaining context.

Q49 Record any comments/additional information shared by the respondent below

Q50 If NMV data was available, why was it not used? (Prompts are below. Record alternative reasons under "other").

▢ Data was from a different fishery/another country etc (4)

▢ Did not know how to use it in a meaningful-way (5)

▢ Other (6) ____

Q51 Record any comments/additional information shared by the respondent below

Q52 In your experience, what would you identify as the key barriers to using NMV?

▢ I do not understand how to incorporate NVM data into decision-making in a meaningful way (1)

▢ There is no reliable or robust sources for NVM data (2)

▢ It costs too much to derive case specific estimates of NMV (3)

▢ Other (4) ____

Q53 Record any comments/additional information shared by the respondent below

Q54 Are you agreeable to us contacting you for further participation in this project (including attendance at workshop, further surveys)

- Definitely yes (1)
- Probably yes (2)
- Might or might not (3)
- Probably not (4)
- Definitely not (5)

Q55 Record any comments/additional information shared by the respondent below

Q56 Do they wish to have a record of the answers we have recorded?

- Yes (confirm email address where they wish this information to be sent) (1) ____
- No (2)

Q57 Record any comments/additional information shared by the respondent below

End of Block: YOUR USE OF ECONOMIC AND SOCIAL DATA

Appendix B: Total Economic Value Framework

The marine environment provides a wide range of ecosystem services. These can largely be considered to have either use values or non-use values (Figure 22). Use values, as the name implies, derive from the use of a resource. Use values may be further considered as either direct use such as fishery harvest, or indirect use such as seagrass habitats which support the fish populations that ultimately benefit commercial, recreational and Indigenous fisheries.

Associated with use values are option values. These are the benefits of not using a resource now, but maintaining the option to use it in the future. Options values also exist with marine resources under conditions of uncertainty, as these assets may have greater value at some point in the future (Fisher 2000). For example, currently underutilised and discarded species may have an option value if a more valuable use can be derived in the future. Biodiversity also has an option value, as future uses for different species may emerge over time. These values are not as often assessed in non-market valuation studies of marine resources.

Marine resources may also produce non-use values (Figure 22). These are usually based around knowledge that the resource continues to exist for use by future generations even if it is never to be experienced directly (e.g. conservation values). This differs from option value, which relates to potential future uses by the current generation. Conservation of iconic species (e.g. whales, seals, dolphins) provide examples of non-use values; and the impact of fishing on these species can thereby impose a cost on society through loss of these values.

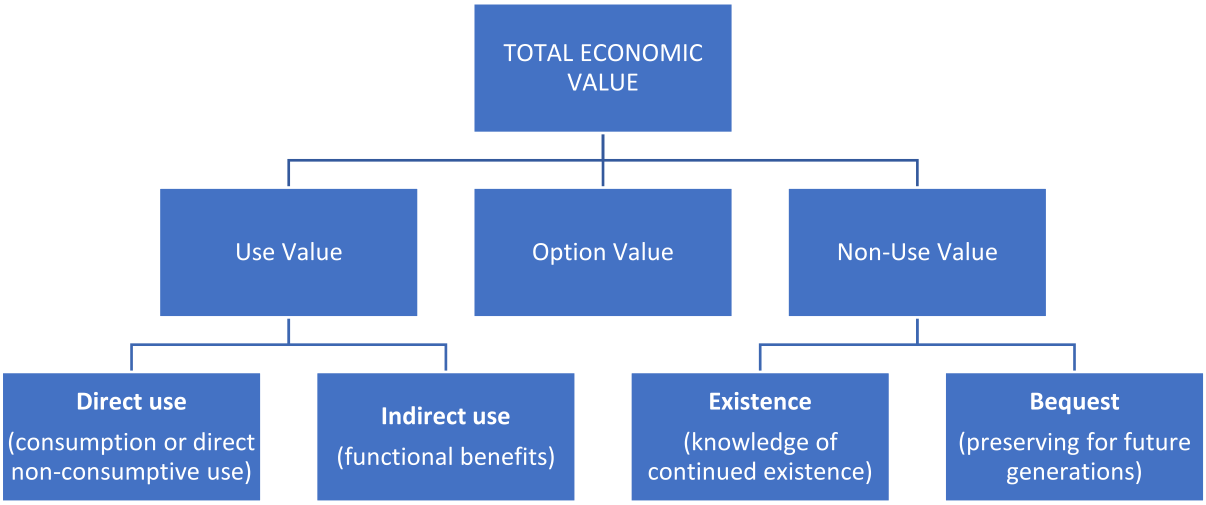


Figure 22. Total economic value framework

The Millennium Ecosystem Assessment (2005) grouped ecosystem services into four broad categories: *provisioning*, such as the production of food and genetic resources; *regulating*, such as the control of water quality and erosion; *supporting*, such as nutrient cycles and primary production; and *cultural*, such as spiritual, educational and recreational benefits. Marine resources may provide several values simultaneously. For example, fish stocks provide recreational use benefits (cultural services), but also provide commercial benefits (provisioning services).

Appendix C: Non-market valuation approaches

Revealed preference methods

Revealed preference (RP) methods are used to estimate use-values that are not captured directly from market data. RP methods are based on observable and actual choices for utility maximization (Parsons 2003). Preferences of non-market values are determined indirectly through consumption of representative market goods (Freeman 2003). More fundamental RP methods include analysis of market or related census data (Brownstone *et al.* 2000). Incorporation of individual consumers' characteristics (e.g. income, age, education) have been increasingly applied to RP studies such as in food consumption (Myrland *et al.* 2000) and education (Jacob and Lefgren 2007). Recent examples pertaining to ecosystem services include user-value for natural resources estimated using the travel cost method (TCM) (Martínez-Espiñeira and Amoako-Tuffour 2008; Martínez-Espiñeira and Amoako-Tuffour 2009; Doshi and Pascoe 2013; Pascoe, Doshi *et al.* 2014), and the impact of floods on house prices estimated through hedonic pricing (HP) (Pryce *et al.* 2011; Bin and Landry 2013).

RP methods have been well established and extensively applied in the context of non-market valuations. However, there are key limitations to the application of these methods to ascertain non-market values. Studies in ecosystem valuation highlight the inability to account for non-use and passive use values. However, the main issue with regards to the context for this study is the inability for RP methods to assess non-market values for hypothetical situations/markets or new products (Louviere *et al.* 2000; Carson *et al.* 2001). The lack of established markets or representative market goods results in the inability to derive estimates (Bunch *et al.* 1993). Also, attempting to estimate proxy values from existing markets is largely uninformative in such cases (Bennett and Blamey 2001).

Hedonic pricing

The hedonic pricing model assumes that consumers' perception of the value of a good derives from the characteristics of the good, based on theoretical foundations developed by Lancaster (1966) and Rosen (1974). By obtaining measures of these characteristics and incorporating them into a regression model, the contribution of each characteristic to the overall price of the good can be derived. In this sense, the value of the characteristic derived from a market value represents a proportion of the market value.

Hedonic price models have been used to estimate the value of a range of environmental attributes. For example, Garrod and Willis (1992), Lake *et al.* (1998) and Hamilton and Morgan (2010) valued different environmental attributes (e.g. woodland, rivers, beach access and views etc.) based on their contribution to real estate (house) values, while Hamilton (2007) valued different coastal access values based on accommodation charges in hotels adjacent to these assets.

Hedonic pricing approaches estimate *marginal values* of an environmental good. That is, the contribution of an additional unit of the environmental good. In the case of fishing, for example, this may represent the marginal value of an additional fish. In the case of aquaculture, for example, this may represent the marginal value of a change in amenity.

Applications in fisheries, aquaculture and marine resource management

Hedonic pricing has been applied in a wide range of fisheries and aquaculture contexts, predominately around the contribution of fish characteristics to the price consumers were willing to pay in the market place. Examples include studies of the influence of landed size on price (Asche, Chen *et al.* 2014; Lee 2014), where the fish was caught (reflecting the environmental conditions of the region) (Asche and Guillen 2012; Asche, Chen *et al.* 2014), and how the fish was caught (Asche and Guillen 2012). Other studies have considered the value of sustainability certification (Roheim, Asche *et al.* 2011).

Other studies have considered the negative impacts of aquaculture on water quality, availability and visual amenity through an examination on house prices in areas close to aquaculture facilities (Huang 1990; Evans, Chen *et al.* 2017). Chen, Lee *et al.* (2006) used such hedonic prices to estimate the optimal level of aquaculture production given the values of the externalities associated with groundwater use and water pollution.

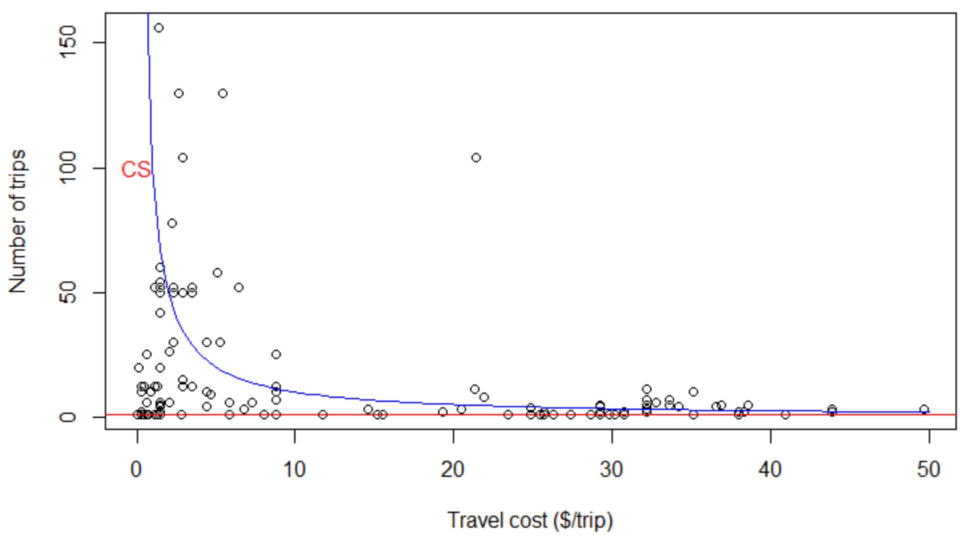
Other studies have used hedonic property pricing to estimate the amenity values of National Wildlife Refuges near urban areas on the eastern coast of the US (Xiangping, Taylor *et al.* 2013) and of wetlands in Perth, Australia (Tapsuwan, Ingram *et al.* 2009).

In recreational fisheries management, Bennear, Stavins *et al.* (2005) used hedonic pricing to estimate the implicit value of a recreational fishing day based on volume and price of recreational fishing licences sold in the US over a 15 year period. They found substantial variation in value between different recreational fisheries across the US, and concluded that the previous practice of benefit transfer potentially introduced substantial bias in the estimation of recreational values. Carter and Liese (2010) used data from the market for offshore charter fishing in the Gulf of Mexico, with the marginal value of sportfishing harvest identified based on differences in charter fees charged and spatial variation in harvest rates and fish sizes (taking account also of other factors such as vessel characteristics and distance travelled).

Travel cost method

The initial theoretical underpinnings of TCM were first proposed in a letter by Hotelling to a policymaker that outlined how the demand for visits to a park could be estimated based on the number of visits, and using the cost of travel and entry taken as a proxy to the price (Smith and Kaoru 1990). This allows for the derivation of a demand curve for the site similar to that shown in Figure 23 (the blue line), from which consumer surplus estimates can be derived. Consumer surplus – the measure of non-market benefits to the site – is the difference between what the visitor would be (theoretically) willing to pay to go to the site and what they are actually required to pay (the red line). This is the area under the curve in Figure 23 bounded by the price level for a single visit (area CS).

Figure 23. Demand curve derived from trip and travel cost data



The travel cost approach does not ask willingness to pay directly but imputes it from the observed behaviour of visitors through an estimated demand function, which relates the number of observed trips to the travel cost incurred. The demand function is estimated at an individual level and assumes that individuals with similar characteristics will respond to the “price” of accessing the recreational site (i.e.

the cost of the trip) in similar ways. In some cases, zonal models have been used to estimate consumer surplus associated with recreational travel (e.g. Fleming and Cook 2008), where the proportion of the population traveling to the site from each zone is used as the trip measure.

The use of TCM in valuation and derivation of demand can be divided into demand for recreation sites, recreation activities and recreational features. In the case of the former, the demand function derived is that for the number of trips to the site regardless of the activities participated in. The separate value of the activities undertaken at the site can also be estimated provided additional information on what visitors did at the sites is available.

Most travel cost models are based on visits to, and data collected from, a single site. However, in many cases, recreational users may visit several different sites over the year. Where the decision to visit different sites is based on different site characteristics, multi-site models can be used to assess the contribution of these characteristics (e.g. catch rates, iconic species or other site attributes) to value, and thereby derive the marginal value of these attributes. Multi-site models can be developed using random utility modelling approaches. These can be also used to derive not just the value of a trip, but also the value of different characteristics of the areas.

Single site models can also be used to estimate some marginal values, such as those associated with different catch rates by individuals. This approach has also been termed the *hedonic travel cost* approach (Brown and Mendelsohn 1984), as it is a technique which reveals how much users are willing to pay for each individual characteristic of outdoor recreation sites based on their cost of access (much the same as attribute values are determined in a hedonic model).

An additional feature of the travel cost model is that, being demand curves, they can also be used to estimate the price elasticity of demand for recreation. That is, they can be used to estimate how the level of demand may change with changes to the travel cost, either through the introduction of a policy instrument (e.g. licence fee) or changes in costs due to changes in fuel prices for example.

Applications in fisheries, aquaculture and marine resource management

The travel cost method has been used extensively to derive values for recreational fishing. A range of values derived from Australian travel cost studies are given in Tables 1 (average value per recreational fishing trip) and Table 2 (marginal value of a fish derived from Australian recreation fishing).

The value of fisheries for tourism and the local coastal communities has also been estimated. Paredes (2020) used the contingent travel cost method to estimate the contribution of supplies of local seafood and the option of seeing the local fishing fleet to the value of a trip to a coastal town in Southeast Queensland. The results of the study indicate that the fishing and seafood industries are of value to day trippers and holiday makers, such that the loss of these industries would reduce the value per trip (on average).

In the coastal and marine environments, the travel cost method has been used to value some key habitats that can be affected by fisheries, such as coral reefs (Seenprachawong 2001; Carr and Mendelsohn 2003; Seenprachawong 2003; Ahmed, Umali *et al.* 2007; Andersson 2007; Andersson 2007; Martínez-Espiñeira and Amoako-Tuffour 2008; Tapsuwan and Asafu-Adjaye 2008; Martínez-Espiñeira and Amoako-Tuffour 2009; Doshi and Pascoe 2013; Pascoe, Doshi *et al.* 2014). These studies have all been conducted overseas.

Stated preference methods

Stated preference (SP) methods are valuation techniques that determine the value of a good or service through reported responses to (hypothetical) changes or scenarios (Bennett and Blamey 2001). Individuals are assumed to account for information regarding various available scenarios of a particular decision-making problem. These agents then evaluate the 'quality' of each alternative based on various attributes, and make trade-offs on levels and positions of these attributes in selecting one of the alternative scenarios (Adamowicz *et al.* 1998). Thus, individuals derive utility from the bundle of characteristics or attributes rather than the good or service itself[1] (Louviere, Hensher *et al.* 2000). The individual choices are then modelled to determine the marginal values of the specific good, service, or attribute, based on the survey sample. These responses are grounded on economic principles of rational choice and utility maximisation.

The theoretical underpinnings of SP methods are based on Random Utility Theory. Initially proposed by Thurstone (1927), this theory assumes that an individual makes choices to maximise their utility based on two types of elements, systematic (known and observable), and random (unknown and unobservable); subject to constraints e.g. income and time endowments. Both good/service characteristics and individual differences (e.g. demographics) form the systematic, explanatory components of the behavioural choices, outside of other unknown factors captured in the random component. As responses are generally based on a finite number of alternative scenarios, SP methods are often modelled based on probabilistic models (e.g. probit, logit, and multinomial logit).

Stated preference valuation studies most often utilize two types of methodologies[2]: (1) contingent valuation methods (CVM) and more recently, (2) discrete choice experiments (DCE[3]).

Contingent valuation method

Initially proposed by Ciriacy-Wantrup (1947), the contingent valuation method (CVM) analyses how much respondents are willing to pay (or accept) for a change in the quality/attribute of a good or service through scenario-based surveys. The estimated economic values for specific attributes are "contingent" on those highlighted in each listed scenario (Carson 2011). The types of questioning employed in CVM is diverse across the studies; they can either be open-ended (e.g. how much a respondent will be willing to pay) (Alvarez-Farizo 1999), referendum-styled (is respondent willing to pay $X?) (Cameron and Huppert 1991), or through bidding (listing progressively increasing WTP values until the response rejects) (Yu and Abler 2010). While the use of such direct survey methods to derive demand schedules was much debated among economists, there has been some admission in the validity of such surveys due to the ability to capture (complete) non-market monetary values and non-use values (Hanemann 1994); as evident in the much quoted statistic of over 7500 studies across 130 countries catalogued by Carson (2011).

The open-ended variant is said to result in inflated WTP estimates due to the lack of realistic boundaries for a respondent both in the laboratory (Neill *et al.* 1994) and field setting, which Loomis, Brown, Lucero, and Peterson (1996) attempted to reduce in the former through more stringent survey questions. However, there has been some evidence that the open-ended CVMs can result in lower WTP estimates compared to bounded varieties (Brown *et al.* 1996). The bidding-styled CVM is largely similar, in that the respondent is not bounded by his/her response for a WTP. The use of a referendum-styled CVM is the most popular variant in the literature, where respondents are given hypothetical choices of paying a stated amount for a particular good/service or change in quality, most often done with dichotomous

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[1] This overview of stated preference theory is an evolution from early definitions of conjoint analysis, particularly by Green and Srinivasan (1978).

[2] Conjoint analyses (CA) are not addressed in this review as a SP method due to it being based purely on mathematical systems rather than individual preferences. However, reference to past CA studies will be discussed if they provide insight into the study or are relevant examples outside of the economics discipline. There has been discussion into how some CA studies resemble DCEs (i.e. Louviere *et al.* 2010) or that DCEs are an evolution of CAs (Bennett and Blamey 2001).

[3] Taking the cue from Carson and Louviere (2011), the term DCE is used rather than the more common 'choice experiment' to avoid confusion with the latter's use in other disciplines, the underlying theories and assumptions, and the interpretation of estimates.

choices. The aggregate WTP is then estimated using random utility models (RUM) using limited dependent variable (LDV) regressions i.e. probit and logit for dichotomous choice, and multinomial variants for multiple alternatives.

Applications in fisheries, aquaculture and marine resource management

As noted in the main section of the report, contingent valuation has been applied internationally in several fisheries studies, primarily with regard to recreational fishing (e.g. Sorg and Loomis 1986; Toivonen, Roth *et al.* 2004; Deely, Hynes *et al.* 2019). Several Australian studies have also used contingent valuation. The FRDC project "Golden fish: evaluating and optimising the biological, social and economic returns of small-scale fisheries" (FRDC 2016-034) is using contingent valuation approaches to value recreational fishers' willingness to pay for additional catch. The aim of the study is to estimate the potential benefits of a stock enhancement program for the recreational fishing sector. Yamazaki, Rust *et al.* (2013) estimated the value of a day's recreational fishing using contingent valuation for two Tasmanian fisheries.

Several contingent valuation studies have also been undertaken with regard to ecosystem impacts associated with aquaculture development. For example, Gunawardena and Rowan (2005) and McDonough, Gallardo *et al.* (2014) estimated the non-market value of mangroves and wetlands displaced by aquaculture in Asia, while Whitmarsh and Wattage (2006) looked at the value of pollution caused by farmed salmon in Scotland. In contrast, Yi (2019) used CVM to estimate the willingness to pay of consumers for sustainable aquaculture products in Korea, and the economic value of environmental improvements in aquaculture areas (Yi and Kim 2020).

Discrete choice experiments

Discrete choice experiments (DCE) estimate the probability of choosing from a given set of alternatives, known as a 'choice set', where the individual receives the highest perceived utility based on both known and unknown elements (Train 2009). The alternatives are characterised by a set of attributes at different levels (Adamowicz, Louviere *et al.* 1998), and individuals make trade-offs across multiple attributes simultaneously (Bennett and Blamey 2001). The marginal utility from each attribute can then be estimated through analysis of these trade-offs (Adamowicz, Louviere *et al.* 1998; Hensher *et al.* 2005). The referendum-styled (dichotomous) CVM is largely similar to DCE techniques. However, unlike the former, DCE employs the use of multiple alternative hypothetical scenarios/choices with varying attribute levels marked with related WTP values, and the specific inclusion of a status quo or 'no-choice' option to avoid issues of forcing (Rolfe and Bennett 2009). The responses are then estimated using multinomial models and the WTP value for specific attributes can be derived from regression estimates. Although there are methodologies (e.g. conjoint analyses) in non-economic studies that employ a similar design (e.g. Green and Srinivasan 1978; Green and Srinivasan 1990), DCEs are specifically based on economic theory and potentially help to elicit choice preferences rather than just estimate aggregate preferences specific for the sample (Adamowicz *et al.* 1998; Louviere, Flynn *et al.* 2010).

DCEs were developed to overcome methodological limitations in transport choice studies (Louviere 1981; Louviere and Hensher 1982) and they continue to be a tool in this field (Hensher *et al.* 1988; Kroes and Sheldon 1988; Hensher 1994). Since then, the use of DCEs in environmental economics literature has been increasing, particularly in conjunction with non-market valuation of natural resources. Some examples include valuation of national parks (Boxall and Adamowicz 2002; Juutinen *et al.* 2011; Chaminuka *et al.* 2012) and forests (Boxall and Macnab 2000; Lehtonen *et al.* 2003; Christie *et al.* 2007). They have also been used to derive consumer preferences for environment-related activities such as rock climbing/hiking (Scarpa and Thiene 2005).

Applications in fisheries, aquaculture and marine resource management

In the marine environment, DCEs have been used for the valuation of coral reefs (Ngazy *et al.* 2005; Wattage *et al.* 2011) and recreational diving at coral reef sites (Ngazy, Jiddawi *et al.* 2005; Parsons and Thur 2007; Doshi *et al.* 2012; Gill *et al.* 2015). Similarly, the value of NSW coastal and marine habitats were valued using DCE (Pascoe, Doshi *et al.* 2019).

DCEs have also been used in commercial fisheries and aquaculture to assess consumer preferences, and willingness to pay for more sustainable fisheries (Lew and Larson 2015; Pascoe, Innes *et al.* 2016; Bronnmann and Asche 2017; Cantillo *et al.* 2020; Xuan and Sandorf 2020). DCEs have also been used to estimate the value of commercial fisheries to tourism (Ropars-Collet, Leplat *et al.* 2015).

In recreational fisheries, DCEs have been used to assess the value of changing bag limits (Lew and Larson 2015) and other characteristics of the fishing trip (Lew and Larson 2012; Golden *et al.* 2019; Carter *et al.* 2020). The value of protecting recreational fish stocks has also been estimated using DCEs (Grilli and Curtis 2020).

Several studies have been undertaken in the US progressively estimating the non-market value of threatened and endangered species using DCE (Wallmo and Lew 2011; Wallmo and Lew 2012; Lew 2015; Wallmo and Lew 2016). These types of values would be of use in addressing the hypothetical example given in Box 3, and were identified as a priority for Australian fisheries and aquaculture management.

Advantages and limitations of stated and revealed preference approaches

All of the above approaches have both advantages and limitations, and their appropriate use will depend on the type of non-market value being assessed.

Revealed preference approaches require a well-functioning market in order to estimate the non-market values. Hedonic pricing is limited to attributes that can be considered to affect the market price for some other commodity. Factors that affect the price of fish, for example sustainability certification, can be readily established. However, determining what this value captures is itself an issue, and the value may represent a bundle of attributes. For example, certification often requires several criteria to be met, including resource sustainability, habitat sustainability (i.e. limited or no adverse impacts), bycatch sustainability and, in some cases, social measures to be met. Separating out these values is generally not feasible unless multiple certification schemes with different criteria are in operation for a given fish species on a given market. Most environmental studies using hedonic pricing, including those relating to aquaculture, are based on real estate values. This assumes a well-functioning property market, that householders are well-informed about differences in environmental quality in different locations, and have similar perception of environmental quality (Muir *et al.* 1999).

While travel cost approaches have a theoretical advantage in that they are based on observed behaviour, there is still lack of consistency in terms of how the travel cost is measured. The estimate of consumer surplus is very sensitive to the cost measure, with higher estimates of travel cost resulting in higher estimates of consumer surplus. The wide variation in recreational values seen in Table 1 reflect these different assumptions.

These measures of travel cost used in the analyses are influenced by assumptions about which costs are included and how they are measured. For example, accommodation costs for multiday trips will reflect also the value derived from the accommodation option (e.g. 5-star resort versus camping area), which are not related to the benefit of the consumer surplus of the activity being considered (e.g. recreational fishing). Similarly, inclusion of different food costs can distort the measure of travel cost, particularly if some individuals dine in restaurants and others bring a packed lunch. There is also no consistent treatment of the value of travel time. Allocation of costs in case of multi-purpose trips requires further assumptions. Even in single day trips, the recreational activity of interest may just be one of several activities undertaken. Allocating the total cost to this one activity will inflate the estimated consumer surplus. Further details on these issues are provided in Appendix D.

Stated preference approaches have a distinct advantage over revealed preference approaches when considering options that are not apparent in the "marketplace". For example, assessing the potential willingness to pay for a certification scheme that does not yet exist, or changes in management options that also have not yet occurred. It can also be used to value components of a system that are not readily observable. For example, the value of a particular species in a range of species caught when such information is not available for other methods (i.e. due to being unavailable).

Several limitations have been highlighted in regard to the use of CVM. These pertain to how responses might be invalid or subject to bias due to respondents misrepresenting their preferences, 'yea saying',[4] being insensitive to the scope, or not reflecting the accessibility to substitutes (Bennett and Blamey 2001). These have resulted in criticism of the methodology (Portney 1994). Also, issues regarding the survey design and administration (Carson, Flores *et al.* 2001; Carson 2011) have suggested the need for alternative methodologies.

There are a number of key benefits of DCEs over CVMs as an SP method of non-market valuation, especially in the context of hypothetical goods. The availability of a detailed account of respondent trade-offs and preferences allows for estimating attribute-specific estimates (Bennett and Blamey 2001). These preferences can then be applied to alternatives that may not be specifically represented in the experiment e.g. the marginal WTP for bycatch reduction of a TEP species can be applied to any similar fishery in which the species is caught. These 'decomposed' estimates can also be used beyond the scope of the DCE, such as in benefit transfer (Morrison *et al.* 2002) and policymaking (Bennett and Blamey 2001). Also, DCEs reduce the severity of yea-saying, and scope and framing issues that plague SP methods compared to CVM (Rolfe *et al.* 2000; Bennett and Blamey 2001; Bennett and Blamey 2001; Rolfe *et al.* 2002). As an SP method of non-market valuation DCEs are subject to hypothetical bias (Hensher 2010). However, studies have suggested methods where DCEs can reduce this bias relative to other SP methods (Morrison *et al.* 1997; Murphy *et al.* 2005).

Nonetheless, DCEs are also subject to limitations. A major issue is the additional cognitive burden that the experiment places on respondents. This is due to the relatively more complex design of DCEs compared to other methods, including CVM. This can result in the respondent becoming confused, fatigued, or frustrated, resulting in improper weighing of the attributes and making inaccurate trade-offs (Bennett and Blamey 2001). Respondents might also not agree (or protest) with the premise of the experiment and/or alternatives, resulting in a similar issue of inaccurate trade-offs and choices (Bennett and Adamowicz 2001). These two limitations can manifest themselves through respondents either ignoring any number of the attributes (including all) or making choices randomly or choosing the same option consistently across the experiment. This results in inaccurate and biased estimates. However, accounting for these issues through the experiment design and modelling can reduce and alleviate them.

Also, despite reducing framing and scope issues from other SP methods, DCE can still be exposed to them. In terms of the former, the context-setting pre-experiment briefing may over-inform respondents and result in upwards bias to estimated values (Rolfe, Bennett *et al.* 2000). However, this unrealistic information overload is likely the result of difficulties in capturing all relevant issues within the limited choice set (Bennett and Blamey 2001), causing practitioners to add complexity and deviate from reality in experiment and survey designs. The hypothetical nature of DCE also leaves it susceptible to issues of scope insensitivity, with respondents showing the same WTP for attributes at different levels due to

[4] 'yea saying' is the tendency of some respondents to conform to perceived expected responses (based on social pressures or norms) even if they do not align with their true belief. For example, some respondents may suggest a WTP value is appropriate if they believe they are expected to, even if they would not actually be WTP that amount in reality (Blamey *et al.* 1999).

an indifference for increasing levels beyond a certain point (Rolfe, Bennett *et al.* 2000; Frontuto *et al.* 2017). The framing issue can be accounted for through a survey and experiment design that appropriately frames the context in a realistic scenario and avoid over-information. The issues from scope insensitivity were assumed to less apparent in this context as the levels were not defined by areas and years, which is more apparent in natural resource based DCEs.

Finally, like all SP methods, DCEs are subject to potential inflated WTP estimates, which may not reflect actual WTP, due to hypothetical bias (Hensher 2010). There have been a number of ways suggested to reduce this effect including benchmarking prices to actual prices (Roe *et al.* 2001; Bergmann *et al.* 2006) or introducing 'cheap talk' scripts to reduce potential inflated WTP estimates (Carlsson *et al.* 2005; Van Loo *et al.* 2011). The latter includes a script informing respondents about avoiding overestimating their value of attributes and also to consider their own household budget constraints where relevant.

Related approaches

Production function approaches

Ecosystem services provide an indirect use function in the production of many fisheries and aquaculture outputs. For example, mangroves provide habitat for juvenile fish which may later recruit to the exploitable stock, while water quality can affect fish health and consequently aquaculture production.

Production functions, where output is modelled as a function of a range of inputs – both fixed and variable – are commonly used in agriculture, aquaculture and fisheries economic analysis. These usually involved only physical inputs (e.g. farms size, labour used, etc), but the analysis can be extended to include environmental variables (e.g. rainfall, water flow, water quality, supporting habitat area, etc).

Production function approaches can be used to estimate the marginal contribution of an ecosystem service to the physical output of a sector – the marginal product. Given exogenous prices, this can then be converted to marginal value product, which reflects the non-market value of the ecosystem service (Point 1994). Given this, their measurement can allow estimates of the non-market value of changes in these assets (e.g. an improvement in water quality).

Applications in fisheries, aquaculture and marine resource management

Production function approaches have often been used to estimate the contribution of habitats to fisheries (Barbier 2000; Aburto-Oropeza *et al.* 2008; Armstrong *et al.* 2016; Kurniawan, Arkham *et al.* 2020), coastal management (Boyer and Polasky 2004) and other resource based industries e.g. forestry (Zhang and Stenger 2015). In Australia, Taylor and Creighton (2018) used a production function approach to estimate the values of coastal wetlands to prawn production in order to justify restoration works in the area. Schrobback, Pascoe *et al.* (2018) used a production function approach to estimate the economic impact of changes in water quality on oyster production.

Limitations

Production function approaches, like revealed preference approaches, are based on actual relationships between inputs and outputs that can be observed in an industry. This provides only one aspect of the non-market value, namely the indirect use value. For example, mangroves provide ecosystem services beyond just that of being a nursery area for commercial fisheries. As a result, they provide a reliable estimate of the ecosystem service being valued (Zhang and Stenger 2015), but just a lower bound estimate of the non-market value of the environmental asset.

In terms of their use for fisheries and aquaculture management, the values reflect changes in market-based values also. For example, changes in water quality may lead to higher revenues in aquaculture, with these being captured directly in the market-based benefit (i.e. the production value). Hence, if assessing potential benefits of water quality improvements and the impact of this on production has already been determined, the non-market value does not need to be additionally included.

Replacement cost and avoided cost approaches

An alternative way to value a service provided by an environmental asset is to estimate the replacement cost of providing that service should the natural asset no longer function properly or no longer exist (Boyer and Polasky 2004). For example, a beach provides a range of ecosystem services including shoreline protection. If the beach was eroded (for example through changes in coastal management), then equivalent coastal protection would require building a seawall, the cost of which can be determined through engineering specifications. Consequently, the value of the coastal protection service offered by the beach is at least equal to the cost of the equivalent engineered service.

Related to the replacement cost approach is the avoided cost. This is the cost that would be incurred if the environmental asset is removed. Building on the above example, the avoided cost of having a beach is the potential cost of property damage from coastal erosion (Raheem, Colt *et al.* 2012).

Applications in fisheries, aquaculture and marine resource management

Several Australian studies have applied the replacement cost approach to value the harvest and consumption of subsistence food items by Indigenous fishers (Gray and Altman 2006; Buchanan, Altman *et al.* 2009; Jackson, Finn *et al.* 2011; Sangha, Stoeckl *et al.* 2019). These studies used market prices to determine equivalent replacement costs for the catch of Indigenous fishers (i.e. what they would have had to pay for the fish if they were bought). The use of market prices was in part to minimise the use of a survey, responses from which they considered variable at best, and to also allow for a straightforward calculation of these values.

Limitations

As with the production function approach, replacement or avoided cost tends to underestimate the actual total economic value of an environmental asset as it focuses on just one service that the asset provides, but may be appropriate for some regulating services such as those illustrated above (Raheem, Colt *et al.* 2012).

Measuring what it costs to replace an ecosystem service is not generally viewed by economists as an appropriate measure of value as people might not be willing to replace the service at the replacement cost (Swinton et al. 2007). As a general rule, comparing replacement values against economic value of other goods and services should be undertaken with caution. Avoidance of this approach is advised if the costs of damage avoidance or replacement substantially differs from the benefits provided (Zhang and Stenger 2015).

Multi-criteria decision analysis

A further alternative approach is to estimate values using preference elicitation techniques commonly employed in multi-criteria decision analysis (MCDA). These approaches, as the name suggests, are aimed at supporting decision-making through assessing stakeholder preferences for different outcomes of management. 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).

While there are several different multicriteria approaches to elicit relative preferences, a commonly used approach to preference elicitation in fisheries and environmental management is the Analytic Hierarchy Process (AHP) (Saaty 1980). AHP is based upon the construction of a series of pair-wise comparison matrices which compare attributes to one another, and a hierarchical structure that groups similar attributes into subgroups. The pair-wise comparison method makes the process of assigning preferences much easier for participants because only two attributes are being compared at any one time rather than all attributes having to be compared with each other simultaneously (such as in the case of choice experiments).

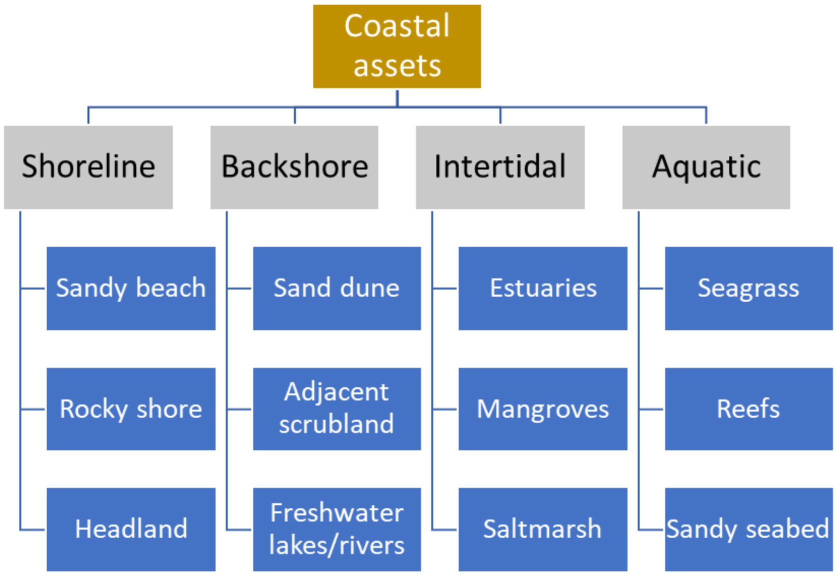
AHP estimates the individual's relative preference for an attribute, relative to the other attributes in the complete set. Two well established premises in economics are, first, that an individual's relative preference for an attribute relates to the relative utility that the individual receives from the attribute (Arrow 1963); and second, that economic value also reflects relative utility (Debreu 1959). Given these two premises, we can assume that the relative preferences also reflect the relative value of the assets to the individual.

AHP has been combined with other measures (e.g. those derived from a choice experiment) to derive non-monetary values for a range of environmental assets. These have largely been to terrestrial environments (Curtis 2004; McDaniels and Trousdale 2005; Moran *et al.* 2007; Martin-Ortega and Berbel 2010), with only limited application in the coastal environment (Wattage 2010; Brocklesby *et al.* 2015). The way these approaches were applied also differed substantially. In most studies, the weights derived from the multi-criteria approach were only used to allocate estimates of total economic value to the component ecosystem services (Curtis 2004; Wattage 2010; Brocklesby, Buchan *et al.* 2015). In contrast, Pascoe, Doshi *et al.* (2019) scaled estimated non-market values from a choice experiment for a key coastal asset to provide estimates for a wider range of assets based on their relative preferences.

The approach has also been used to derive values relating to Indigenous communities. McDaniels and Trousdale (2005) extrapolated market-based measures of economic loss incurred by an Indigenous community as a result of petroleum development on their lands based on a multi-criteria analysis of the relative importance of economic, cultural (e.g. traditional and spiritual sites), community (e.g. health and social cohesion) and environmental losses.

A feature of AHP is, as the name suggests, its hierarchical structure. That is, preferences are determined first at an aggregate level across broad categories, and then for each sub-category. This reduces the total number of pairwise comparisons that need to be made. For example, Pascoe, Doshi *et al.* (2019) developed a hierarchy linking different coastal and marine assets (Figure 24), with the four broad categories based on the relative location of the assets, then the individual components of each of these broader categories compared within the category.

Figure 24. Example hierarchy of coastal natural capital assets

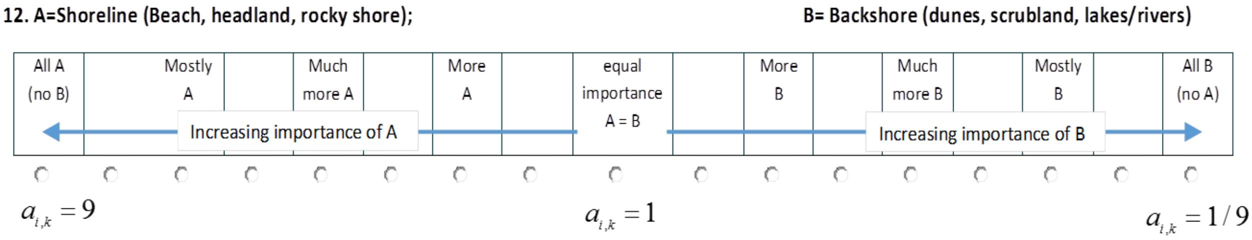


Source: Pascoe, Doshi *et al.* (2019)

The most common approach to estimating preferences using AHP is to use pair-wise comparisons. Preferences are expressed on a nine-point scale, with a 1 indicating equal preference, and a 9 indicating an extreme preference for one of the sub-components. Preferences are assumed symmetrical, such that if A against B has a preference of $a_{AB} = 9$, then $a_{BA} = 1/a_{AB} = 1/9$.

An example of this for the above hierarchy is shown in Figure 25 for two broad asset categories shoreline and backshore. In this case, considering only Shoreline as important would result in a preference score of 9, while considering only Backshore assets as important would result in a preference score of 1/9.

Figure 25. Example of bivariate comparison used in the coastal asset study



Source: Pascoe, Doshi *et al.* (2019)

Variants of the AHP process have also been developed, particularly designed to simplify online survey applications (Pascoe, Cannard *et al.* 2019). Further details of these are given in Appendix D.

Applications in fisheries, aquaculture and marine resource management

AHP is one of the most common approaches used for preference elicitation in a wide range of applied environmental and natural resource case studies (Deng *et al.* 2002; Hajkowicz and Collins 2007; Huang *et al.* 2011). The link between AHP derived weights and monetary values has also been used in several cases, with the relative utility determined by AHP used to derive monetary values of non-monetary assets.

AHP has been used in a number of fisheries applications, primarily to determine preferences for different fisheries management objectives (Leung *et al.* 1998; Soma 2003; Mardle *et al.* 2004; Wattage and Mardle 2005; Nielsen and Mathiesen 2006; Himes 2007; Pascoe *et al.* 2009; Baby 2013; Pascoe, Dichmont *et al.* 2013; Jennings, Pascoe *et al.* 2016).

AHP has a fairly broad applicability for fisheries management. For example, it has also been used to help determine fishing harbour management and planning given environmental, social and economic considerations (Chen *et al.* 2019). Weights derived using AHP can be used for determining optimal management strategies across a range of different objectives (Dichmont, Pascoe *et al.* 2013; Pascoe, Cannard *et al.* 2019), and can be used to derive utility measures for use in cost-utility analyses of management options.[5]

AHP has also been extensively applied in aquaculture. Applications include assessments of social acceptability of different management strategies in order to develop a social licence to operate (Whitmarsh and Wattage 2006; Whitmarsh and Palmieri 2009; Whitmarsh and Palmieri 2011) or certification (Jong-Seok *et al.* 2014; Jong-Seok *et al.* 2014), and identifying best sites for sustainable aquaculture (Halide *et al.* 2009; Hadipour *et al.* 2015).

[5] To date, no cost utility studies have been undertaken in fisheries, but have been undertaken in other resource-based industries (e.g. Hajkowicz *et al.* 2008; Marinoni *et al.* 2011).

Benefit transfer

An alternative to development of site-specific or species-specific values is to apply benefit transfer techniques. These involve the use of empirical value estimates from a previous study undertaken in a similar context, and applying these to the case study in question (Boutwell and Westra 2013). For benefit transfer to be valid, the study site from which the estimates are derived has to be similar to the site it is being applied to.

Benefit transfer first requires estimating a transfer function, usually developed through meta-analysis to isolate the effects of the different study-site specific characteristics on the resultant values. Several meta-analysis studies have been undertaken linking non-market values of wetlands (derived through stated preference, revealed preference and/or production function approaches) to the characteristics of the study and the ecosystem services produced, including fisheries production and fisheries nursery services (e.g. Brander, Florax *et al.* 2006; Brander *et al.* 2012; Camacho-Valdez *et al.* 2013; Chaikumbung *et al.* 2016). These have found variable reliability of the approach, with one study finding errors of around 75% (Brander, Florax *et al.* 2006) and another errors of around 17% (Chaikumbung, Doucouliagos *et al.* 2016). Other studies have focused on impacts of water quality on recreational fishing, using meta-analysis to derive the appropriate values (Johnston, Besedin *et al.* 2005).

Applications in fisheries, aquaculture and marine resource management

Several studies have used meta-analysis to derive values of recreational fish that can be used for benefit transfer purposes. For example, Johnston, Ranson *et al.* (2006) estimated that the real WTP per fish over the sample ranged from US$0.048 to US$612.79, with a mean of US$16.82 (June 2003 values). The meta-data were drawn from 391 observations from 48 non-market valuation studies that estimate the marginal value (or WTP) that anglers place on catching an additional fish or allow such a value to be calculated. The model results suggest that WTP is systematically influenced by both methodological variables and by resource, context, and angler characteristics. Similarly, Mazzotta, Wainger *et al.* (2015) used benefit transfer approaches to estimate the costs to recreational fishing from mining operations that affect stock abundance. The estimated values per species using meta-analysis, and found that the values for freshwater bass species ranges from around $15 to around $22 (2012 US dollars) per additional fish caught, and the value of other species range from around $3.80 to around $4.50 per additional fish caught. Sturtevant, Johnson *et al.* (1995) used meta-analysis as a benefit transfer approach for deriving values of freshwater recreational fishing.

Combining methods

The above studies are not necessarily required to be used in isolation, and several studies have used combinations of approaches to derive appropriate values.

For example, Rolfe and Prayaga (2007) combined travel cost models with contingent valuation methods to determine the current value of recreational fishing and the potential changes in values due to site improvements. Similarly, Prayaga, Rolfe *et al.* (2010) combined travel cost models with contingent behaviour models (related to stated preference approaches but based on behavioural change e.g. number of trips) to estimate changes in recreational fishing demand along the Great Barrier Reef. While not fisheries related, Pascoe, Doshi *et al.* (2019) combined multicriteria decision analysis approaches with choice experiments to derive values for a wide range of coastal and marine environmental assets.

Appendix D: Technical details on non-market estimation methods

Travel cost models

The visitor's demand for the activity, represented by the number of trips undertaken, is assumed to be a function of the cost of travel (taken as a proxy measure of its price), the socioeconomic characteristics of the visitor (e.g. education level and income) and the characteristics of the location. The generic demand curve for an individual *i* can be given as

$$x_i = f(z_i) + \varepsilon \tag{1}$$

where x_i is the number of trips to the site undertaken, and z_i is a vector of factors influencing demand, such as travel cost, individual characteristics such as income, site characteristics (if available), and ε is a random error term assumed to be independent and identically distributed (iid).

The dependent variable – the number of trips – has a non-negative integer distribution and therefore ordinary least squares (OLS) regression methods are inappropriate for estimating the model (Haab and McConnell 2002). Instead, count models are generally considered more appropriate (Creel and Loomis 1990),[1] which in principle estimate the probability of a visitor choosing to visit the beach. If the probability of visiting a beach on any particular day is small, constant and independent of previous decisions, and if the season length is relatively large, then the distribution of trips will approximate a Poisson distribution (Hellerstein 1991), and the probability of a visit can be given by

$$Pr(x_i = n) = \frac{e^{-\lambda_i}\lambda_i^n}{n!},\ n = 0,1,2,\ldots \tag{2}$$

where n is the observed number of trips and the parameter λ is both the mean and variance of the distribution, commonly specified as an exponential function (Haab and McConnell 2002) and given by $\lambda_i = exp(z_i\beta) = E(x_i|z_i\beta)$ where β is a vector of the unknown parameters associated with each explanatory variable *z*.

A common problem experienced with travel cost models in practice, however, is that the data are not equidispersed, such that the observed variance and mean may differ. Where the variance exceeds the mean, as is often the case in recreational travel cost models, the data is said to be subject to overdispersion and an alternative distributional assumption may be required. Failure to account for overdispersion in models using count data can have serious consequences for estimation and inference (Grogger and Carson 1991).

While several alternatives exist, a common approach is to use the negative binomial model.[2] This has a variance $var(x_i|z_i\beta) = \lambda_i(1 + \alpha\lambda_i)$, where α (the dispersion parameter) is a measure of the degree to which the conditional variance exceeds the conditional mean (Haab and McConnell 2002). If $\alpha > 0$,

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[1] While count models are the most commonly employed, more recently travel cost analyses have also being undertaken using a random utility modelling framework, where choice of recreational location is also taken into consideration (e.g. Yeh, Haab *et al.* 2006; Raguragavan, Hailu *et al.* 2013).

[2] The negative binomial distribution derives from the Poisson distribution through the introduction of a parameter α that may vary randomly allowing for inter-person heterogeneity. If this random variable is assumed to have a gamma distribution, then integrating over this variable leads to the negative binomial model. For further details, see Cameron and Trivedi (1986).

then overdispersion exists and the Poisson model should be rejected in favour of the negative binomial. The probability function for the negative binomial is given by:

$$P * r(x_i) = \frac{\Gamma(x_i+\alpha^{-1})}{\Gamma(x_i+1)\Gamma(\alpha^{-1})}\left(\frac{\alpha^{-1}}{\alpha^{-1}+\lambda_i v}\right)^{\alpha^{-1}}\left(\frac{\lambda_i v}{\alpha^{-1}+\lambda_i v}\right)^{x_i} \quad (3)$$

where $\Gamma(\cdot)$is the gamma probability density function evaluated at $(\cdot)$, and $v > 0$ is an independently and identically distributed random variable with density $g(v|\alpha)$ (Martínez-Espiñeira and Amoako-Tuffour 2008). This collapses to the standard Poisson distribution when $\alpha = 0$.

In most travel cost studies, data are collected on-site from individuals who have definitely undertaken at least one trip (as they were there). This introduces two potential biases; truncation, as the data are truncated at 1 (as no zeros are observed); and endogenous stratification, where individuals who visit the site frequently are more likely to be sampled than people who go to the site only occasionally (Shaw 1988). Methods for correcting the bias this introduces are well established and have been validated empirically (Martínez-Espiñeira *et al.* 2008).

For data collected from the general public, where a proportion of the sample may not have undertaken any beach visits, a different potential problem exists, the issue of zero-inflation. Two approaches are available to address zero-inflation, each with different underlying assumptions. The first approach assumes that the excess presence of zeros is a sampling issue, and estimates the probability of a true or false zero as well as negative binomial model (the zero-inflated mixture model). The second approach (the hurdle count model) is a two stage model that first estimates the probability that a trip would be undertaken using a binary choice model (i.e. $P(x_i > 0) = F(z_i)$, where z_i is, again, the vector of explanatory variables), then estimates the truncated negative binomial for those trips that are undertaken, again using Equation 3 (Rose *et al.* 2006; Zuur *et al.* 2009). The hurdle model has been used in several previous studies to estimate a travel cost model in the presence of excess zeros (e.g. Bilgic and Florkowski 2007; Carpio *et al.* 2008; Anderson 2010), and has also been found to perform better than other approaches when the data are characterised by the simultaneous presence of both excess zeros and overdispersion (Gurmu and Trivedi 1996).

The consumer surplus associated with undertaking the recreational activity is the area under the demand curve, which can be shown to reduce to $-1/\beta_1$, where β_1 is the estimated parameter associated with travel cost variable. Further, the marginal value of an attribute in the model (e.g. the value of an additional trip characteristic) can be estimated by $-\beta_i/\beta_1$ where β_i is the coefficient relating to attribute i (i > 1) (Haab and McConnell 2002). This approach has also been termed the hedonic travel cost approach (Brown and Mendelsohn 1984), as it is a technique which reveals how much users are willing to pay for each individual characteristics of outdoor recreation sites based on their cost of access (much the same was as attribute values are determined in a hedonic model).

Implicit in the derivation of the above travel cost model is the assumption that individuals are travelling to a single destination to undertake a single activity. In many instances, individuals may undertake a trip for multiple purposes, with the activity of interest being just one of several activities undertaken. Several different approaches have been applied to separate out the different components of a multi-purpose trip, including allocating costs based on the proportion of total trip time spent at the destination (or activity) of interest (e.g. Farr *et al.* 2011) or a weighting based on the individual's perception of the importance of the activity/destination to their overall trip (e.g. Martínez-Espiñeira and Amoako-Tuffour 2009; Pascoe, Doshi *et al.* 2014). Others include a dummy variable to differentiate multi-purpose and single purpose trips, allowing a different estimate of consumer surplus for each trip type (Parsons and Wilson 1997; Loomis 2006).

A variation to the multi-purpose trip issue is that, in some cases, a destination may offer multiple recreational opportunities, the availability of which may affect the demand of individuals differently. Different individuals may undertake one or several of these. While weighting approaches and separate models could potentially be applied to each activity, several studies have treated these multiple activities in travel cost models through incorporating each activity as a dummy variable in the model, with respondents indicating which activities they participated in, and a single model estimated (e.g. Siderelis and Moore 1995; Englin and Moeltner 2004; Shrestha *et al.* 2007; Prayaga 2017).

Travel time is often considered as a component of the travel cost, as time spent travelling has an opportunity cost (i.e. something else could be done with that time if not travelling). However, there is still contention as how the opportunity cost of travel time should be measured if it is included in the model, or whether this should be included in the model at all. In some cases, assumptions as to the value of travel time have been made based on a proportion of the wage rate. However, there is significant variation in the literature as to what proportion of the wage rate, and which wage rate, should be used. For example, Rolfe and Dyack (2011) assumed that travel time cost was equal to one third of the Australian average wage rate, whereas Ezzy, Scarborough *et al.* (2012) used the average non-managerial full-time employee wage rate. Other studies suggest that the appropriate proportion of income should be based on an empirical assessment (McConnell and Strand 1981). Based on this, travel time was found to have zero cost by Pascoe (2019).

Choice experiments

General econometric specification

The econometrics underlying the estimation of choice models is fairly well established (Hensher *et al.* 2005). These models are commonly estimated using a random parameters (mixed) multinomial logit model (MMNL). Following Hensher and Greene (2003), a sampled individual, i, faces a choice between several alternatives, j. These alternatives have different levels of several attributes, A, which affects the level of utility, U, derived from each alternative, and a different cost, C, associated with realising these benefits. The sampled individuals also have different socio-economic characteristics, X, which may also affect their level of utility.

The relative utility of each alternative can be given by

$$U_{i,j} = \beta_0 + \beta_c C_j + \sum_a \beta_a A_{a,j} + \sum_x \beta_x X_{x,i} + \varepsilon_{i,j} \tag{3}$$

where $\varepsilon_{i,j}$ is a random error term that is assumed to be independent and identically distributed (IID). This latter assumption is restrictive as it does not allow for the error component of different alternatives to be correlated, which may be the case. To allow for this, the error function can be modified with the addition of a second random term, $\eta_{i,j}$. The distribution of this term varies over individuals and alternatives depends on the underlying parameters and observed data relating to alternative j and individual i (Hensher and Greene 2003). The original error term, $\varepsilon_{i,j}$, is again a random term with zero mean that is IID over alternatives and does not depend on underlying parameters or data (Hensher and Greene 2003). Utility in such a model is given by

$$U_{i,j} = \beta_0 + \beta_c C_j + \sum_a \beta_a A_{a,j} + \sum_x \beta_x X_{x,i} + \varepsilon_{i,j} + \eta_{i,j} \tag{4}$$

Given the set of observed choices, the marginal willingness to pay of an attribute, a, can be estimated from the coefficients of the MMNL regression,

$$WTP_a = -\frac{\beta_a}{\beta_c} \tag{5}$$

where the numerator is the coefficient of the attribute of interest and the denominator is the coefficient of the cost of the alternative. The negative sign is necessary given the expected value of the cost coefficient will be negative (Hensher, Rose *et al.* 2005).

Choice experiment design

The implementation of a discrete choice experiment requires comparing options with different levels of each asset being considered, along with different costs associated with that option.

An example of key attributes used in the choice experiment are presented in Table 12, based on the study of Pascoe, Doshi *et al.* (2019). The key assets selected were those most beach visitors would be familiar with (Maguire *et al.* 2011), and would most likely also occur in a coastal reserve.

Table 12. Coastal reserve attributes used for the choice experiment

Attribute	Attribute levels	
	Protection options	The “none” option
Cost ($/quarter)[a]	$10, $25, $40, $50	$0
Sand dunes and adjacent scrubland (Ha)	10, 20, 40, 50	0
Headland and rocky shoreline (Ha)	10, 20, 40, 50	0
Sandy Beach (Ha)	10, 20, 40, 50	0

a) All values are in 2016-17 prices.

The choice experiment requires a ‘realistic’ scenario to try and avoid hypothetical bias. In Pascoe, Doshi *et al.* (2019), the scenario presented to respondents was the creation of a new reserve to protect the key coastal habitats from future development, with the costs representing acquisition, rehabilitation and ongoing protection of differing areas of different combinations of the three habitats. No change in the direct use of the areas by the local residents was expected through the creation of the reserve (i.e. it was not designed to enhance recreational experiences). The payment mechanism (cost) was presented as an environmental levy to be included in quarterly rates indefinitely. For renters, it was assumed that this cost would be passed through in terms of higher rent. The range of cost values used in the choice sets were derived from a previous study of marine reserves in Australia (Burton *et al.* 2016).

Given the number of attributes and levels in Table 12 relating to the reserve options, there are 256 potential combinations (4^4) that each hypothetical reserve could present for the comparisons. If all possible combinations were to be compared, potential respondents would be presented with an unrealistically large number of comparisons. An appropriate experimental design can provide sufficient information to derive the model parameters without requiring all possible combinations to be presented. A number of different design options are available which aim to ensure sufficient variability across the alternative to identify each of the variables in the model and maintaining orthogonality (i.e. little or no correlation between the variables in the alternatives to be considered).

An alternative to orthogonal designs are efficient designs (Rose et al. 2009). Given the functional form of the final model, and prior estimates of the parameter values, the combinations required to minimise the potential standard errors of the estimated parameters in the model can be derived (efficient design). The D-efficiency of a model is measured through the value of the D-error – the determinant of the asymptotic variance-covariance matrix derived through Monte Carlo simulations based on prior estimates of the parameter values (Rose and Bliemer 2009) and the constraints imposed on the choice sets (e.g. number of blocks and choice sets within each block). The design is said to be D-optimal if it achieves the minimum D-error. However, convergence through simulation may take considerable resources and there is no guarantee that a D-optimal design can be found; hence a D-efficient design is one with a low D-error but not necessarily D-optimal (Rose and Bliemer 2013).

A D-efficient design can be derived using NGENE (Choice Metrics Pty Ltd 2012), a software package developed specifically to derive optimal choice sets for choice experiments. Naive expectations can be imposed by imposing parameter values of either plus or minus 1, reflecting the expected signs on the parameters (but naïve estimates of the parameters).

An example of one of the choice sets used in Pascoe, Doshi *et al.* (2019) is given in Figure 26. Individual respondents were asked to choose which option they preferred (if any) taking into account the characteristics of the area and its cost.

Figure 26. Example of a choice set used in the survey

		Area A	Area B	Area C	None
Sandy Beach (ha)		10	50	40	0
Dunes and adjacent scrubland (ha)		20	50	10	0
Headland and rocky shoreline (ha)		40	40	20	0
Cost ($)		10	50	10	0
Choice					

Photo credits: Sandy beach: CSIRO; Dunes and adjacent scrubland: Fiona Blandford, Birdlife Australia; Headland: Aussie Towns (Aussietowns.com.au)

Analytic Hierarchy Process (AHP)

For each set of comparisons, a matrix of scores can be developed, given by:

$$A = \begin{bmatrix} a_{11} & a_{12} & \dots & a_{1n} \\ a_{21} & a_{22} & \dots & a_{2n} \\ \dots & \dots & \dots & \dots \\ a_{n1} & a_{n2} & \dots & a_{nn} \end{bmatrix} \qquad (1)$$

There are two general approaches used for determining the weights; the original eigenvalue method (EM) developed by Saaty (1980)[3] and the Geometric Mean Method (GMM) developed by Crawford and Williams (1985). While the former approach has been employed in a wider range of coastal and resource management studies, the latter approach has been found to be less susceptible to influence from extreme preferences, as well as having better performance around other aspects of theoretical consistency (e.g. less susceptible to rank reversibility if the preference set changes, and greater transitivity properties) (Aguarón and Moreno-Jiménez 2000). The objectives rankings using the GMM are determined by

$$\omega_i = \frac{\left(\prod_{k=1}^{n} a_{i,k}\right)^{1/n}}{\sum_k \left(\prod_{k=1}^{n} a_{i,k}\right)^{1/n}} \qquad (2)$$

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

[3] Alternative approaches have also been used to derive the weights, with the row geometric method (Crawford and Williams 1985) gaining increasing interest (Aguarón and Moreno-Jiménez 2000).

need to be made, as only sub-components at the same level and within the same broader sub-component need to be compared.

A challenge facing the use of AHP is the propensity for respondents to be inconsistent in their responses (Bodin and Gass 2003; Baby 2013). In the case of perfect consistency, we would expect the ratio of the weights $A/B * B/C = A/C$ (Ishizaka and Lusti 2004). Such a condition may not hold due to several reasons, mostly relating to human error when undertaking multiple different bivariate comparisons (Castaño-Isaza, Newball *et al.* 2015). At an extreme level, this may result in a circular triad, where preferences for A>B>C>A. With the exception of circular triads, information on preference structures can still be obtained even in light of some level of inconsistency (e.g. A>B>C even if $A/B * B/C \neq A/C$). Several approaches have been undertaken to deal with inconsistency, including excluding highly inconsistent responses or adjusting responses through iterative approaches to ensure preference structures are maintained (Finan and Hurley 1997).

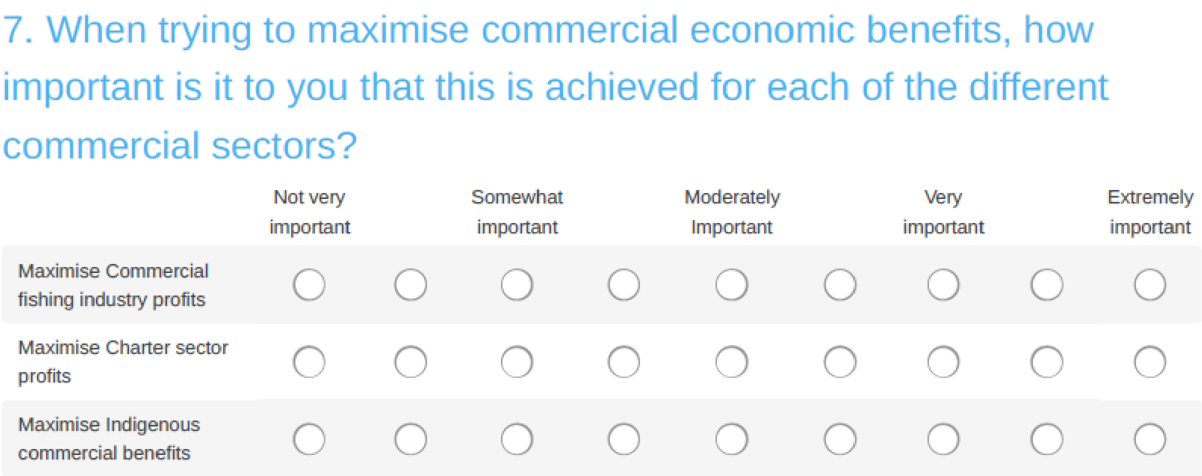
The preference weights derived through AHP represent values between 0 and 1, summing to 1. The preference scores reflect the value of the assets relative to the value of reference asset.

Modified AHP

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

The Modified AHP approach (Pascoe, Cannard *et al.* 2019) was developed to avoid some of these pitfalls by modifying the way in which the data are collected and analysed, taking into account the symmetry assumption underlying AHP. Instead of bivariate comparisons as in the standard AHP approach, respondents are 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 it allows five categories to be defined with mid-points between them, and is also consistent with the traditional AHP approach. An example of one of the questions is presented in Figure 27.

Figure 27. Example question in the objective importance survey


7. When trying to maximise commercial economic benefits, how important is it to you that this is achieved for each of the different commercial sectors?

	Not very important		Somewhat important		Moderately Important		Very important		Extremely important
Maximise Commercial fishing industry profits	○	○	○	○	○	○	○	○	○
Maximise Charter sector profits	○	○	○	○	○	○	○	○	○
Maximise Indigenous commercial benefits	○	○	○	○	○	○	○	○	○

Source: Pascoe, Cannard *et al.* (2019)

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

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

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

$$a_{i,j} = \begin{cases} a_i - a_j + 1 & if \ a_i - a_j > 0 \\ \dfrac{-1}{a_i - a_j - 1} & if \ a_i - a_j \le 0 \end{cases} \quad and \quad a_{j,i} = \frac{1}{a_{i,j}} \tag{2}$$

There are two general approaches used for determining the weights: the original eigenvalue method (EM) developed by Saaty (1980)[4] and the Geometric Mean Method (GMM) developed by Crawford and Williams (1985). While the former approach has been employed in a wider range of coastal and resource management studies, the latter has been found to be less susceptible to influence from extreme preferences, as well as having better performance around other aspects of theoretical consistency (e.g. less susceptible to rank reversibility if the preference set changes, and greater transitivity properties) (Aguarón and Moreno-Jiménez 2000).

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

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

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

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

[4] Alternative approaches have also been used to derive the weights, with the row geometric method (Crawford and Williams 1985) gaining increasing interest (Aguarón and Moreno-Jiménez 2000).

Appendix E: Using Non-Market Values in Cost-Benefit Analysis in Australian Fisheries and Aquaculture

<u>THIS APPENDIX IS ALSO AVAILABLE AS A STAND-ALONE DOCUMENT FROM THE FRDC WEBSITE.</u>

Using non-market values in cost-benefit analysis in Australian fisheries and aquaculture

2020

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Introduction

The marine environment produces a wide range of ecosystem services, many of which may be affected by fisheries and aquaculture management, but are not routinely considered in management decision making.

These may include cultural services such as recreational fishing, aesthetics and heritage values as well as supporting services for a range of marine species that are harvested commercially, either through wild caught fisheries or aquaculture, or are of conservation value to society *in situ* (e.g. seals, seabirds and dolphins).

There is increasing need for fisheries and aquaculture management to take account of the cumulative environmental, economic and social impacts of all human uses of marine resources. At an operational level, this shifts decision making beyond simple financial analysis, that accounts for the direct monetary effects of management decisions on industry, to consideration of an economic evaluation framework (e.g. cost-benefit analysis) that also captures the broader non-monetary impacts. Non-monetary impacts are captured by non-market values which place a dollar value ($AUD) on environmental and social costs and benefits that are also incurred as a result of fisheries and aquaculture management, but do not have an explicit market price.

This document is aimed at fisheries and aquaculture managers and policy-makers and provides an introduction to:

- the role of cost-benefit analysis and non-market values in fisheries and aquaculture management decision making;
- when non-market valuation needs to be undertaken and examples as to how they could be applied; and
- obtaining appropriate non-market values.

Realising the objectives of fisheries and aquaculture management

There has been a shift in focus within natural resources management, which is also reflected in fisheries and aquaculture management, from a purely commercial perspective to a broader social perspective reflecting the adoption of the principles of Ecologically Sustainable Development (ESD).

Managers of marine resources need to be cognisant of the costs and benefits associated with potential impacts of fishing and aquaculture activities on the wider ecosystem and community including (i) environmental and ecosystem impacts (for example, water and habitat quality, carbon footprint and visual amenity) and (ii) recreational, cultural, and social impacts. Furthermore, they have to balance the competing uses and users of marine resources across both current and future generations. This can be addressed by ensuring that fisheries and aquaculture management decisions account for all benefits and costs associated with the utilisation of marine resources, thereby ensuring that total benefits to society are maximised.

THE ROLE OF COST-BENEFIT ANALYSIS

Cost-benefit analysis (CBA) is an economic evaluation framework that quantitatively captures the broader costs and benefits of using ecosystem services and uses a standardised measure or numeraire (e.g. $AUD). This allows direct comparison between different management options.

CBA can be thought of as an extension of a financial analysis. A financial analysis only accounts for the direct dollar effects of management decisions on an industry, where market prices are the basis for valuing the policy and its impacts. For example, the impact of a regulation change on the aquaculture industry may reduce a farm's output. The financial impact would be measured by the loss in farm profit (as determined by the market price of the farm's output and cost of inputs used). CBA extends this analysis through the inclusion of the non-market costs and benefits on the wider community that also results from the regulation change. The wider impacts would need to be explicitly identified and valued as there would be no explicit dollar value for them (e.g. improved recreational experience of beach users). By including all relevant costs and benefits CBA provides an absolute measure of the dollar value of the total economic and social impact of a policy outcome.

CBA is regularly used to choose between policy alternatives in natural resource management, health, education and transport. In fisheries and aquaculture decision making its use is less systematic, although it is mandated in certain States. The use of CBA to inform decisions about the management of marine resources aligns to the broader social perspective reflecting the principles of Ecologically Sustainable Development.

WHAT IS TOTAL ECONOMIC VALUE?

Using CBA, management decisions are informed by the concept of total economic value. Total economic value captures:

- use values (e.g. commercially and recreationally harvested fish);
- non-use values (e.g. continued existence of an endangered species); and
- option values (e.g. the benefits of not using a resource now, but maintaining the option to use it in the future)

Some use values are captured through market prices, for example commercial fishing and aquaculture. But other use values, such as the fish caught by a recreational fisher, do not have an explicit market price. These values, as well as non-use values, generally have to be estimated. Option values also exist within the marine environment. These are the benefits of not using a resource now, but maintaining the option to use it in the future. For example, currently underutilised and discarded species may have an option value if a more valuable use for that species can be derived in the future. Biodiversity also has an option value, as future uses for different species may emerge over time.

What are non-market values?

CBA and the non-market values that are used in CBA have their foundations in economics.

It is well understood that market prices can be used to determine the economic gain or loss of a change. However, many ecosystem services provided by marine resources are not bought and sold in markets, but generate satisfaction or utility to individuals in the broader community. Examples include:

- recreational fishing (the experience);
- catch of an additional recreational fish;
- visual amenity of coastal areas; and
- continued existence of a species or habitat.

Non-market values capture the satisfaction or utility related to the use of these services and represent this as a monetary equivalent.

> Including both market and non-market values ensures that all relevant costs and benefits are included in the decision making process

Inclusion of non-market values in CBA effectively provides a "voice", expressed as a monetary value, to impacts (i.e. the changes in the level of these services) that might otherwise be ignored.

A relevant non-market cost or benefit that is not explicitly valued still effectively assumes a "value" in decision making. This may equate to zero if the cost or benefit is ignored, or be exaggerated through the influence of interest groups in the political process beyond its non-market value.

Examples of the role of non-market values in fisheries and aquaculture management

Marine resources provide a stream of goods and services, some of which are directly beneficial to humans. The demand for these goods and services is characterised by competing and often mutually exclusive uses by a diverse range of users.

For example, the marine environment provides supporting services for a wide range of species, affecting biodiversity. Some of these species may be incidentally caught and discarded by commercial fishers and/or recreational fishers, impacting their abundance and indirectly the species that are connected to them through the ecosystem. Intensive aquaculture developments in an area may impact biodiversity through, for example, habitat destruction and water pollution [1].

Other recreational uses (such as scuba diving) may also benefit from increased biodiversity, the benefits of which will also be affected by fishing and aquaculture.

Protecting biodiversity may mean restricting commercial and recreational fishing and aquaculture activity. Determining the balance of these activities requires an understanding of the non-market values of the change in biodiversity as well as the market values generated through their use.

> Ignoring non-market values in the management of marine resources can result in inefficient resource allocation and leave society worse off

Examples of the types of fisheries and aquaculture management decisions and the non-market values that could help inform these decisions are summarised in Table 1. Three typical management scenarios are then examined in more detail.

Table 1. Typical Fisheries and Aquaculture Management Decisions and Non-market Values

		Allocation	Spatial	Seasonal	Catch/ production
		Who	Where	When	How much
Types of NMV that would be required	**FISHERIES**				
	An individual fish to a recreational fisher	✓			✓
	Fish to indigenous communities	✓	✓		✓
	Recreational fishing experience		✓		✓
	Bycatch and discarding		✓	✓	✓
	Habitats affected by fishing		✓		✓
	TEP species bycatch (e.g. dolphins)		✓	✓	✓
	AQUACULTURE				
	Amenity values (e.g. aesthetic value)		✓		✓
	Changes in water quality		✓		✓
	Habitats affected by aquaculture		✓		✓

AQUACULTURE PRODUCTION AND IMPROVED BIOFOULING MANAGEMENT

Sea-based aquaculture sometimes produces effluent that enters the marine environment directly affecting water quality, and potentially leading to adverse environmental impacts such as eutrophication and biodiversity loss. Management of effluent, such as the use of biofouling coatings and physical removal of biofouling from the nets, imposes costs on the industry. However, failure to undertake biofouling management incurs costs on the broader community through the loss of non-market value associated with reduced water quality (and the associated impacts on biodiversity).

While there may be some benefits to the industry through improved fish health, most of the benefits associated with effective biofouling management accrue to the broader community through the reduced negative impacts.

Determining the appropriate amount (or type) of biofouling management requires an understanding of the monetary costs to the industry as well as the non-monetary costs to the broader community of not undertaking management.

ALLOCATION DECISIONS AND THE VALUE OF A RECREATIONAL FISH

The need to develop specific allocations of fisheries resources between commercial and recreational fishers is increasingly becoming an issue in many Australian fisheries.

From a societal perspective, any policy change that involves a reallocation of marine resources across competing uses should achieve an improvement in total net benefits to society. Following this argument, the question then becomes: will a reallocation generate additional benefits to the recreational sector that are larger than the additional costs to the commercial sector?

In order to do this, fisheries managers need to know the value of an additional fish caught by a commercial fisher and a recreational fisher. The former can be derived through market-based information, the latter requires non-market value estimations.

Without the non-market value of recreational fish catch, the allocation that provides the most net benefits to society cannot be determined. Further details on the optimal allocation process can be found in Edwards [2] and Plummer, Morrison and Steiner [3].

SPATIAL AND GEAR OPTIONS TO BALANCE ENVIRONMENTAL AND ECONOMIC OUTCOMES

A hypothetical scenario is presented where managers are confronted with a problem of dolphin bycatch. The current management results in six dolphin deaths a year. In this example, we assume that there are only two alternative management options that could be applied to reduce this problem. Option 1 involves a series of area closures that modelling suggests would reduce the bycatch of dolphins from six to three, but would also result in the bycatch of an additional three turtles. The cost to the industry of Option 1 is zero. Option 2 involves the use of area closures and new bycatch reduction technology which reduces bycatch of all species to zero, but results in an additional $1m a year in costs to the industry (Table 2).

Table 2. Cost of Dolphin Bycatch Management Options

			Options	
		Current management	1	2
Bycatch	Dolphins	6	3	0
	Turtles	0	3	0
	Costs	**0**	**0**	**$1m**

The key questions facing managers are: is it better to catch three turtles and save three dolphins; or is the benefit of saving all of these species to society (i.e. reduce bycatch to zero) worth more than the $1m additional cost imposed on the industry? From a societal perspective, if the value of saving six dolphins was more than the $1m additional cost to the industry, then Option 2 would be preferable to current management. Similarly, if the value of a turtle was less than the value of a dolphin (irrespective of the value of a dolphin), then Option 1 would also be preferable to current management. It may also be the case that current management is the most economically efficient outcome, depending on the relative and absolute values of the different species.

Without non-market values for the dolphins and turtles, the option that provides the most net benefits to society cannot be determined.

> Without non-market values, the option that provides the most net benefits to society cannot be determined

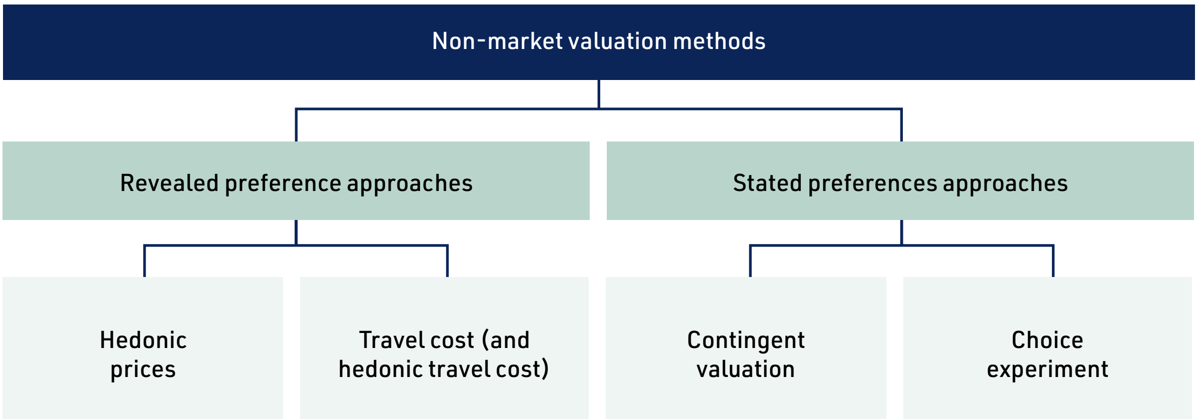
Where do non-market values come from?

Non-market values are estimated using data that have been collected for this purpose.

Estimation of non-market values requires specialist skill and it is not expected that fisheries and aquaculture managers would derive these values themselves, but instead team up with an economic research group with the appropriate skills. However, a broad understanding of the types of estimation processes is useful for managers as different approaches have different pros and cons.

Non-market valuation methods are broadly classified into revealed preference and stated preference methods (Table 3).

Table 3. Non-market Valuation Methods



REVEALED PREFERENCE METHODS

Revealed preference methods are based on observed behaviour and contain an element of market value in the derivation of the non-market values. They are restricted to the estimation of use values. These approaches include hedonic pricing and travel cost approaches.

STATED PREFERENCE METHODS

Stated preference methods estimate willingness to pay for hypothetical changes in resource allocations presented in a survey in the form of alternative management options. The most prominent stated preference methods are contingent valuation and discrete choice experiments.

> Estimation of non-market values requires primary data collection and a specific set of technical skills to derive the values. Best practice in non-market valuation does not come cheap

Benefit transfer

Benefit transfer involves the use of existing non-market values and adjusting these values as necessary for use in fisheries and aquaculture decision making.

Whilst benefit transfer sounds intuitive and simple, it:

- requires the existence of technically competent source studies (also referred to as primary studies) that have estimated non-market values for relevant benefits/costs in contexts where biophysical conditions, type and scale of the change and population characteristics are similar; and
- involves rigorous statistical techniques to adjust for any differences between the settings in which the values were originally estimated and the one in which they are going to be used.

Benefit transfer are generally evaluated in terms of validity and reliability.

Validity can be largely viewed as a measurement error problem resulting from systematic errors in the underlying source study estimates.

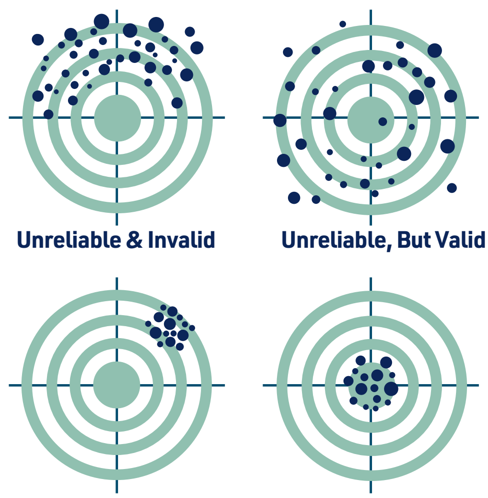
Reliability occurs when the generalization error is small (transfer is accurate).

The effects of unreliable and invalid estimation of the non-market value using benefit transfer is illustrated in Figure 1.

In practice, most cases of benefit transfer involve variations between sites, populations, context and scale. Combined with the possibility of unrepresentative or unreliable source studies, the potential for invalid and unreliable benefit transfers can be large.

Benefit transfer does have to be approached with care. There are a number of common pit-falls with using values from different studies. As with the estimation of non-market values, benefit transfer requires specialist expertise, and again managers should team up with an economics research group with the appropriate skills to do this.

Figure 1. Reliability and Validity



Adapted from Diman [4]

> Best practise in benefit transfer can provide a cost-effective approach to deriving non-market values without requiring direct value estimation

WHERE CAN WE FIND VALUES FOR BENEFIT TRANSFER?

There are a number of free to access online databases in the area of environmental valuation. A key objective of these databases is to broaden access to valuation studies and facilitate the use of existing non-market values in decision making through benefit transfer. These are international databases that include some Australian non-market valuation studies, but these are generally limited in number. These databases are constantly evolving, and will include future Australian case studies as they are undertaken.

The Environmental Valuation Reference Inventory (EVRI) is the most comprehensive searchable online database of environmental valuation studies developed by Environment and Climate Change Canada in collaboration with international valuation experts and organisations. EVRI is now the largest database of its kind. While broader than marine based values it contains relatively up to date studies from Australia and elsewhere.

How to know when to estimate non-market values?

The reality is that estimation of non-market values is not costless. Benefit transfer offers a less expensive alternative to estimating non-market values but is not always feasible.

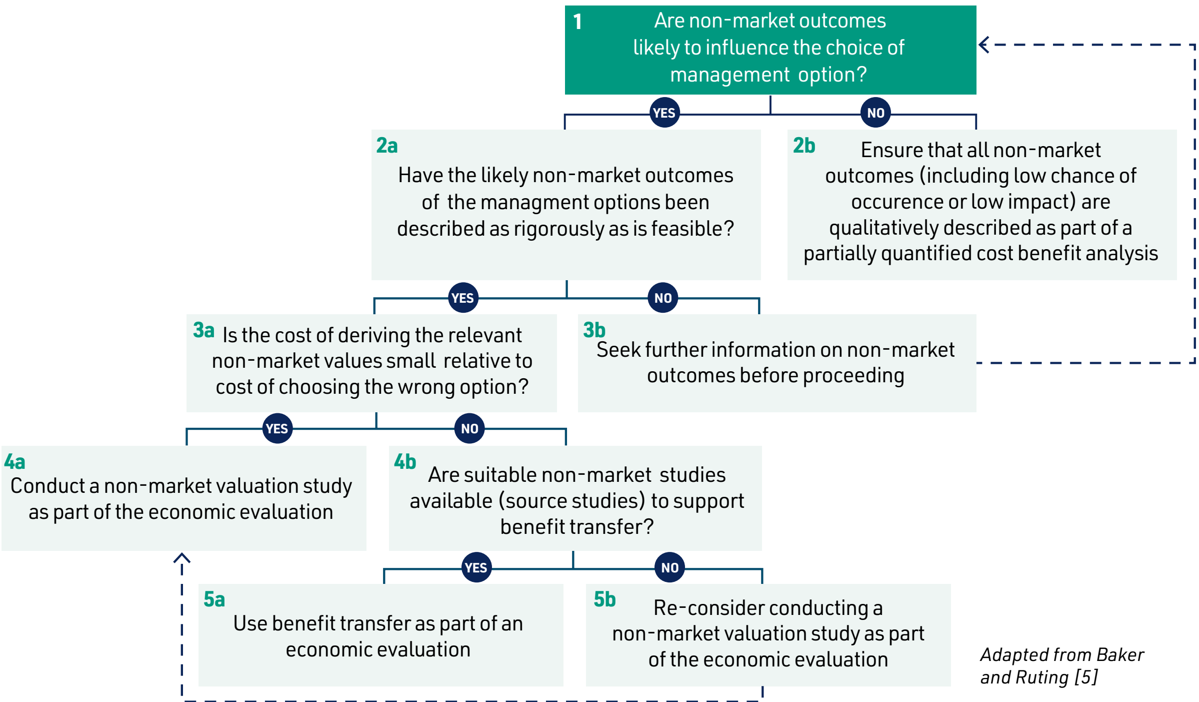
The flow chart (Figure 2) provides guidance with respect to whether non-market values would be appropriate for the management options under consideration, and if so, whether to estimate primary values or use benefit transfer.

To illustrate how the flow chart works, we will re-examine the earlier example where managers of a fishery were confronted with a problem of dolphin bycatch.

To recap, the current management strategy results in six dolphin deaths a year. There were two alternative management options to reduce this problem. Option 1 involved a series of area closures that modelling suggests would reduce the bycatch of dolphins from six to three, but would also result in the bycatch of an additional three turtles. Option 2 involved the use of spatial closures and new bycatch reduction technology which reduces bycatch of all species to zero, but adds an additional $1m a year in costs to the industry.

Starting at Box 1, it has been determined that non-market values for averting dolphin and turtle bycatch are likely to influence the choice of management options and the likely non-market outcomes of management options have all been described as rigorously as is feasible (Box 2a). The cost of deriving non-market values for dolphins and turtles is not known without going out to tender (Box 3a). Therefore, the decision is taken to research if there are suitable non-market studies available to support benefit transfer (Box 4b). A review finds no suitable studies to support benefit transfer. The decision is now to reconsider conducting a primary study to estimate appropriate non-market values for dolphin and turtle bycatch (Box 5b).

Figure 2. Estimate or Borrow Non-market Values?



Adapted from Baker and Ruting [5]

IS NON-MARKET VALUATION ALWAYS REQUIRED IN FISHERIES AND AQUACULTURE DECISION MAKING THAT IMPACTS MARINE ECOSYTEMS?

When decision making impacts marine ecosystems both market (financial values) and non-market values need to be explicitly considered. However, this does not necessarily always require non-market valuation to be undertaken.

Non-market values are required when the impacts affect the policy outcome. In some cases, there may be alternatives that remove the need to consider non-market values. For example, an aquaculture company plans to establish an aquaculture farm in a recreational area popular with locals.

Let's assume an environmental impact analysis suggests that once in operation, the farm would reduce the water quality in the estuary and the nearby beaches. Additionally, the farm operation would produce significant noise levels. Both, the reduced water quality and the noise would diminish the recreational experience from visiting the estuary and beaches. The farm should only be established if the expected commercial net benefits generated through the farm outweigh the non-market costs associated with the diminished recreational opportunities (a social cost). However, an alternative site may maintain the commercial benefits of the farm without any significant impact on recreational experience. In this case, further viable site options should be rigorously investigated prior to conducting any further analysis.

This example highlights the importance of considering all feasible options prior to undertaking non-market valuation.

Other approaches compared to cost-benefit analysis

Expenditure studies, well-being studies, triple bottom line approaches (TBL), multi-criteria decision analysis (MCDA) and CBA are all approaches used in the management of fisheries and aquaculture to capture the broader societal perspective of Ecologically Sustainable Development.

Prima facie there is some degree of overlap between each of these approaches. However, each approach informs a different element of fisheries and aquaculture management objectives (Table 4) and should be considered complementary rather than substitutable in the decision making process.

Table 4. How Fisheries and Aquaculture Management Objectives are Informed

	Expenditure Study	Well-being study	TBL	MCDA	CBA
Contribution to regional income & employment	✓	✓			
Social outcomes and impacts to regional communities & industry		✓	✓		
Articulation of economic, social & environmental impacts			✓	✓	✓
Assesses explicit tradeoff between policy alternatives				✓	✓
Ranks policy alternatives				✓	✓
Absolute measure of policy outcomes in $					✓

Recreational fishing expenditure surveys, commercial fishing expenditure studies and well-being studies to date have focused on identifying and measuring the impacts (both direct and indirect) of utilising marine ecosystem services and the contribution of the different sectors to local communities.

The triple bottom line (TBL) approach is the general framework used to assess policy outcomes against economic, social, and environmental dimensions. TBL requires articulation of impacts in each of these dimensions, which are then individually assessed. However, optimal decisions require the trade-off between these dimensions to be considered. Approaches that explicitly assess trade-offs between different components of the triple bottom line fall into two "camps" – multi-criteria decision analysis (MCDA) and CBA.

MCDA supports decision making through assessing stakeholder preferences for different outcomes of management options and provides an explicit and relative ranking of different policy outcomes. With MCDA, many impacts are assessed semi-quantitatively through expert opinion, allowing measures that are often not commensurate to be combined in a common framework (e.g. social impacts as well as economic and environmental). Whilst MCDA can identify which option is most preferred, it does not identify if the benefits exceed the cost of the preferred option.

In contrast, CBA when conducted appropriately, captures all use and non-use values, ensures transparency as to the costs and benefits of different management options and provides a measure of the total economic value of the use of ecosystem services. From this, the optimal allocation of the marine resource for the whole of society can be determined.

Summary

The use of non-market valuation is not unique to fisheries and aquaculture management decision making and is used to inform decision making across a range of policy areas in natural resource management including water resources, forestry, national park management and coastal management.

To date, non-market valuation has not been used widely in management decision making and policy analysis in Australian fisheries and aquaculture. Whilst some attention has been focused on the estimation of non-market values of recreational fishing, only limited attempts to date have been made to use these values in supporting management decision making. Many other key non-market values have not been quantified, and their use in fisheries and aquaculture management has not been fully explored.

Estimation of non-market values can require additional primary research. When done poorly, this can undermine the scientific rigour of decisions and reporting, weakening trust in practitioners and users of non-market values and in decision making.

In some cases, existing values can be used but this requires caution as poor benefit transfer practice can lead to inappropriate decisions being made. In addition to issues of identifying relevant source studies that are methodologically rigorous, the use of older studies can also be problematic. Non-market values are likely to change over time and should be used with caution.

Further details on all of the issues discussed above can be found in the full project report [6].

Other resources on the use of non-market valuation

For those seeking to extend their knowledge in this area, the following resources, whilst not necessarily specific to fisheries and aquaculture management in Australia, are a good place to start.

Baker, R., & Ruting, B. (2014). *Environmental policy analysis: A guide to non-market valuation.* Productivity Commission Staff Working Paper. Canberra, Australia.

Clough, P., & Bealing, M. (2018). *What's the use of non-use values? Non-Use Values and the Investment Statement.* NZ Institute of Economic Research. Wellington, NZ.

National Ocean Economics Program (NOEP). (2012). *Environmental and recreational (non-market) values.* Center for the Blue Economy, Middlebury Institute of International Studies at Monterey.

Acknowledgements

We would like to thank the anonymous reviewers whose comments and suggestions have helped improve this document.

Graphic design by Tracy at smalldesign.com

This document is an output of the project Non-market values to inform decision-making and reporting in fisheries and aquaculture – an audit and gap analysis (2018-068) which is supported by the Fisheries Research and Development Corporation (FRDC) on behalf of the Australian Government.

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Using non-market values in cost-benefit analysis in Australian fisheries and aquaculture

ISBN 978-1-925553-29-1

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

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