ICES Journal of Marine Science, 2026, Vol. 83, Issue 5, fsag077
https://doi.org/10.1093/icesjms/fsag077
Review Article

A guide to navigating the complexities of quantitative trade-off analysis

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Abstract

Trade-offs are fundamental in natural resource management, being either implicit or explicit in every decision relating to human activity undertaken. Managing for sustainable use of resources typically involves the need to account for three interconnected dimensions: the state of ecosystems, economic productivity, and social infrastructure. Alternative management strategies often result in different combinations of outcomes in each of these three dimensions and hence are likely to involve trade-offs which must be evaluated to determine the most appropriate overall strategy from a broad societal perspective. If the different dimensions are considered separately, or if certain dimensions are neglected in assessing the outcomes of different courses of action, there is a risk that at least some of the sustainable development objectives may not be achieved. Conceptual frameworks have been developed to address the integrated evaluation of ecological, economic, and social/cultural aspects of ecosystem-based management, and a diverse set of quantitative methods have been proposed to support their implementation, across a range of disciplinary fields. The aim of this paper is to propose a decision map to help researchers and decision-makers choose appropriate methods according to the context of the assessment. We identify the different methods that can be used, starting from standard cost-benefit analysis, and expanding to alternative approaches, and provide illustrations of how the methods can be implemented, highlighting their advantages and limitations.

Keywords trade-off analysis, marine social-ecological systems, decision support, cost-benefit analysis, multi-criteria analysis, coviability analysis, goal programming

Introduction

The concept of sustainable development has been adopted by many nations as a guiding principle for natural resource and conservation management (United Nations 2015). However, the actual implementation of sustainable development policies is not without controversy, as balancing the protection of ecosystems with improving economic performance and social well-being presents many practical challenges (Link et al. 2017). In addition, voices are being raised calling for a reassessment of the very

Received: 10 September 2025. **Revised:** 12 February 2026. **Accepted:** 20 April 2026

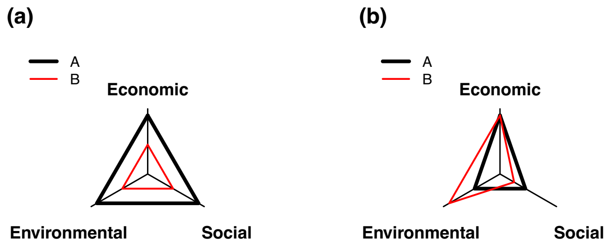

Figure 1 Comparing alternative courses of action integrating environmental, economic, and social dimensions. (a) Scenario where one option (A) dominates the other (B) across all dimensions; (b) scenario where no option dominates across all dimensions.

concept of sustainability (Priddat and Schlaudt 2025). Nonetheless, sustainable development remains a framework that aims to enhance three interconnected components, namely natural ecosystems, economic growth, and social infrastructure, often collectively known as the triple bottom line (TBL; Elkington 1994). In addition, the importance of the institutional dimensions of sustainable development has also been increasingly emphasized (Stephenson et al. 2017, 2018), as they provide the frameworks and mechanisms required for effective governance, policy implementation, and stakeholder collaboration.

Trade-offs are fundamental in natural resource management, being either implicit or explicit in every human activity undertaken. Alternative management strategies often result in different combinations of outcomes for each of the three TBL components and hence involve trade-offs which must be evaluated to determine the most appropriate overall strategy from the broader societal perspective. For example, in Fig. 1a, option A clearly dominates option B in all aspects so choosing between them is straightforward. However, in Fig. 1b, no single option dominates outcomes in all three domains, so some form of overall assessment that captures the relative importance of the different domains is required to determine which option managers could choose according to their preferences and those of the impacted stakeholders.

Historically, the three dimensions of trade-off analysis have mainly been assessed in isolation. This is unsurprising, given the highly segmented structure of knowledge production, and the high level of skills required for such assessments. For example, environmental impact assessment has a long-standing tradition in assessing development proposals under the assumption that economic outcomes are the concern of the developer rather than the decision maker. More recently, social impact assessment has been included to assess the impacts of different alternatives of natural resource management (Murphy et al. 2022, Dawson and Suich 2025). Economic analyses have also been commonly undertaken, in some cases including environmental costs but often ignoring social issues such as distributional outcomes of alternative proposals.

As illustrated in Fig. 2, if the different dimensions are considered separately, if certain dimensions are neglected in assessments, and/or if methods for integrated assessments highlighting inherent trade-offs are not identified, it is very likely that some of the sustainable development objectives will not be achieved. In particular, overlooking important social, economic, or institutional issues may lead to low acceptability of management measures and low compliance (Jagers et al. 2012). In addition, any assessment relies on at least implicit assumptions about the relative importance of these dimensions, and about how such dimensions are linked. Hence, in any evaluation attempting to integrate multiple dimensions, there is a need to explicitly define the processes by which these dimensions are being combined or aggregated (ICES 2023). This applies equally to the assessment of trade-offs across each of the three components of the TBL (see e.g. Thébaud et al. 2014).

In the context of sustainable development, there is thus a need for approaches that look at trade-offs in an integrated manner; i.e. that cover the full values of ecological, economic, social/cultural, and institutional aspects. While a growing number of studies have proposed conceptual frameworks for integrated assessments (see e.g. Stephenson et al. 2017, 2018 or Kempf et al. 2016 with reference to fisheries policy and management), there is a need to review the different quantitative methods available to carry out such analyses across a range of disciplinary fields. This is because the question of which method to use is far from trivial, and there is no single best method available. Depending on the context, the data available and the objectives, different methods will be more appropriate. The question is particularly relevant in the marine domain, as was highlighted by the ICES working group on economics (Thébaud et al. 2023).

The aim of this paper is to identify the different quantitative methods that can be used for trade-off analysis for the assessment of alternative policies applying to marine social–ecological systems, starting from the traditional perspective of cost-benefit analysis (CBA) and expanding to discuss alternative approaches that can be used, and to highlight their advantages and limitations. We propose a decision map to help researchers and decision-makers choose appropriate methods according to the context of their assessments. We use a stylized example to illustrate the different situations the researcher may be in, and the approach to selecting the methods. We complement this with a set of references to real-life case studies, for the reader to refer to for illustration of the practical implementation of the method discussed. In addition, within the discussion of each approach we provide citations to detailed manuals that analysts can use to develop trade-off analyses tailored to the particular use case at hand.

A decision-map for trade-off analysis

An initial step in integrated assessment is to identify the objectives, and underlying values, associated with the natural resource under management. These objectives should identify the resource dimensions recognized by stakeholders as necessary to assess the outcomes of alternative management strategies and scenarios. Once these are identified, a diverse toolkit is available to perform a trade-off analysis. In some cases, a single method of analysis will suffice, while others will require mixed approaches, possibly including a combination of quantitative and deliberative approaches. The choice of which approach to apply will depend on the structure of the management objectives, the data available, the characteristics and knowledge of the decision makers, the capacity for the different methods to assess performance against the objectives, etc.

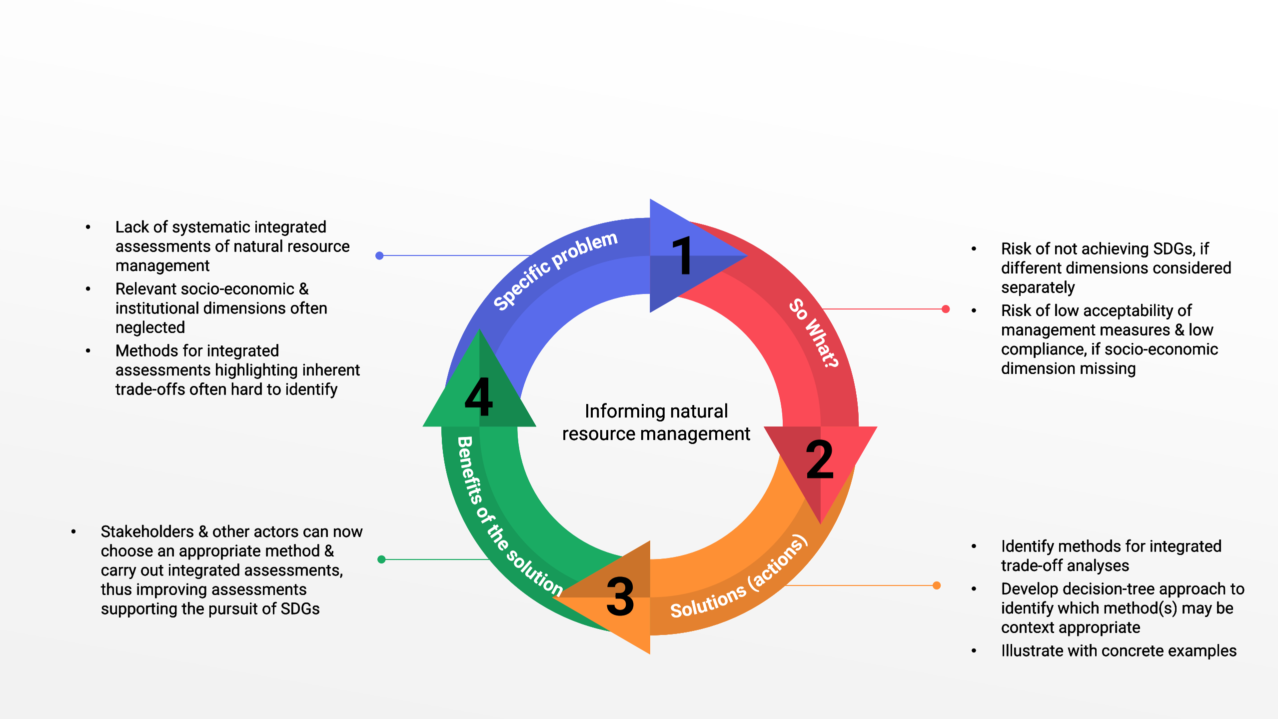

Figure 2 Why quantitative trade-off analysis is required to inform natural resources management in pursuit of the sustainable development goals (SDGs).

Existing approaches for assessing trade-offs range from multiple indicator analyses based on qualitative information (referred to hereafter as the traffic light approach for convenience) to quantitative measures aimed at integrating all impacts into a single value. In particular, CBA converts all impacts into economic values (expressed in monetary terms) that may be added to determine the option that produces the greatest net-benefits (see Measuring welfare-outcomes through Cost-Benefit Analysis - section 2.2) Between these extremes are several semi-quantitative multi-criteria analysis (MCA) approaches that aim to capture the values and preferences of stakeholders in different ways (see e.g. Mendoza & Martins 2006). These include deliberative multi-criteria evaluation—a participatory approach aimed at deriving a consensus amongst multiple stakeholders; MCA that derives preferences from stakeholders and used to weight outcome measures to rank options; and cost-effectiveness analysis that combines a monetary measure of costs with a non-monetary and potentially multi-criteria measure of outcomes.

There is substantial debate between proponents of the different applied approaches about their relative merits. Some economists (e.g. Dobes & Bennett 2009) argue that only CBA is appropriate, as it takes the perspective of society as a whole, whereas multi-criteria approaches generally rely on groups of self-selected stakeholders with potentially vested interests, and hence may introduce bias into the analysis. Others (Bebbington et al. 2007, Spangenberg & Settele 2010) argue that the use of economic values for environmental assets is questionable as the assumptions underlying the valuation are often unrealistic and the derived values themselves are often method-dependent, with alternative approaches providing widely varying values. Global initiatives such as The Economics of Ecosystems and Biodiversity (TEEB, https://www.unep.org/topics/teeb) have aimed to reduce this distrust, at least with respect to the valuation of ecosystem services. However, appropriate methods for quantifying and capturing many social and cultural impacts in economic value terms remain elusive (Daniel et al. 2012).

The choice of which resource management plan is best is often not the prerogative of a single decision-maker. In fact, there is an increasing trend for co-management in natural resource and conservation management (Armitage et al. 2009). This is because, in many cases, a large number of individuals and groups interact with the natural resources, and policy needs to influence the behaviour of these groups if environmental, economic, and social objectives are to be achieved. Stakeholders generally also want to have their views recognized explicitly in some form, often preferring more participatory approaches to decision making, which can improve the relevance, perceived legitimacy, and enforceability of collective decisions being made (e.g. Bellanger et al. 2020). Co-management brings with it a broader range of knowledge at different scales, with the aim to develop better management plans that are more likely to be supported by industry participants and/or society (Berkes 2009). However, while a strength of co-management lies in this broad range of expertise, some stakeholders may have limited understanding of key indicators, or potentially deeply rooted distrust of certain assessment methods. In addition, co-management approaches may present problems such as rent-seeking, in which stakeholders look to manipulate the management system for their own benefit (Jones and Seara 2020, Bellanger et al. 2021).

Whatever the context for deciding on the methods that will be used to inform trade-off decisions, transparency in the process by which such decisions are made, as well as with respect to the ways in which they are implemented, are likely key to the perceived legitimacy and relevance of the information produced. We propose

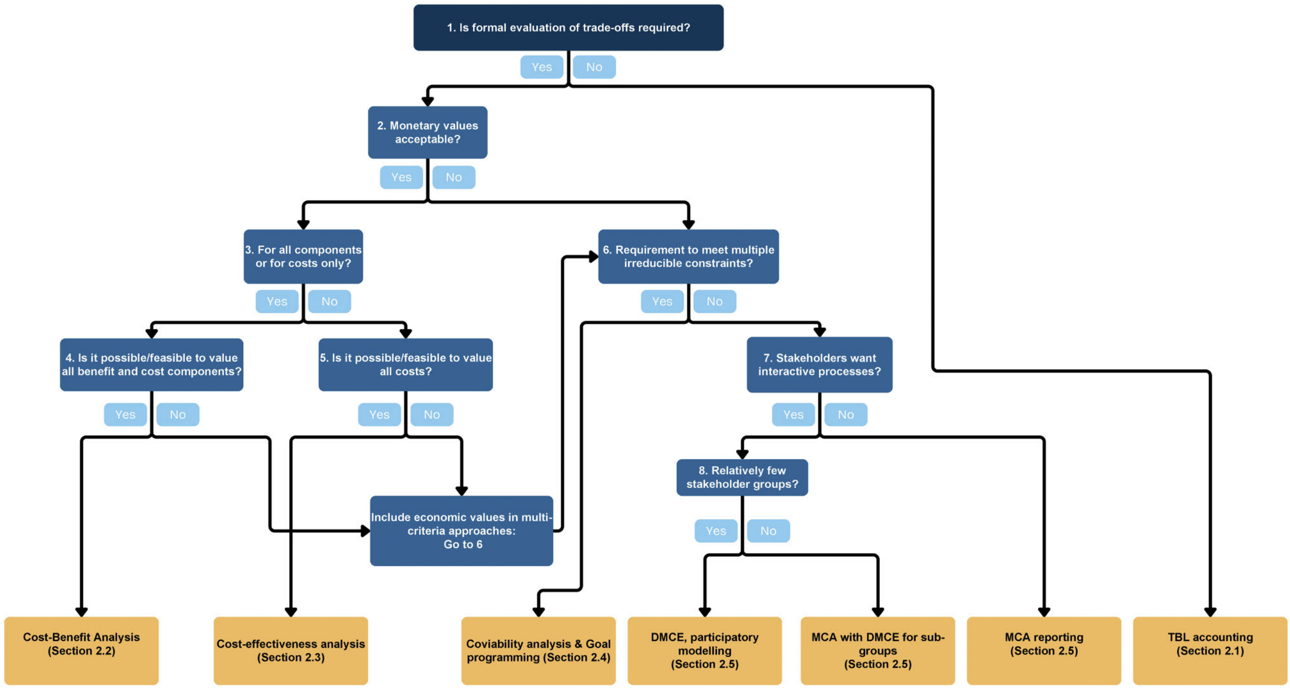

Figure 3 Mapping out the questions associated with choosing the most appropriate method. MCA stands for multi-criteria analysis, DMCE for deliberative multi-criteria evaluation, and TBL for triple bottom line (references in the last row denote the section in the main text in which the node is discussed).

a decision-map to identify appropriate assessment options given the context of the assessment and the views/objectives of the decision makers as well as identify issues which can affect the choice of method. Starting from economic welfare assessment as a reference approach to trade-off assessments, where monetary values are considered acceptable and are available, we explore the alternative assessment options which may be relevant under alternative circumstances. We take the view that each approach is potentially valid, but some approaches may be found to be more appropriate or more acceptable under certain circumstances. All methods include, at least initially, separate measures for each of the three main assessment components (social, economic, and ecological). Where they mostly differ is in the way in which they are measured, as well as how, or if, they are aggregated to capture the net results given trade-offs among the three dimensions.

The key steps we identify in carrying out a trade-off analysis involve answering the following eight questions (Fig. 3):

1. Is a formal evaluation of trade-offs required?
2. Are monetary values acceptable?
3. Are monetary values acceptable for all components or for costs only?
4. Is it possible/feasible to value all components?
5. Is it possible/feasible to value all costs?
6. Is there a requirement to meet multiple, irreducible constraints?
7. Do stakeholders want an interactive process?
8. Are there relatively few stakeholder groups?

The first question to ask (step 1 in Fig. 3) is whether a formal, quantitative assessment of trade-offs is required to make the decision. In some cases, trading off one outcome directly against another explicitly within the analysis is not considered appropriate or useful, and decision makers may prefer to make these trade-offs subjectively rather than be guided by objective weightings. This may be due to lack of trust in a single aggregated metric (e.g. due to lack of transparency of the aggregation procedure), or to the lack of agreement as to how trade-offs should be represented. It may also result from legal requirements forbidding certain courses of action, independently of their outcomes.

When a formal trade-off analysis is required, if monetary values are acceptable (response is 'yes' at step 2 in Fig. 3), the approach will depend on the scope that monetary valuation can encompass. Monetary values may be deemed unacceptable, however, on legal or ethical grounds (e.g. as when attempting to place a monetary value on endangered species or on resources of cultural and spiritual significance to indigenous groups). Where monetary values are considered acceptable for all components (response is 'yes' at step 3 in Fig. 3), the approach will depend on whether it is possible to derive reliable monetary values for all costs and benefits. If this is possible (response is 'yes' at step 4 in Fig. 3), then CBA will be the preferred, gold standard, approach in economics (Coglan et al. 2021).

When it is not possible to derive monetary values for all costs and benefits (response is 'no' at step 4 in Fig. 3), then a mixed approach incorporating available monetary values into an MCA framework may be appropriate (step 6 in Fig. 3 and section 3.5).

The actual use of economic values in natural resource management is in fact often limited in practice (Marre et al. 2016). This may be due to a lack of trust in these values (Driml 1997, Laurans et al. 2013). It may also result from the difficulty in establishing these values empirically (Pendleton et al. 2016) and from the fact that different types of values (e.g. use value and existence value) are not viewed as equally legitimate by decision-makers. In some situations, it may be legally prohibited to use monetary values in assessing trade-offs associated with natural management options. For example, in Committee for Humane Legislation, Inc. v. Richardson (540 F.2d 1141 D.C. Cir. 1976) the Washington DC Court of Appeals ruled that costs could not be considered in regard to the prohibition on killing porpoises in the tuna purse-seine fishery.

In many cases, monetary values will actually be considered acceptable or reliable for costs but not benefits (response is 'no' at step 3 in Fig. 3). Cost information may be readily available (response is 'yes' at step 5 in Fig. 3) while monetary values for outcomes may be uncertain or too costly to estimate.

If monetary values are generally not considered acceptable as a basis to assess trade-offs (response is 'no' at step 2 in Fig. 3), the assessment approach will lean towards the use of MCA methods. These techniques and tools, varying in complexity, are specifically designed to explicitly account for multiple objectives and criteria (or attributes) in decision-making problems (Dean 2020). These approaches may be most appropriate in many contexts but also face numerous challenges in implementation.

A key question to ask before selecting an approach is whether the decision context lends itself to the specification of constraints that relate to multiple, irreducible dimensions of the system under consideration (situations as the one illustrated in Fig. 1b). If this is the case (response is 'yes' at step 6 in Fig. 3), the coviability approach may provide a suitable alternative, which combines the application of multi-criteria evaluation with the identification of feasible pathways for management. Although considered an approach under the MCA umbrella, we discuss coviability separately here because of the fact that, unlike the majority of MCA approaches, coviability identifies a set of strategies which concurrently satisfy all the system constraints, without a prescribed final ranking.

If it seems difficult to identify adequate performance metrics and references or threshold levels allowing implementing coviability analysis (response is 'no' at step 6 in Fig. 3), one can turn to one of the multiple MCA methods. If an interactive process is not required (response is 'no' at step 7 in Fig. 3), standard MCA reporting approaches can be used.

If the decision-making process is such that trade-off assessment requires an interactive process (response is 'yes' at step 7 in Fig. 3), the approach will likely depend on the number of stakeholder groups (step 8 in Fig. 3).

The alternative approaches to carrying out a trade-off analysis

Following the decision map (Fig. 3), we describe the alternatives that the analyst may consider in response to the question being asked at each node. It is important to understand the context in which a formal trade-off analysis is required as this will shape the approach to the analysis, and the selection of the methods used. A range of situations may exist, from decision systems where a question is placed to the assessment team, which is left to decide on the approach and methods used, to decision-making processes where collaborative scientific evaluation is the rule. The latter situations are likely to involve a number of iterations between the team in charge of the assessment and the decision-making body. Whatever the context, the first step will involve scoping the need for evaluation, including the identification of the evaluation dimensions that should be considered, and system components that should be evaluated. This scoping phase in itself requires that clear objectives are available and will usually only be possible in a well-defined decision-making setting.

> Box 1: A stylized example
>
> For the sake of illustration, assume that a fisheries management agency must choose between two mutually exclusive policy alternatives X and Y. Alternative X would allow the commercial fishers to harvest an annual quota of 1000 tonnes but have a detrimental impact on the population of a certain species of marine mammal, reducing the population by 15% (50 individuals) due to bycatch mortality. Alternatively, the manager may decide to implement policy Y (the status quo), which would keep the harvest unchanged with respect to the previous year and allow a catch of only 200 tonnes, but would have no anticipated impacts on the marine mammal population.
>
> In this simple example, the trade-off is clear: the benefits associated with higher landings versus the costs associated with a reduction in the marine mammal population. However, as discussed next, even in this simplified world, the determination of these benefits and costs can be challenging to assess and will typically depend on the criteria adopted for the analysis. While this is a simple example, alternatives X and Y could, more generally, reflect the priorities of different sectors (not necessarily those of fishery management or fishers), and the decision as to which alternative should be implemented should ideally incorporate the broader human dimensions of the system.
>
> How can trade-off analysis help inform the decision on what policy to adopt? The first step is to decide which criterion to use to compare the two policy alternatives. Once that determination is made, the methods associated with each approach described in Fig. 3 will help identify the data necessary to conduct the analyses.

Applying triple bottom line accounting (section 2.1)

If there is still a requirement for some quantitative information to be made available to inform trade-off decisions, the traditional 'triple bottom line' accounting (TBL) with separate accounting-based measures for each component will usually be the default approach. While each outcome may be represented by a numerical measure, the use of 'traffic lights' (i.e. green, amber, and red) is often used to provide a visual guide to the acceptability (or otherwise) of each particular outcome (Glasson 2008). Savitz (2013) provides a detailed discussion of TBL.

Anderson et al. (2015) developed a set of indicators to assess TBL performance of fisheries. These have been applied to a wide range of fisheries internationally (e.g. Danielsen et al. 2021, Liu et al. 2021). Asche et al. (2018) compared the TBL outcomes of 121 fisheries worldwide—derived using the performance indicators—and found that, when effectively managed, all three domains may be improved simultaneously. Conversely, Brown et al. (2018) estimated TBL outcomes [biomass (environment); profitability (economic); and equity (social)] of stock recovery management options in the presence of fleet heterogeneity using a simulation modelling approach. They found that the trade-offs between the outcomes differed substantially depending on the management option simulated.

Measuring welfare outcomes through cost-benefit analysis (section 2.2)

CBA is required to assess policy trade-offs in numerous countries, including the United States through Executive Order 12866, the United Kingdom through the Green Book, and through the European Union's Better Regulations Guidelines. The OECD (2018) provides a detailed open-access manual for the application of CBA. Even though such analysis traditionally focuses on aggregate welfare outcomes using monetary measures, an important dimension may be to consider the distribution of benefits and costs, as the question of winners and losers (across people, companies, social groups, regions, and countries) is often a core issue in trade-off assessment. A standard approach to such an issue in economics is the use of a so-called 'Kaldor-Hicks efficiency criterion'. Under this criterion, a policy change is beneficial if the potential winners from the change could compensate the losers and still be better off. Kaldor-Hicks efficiency is used, e.g., to evaluate environmental policies by the US Environmental Protection Agency (EPA 2024).

Generally, and particularly in the context of ecosystem-based fisheries management, all the impacts of a policy on the complex of species and habitats in the ecosystem should (ideally) be identified and valued. This requires (i) for the science on the ecosystem linkages to be in place in order to describe the anticipated changes associated from each policy option, and (ii) economic valuation of those identified changes. The latter, while difficult, is done using various techniques that fall into two categories: revealed preference approaches (that rely on actual agents' decisions to infer the non-market values implicit in those choices), and stated preference approaches (that rely on surveys and hypothetical choices). The main data required, particularly in designing valuation surveys, is a clear description of the changes induced by each policy option, and demographic information for survey respondents (see e.g. Champ et al. 2017).

Under this framework, the alternative with the highest net-benefit would be the alternative that will serve as the reference or benchmark against which any other choice options will be compared.

Box 2: CBA in our stylized example

The standard economic approach (the gold standard) to deal with trade-offs is to estimate the monetary value, the so-called net-benefit or welfare, of alternatives X and Y. The recommendation from an economic efficiency standpoint is to then pick the alternative with the highest net-benefit. How would you proceed to come up with the monetary value for the net-benefits of the two policies in the example?

Welfare derived from the fishing industry is the 'value for consumers and producers' of an increase in annual landings. Termed consumer and producer surplus, this is the difference between the total cost to produce, and the total value society places on, a good. On the other hand, economists would conduct surveys in order to assess the economic value (willingness to pay) society attaches to preserving the marine mammal population. A type of stated preference study, surveys are used to estimate the value of conservation, recreation, culture, and a host of other important services which cannot be bought or sold directly in a market. While these non-market valuation methods present challenges and are not free from criticism, they allow the costs of losing part of the marine mammal population to be estimated in monetary-equivalence. The underlying assumption of these methods is that for something to have value, somebody must be willing to pay for it. Monetization (i.e. expressing benefits and costs in the same unit) allows economists to explicitly account for the trade-off involved in each policy when computing net-benefits, since costs are directly subtracted from benefits (for further details on non-market valuation, see McConnell and Habb 2002).

In our stylized example, denote the benefits and costs of each policy, respectively, by B_i and C_i where the sub-index refers to the policy, $i = X, Y$. Then, from an economic efficiency standpoint, policy X will be preferred to policy Y if and only if $Bx - Cx > B_Y$ (since policy Y will not hurt the marine mammal population, it follows that its monetized costs are $C_Y = 0$). The higher the net-benefit from an alternative, the more efficient the use of scarce inputs. Thus, policy X would combine a larger monetary value in the form of benefits for the fishing industry (and possible consumers) but it would incur a cost in terms of marine mammal population loss when compared to alternative Y. This cost C_X is directly subtracted from the benefits to arrive at the net-benefits of the policy, $NB_X = Bx - Cx$. Only if the net-benefits of policy X exceed those of Y, $NBx - NBy > 0$, would the former be preferred from an economic efficiency standpoint (see Pearce et al. 2006 for additional details on CBA). Importantly, note that if the monetised costs of reducing the marine mammal population were not specified and were ignored, alternative X would always be preferred as $B_X > B_Y$ (as policy X allows significantly more harvest than policy Y).

A real-world example of an application of CBA to fisheries management is provided by Holzer and McConnell (2017), which studied the implementation of a bioeconomic model for use in the summer flounder, black sea bass, and scup recreational fishery in the Mid-Atlantic Region of the United States. Relying on data from a choice experiment survey to estimate anglers' preferences on different trip attributes, they integrate those preferences into a biological model in order to simulate fisheries outcomes such as number of trips, landings, discards, and welfare under different policy scenarios. The model is currently used by the Mid-Atlantic Fishery Management Council to set regulations. Another example is provided by Lee et al. (2017), who conducted a similar study in the context of cod and haddock in New England. Their bioeconomic model is currently used by the New England Fishery Management Council to set regulations for the cod and haddock recreational fishery.

The CBA approach may still be used if constraints exist that the decision-makers wish to impose on the domain of possible outcomes (e.g. imposing a minimum biomass level for a harvested resource, as a precautionary measure). In this case, welfare evaluations can still be carried out following the same approach and will lead to the identification of options that meet the constraints imposed, the welfare costs of which can also be determined.

Working with impacts and costs: cost-effectiveness analysis (section 2.3)

In cases where only impacts and costs are considered but only cost can be monetized, cost-effectiveness analysis provides a means to prioritize management strategies on a cost per unit outcome basis, where the outcome measures are relatively homogeneous, which is a key assumption of the methodology. In such approaches, the target impact of the management option is assumed desirable, and options are compared in terms of their costs to achieve this impact. Levin and McEwan (2001) present a detailed manual on the application of CEA.

> Box 3: Cost-effectiveness analysis in our stylized example
>
> In terms of the stylized example, assume the manager has decided to implement policy X and increase annual fishery harvest to 1000 tonnes, while limiting the concomitant mortality of marine mammals to a maximum of 50 individuals. A set of policies aimed at limiting the level of bycatch of marine mammals could be formulated, and compared according to their aggregate costs.
>
> Option A (command and control)—this option increases harvest to 1000 tonnes but uses gear restriction to limit bycatch. Specifically, to limit bycatch mortality of the marine mammal population to 50 individuals, all 200 vessels must install a specified bycatch reduction device at a cost of \$c/vessel. It is assumed that this enables reducing bycatch mortality to the maximum 50 individuals, while still harvesting 1000 tonnes of target fish.
>
> Option B (market-based)—this option also increases fishery harvest to 1000 tonnes, but relies on markets to limit bycatch. Concretely, in order to limit the negative impact on the marine mammal population to 50 units, the manager allocates 50 units of bycatch quota among vessels (at no cost) allowing them to trade these 'bycatch rights'.
>
> Under option B the bycatch quota is expected to be sold (or rented out) by vessels that are able to reduce bycatch at low cost (because of configuration of their gear, location of their fishing grounds, skill, etc.) and bought or leased by captains for which bycatch avoidance is costly. The market of bycatch rights allows bycatch reduction to be conducted primarily by those vessels that are efficient at it, reducing the overall cost of abatement. Again, it is assumed that such abatement would still enable catching 1000 tonnes of target fish.
>
> In order to determine the most cost-effective option to achieve the increase in commercial quota to 1000 tonnes while limiting the mortality of marine mammals to 50 individuals, we proceed to compare the cost of option A, given by the number of vessels times the cost per vessel of installing the new device, namely $C_A = 200{*}\$c$, with the cost of option B given by the fleetwide bycatch avoidance costs when bycatch quota is traded freely; called it C_B. Thus, the relevant comparison is between C_A and C_B, where in the absence of transaction costs, C_B will be typically be lower than C_A.

There is an abundant literature on the application of cost-effectiveness analysis in marine resources management. Kronbak and Vestergaard (2013) developed an environmental cost-effectiveness analysis to include intangible benefits in intertemporal natural resource issues. Their example was applied to the Danish mixed trawl fisheries in Kattegat and Skaggerak where the cod stock was determined to be in a poor state and in need of intervention. Two potential selective gears were assessed upon their impact on the eventual equilibrium of the focus fish stock allowing the evaluation of a cost per unit of biomass for each selectivity measure.

Tulloch et al. (2020) utilized return on investment to identify cost-effective measures to reduce cetacean bycatch in a range of Australian fisheries. Three measures were assessed, spatial closures, acoustic deterrents, and gear modifications, in terms of their cost-effectiveness to achieve two outcomes: (i) to lower the cost of bycatch reduction in fisheries, and (ii) solely to protect whale and dolphin species. Spatial and temporal mapping identified the variations in the location and seasonality of optimal cost-effective management measures. Sites could then be prioritized based on the objectives of minimizing costs or maximizing biodiversity outcomes.

Holzer and Byler (2019) studied the Deepwater Horizon Oceanic Fish Restoration Project, developed by the National Oceanic and Atmospheric Administration to reduce bycatch in the Gulf of Mexico Highly Migratory Species fishery. Part of the restoration efforts undertaken after the 2010 Deepwater Horizon oil spill, the project offered vessel owners compensation to voluntarily refrain from using pelagic longline gear for the first 6 months of the year, and for adopting alternative fishing gear that results in low bycatch mortality. Cost-effectiveness in the recruitment of participants and ultimately on bycatch reduction was achieved by running uniform-price auctions, a multi-unit auction in which all winning bidders are paid the same price per unit, the highest losing bid.

If quality information on costs and/or outcomes is not available (response is 'no' at step 5 in Fig. 3), then again, a mixed approach incorporating available monetary values into MCA approaches may be appropriate (step 6 in Fig. 3 and Section 3.5).

Meeting multiple irreducible constraints: the coviability assessment approach and goal programming (section 2.4)

In some cases, the decision problem may be defined in terms of identifying the strategies that will maintain a system within a set of predefined constraints reflecting the different objectives associated with the management options (Doyen et al. 2017). De Lara and Doyen (2008) provide a manual on the history and implementation of coviability analysis to the management of natural resources. To illustrate, a biological objective may be to avoid reducing biomass levels of a harvested fish population below a minimum level under which the recovery of this biomass may be compromised or to set a maximum level of mortality on a protected species. A social objective may be to ensure a minimum number of economically viable participants in an industry or to ensure that the diversity of fishing fleets operating in a fishery is maintained across multiple segments. An economic objective may be to guarantee that the fishery generates a minimum level of aggregate profits. Coviability analysis identifies a set of strategies that are compatible with these constraints by eliminating options that are incompatible, and is closely associated with the concept of safe and just operating space (see e.g. Péreau et al. 2012, Raworth 2012). This narrows the range of options for decision makers to consider without a need for an explicit ranking of these options, although they can be compared by any of the metrics used to assess the performance of the fishery system (e.g. total fishery profit, capacity to maintain stocks above biological limit reference points, capacity to maintain diverse economically viable fleets).

Box 4: Coviability analysis in our stylized example

In our example, application of a coviability approach could be warranted, if it proves easier to bring stakeholders together to discuss the minimum viable levels that should be met for ecological, economic, and social impacts of the management options. This could involve identifying, for instance, three categories of constraints: (i) on the overall bycatch of the protected marine mammal, (ii) on the number of economically viable boats operating in the fleet (taking into account the differences in economic efficiency and ability to avoid marine mammal bycatch between different categories of vessels), and (iii) on the minimum stock size of the target fish species. Alternative management policies could then be examined, knowing the bioeconomic dynamics of the fishery and the ensuing impacts on these three dimensions, seeking to identify management options that enable this set of constraints to be met over time (so-called eco-viable management strategies).

An example of applying coviability analysis is provided by the study of Briton et al. (2023) of the trade-offs associated with the setting of catch possibilities in the Australian Southern and Eastern Scalefish and Shark Fishery, which aimed to develop an operational approach for comparing trade-offs of not only biological and economic objectives, but also social objectives, in this case for maintaining affordable fish prices for the Australian public. Building on a stochastic bioeconomic model of the fishery integrating interactions between multiple fish stocks and fleets, as well as key dynamic processes such as fishing effort allocation, quota trading, and price responses to variation in landings, a set of viability constraints were defined which reflect the above objectives, building on the available biological and economic information. The model developed can be used to analyse the capacity of alternative policy options relating to the levels of Total Allowable Catch limitations (used as control variables in the analysis), for target and bycatch species, to simultaneously meet the set of constraints, as well as allowing identifying so-called eco-viable management strategies which maximize the probability that all constraints will be met.

Another example is the study of Gourguet et al. (2016), the purpose of which was to formally assess the trade-offs associated with balancing biological, economic and non-target species conservation objectives, using the Australian Northern Prawn Fishery (NPF), which is one of the most valuable federally managed commercial fisheries in Australia, as a case study. A similar stochastic coviability assessment of the fishery was developed and used to show that due to the variability in the interactions between the fishery and the ecosystem, current effort-based management strategies were characterized by biological and economic risks. Results highlighted the trade-offs between respecting biological, economic, and non-target species conservation constraints at each point in time with a high probability and maximizing the net present value of the fishery.

The main difference with constrained welfare assessments (see Section 3.2) lies in the fact that the method will usually not lead to the identification of a single, best decision schedule, but to a family of management options which meet the viability criteria, allowing some bargaining space to remain for stakeholders with respect to the (subjective) advantages of alternative courses of action.

A related approach is goal programming, an optimization modelling approach that can be used to identify optimal levels of various types of management levers (or control variables) in a fishery (e.g. fleet size, structure, and effort allocation) that best achieve (i.e. minimizes deviations from) a set of pre-defined goals associated with each objective: for example, target employment level, profitability, and biomass. As with the coviability approach, minimum acceptable levels of each objective outcome can be imposed, as well as target levels.

The approach estimates the level of the key control variables that minimize the weighted sum of deviations from the target levels of the set of objectives. Pascoe and Mardle (2001) developed a goal-programming bioeconomic model of the multi-species multi-gear fisheries of the English Channel to assess the optimal UK and French fleet size and effort distribution that best achieved profitability, employment, and relative stability objective targets. Similarly, Kjaersgaard et al. (2007) developed a goal-programming bioeconomic model of the Danish industrial fishery in the North Sea to consider implications of different preference structures of different stakeholder groups with regard to economic, political, and biological objectives on optimal fleet structure.

Both coviability analysis and goal programming provide different information to policy makers. The former is best for determining the likelihood that a harvest strategy will be able to achieve each objective, while the latter is best for providing an indication as to the best 'end point' that a harvest strategy should aim to achieve, taking into account the trade-offs between objectives.

Working with multi-criteria analyses (section 2.5)

There is a broad range of MCA methodologies, and Dodgson et al. (2009) provide a manual on their implementation. In a case study, which parallels our stylized example, Walden and DePiper (2023) use Data Envelopment Analysis (DEA) to estimate implicit weights and trade-offs from the historical performance of fishery revenue, right whale abundance, and recreational fisheries in the Northeast US to develop a single integrated performance indicator. Their analysis indicates that system performance has not significantly improved over time once environmental quality is also considered, and highlights how historical data can be used to empirically estimate weights across multiple criteria.

Box 5: MCA in our stylized example

In our example, managers may also want to consider the increase in regional jobs associated with a higher annual quota under alternative *X* compared with alternative *Y*, which leaves both the number of jobs and the marine mammal population unchanged. Regional economic impacts (such as employment) can be estimated using input–output models, which capture the multiplier or ripple effect of an additional dollar spent in the region. In our example, increasing annual harvest from 200 to 1000 tonnes increases the number of full-time jobs from 1500 to 1900. The comparison between alternatives *X* and *Y* requires the 400 additional jobs to be weighed against 50 fewer marine mammals, without the help of a common metric.

The 'best' option will depend on the subjective assessment of the relative importance of both variables by the parties making the decisions. As such, the choice of weights placed on each variable plays a central role in the ranking of these outcomes. Where explicit weights between different objectives are used, this may be equivalent to implicitly allocating a monetary value to the different components being weighted.

Similarly, Pascoe et al. (2023) used DEA to assess management options for a range of commercial and recreational fisheries with multiple objectives, while Pascoe et al. (2020) used DEA to compare different harvest strategies against multiple objectives. A key advantage of DEA as an MCA approach is that it is unit invariant. That is, it can encompass different variable scales and types (e.g. profit in Euros, employment in job numbers, bycatch in tonnes) without the need for normalization.

Other forms of MCA compare relative deviations from current conditions, weighting these deviations to provide an overall weighted average deviation (either positive or negative) associated with each management alternative. Dichmont et al. (2013) and Pascoe et al. (2019) used this approach to assess management options for two fisheries in Queensland across a range of environmental, economic, and social objectives. Similarly, Rossetto et al. (2015) used (non-linear) normalized values of income, employment, fishing mortality, discards, and other outcomes, aggregated using stakeholder preference weights, to assess different management scenarios for Mediterranean demersal fisheries.

Where interactive processes are required, and with only a small number of stakeholder groups, participatory MCA approaches such as deliberative MCA may be appropriate, especially if combined with broader MCA approaches (Rauschmayer et al., 2006; Proctor, 2006). If the number of stakeholder groups is large, MCA with deliberative multi-criteria evaluation (DMCE) for subgroups (Mavrommati et al. 2016) may offer an alternative. MCA approaches can, in fact, also encompass large sets of stakeholders, as well as elicit preferences from the general public (Stagl 2006, Dichmont et al. 2013).

Box 6: Participatory MCA approaches

In our example, we could assume that stakeholders require a deliberative process for carrying out MCA to inform management choices. This could, in particular, help to identify weights used in balancing the outcomes across dimensions considered in the analysis. If the number of stakeholders is small, the participatory process could include consensus building through processes such as mediated modelling or the 'consensus conference'; approaches where stakeholders and scientists collaborate to develop a common understanding of the system under consideration and the key system components required to answer the management question at hand. In larger groups with diverse stakeholder interests, consensus might not be possible. In these situations, heterogeneity in the weights and values across stakeholder groups can be used to not only assess the sensitivity of results to alternate weighting schemes but also scope out alternate system states from which to assess the robustness of management performance.

Discussion

Increasingly, science in support of natural resource management and conservation planning is required to contribute to the evaluation of alternative policy options. This has led to a growing field of work at the science-policy interface, building on methods of decision theory and economics to develop approaches that provide the integrated assessments required. While there are many arguments between proponents of each approach, we take the view that all of the approaches are variations of a common theme. There exists a continuum, from standard evaluation of monetary costs and benefits, through analyses that focus on the costs of achieving a particular objective, to the use quantitative assessments such as coviability and goal-programming approaches to address multiple objectives, and to the use multi-criteria assessments with a range of quantitative aggregation methods, or of deliberative approaches, to identify preferred options. Which approach is more appropriate depends on the context, scale, and the availability of existing data and costs of new data collection.

The key role of scoping

With any assessment, it is not feasible to capture all costs and benefits. Which costs and benefits are considered—and how they are captured—is largely context specific. Multicriteria approaches utilize an explicit set of objectives against which outcomes are measured, and help refine the information that is required for the assessment. However, a key criticism of MCA is the potential for included stakeholders to influence both the objective set and objective weightings, which, in turn, may bias the outcomes (Dobes & Bennett 2009). Choice of which costs and benefits to include in the CBA, as well as scale issues (i.e. considering local, national, or international costs and/or benefits) can equally affect the outcome (Bebbington et al. 2007). Hence, how the methods are undertaken is as important as which method is applied. Any approach thus implies value judgments, and whether these judgements can be perceived as a legitimate basis for decision-making is a key aspect of selecting the approach to trade-off assessment.

Data requirements

Data requirements will depend on the criterion adopted to conduct the analysis. As discussed earlier, welfare analysis (i.e. CBA) relies on the quantification of all the benefits and the costs associated with each policy alternative. In the case of commercial fishing, net-benefits associated with the commercialization of the target species entails estimating consumer and producer surpluses. Consumer surplus measures the difference between what consumers are willing to pay for the catch and what they actually must pay in the market. Producer surplus, on the other hand, measures the difference between what the market pays for the catch and the costs of harvesting the market equilibrium level of output. Computation of the consumer surplus requires estimation of the demand for the species of interest (or system of demands in a multispecies fishery). In turn, demand estimation requires data on quantities demanded at different prices. Calculation of producer surplus, on the other hand, relies on estimation of the market supply, which requires data on quantities supplied at different prices. Supply and demand functions are typically estimated jointly using market equilibrium data on prices and quantities, to ensure unbiased estimates of both functions. When a fishery produces only a very small portion of the total global production of a good, the effect of policy changes on consumers are often so small that they can be ignored. In these scenarios, economists estimate (when the data are available) vessel-level profit functions, that is, the relationship between net revenues per trip (revenues minus trip costs) and prices of both inputs and outputs for each gear type in the fishery. This approach requires data on revenues, variable costs (ice, fuel, bait, quota rental costs, etc.), and prices of all species landed and of inputs used at the trip level for all participants in the fishery. When coupled with information on fixed costs (such as repair costs, insurance, etc.) these estimated profit functions allow the manager to compare participation and total net-benefits for each policy alternative under consideration.

MCA approaches may rely on all the data available to characterize the dimensions that are considered important for ecological, economic, and social reasons. In practice, while large sets of data may be available regarding the ecosystem state, it may often be the case that social and economic information is less readily available, at the scales allowing direct inclusion in MCA evaluations devoted to marine resource management problems, leading to a need to collect *ad hoc* information.

A key advantage of MCA-based approaches is that they can be directly integrated into fishery bioeconomic models (such as coviability analysis and goal programming) and/or use model-produced outputs to assess different management strategies across the range of objectives. They can also integrate other considerations such as regional economic impacts derived from input–output models, e.g. which may also use the outcomes from the multi-objective bioeconomic models as inputs.

Regardless of what decision framework is employed, sensitivity analysis is a best practice used to assess the robustness of policy rankings in economic trade-off analysis (e.g. Renda et al. 2013, OECD 2018). Sensitivity analysis can address uncertainty in any metric utilized in the decision criteria employed, including those stemming from economic cost and benefit estimates, weights across alternate objectives, imperfect knowledge of future environmental states, and any other source that lends uncertainty to the analysis.

The importance of the decision context

Who makes the decision is also an important consideration. A key determinant in Figure 2 is the acceptability of monetary values for all costs and benefits. Despite the long-established history of CBA, there remains substantial scepticism about the reliability of non-market values in natural resource and conservation management (Driml 1997, Laurans et al. 2013). While there are well established and accepted approaches for deriving environmental non-market values, assigning monetary value to social outcomes is still problematic. Indeed, there may be more estimation uncertainty with non-market values than with market values, and communicating this in a CBA may be challenging. In co-management systems, there may be no single decision maker, and the consensus decision will be based on stakeholder views as much (if not more) than external valuation studies. The result of even a well-conducted CBA may have limited influence in such a situation, whereas capturing these stakeholder views in the analysis is a substantial strength of MCA in these circumstances. In some cases, CBA can in fact be used as a way of capturing and communicating the economic consequences of a policy, in an orderly manner, and it is the process of conducting a CBA and the data collected that may be more important than the final analytical outcome.

The costs of undertaking the different approaches may be significantly different, particularly where non-market costs and benefits are key considerations. Valuation studies can be expensive to conduct (although maybe not compared to many environmental analyses) and require expertise that precludes involvement of stakeholders at an operational level. The use of benefit transfer techniques reduces the costs substantially but introduces sources of uncertainty into the analysis that is sometimes difficult to quantify (Plummer 2009). With small-scale issues (e.g. management of a particular fishery), participatory MCA approaches may be both more cost-effective and also more acceptable than an externally conducted CBA, although participatory CBA could in principle also be conducted. For large-scale issues (e.g. implementing a national marine policy), the larger number of potential stakeholders may make participatory MCA impractical if there is too much heterogeneity in views, while the costs of undertaking CBA may be less a deterrent given the scale of the potential benefits and costs being considered.

Optimally, the approach will be selected first, and the timeline and other management milestones will be scoped appropriately as a second step to match the needs of the analysis. However, management actions are often undertaken within predefined timelines that can also impact the choice of approach employed and constrain the scope of the question/analysis itself. Gathering new information, including through stakeholder engagement, will extend the timeline for any of the approaches discussed. From a practical perspective, this means that analysts should aim for a level of trade-off analysis appropriate for the question, timeline, and data at hand. Like many other pursuits, economic analysis can face decreasing marginal returns and it is often better to engage with imperfect data and tools than to fully retreat from a policy question of importance (Leman and Nelson 1981). The level of data availability, in conjunction with the timeline, will also inform the selection of an approach to support trade-off analysis. Strategic decision-making can also provide an opportunity for managers to test unfamiliar trade-off methodologies under lower political pressure than would come with providing tactical advice. In addition, regulatory mandates, executive orders, national standards, and other governmental policies and guidelines can play an important role in facilitating the engagement in, and uptake of, formal trade-off analysis for management purposes. Finally, it must be stressed that some degree of redundancy, where different trade-off approaches are implemented to study the same problem, may prove useful in fostering stakeholders' trust in the findings, and in reducing the uncertainty of possible outcomes.

Concluding remarks

There are a wide range of methods available for assessing fisheries management options in the light of multiple objectives. Ensuring the best option identified is implemented will largely depend on the degree to which stakeholders and managers accept the results of the analysis. Distrust in a method will result in a distrust of the selected option, especially if it does not agree with the a priori expectations around outcomes (i.e. does not produce their preferred outcome).

We have presented a framework for selecting a method of analysis based primarily on acceptability of the key assumptions underpinning the method and processes. Foremost of these is trust in the integration process itself. Reporting only at the TBL level gives decision makers ultimate responsibility for the implicit aggregation inherent in their final decisions, replacing objectivity with subjective decision making. Economic-based methods (CBA and CEA) provide potentially the greatest objectivity, although the need to derive economic values associated with some outcomes—particularly social and environmental—may not be acceptable to all stakeholders. The MCA approaches offer the greatest potential for stakeholder input through the derivation of importance weights associated with each outcome (either through participatory processes or through less direct involvement through surveys). Again, this may introduce subjectivity into the decision-making process, although provided appropriate consultation with stakeholders is undertaken this may be reduced.

Whichever method is adopted to assess management outcomes, the move to multi-objective decision-making results in other benefits. Foremost of these is the greater need to undertake greater interdisciplinary collaboration, as determining environmental, economic, and social outcomes requires input from natural scientists, as well as from economists and other social scientists, to develop the appropriate models and measures. This greater interdisciplinary collaboration, combined with greater stakeholder involvement, is also more likely to lead to improved management options to be assessed as well as an improved assessment process, ultimately leading to improved decision-making processes.

Acknowledgements

The authors are grateful to the ICES secretariat and the chairs of the ICES working group on Economics (WGECON) for their support in the organisation of the working sessions which led to the preparation of this manuscript. We thank Dr Linwood Pendleton for his feedback on previous versions of this manuscript.

Author contributions

Original conceptualization and writing of original draft: O.T. and S.P. Adaptation of methodology and illustration: O.T., J.H., G.D., M.K., and S.G. Review of methods and selection of case studies. Writing—review and editing: all.

Conflicts of interest

The author(s) declare(s) that there is no conflict of interest.

Funding

None declared.

Data availability

No new data were generated or analysed in support of this research.

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Handling editor: Kanae Tokunaga

Received: 10 September 2025. **Revised:** 12 February 2026. **Accepted:** 20 April 2026