# ENHANCING BENEFITS EVALUATION FOR WATER RESOURCES PROJECTS: TOWARDS A MORE COMPREHENSIVE APPROACH FOR NATURE-BASED SOLUTIONS

*Planning and Valuation Methods for Case Study Analysis*

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# PREFACE

To accelerate progress and delivery of new and enhanced infrastructure projects for navigation, flood risk management, water operations, and ecosystem restoration consistent with its Engineering With Nature® (EWN) initiative, the U.S. Army Corps of Engineers (USACE) has engaged in a collaborative effort with The Water Institute of the Gulf (the Institute) to conduct policy research for fully evaluating the benefits of EWN strategies and projects, to include Natural Infrastructure, Natural and Nature-Based Features and other Nature-Based Solutions (NBS).

This document is the third in a series of reports produced as part of this collaborative effort. It is intended to provide methodological guidance for the retrospective case study analysis and should be of interest to technical readers at USACE, other federal agencies, and interested stakeholders. The report describes recent and relevant decision analysis methods designed to inform decision makers when considering or trading off among multiple objectives when prioritizing water resources projects. It also includes a review of relevant valuation methodologies in recent economics and ecological management literature, leading to specific guidance for the subsequent analysis to follow. The review includes: a) a description of each method; b) the suitability of each method for estimating particular classes of benefits; c) the data necessary to implement the methods; d) the advantages and disadvantages of each method including examples of where it has been successfully applied; and e) inherent assumptions of different methodologies and implications for outcome. The study team draws on this review to provide a guide for the case study analysis that follows, documented in a separate report.

Questions about this research should be directed to the project lead and Director of Planning and Policy Research at the Institute, Jordan Fischbach (jfischbach@thewaterinstitute.org).

# EXECUTIVE SUMMARY

The U.S. Army Corps of Engineers (USACE) Civil Works program is tasked with development and management of the Nation’s water resources, including maintaining navigable waterways, managing flood risk, and restoring aquatic ecosystems. Projects to advance these objectives must be selected based on weighing overall costs and benefits, which is conducted within USACE using a benefit-cost analysis (BCA). There are several challenges and limitations to conducting this evaluation, however:

- Natural ecosystems and engineered solutions provide value to people that span economic and socio-cultural benefits. Methods for placing economic value on ecosystem goods and services have expanded beyond what is typically used in USACE BCA, but even the most advanced monetization approaches may not fully account for the range of benefits a project may provide to communities and the environment.
- Significant variation exists in the type, function, quality, and associated human value that projects can have across different geographies, ecosystem types, and human communities, making it challenging to develop a uniform approach that can be applied across USACE.
- All approaches for quantifying project costs and benefits have inherent uncertainty, from the conceptual (such as the full suite of long-term costs and benefits to include and the spatial area or subpopulations over which to consider project impacts) to the specifics of monetization (such as future market conditions or likely cost overruns). Environmental benefits and costs are often excluded due to a perception of high uncertainty in quantification and monetization. However, this approach implicitly sets these values to zero, which is likely inaccurate, and also may not account for the uncertainty also present when estimating other types of benefit or cost.
- As a Federal agency and steward of public resources, USACE BCA approaches should be transparent, comprehensive in evaluating tradeoffs across interests, include consideration of local interests, and be consistent with legislative mandates established for the agency. This task can be made challenging by the diversity of stakeholders impacted by USACE projects and perceived difficulties in eliciting and incorporating local knowledge into decision-making in scientifically rigorous way.

The objective of the work described here was to develop a methodology that could address these challenges as part of a more complete and holistic project evaluation than approaches that rely exclusively on monetized cost-benefit analysis. This report was guided by these research questions:

- What methods would be relevant and practical to incorporate into project planning and implementation to support more holistic evaluation of monetized and non-monetized outcomes from USACE projects?
- What methodologies should USACE consider to better evaluate (calculate) the environmental and social benefits and costs from water resources projects?

To address these questions, the study team reviewed methods to support decision making across multiple objectives, explicitly account for long-term uncertainty, and/or consider a range of societal values or differing priorities among different communities. Drawing from these approaches, the team developed a method for evaluating the costs and benefits of USACE projects that considers both monetized and non-monetized metrics. Lastly, the team reviewed relevant valuation methods in economics and ecological management literature that can be used within that evaluation framework.

The study team identified several preliminary conclusions from this methodology review:

- **Integrated analysis provides a more complete evaluation of project benefits.** Exclusively focusing on economic analysis for project evaluation can exclude relevant benefits, but methods exist for more fully capturing the monetized and non-monetized benefits of water resource projects. An integrated analysis that simultaneously considers economic benefits alongside environmental and social benefits can provide a more complete valuation of project costs and benefits. This approach has already been applied for select USACE projects, suggesting that USACE planning practice can evolve to apply integrated analysis and more fully consider a wide range of economic, social, and environmental benefits and costs.
- **Leveraging multiple methodologies can allow benefits and costs to be evaluated across the diverse range of USACE projects.** The methods reviewed in this report provide a menu of options for the subsequent case study analysis and for USACE planners to draw upon in future feasibility studies. Suitable quantitative methods can consider benefits in both monetized and non-monetized terms, and qualitative methods that can capture local values and traditional knowledge may also play an important role. The diversity of available methods for environmental and social benefit estimation can enable location-specific data—which will vary in availability by project and location—to be used in conjunction with other approaches to inform ecosystem services analysis across the range of USACE mission areas and projects.
- **Enabling flexibility in evaluation methods and providing support in their application can support more widespread use within USACE.** Given the range of available methods for more integrated and comprehensive analysis, USACE practitioners need flexibility to employ methods appropriate for individual projects and may benefit from additional resources to successfully employ these methods at scale. Additional resources could include expanded guidance, technical expertise, and programmatic funding to engage with and gather data from communities of interest in advance of a Congressionally authorized study process. Added flexibility would allow districts to select the best available methods and evaluation approach given the water resources challenge, local geography and community context, and non-federal sponsor interest and capacity.

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# LIST OF ACRONYMS

| Acronym | Term |
|---|---|
| AM | Adaptive Management |
| ARIES | Artificial Intelligence for Ecosystem Services |
| BCA | Benefit-Cost Analysis |
| BRI | Benefit Relevant Indicator |
| DMDU | Decision Making under Deep Uncertainty |
| EAB | Environmental Advisory Board |
| EEIRP | Evaluation of Environmental Investments Research Program |
| EMRRP | Ecosystem Management and Restoration Research Program |
| ERDC | Engineer Research and Development Center |
| EWN | Engineering with Nature |
| FEMA | Federal Emergency Management Agency |
| InVest | Integrated Valuation of Ecosystem Services and Tradeoffs |
| IPBES | Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services |
| IWR | Institute for Water Resources |
| MCDA | Multi-Criteria Decision Analysis |
| MEA | Millennium Ecosystem Assessment |
| MODA | Multi-Objective Decision Analysis |
| MORDM | Many Objective Robust Decision Making |
| NBS | Nature Based Solutions |
| NCP | Nature’s Contributions to People |
| NED | National Economic Development |
| NNBF | Natural and Nature Based Features |
| NOAA | National Oceanic and Atmospheric Administration |
| OMB | Office of Management and Budget |
| PR&G | Principles, Requirements and Guidelines |
| PrOACT | Problem, Objectives, Alternatives, Consequence Analysis, Tradeoffs |

| Acronym | Term |
|---|---|
| RED | Regional Economic Development |
| SDM | Structured Decision Making |
| SMART. | Specific, Measurable, Achievable, Relevant, Timeline realistic |
| SROI | Social Return on Investment |
| TEEB | The Economics of Ecosystems and Biodiversity |
| USACE | United States Army Corps of Engineers |
| USFS | United States Fish and Wildlife Service |
| USGS | United States Geological Survey |
| WRDA 1974 | Water Resources Development Act of 1974 |

# 1.0 INTRODUCTION

To accelerate progress and delivery of new and enhanced infrastructure projects for navigation, flood risk management, water operations, and ecosystem restoration consistent with its Engineering With Nature® (EWN) initiative, the U.S. Army Corps of Engineers (USACE) has engaged in a collaborative effort with The Water Institute of the Gulf (the Institute) to conduct policy research for fully evaluating the benefits of EWN strategies and projects, to include Natural Infrastructure, Natural[1] and Nature-Based Features (USACE, 2021), and other Nature-Based Solutions (NBS). Throughout this document, these techniques will be referred to using the umbrella term "NBS".

This document is the third in a series of reports produced as part of this collaborative effort. Here, the study team seeks to answer key questions identified in the overview report (Ehrenwerth et al., 2022): What methods are relevant and practical to incorporate into project planning and implementation to support more holistic evaluation of monetized and non-monetized outcomes from USACE projects? What methodologies should USACE consider to better evaluate (calculate) the environmental and social benefits and costs from water resources projects?

The USACE Civil Works program is tasked with development and management of the Nation's water resources, including maintaining navigable waterways, managing flood risk, and restoring aquatic ecosystems. Projects to advance these objectives must be selected based on weighing overall costs and benefits, which is conducted within USACE using a benefit-cost analysis (BCA). There are several challenges and limitations to conducting this evaluation, however:

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- All approaches for quantifying project costs and benefits have inherent uncertainty, from the conceptual (such as the full suite of long-term costs and benefits to include and the spatial area or subpopulations over which to consider project impacts) to the specifics of monetization (such as future market conditions or likely cost overruns). Environmental benefits and costs are often

[1] "[N]atural features are created and evolve over time through the actions of physical, biological, geologic, and chemical processes operating in nature… [c]onversely, nature-based features are those that may mimic characteristics of natural features, but are created by human design, engineering, and construction to provide specific services such as coastal risk reduction" (Bridges et al., 2015).

excluded due to a perception of high uncertainty in quantification and monetization. However, this approach implicitly sets these values to zero, which is likely inaccurate, and also may not account for the uncertainty also present when estimating other types of benefit or cost.

- As a Federal agency and steward of public resources, USACE BCA approaches should be transparent, comprehensive in evaluating tradeoffs across interests, include consideration of local interests, and be consistent with legislative mandates established for the agency. This task can be made challenging by the diversity of stakeholders impacted by USACE projects and perceived difficulties in eliciting and incorporating local knowledge into decision-making in scientifically rigorous way.

BCA also has other inherent limitations. The approach uses monetized valuation as way to bring together and implicitly weight different effects into a single decision metric, but this "black box" valuation summary can mask important tradeoffs. It also does not necessarily address the distribution of benefits across different populations or other social objectives (e.g., considerations of social equity). Furthermore, BCA approaches typically use post-hoc sensitivity analysis to consider uncertainty rather than incorporating uncertainty from the outset as with scenario-based methods.

BCA is only one of many potential approaches to support water resources decision analysis, however.[2] Methods exist that are designed to support decision making across multiple objectives and that explicitly account for uncertainty, thereby having the potential address many of the challenges and limitations inherent in project evaluation based solely on monetized valuation. These approaches can incorporate both monetary and non-monetary metrics for assessing project impacts and necessitate developing non-monetary metrics to capture other types of benefits and costs that are relevant to decision making.

## 1.1 QUANTIFYING ECONOMIC, ENVIRONMENTAL, AND SOCIAL BENEFITS

As discussed in Ehrenwerth et al. (2022), there has been significant evolution in USACE's mission and in the laws and policies governing the valuation of USACE projects. The science of evaluating economic, environmental, and social benefits has also evolved considerable over time (see Appendix A), with new approaches and perspectives being developed that can potentially be used to more comprehensively evaluate the ecosystem services provided by USACE projects in general and NBS specifically than through the exclusive use of BCA. Ecosystem services are defined as the flows of services from environmental features, or natural capital, that provide direct or indirect benefits to humans, and are thus

[2] A related technique, cost effectiveness analysis, assumes what is to be accomplished, and compares the costs of achieving it, with the least expensive option preferred. In other words, benefits are not monetized, and in many applications, any cobenefits not explicitly contained in the objective are implicitly valued at zero with an assumed (identical) level of benefit across alternatives. For formal Federal and Army guidance about economic analysis, see The Department of the Army, Headquarters, "Economic Analysis: Description and Methods" Pamphlet 415–3, September 28, 2018 (Department of the Army, 2018) and OMB, Circular A-4, Regulatory Analysis, September 17, 2003 (OMB, 2003).

valuable.[3] For the ecosystem service to be provided or supported, there has to be an ecosystem that supports a relevant biophysical structure or process (e.g., presence of trees) that results in an ecosystem function (e.g., reduce water flow or wave height) that provides benefits to human communities (e.g., reduced flooding and property loss; Crossman et al. [2013]; TEEB [2009]).

In recent decades, the science necessary to better understand and quantify the ecosystem services provided by NBS has advanced, in terms of both the modeling of biophysical systems and the welfare effects of changes in the provision of private and public ecosystem services. The complexity of the systems under consideration and the many differences between human communities implies that biophysical effects—and their valuation—are likely to vary from place to place. Study resources are limited and it may be beyond the capacity of planners under current policy practice and budget constraints to develop location-specific models or collect the quantitative and qualitative data necessary to identify and value these effects using primary studies.

Because ignoring the environmental co-benefits of NBS leads to risk that chosen solutions are suboptimal from a societal perspective, analysis techniques that rely more heavily on expert elicitation and/or secondary data may provide opportunities for more comprehensive project evaluation. These methods may be less familiar to practitioners and therefore perceived as more difficult to apply (or less robust) than standard USACE valuation techniques, however. This report, along with the others in this series, are designed to provide resources that can support more comprehensive evaluation of ecosystem service benefits from NBS.

## 1.2 PURPOSE AND ORGANIZATION OF THIS REPORT

This report is intended to provide specific methodological guidance for the retrospective case study analysis, which will be documented in a separate forthcoming capstone report. The next chapter of this report provides an overview of decision analysis approaches that provide opportunities for project evaluation that can inform project selection while considering multiple objectives and the potential synergies and tradeoffs between them. Chapter 3.0 draws from these approaches to outline a framework for using multi-objective analysis in conjunction with BCA that USACE can potentially use to evaluate projects. Chapters 4.0 and 5.0 then describe non-valuation metrics and relevant methodologies for valuation (in dollar terms), respectively, which could be used within that framework to inform more holistic consideration of NBS approaches. These chapters include: a) a description of each method; b) the suitability of each method for estimating particular classes of benefits; c) the data necessary to implement the methods; d) the advantages and disadvantages of each method including examples of where it has been successfully applied; and e) inherent assumptions of different methodologies and implications for

[3] In this formulation, “benefits” are defined as the goods and services that increase an individual’s utility, and their value is defined as what an individual would give up in order obtain them (or how an individual must be compensated if she were to lose them) in order to stay at the same utility level.

outcome. A brief conclusion and considerations for the case study analysis is provided in Chapter 6.0. Additional background on ecosystem service frameworks and methods is provided in Appendix A.

# 2.0 DECISION SUPPORT APPROACHES AND TOOLS

## 2.1 OVERVIEW

USACE has historically relied on monetized valuation of goods and services to select project alternatives. However, these approaches cannot capture the full range of benefits and costs from a given project, particularly those that incorporate NBS. Technical limitations, time and resource constraints, and the theoretical underpinnings of nature's benefit to people all point to the need for a broader decision framework that can consider project consequences in non-monetized terms and use these outputs to help prioritize different approaches or consider key tradeoffs. The need for alternate frameworks is further reinforced when including the social impacts of proposed alternatives, because benefits and costs can vary significantly across different communities and traditional BCA methods can exacerbate historical inequities.

Uncertainty about future benefits and costs is another key component of infrastructure decision making that valuation and traditional BCA comparisons alone may not address. The assumptions required for valuation may lead to underestimating or ignoring uncertainties essential to a long-term infrastructure decision. As a result, additional frameworks and tools are needed to consider long-term uncertainty in project performance and cost directly in USACE planning studies.

The limitations of a purely valuation-based approach have led to the development of techniques that consider multiple criteria in making decisions. These types of approaches have been applied to support a range of consequential decisions, such as considering factors of blast protection, rollover risk, and fuel consumption in the design of military ground vehicles (Hoffenson et al., 2014) or Army helicopter fleet modernization (Prueitt, 2000). In this chapter, an overview is provided of decision analysis approaches designed to address multiple objectives measured in different units, consider tradeoffs across these multiple objectives, and use scenario analysis to incorporate future uncertainty explicitly and build towards alternatives that are more robust to uncertainty related to project performance or cost. For a more extensive discussion on additional methods, see Harrison, et al. (2018).

## 2.2 MULTI-CRITERIA AND MULTI-OBJECTIVE DECISION ANALYSIS

Multi-criteria decision analysis (MCDA) and multi-objective decision analysis (MODA) are overarching terms describing a suite of different approaches for systematically evaluating the performance of alternatives across objectives, assessing tradeoffs, and testing the robustness of a decision to unknowns and uncertainties (Adem Esmail & Geneletti, 2018; Belton & Steward, 2002; Kiker et al., 2005). MCDA/MODA was developed to improve decision-making in cases where complex decisions with multiple, potentially competing objectives make it challenging for individual or groups of decision-makers to select an alternative for optimizing outcomes without the support of an analysis framework. USACE practitioners have successfully applied MCDA/MODA in select cases for project evaluation. For example, MCDA was used as part of a risk-informed decision framework to evaluate alternatives to advance public health and safety, storm damage reduction, coastal ecosystem and habitat preservation, and cultural resource protection objectives under the Louisiana Coastal Protection and Restoration project (USACE, 2009). Although there are multiple variations of MCDA/MODA approaches, the stages of an

MCDA/MODA decision making process can be generalized as: (1) decision context and structuring; (2) analysis; and (3) making the decision (Adem Esmail & Geneletti, 2018).

### 2.2.1 Decision Context and Structuring

The first stage of MCDA/MODA includes articulating a set of potential alternatives (decision choices) that could potentially result in desired outcomes, then formulating criteria against which to benchmark success in achieving those outcomes. Alternatives may be identified prior to initiation of an MCDA/MODA approach (Adem Esmail & Geneletti, 2018). The criteria for evaluating alternatives are identified by the decision-maker and may include eliciting input from other stakeholders (Adem Esmail & Geneletti, 2018; Kiker et al., 2005; Velasquez & Hester, 2013). For example, criteria identified for siting of dredge material placement might include capacity and ownership of disposal sites, proximity to the dredging location, and regulatory concerns (Suedel et al., 2008, 2009).

### 2.2.2 Analysis

The analysis stage of MCDA/MODA includes assessing the outcomes of the set of alternatives by calculating/predicting the value of the identified criteria; weighting individual criteria based on decision-maker and stakeholder priorities; aggregating the criteria values to create a combined score(s) for each alternative; and conducting sensitivity analysis to determine how robust the criteria values and combined expected utility are to uncertainties (Adem Esmail & Geneletti, 2018; Belton & Steward, 2002; Kiker et al., 2005). For example, a simple method for combining criteria to calculate scores for alternatives would be to calculate the weighted sum of individual criteria values, with the weights elicited from the decision-maker or stakeholders based on the relative importance of each. However, a wide range of quantitative methods for combining criteria and testing the sensitivity of the combined score(s) to underlying uncertainties have been developed and have been reviewed elsewhere (Adem Esmail & Geneletti, 2018; Belton & Steward, 2002; Kiker et al., 2005; Linkov et al., 2006b).

### 2.2.3 Making the Decision

The last stage includes ranking the alternative(s) through use of quantitative analysis to optimize the predicted outcomes. As with criteria calculation and combination, there are a wide range of techniques and quantitative tools that can be applied to calculate the range of possible outcomes over a range of alternatives and uncertainties and select the alternative or portfolios of alternatives that are most likely to produce positive outcomes across the identified criteria and/or that are the most robust to the under lying uncertainties (Edwards et al., 2007; Linkov et al., 2006b).

Because MCDA/MODA approaches span a range of calculation methods and the criteria for alternative evaluation are defined as part of the decision-making/analysis process, it is applicable to a range of problems. MCDA/MODA has been widely applied in environmental management (Kiker et al., 2005), with specific examples including disposal of contaminated sediments (Linkov et al., 2006b); placement of dredged material (Suedel et al., 2008); and ecosystem restoration (Convertino et al., 2013). Although decision-making processes within USACE (and other federal agencies) are dictated by regulation, policy, or agency-level guidance, MCDA/MODA has been applied and utilized in some cases with USACE to evaluate multicriteria benefits while still being consistent with the prescribed Principles and Guidelines (Linkov et al., 2006c, 2006a; USACE, 2009).

One of the benefits of MCDA/MODA is that it enables multiple objectives to be transparently and systematically evaluated, making it well-suited for complex problems with multiple stakeholders (Kiker et al., 2005). Data requirements for MCDA/MODA will vary depending on the specific methodology chosen and the underlying needs of the decision-maker; however, there must be sufficient data at the appropriate spatial and temporal resolution to calculate criteria and optimize outcomes across the range of alternatives being considered.

MCDA/MODA has several limitations that should also be considered prior to practical application. Outcome evaluation strongly depends on criteria selection, which meta-analysis suggests can vary widely across MCDA/MODA applied to inform similar decisions and may double count effects or include implicit biases if not developed carefully (Wahlster et al., 2015). As the decision becomes more complex and the number of criteria increases, it can be challenging to elicit subjective ranking or rating preferences from decision-makers and stakeholders and incorporate them into tradeoff analysis (Kiker et al., 2005; Wahlster et al., 2015). The quantification of outcomes through a set of discrete metrics may result in loss of information relevant to the decision, either because there are interrelated considerations that cannot be captured in multiple metrics without double counting or because numerical metrics cannot fully capture the value of potential alternatives from the perspective of the decision-maker (Baltussen et al., 2019). Lastly, aggregate scores are sensitive to individual metric weights (Abdullah et al., 2021; Baltussen et al., 2019), which are subjective and may vary even when elicited from the same set of decision-makers. This aggregation can mask decision-relevant information or outcomes for specific objectives and/or undue sensitivity to weighting factors.

## 2.3 STRUCTURED DECISION-MAKING

Structured decision-making (SDM) is an approach for making complex decisions that includes tradeoff analysis and optimization (Gregory et al., 2012; Hammond et al., 1999). SDM builds on and can incorporate a variety of tools for decision analysis and can be considered an extension of MCDA/MODA. In comparison, however, SDM has a more structured approach to problem definition and decomposition than most MCDA/MODA approaches, with the tools used for consequence and tradeoff analysis selected after (and based on) the problem definition (Gregory et al., 2012).

SDM draws underlying principles from the fields of decision and risk analysis, management science, and behavioral psychology, and was developed to support value- and objective-focused decision-making through a process that is deliberately designed to (1) mitigate human biases that tend to limit identification and consideration of a complete range of decision alternatives; (2) incorporate prediction of decision outcomes to support robust choices; and (3) include identification and evaluation of relevant uncertainties (Gregory et al., 2012; Gregory & Keeney, 2002).

SDM is particularly well-suited for applications where there are multiple decision-makers or stakeholders because the structure enables a transparent decision-making process, includes explicit identification and consideration of multiple objectives, and incorporates tradeoff analysis to analyze and ultimately optimize decision outcomes across competing objectives (Robinson & Fuller, 2016). As such, SDM has been used to support a variety of decision-making and planning processes with example applications including informing mid-construction response to storm impacts in a barrier island restoration project (Dalyander et al., 2016); evaluating impacts of flow releases from dams (DeWeber & Peterson, 2020); and management of invasive species (Runge et al., 2011).

SDM is conducted through the PrOACT framework (Figure 2-1), which includes Gregory et al. (2012):

1. Define the **Pr**oblem: articulation of the issue to be resolved, the scope of the decision context, and the relevant decision-makers and stakeholders. In practical application to complex problems, previously unknown lack of clarity on the bounds of the decision and/or the role of multiple decision-makers is often identified and resolved in this step.
2. Identify Fundamental **O**bjectives: explicitly identifying positive (or negative) desired outcomes that decision-makers and stakeholder would like to maximize (or minimize/avoid). Objectives may be linked to specific metrics that are used to predict (or later track) success.
3. Identify **A**lternatives: development of a list of potential alternatives (i.e., decision options) that have the potential to advance the fundamental objectives identified in (2). This step is conducted after objective identification to encourage values-focused thinking.
4. Conduct **C**onsequence analysis: prediction of the outcomes of the alternatives identified in (3) through the lens of the objectives identified in (2). SDM allows for broad flexibility in the tools used for this step and may include complex deterministic or probabilistic numerical models, simple desktop models, expert elicitation through structured input such as defined impact scales, or a combination of approaches.
5. Evaluate and optimize **Tr**adeoffs: assess the positive and negative tradeoffs of the alternatives identified in (3) for the complete set of objectives articulated in (2) using the consequence analysis conducted in (4). SDM allows for flexibility in choice of tools used in this step, as well, and may incorporate quantitative techniques that optimize outcomes across metrics associated with each objective.

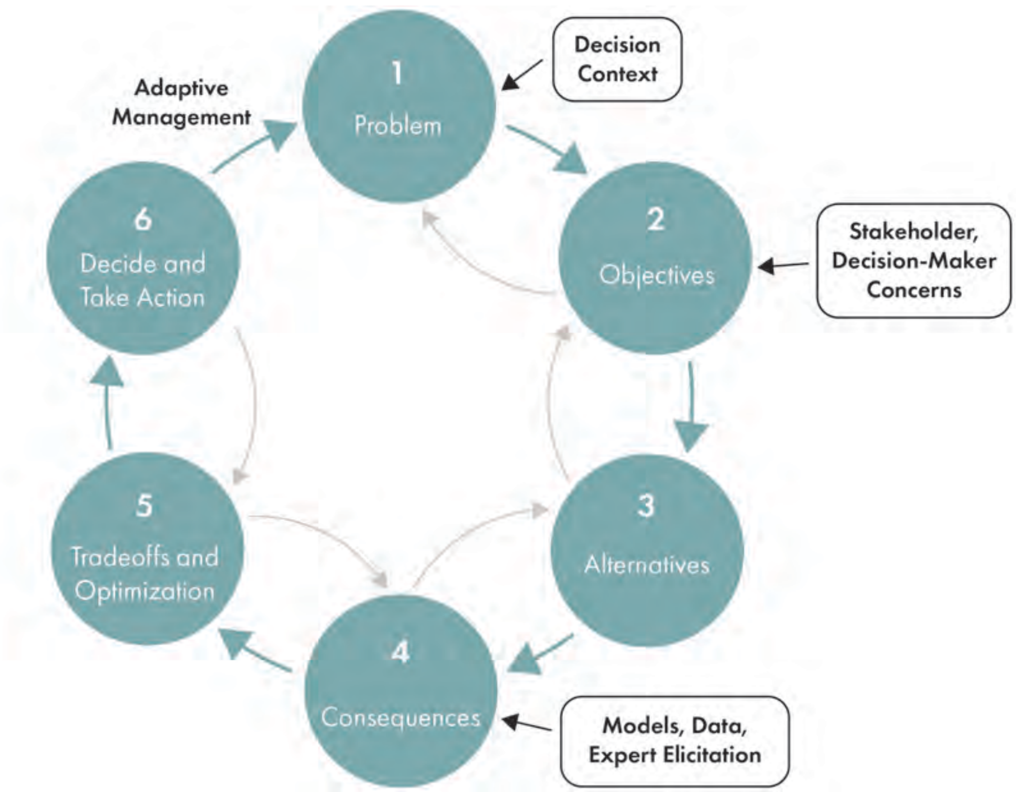


*Figure 2-1. Visual representation of the PrOACT cycle used in Structured Decision Making. Source: adapted from Dalyander et al. (2021).*

SDM is a process for implementing adaptive management (AM), which can integrate with the USACE feasibility study process and be of particular value in the case of ecosystem restoration projects (Fischenich et al., 2019). In an AM application, PrOACT becomes a loop wherein future iterations restart with revisiting and refining the problem statement. SDM supports both passive AM, where monitoring of outcomes is used to reduce uncertainties in the consequence and tradeoff analysis, and active AM, where optimization of alternatives includes explicit evaluation of the value of information and targeted action to reduce critical uncertainties (Williams, 2011).

One of benefits of SDM is that the framework is scalable and adaptable, which allows the scope and complexity of the decision—as well timing and budget constraints—to be considered when selecting the tools that are used to support the consequence analysis and tradeoff optimization components of PrOACT. This adaptability also limits constraints in terms of required data or model output. Expert elicitation may be used for consequence analysis as an objective, structured, and systematic process for evaluating consequences when data are unavailable, while more complex models and tools can be used where available and appropriate for the scope and timeline of the decision (Hammond et al., 1999).

Because SDM relies on transparency of objectives and other considerations in the decision-making process, however, it is ill-suited to supporting decisions where stakeholders may have objectives they are unwilling to discuss openly. In addition, SDM encompasses the complete decision-making process and may therefore not integrate seamlessly into applications where portions of that process are dictated by established legal or policy constraints, such as USACE planning studies. However, the flexibility of the approach can allow for modification and adaptation to these cases, such as incorporating those constraints into definition of the decision context and/or identification of objectives and alternatives.

## 2.4 METHODS FOR DECISION MAKING UNDER DEEP UNCERTAINTY

Another relevant set of approaches, related to MCDA/MODA and SDM, are methods for decision making under deep uncertainty (DMDU; Marchau et al., 2019). These are methods designed to inform decisions where the decision maker and key stakeholders do not know—or cannot agree on—predictions about future conditions that directly affect the success or failure of a project or plan. In these cases, DMDU methods are designed to help the decision maker identify strategies that are robust to uncertain future conditions (Lempert, 2019; Lempert et al., 2006). Robust strategies are designed to perform reasonably well (i.e., satisfice) under a wide range of plausible future conditions when faced with deep uncertainty (Lempert et al., 2003).

Robust strategies often include adaptive elements or multiple pathways that are pre-defined in response to observed conditions over time (Haasnoot et al., 2013). Adaptive strategies also include key signposts (Dewar et al., 1993) or “tipping points” that signal when the decision maker should switch from one pathway to another, or call an optional strategy component (Groves et al., 2015; Kwakkel et al., 2015). Key methods within this toolkit include Robust Decision Making (Lempert, 2019), Decision Scaling (Brown et al., 2012), Dynamic Adaptive Policy Pathways (Haasnoot et al., 2013, 2019), and InfoGap (Ben-Haim, 2019; Korteling et al., 2013).

DMDU methods often rely on exploratory analysis with simulation models, simulating hundreds to millions of plausible futures and evaluating the potential performance of strategies across the range of scenarios (Groves & Lempert, 2007). Sometimes referred to as “running the analysis backwards,” this process entails starting with a strategy, systematically stress-testing it against a range of conditions, and then determining in what conditions the strategy meets or fails to meet its goals (Lempert et al., 2013). The resulting dataset—an ensemble of “what if” projections—is then explored using visualization and clustering algorithms in order to identify key uncertainties that most often lead to strategy success or failure (Bryant & Lempert, 2010; Dalal et al., 2013; Groves & Lempert, 2007). These can then be described as “decision relevant” scenarios, intended to inform ongoing deliberations or the identification of additional strategy augments.

Recent advances have expanded this toolkit to simultaneously consider multiple objectives (i.e., Many Objective Robust Decision Making, or MORDM), and have also included “policy search” with optimization algorithms as part of the process to allow decision makers to consider tradeoffs among objectives within a set of potentially robust strategies (Hadka et al., 2015; Kasprzyk et al., 2013).

DMDU methods offer a number of potential advantages for USACE decision making. They have been developed and applied across a number of water resources applications, including water supply, reservoir operations and dam safety, and coastal and riverine flood risk management, and can already leverage models and approaches relevant for USACE decisions. They are intended to support participatory, iterative, and multi-objective planning under uncertainty, and address uncertainty at the outset of the decision analysis to ensure that it directly informs tradeoff analysis, policy prioritization, and selection. DMDU methods represent an emerging best practice for global water resources planning when accounting for uncertain climate change, land use, and other critical long-term drivers of change.

However, there remain several potential challenges to applying DMDU methods to a USACE planning study. First, these methods often require one or more simulation models representing the biophysical and/or economic and social systems of interest to evaluate potential outcomes under different scenario conditions or with various proposed alternatives in place. Even if such models exist or are developed or adapted in the course of a study, they must also be able to support ensemble analysis (i.e., hundreds to millions of model runs), which can present a significant technical challenge or require the use of high-performance computing resources. Given the technical and computational requirements, these methods may be difficult to implement within a time- and resource-constrained 3x3x3 USACE feasibility study while also meeting other study requirements.

In addition, these methods are intended to be iterative, with earlier round(s) of analysis contributing to revised project formulations that balance across multiple objectives and are more robust to identified key uncertainties. They also often point to adaptive approaches, as noted earlier, which can entail specifying different adaptive pathways as part of a single plan and delaying key investment decisions until there is greater certainty regarding which future scenario will come to pass. As with the technical constraints, both the iterative nature of DMDU analysis and the move towards adaptive strategies are not well aligned with current USACE project formulation, analysis, or implementation. Aligning USACE planning with DMDU best practices would likely require changes to USACE process beyond project evaluation, and this extends beyond the scope of the present case study analysis.

## 2.5 SOCIAL RETURN ON INVESTMENT AND PARTICIPATORY PLANNING

### 2.5.1 Using Social Return on Investment to Improve Assessment of Ecosystem Services and More Fully Account for Social Value

Natural ecosystems provide a number of functions that can potentially be used by communities as ecosystem services that have both economic and socio-cultural benefits. As noted in Chapter 1.0, human communities rely on natural ecosystems for a wide range of services, including: provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services, such as nutrient cycling and carbon sequestration (Millennium Ecosystem Assessment [MEA], 2005).

Additionally, this linkage between human communities and natural ecosystems is directly linked to community resilience. This linkage is often seen as one that is economic or market-based. The maintenance and restoration of ecological integrity and the use of green infrastructure, for example, has high potential for supporting community resilience through the provision of ecosystem services, such as reducing the direct impacts of waves, storm surge, and marginal erosion; providing essential habitat for juvenile and adult fisheries species, ducks, and other hunted species; and potential revenue raising functions such as nutrient and carbon sequestration (Carruthers et al., 2017). The valuation of these services is the providence of the methods discussed in Chapter 2.0.

However, the value generated by ecosystem services may extend beyond the typical microeconomic welfare concepts which measure individual willingness to pay. For example, healthy ecosystems have tremendous potential to generate a wide range of social and cultural outcomes, reduce inequality and environmental degradation, and improve community wellbeing. To the extent that, for example, survey respondents are either unaware or uncertain about the community-level outcomes supported by ecosystems, traditional valuation methods may fail to capture the full range of benefits, nor are they appropriate for estimating the economic impacts of nature-based projects.[4]

In order to examine impacts to community resilience and to more fully understand the spectrum of social, environmental, and economic benefits and costs, traditional ecosystem services valuation can be supplemented using a Social Return on Investment (SROI) analysis (Hemmerling et al., 2017a; SROI Network, 2012). Building on the concept of social accounting, SROI analysis is a conceptual and quantitative approach that incorporates social and environmental values into a traditional economic-only BCA that is conceptually familiar to policy makers and the general public (Pattison-Williams et al., 2018).

SROI is defined as “a framework for measuring and accounting for the much broader concept of value. It seeks to reduce inequality and environmental degradation and improve wellbeing by incorporating social, environmental, and economic costs and benefits” (Nicholls et al., 2012). The assessment process is based on understanding, measuring, and reporting the social, economic, and environmental value created by projects and policies and can be used to retrospectively measure outcomes that have already occurred (evaluative-type) or prospectively predict how much value will be generated if the intervention meets its intended outcomes (forecast-type). SROI is markedly different from the traditional concept of return on investment in that it accounts for both financial costs and benefits and the broader societal costs and benefits experienced at the local and regional levels (Varua & Stenberg, 2009). Such societal costs and benefits cannot typically be fully identified through purely quantitative means and models, though in principle many of them could be valued using the methods in Chapter 2.

SROI differs from a more traditional ecosystem valuation approach primarily in the participatory nature of the identification of the outcomes to be valued (including, but not necessarily limited to, ecosystem services), the potential inclusion of economic, social, and cultural outcomes, incorporating qualitative data into a narrative about how a project affects stakeholders, and the focus on the distribution of outcomes and impacts across different stakeholder groups. As such, it may be viewed as a complementary method that adds local context, richness, and narrative to standard valuation studies.

To fully understand the impacts of projects and policies requires meaningful engagement with multiple stakeholders and a representation of stakeholder benefits in ways that are unique to the stakeholders themselves (Banke-Thomas et al., 2015). In the SROI process, qualitative data analysis classifies differences in the ways stakeholder groups potentially impacted by ecological restoration projects engage with the project sites and identifies a suite of outcomes unique to each stakeholder group. Identifying these outcomes is integral to defining both the specific objectives and variables needed to develop a comprehensive monitoring framework (Hemmerling & Barra, 2017). Project outcomes are derived from

[4] Economic impacts are changes in observable outcomes related to gross regional product (such as output, employment, and value-added), and are distinct from the welfare implications (as measured by, e.g., willingness to pay) of a project.

direct engagement with local knowledge experts and then financial proxies assign a value to these outcomes, based on valuation traditions in environmental and health economics fields. Suitable financial proxies are generally identified via primary data collection, for example, average wages and cost-per-hire data, and through an exploratory desk-based literature search of academic, public sector and social enterprise publications relating to the measurement of social impact (Watson & Whitley, 2017).

Traditionally viewed as a shortcoming in social valuation studies, recent advances have led to the development of a wide range of financial proxies that allow for a deeper understanding and assessment of the value of subjective wellbeing (Fujiwara, 2019; Trotter et al., 2014). When these financial proxies are inputted alongside the relevant outcomes identified through direct engagement with impacted stakeholders, the SROI process allows for a direct calculation of the value created by an outcome for each stakeholder group. Subsequent analyses allow calculation of a singular, socially inclusive return on investment ratio that captures both positive and negative outcomes (Watson & Whitley, 2017). This knowledge can help to bound the uncertainty of a purely quantitative analysis and therefore makes it useful in setting public policy and making cost-benefit decisions between different environmental interventions. The suite of methodologies used in this research can be translated into a longer-term monitoring program, tracking where and how different economically and geographically situated communities are unequally impacted by the changing material conditions that accompany restoration projects, with NBS or otherwise, over time.

### 2.5.2 Building Social Equity and Justice through Engagement of Impacted Stakeholders in Ecosystem Services Assessments

The inclusion of potentially impacted stakeholders is a key aspect of the SROI process which recognizes residents as unique sensors of their environment. In mapping potential outcomes of projects and policies, SROI uses public participation to understand and integrate the local, traditional, and historical ecological knowledge of local knowledge experts. This is fully in keeping with federal guidelines that call for the cooperative conservation of the landscape, including Presidential Executive Order 13352, which calls for the Departments of the Interior, Agriculture, Commerce, and Defense and the Environmental Protection Agency to implement laws relating to the environment and natural resources in a manner that promotes cooperative conservation, with an emphasis on appropriate inclusion of local participation in federal decision making.

NBS projects, which are expected to generate a suite of market and nonmarket-based outcomes, can be enhanced by the types of cooperative conservation methods typified by SROI. From the included stakeholders, information such as inputs required for project implementation (costs, time, etc.), perceived changes experienced by the stakeholders as a result of the project, outcomes benefited or otherwise from the intervention, duration of the outcome, relative importance or prioritization of these outcomes, changes likely to have occurred in the absence of the intervention and other factors contributing to the changes identified are required elements of an SROI impact map (Banke-Thomas et al., 2015).

Although a combination of numerical modeling and economic analyses can predict some of the expected changes that will likely occur as a result of NBS, there are other impacts to the ecosystem and the human communities that are much more complex and difficult to predict (Peyronnin et al., 2017). For example, cypress reforestation projects in Louisiana have been found to provide new educational opportunities for neighborhood schools as well as an opportunity for nearby Indigenous residents to recover a portion of their cultural heritage lost when the cypress forests were denuded (Hemmerling et al., 2017a). Yet in some cases, there may be negative externalities associated with the projects ranging from safety issues related to increased vehicle traffic during construction to the impacts of changing viewscapes and land uses on neighboring landowners. Such impacts are highly uncertain and not readily modeled, even though in theory they could be valued by nonmarket valuation methods. So, while it is acknowledged that the use of NBS to restore ecosystems and protect communities has the potential to impact local communities and economies both positively and negatively, the extent of these impacts is not as well known, which precludes their monetization via more traditional methods.

This situation is beginning to change as improving technology and methodological advances are increasingly allowing for the input of qualitative local knowledge into mathematical models. Innovations in geospatial technologies, a growing acceptance of mixed methods research, and an increasing awareness of the validity and importance of local knowledge are driving many of these advances (Curtis et al., 2018). There is a growing body of literature on the potential of combining local knowledge systems with technical scientific knowledge to manage both ecosystems and resources, including the evaluation of climate change impacts and the management of fisheries, biodiversity, and landscape dynamics (Folke et al., 2005). These advances have provided tangible ways to evaluate the potential outcomes and shortcomings of ongoing and planned restoration and protection projects, allowing coastal planners to make adjustments that respond to the real-time needs of impacted communities (Hemmerling et al., 2020; Hemmerling & Barra, 2017).

Recent restoration work conducted in coastal Louisiana, for example, used qualitative research to inform the calculation of economic, recreational, cultural, educational, and ecological values of ecological restoration projects on numerous stakeholder groups (Hemmerling et al., 2017b, 2017a). Interviews, survey methods, and focus groups were centered around these discrete topics to develop a consistent analysis across groups and a framework for future research and monitoring. Conversations with participants were analyzed to determine which qualities or concerns were important to participants as well as how they weighted different social and environmental values derived from the restoration projects. The qualitative data derived through this process can provide new insight into the social impacts and outcomes of NBS projects that cannot be gained through traditional scientific approaches and identify potential inequities in the distribution of costs and benefits (Hemmerling et al., 2020).

### 2.5.3 Challenges and Opportunities for USACE Planning Studies

Despite the increasing legitimacy of incorporating local knowledge into the planning process, however, it has not been broadly implemented at scale. This can be attributed in part to the perceived difficulties in achieving scientifically rigorous, replicable, and widely accessible methods of qualitative data collection. In large part, projects that have taken such an approach have been wholly qualitative in nature, which though valid, are still not as readily accepted by scientists, engineers, and policy makers. Even in collaborative management processes specifically designed to integrate local and technical knowledge, it is often difficult for scientists and engineers who are trained to consider data as something to be input into models rather than something shaped by sociocultural processes to interpret local knowledge as anything other than "public input" (Barra et al., 2020).

In spite of these challenges, it is necessary that planners account for these data in a fully integrated manner in order to build a stronger evidence base and guide the development of significant water resources projects. USACE explicitly states that project planners should learn from stakeholders with diverse perspectives and incorporate the scientific, technical, and social information that these stakeholders are able to provide into the planning process, specifically defining stakeholders as any member of the public that might be able to affect, are affected by, or are interested in, the results of the Corps' planning process (USACE, 2019). Research suggests that when residents and other members of the public are not included as full partners in the planning process and their local knowledge not incorporated, the risk of introducing or exacerbating social justice issues increases (Hemmerling et al., 2020). Further, if residents' voices are heard but do not impact the process, then the process will fail to even begin to address deep-seated justice issues. By incorporating data derived from two-way dialogue with local knowledge experts into the planning process, project managers will be able to more effectively adapt to local needs and changing circumstances, particularly when knowledge is transferred horizontally between stakeholder groups and vertically to higher institutional levels (Zedler, 2017).[5]

In order to quantify locally specific social impacts and develop a framework amenable to measuring social change resulting from the use of NBS, USACE recommends that project planners utilize stakeholders' data and knowledge to help identify the problems, opportunities, objectives, and constraint, recognizing that local knowledge is critical for inventorying and forecasting future conditions in the area (USACE, 2019). Recent assessments of NBS and other restoration projects have found that qualitative data collected from focus groups, surveys, and one-on-one interviews with key stakeholders and analyzed in a scientifically rigorous and replicable fashion can be used to develop empirically grounded forecast and retrospective assessments of protection and restoration projects (Hemmerling & Barra, 2017). Empirically derived information on residents' perceptions of the values—positive, negative, or neutral—of NBS projects grounds anticipated social impacts in the material experiences of the residents themselves. While USACE retains final decision-making responsibility for actions within its authority, stakeholder engagement, collaboration, and coordination will improve the quality of that decision-making and increase the legitimacy of the decision reached, resulting in actions that are implementable and sustainable (USACE, 2019).

[5] A related issue is the belief that a technical economic or related analysis is purely "objective," in the sense that multiple qualified practitioners, given the same data, would produce identical results. In practice, however, many subjective decisions are made over the course of an analysis (such as appropriate parameter values, expectations about future economic conditions, distributional assumptions about uncertainty, the appropriate discount rate, etc.) that could significantly impact the results. Interaction with stakeholders is one way to ensure that such subjective decisions are aligned with the values and preferences of the potentially affected communities.

# 3.0 INCORPORATING ECOSYSTEM SERVICE VALUATION INTO BENEFIT-COST ANALYSIS

## 3.1 OVERVIEW

The previous chapter described decision-support approaches such as MCDA/MODA, SDM, and DMDU that can consider tradeoffs across multiple objectives without necessarily monetizing the outcomes, and SROI that has the potential to augment these techniques through increased participation of stakeholder groups in defining their outcomes of interest (including economic, environmental, and social outcomes) and add narrative context about the likely impacts of projects or programs.

This chapter builds from these approaches to outline a method for integrated, multi-objective project evaluation that incorporates BCA as one component of a more holistic approach to considering salient benefits and costs of projects. Given feasibility constraints, uncertainties, information gaps, and other hurdles, no analysis can be completely comprehensively and equivalently applied to all relevant benefits and costs. Therefore, an analysis "funnel" is proposed that walks analysts through an open and transparent analysis process that quantifies and monetizes as many benefits and costs as are feasible, but also provides opportunities to incorporate concepts from the other decision-support approaches discussed in this report.

The analysis funnel provides a structure in which a wide array of relevant benefits and costs are considered conceptually, with methods applied to 1) measure non-monetizable categories (usually measured in physical units), and 2) monetize those that can be, according to what is appropriate and possible. In this way,  projects can be holistically evaluated (using all levels of the analysis funnel) even if some benefits and costs cannot be monetized.

## 3.2 "FUNNEL" CONCEPT

The concept of an analysis "funnel" is used to organize the key steps in incorporating cobenefits into a systematic project benefits evaluation.[6] The analogy is apt because during the first steps of the process (including defining the system[s] of interest and identifying the hypotheses of casual changes from project to benefits and costs from a particular service), the options including benefits are broad and are narrowed

[6] The funnel analogy is borrowed from the marketing literature, in which potential customers are at the "top" of the funnel, and each step in the marketing process moves a certain percentage of those in the current step to the next step, with the objective of completing a sale (the "bottom" of the funnel). The U.S. Army uses the funnel concept for their marketing activities. In this case, the "sale", or ultimate objective, is the monetization of the benefits and costs; however, not all project costs and benefits can be monetized. As such, the "top" of the funnel is a comprehensive but non-quantitative conceptualization of all the benefits and costs of the project, while the middle of funnel excludes those benefits and costs for which there is little to no data for use in quantification. The bottom of the funnel includes the most restrictive set of benefits and costs which can be monetized.

to match the context and constraints of the project and/or study. Figure 3-1 summarizes this conceptual approach.

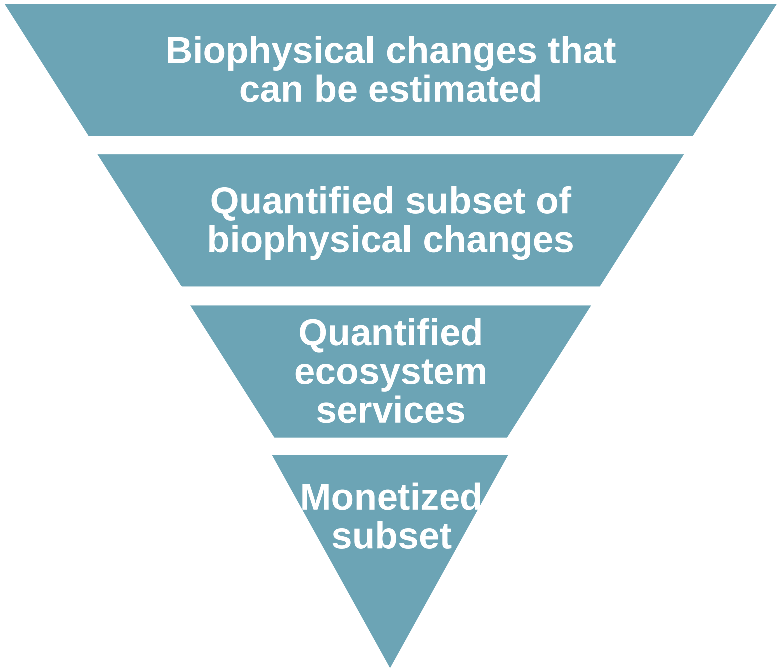


*Figure 3-1. Flowchart showing analysis funnel concept*

As the analysis proceeds, data and information become limiting factors, and only a subset of the identified outcomes can be physically quantified, and of those, only a smaller subset are likely to be accurately monetized. Thus, the subset of monetizable outcomes at the bottom (narrow part) of the funnel is potentially quite a bit smaller than the set of hypothesized outcomes defined at the top. But by proceeding in this step-by-step fashion, analysts provide a clear and transparent picture of the likely *qualitative* effects of a project through the causal chain/impact map/theory of change, provide as much information as is feasible about the *quantitative* physical effects of the project, provide as much information as is feasible about the welfare effects of the project, and perhaps most importantly, provides information about what is missing during each stage of the process. It should also be noted that this process could easily accommodate concepts from MCDA/MODA, SDM, DMDU, and SROI as either complementary stand-alone analyses or by incorporating some of their principles into the steps documented in the next section.

## 3.3 KEY STEPS TO INTEGRATE COBENEFITS WITH BENEFIT-COST ANALYSIS

In the subsections that follow, the key steps (progressing from the top to the bottom of the analysis funnel) are discussed for integrating ecosystem services and other cobenefits with BCA:

1. Define the system(s) of interest
2. Identify the conceptual hypotheses for potential biophysical and cultural/social changes
3. Determine appropriate estimation approaches for biophysical and cultural/social changes

4. Determine the analysis approach for the biophysical and cultural/social changes that can be quantified
5. Monetize the subset of changes that are suitable for quantification and valuation.

Although most ecosystem services and other cobenefits are not traded in formal markets in the same ways as man-made private goods, there is still a demand side and supply side that must come together in order to create benefits and costs (Mandle et al., 2020). Ecosystem services and other cobenefits must be linked to an identifiable set of human beneficiaries; changes in conditions or processes of ecosystems that cannot be linked to the welfare of identifiable beneficiary groups are not "ecosystem services" per se.[7] Furthermore, changing the supply of a *potential* ecosystem service is not sufficient to generate benefits and costs. Instead, there must be a lack of physical and institutional access constraints that prevents stakeholders from realizing those benefits.

The following subsections detail the recommended steps to incorporate ecosystem services and other cobenefits into BCA.

### 3.3.1 Define the System(s) of Interest

The first step in incorporating cobenefits, especially those associated with ecosystem services, is to conceptually define the system of interest. In particular, clearly documenting the boundaries and interlinkages of the biophysical system(s) that are expected to change and the human system that might be affected by those changes provides limits to the scope of any analysis, and a conceptual framework that represents the supply and demand sides of ecosystem service provision.

When distributional considerations are important, subpopulations that are hypothesized to be significantly or differentially affected by a project should be identified and engaged early. Involving and engaging key stakeholders that are expected to be affected by a project is vital to understanding what changes may occur as a result of the project. Direct engagement can be used to help articulate how change is created and then evaluate this through evidence gathered, recognizing positive and negative changes as well as those that are intended and unintended. The social value framework focuses on answering five key questions to define the system(s) of interest (Table 3-1). This is an important step in identifying what gets measured and how it is measured.

[7] Per the SROI method, it is also the case that subgroups of stakeholders may not be affected by or value the same change in the same manner, nor may they face the same access constraints.

*Table 3-1. The social value framework*

| Question | Definition |
|---|---|
| Who changes? | Taking account of all the people, organizations, and environments affected significantly |
| How do they change? | Focusing on all the important positive and negative changes that take place, not just what was intended |
| How do you know? | Gathering evidence to go beyond individual opinion |
| How much is you? | Taking account of all the other influences that might have changed things for the better (or worse) |
| How important are the changes? | Understanding the relative value of the outcomes to all the people, organizations, and environments affected |

*Source: SROI Network (2012).*

### 3.3.2 Identify Conceptual Hypotheses for Potential Biophysical and Cultural/Social Changes

Once the coupled natural-human system has been conceptualized and boundaries are established, analysts should map the expected biophysical changes (to, e.g., natural capital stocks) to changes in goods and services provided or available to the identified subpopulations. These changes should be identified relative to a well-defined baseline (or set of baselines to account for uncertainty). This baseline should include appropriate assumptions for both the ecological and the economic system. In addition, it should be noted that depending on the nature of the goods and services, the "serviceshed" of each may differ. For example, pure public goods or bads with global scope (e.g., atmospheric carbon) theoretically affect the world's population; however, depending on system boundaries established in the first step (3.3.1), only a subset of this population may be relevant. Conversely, provision of a purely private good (e.g., recreational opportunities on private land) may have a limited serviceshed of just one household.

Benefit Relevant Indicators (BRIs; described in more detail in Chapter 4.0) are of particular use in conceptualizing ecological changes. BRIs are "things valued by people….because they have a direct causal impact on human welfare" (Olander et al., 2015, 2017a). In providing a link between ecological change and the human system, both supply and demand-side considerations are considered. In some cases, BRIs may be used in conjunction with willingness-to-pay information in order to monetize benefits and costs associated with biophysical and cultural/social changes.

Causal chain diagrams are a convenient way to map the assumed or expected effects of a management change to changes in human welfare (see Figure 4-1). A causal chain diagram is a visual representation of the causal links between an action that changes ecological function in some manner, and the change in ecological function changes the provision of one or more ecosystem services (perhaps as measured by BRIs).[8] The ecosystem services changes are the physical "quantities" that generate benefits or costs, and

[8] A similar causal chain can be used for social changes.

preference information (in the form of willingness to pay) can be used to value changes in those quantities. Consistent with SROI, different stakeholder groups could be incorporated, and the causal chain diagrams could be assembled using stakeholder information.

### 3.3.3 Determine Appropriate Estimation Approaches for Biophysical and Cultural/Social Changes

Once the conceptualization of changes has been developed, the next step is to determine the feasibility of estimating the changes in ecosystem function and ecosystem services expected to be causally related to the management action. In some cases, available evidence about the linkages can be strong, suggesting that past research has either quantified or provided good theoretical evidence of the management action to ecosystem function to ecosystem service links. In other cases, one or more of these links may be missing due to a lack of statistical or structural models describing the causal chain (Olander et al. 2017).

This evaluation can lead to several outcomes. For some project-driven impacts, it may be possible to quantify ecosystem service changes relative to the baseline and a formal estimate of ecosystem cobenefits. In this case, these benefits continue through the analysis funnel. However, there may be strong empirical or qualitative evidence supporting project-driven biophysical outcomes without a means to robustly calculate a relevant metric to quantify that change. In these cases, decision support approaches described in Chapter 2.0 (e.g., MCDA/MODA) can be considered, including the potential to qualitatively evaluate outcomes through expert elicitation. Alternatively, narratives and supporting documentation can be developed to characterize these benefits and augment quantitative metric calculation and BCA in project evaluation.

### 3.3.4 Determine the Analysis Approach for the Biophysical and Cultural/Social Changes that can be Quantified

Closely related to the feasibility of estimating biophysical changes is the means of doing so. At this stage, biophysical and cultural/social changes are evaluated to determine what appropriate quantitative metrics exist and can be calculated to evaluate project benefits and costs. Because biophysical changes in ecosystem services are "quantities" in BCA, quantitative estimation is needed to use this approach. This suggests using new or pre-existing empirical or structural models or relationships to precisely estimate changes in BRIs that are cardinal in nature. In some cases, more than one option may be available. Benefits that meet these criteria can continue to the next level of the analysis funnel.

In cases where cardinal estimation is not possible (due to, for instance, a lack of required input data, missing links in the casual chain, or a lack of available models or relationships), decision-support methods that require only physical information, such as MCDA/MODA, SDM, and DMDU, can be used to evaluate tradeoffs. In these cases, quantitative metrics that can be calculated based on available information can be used in the analysis, and/or supplemented with categorical evaluation and expert elicitation. In this manner, these benefits and costs can be considered as part of a comprehensive evaluation of project outcomes rather than being strictly limited by constraints on available information to support a BCA.

### 3.3.5 Monetize Subset of Changes Suitable for Quantification and Valuation

For those ecosystem service changes for which cardinal, quantitative estimates are available and for which willingness to pay or other welfare information is available, changes in cobenefits (relative to the baseline) can be monetized (see Chapter 5.0). When using secondary valuation techniques (such as benefit transfer), analysts should be cognizant of differences across study contexts (including, but not limited to, differences in serviceshed, quantities, the available of substitutes) that might affect unit-level (e.g., willingness to pay per household) and overall (the sum of individual willingness to pay across the serviceshed) value. Given uncertainty over both physical and monetary values, assumptions should be clearly stated and robustness/sensitivity analysis should be used to both indicate this uncertainty and test the sensitivity of the analysis to alternative assumptions (Johnston et al., 2021).[9]

For the subset of effects that can be quantified and monetized, the sum of individual or household willingness to pay can directly enter the BCA as a benefit of the project. Negative benefits should be entered as costs.

## 3.4 SUMMARY

Limiting project evaluation to those benefits that can be monetized through a BCA—and for which the requisite data are available to perform those calculations—excludes important considerations relevant to the welfare of communities and subcommunities affected by management actions. As such, it is worth the additional effort and cost to incorporate information about changes in cobenefits when that inclusion has the chance of leading to a different decision than if those cobenefits were excluded from the analysis. Here, a funnel methodology has been developed that enables more comprehensive consideration of project costs and benefits regardless of constraints on available input information that may inhibit comprehensive evaluation with BCA:

- Define the system(s) of interest
- Identify the conceptual hypotheses for potential biophysical and cultural/social changes
- Determine appropriate estimation approaches for biophysical and cultural/social changes
- Determine the analysis approach for the biophysical and cultural/social changes that can be quantified
- Monetize subset of changes suitable for quantification and valuation.

Methods such as the identification and evaluation of BRIs (described in more detail in Chapter 4.0) enable quantification of benefits that cannot be monetized. Decision analysis approaches such as MCDA/MODA (and other techniques described in Chapter 2.0) allow the outcomes of that analysis to be

[9] A complement to this approach, especially when comparing projects, is to ask what the value of the non-monetized benefits or costs would need to be in order to change a decision. Often, this more limited information can be instructive for decision-makers who may not have precise valuation information but have a sense of "orders of magnitude".

incorporated along with categorical metrics, including those derived from expert elicitation, as part of tradeoff analysis. These results can then be incorporated along with BCA as part of a project evaluation approach that more comprehensively evaluates the range of impacts to communities and the environment.

# 4.0 EVALUATION OF ECOSYSTEM SERVICES THROUGH NON-MONETIZED METRICS

## 4.1 OVERVIEW

The integrated evaluation "funnel" described in Chapter 3.0 provides a mechanism through which project benefits, including for NBS, can be more comprehensively evaluated in cases where data and methods are not available or appropriate for monetized valuation. Quantifiable metrics provide one mechanism for evaluating non-monetized benefits as part of this integrated approach, recognizing that metrics can also be complemented with techniques such as expert elicitation as part of a comprehensive approach to evaluating ecosystem services.[10]

One category of metric that is particularly relevant to NBS is BRIs. BRIs quantify how ecological changes are relevant to humans so as to provide greater insight into tradeoffs (USACE, 2016). BRIs provide a mechanism through which data and information can be translated into metrics that quantify direct impacts to humans. For example, a dam's impact can be quantified by combining 'volume of water storage' (which does not inherently tie to societal value) with information about rate or type of human use to generate 'period that a particular user would be supported by available water'. The generated BRI is the period of time a user can be supported, which directly quantifies the 'value' of the stored water to the end user (Olander et al., 2017b). BRIs build on previous approaches that link an ecosystem function (ecological resource) with the human beneficiaries (Borsuk et al. n.d.). They aim to demonstrate and quantify benefits as a direct result of increases or decreases in an ecological function (Olander et al., 2017b, 2018).

BRIs are closely linked to the theory and methods presented in Chapters 2.0 and 3.0. As the empirical measure of ecological endpoints or final goods and services, they are the quantities associated with changes directly valued by stakeholders. As such, they are measures that, in theory, could be used to represent changes in ecosystem services or social impacts generated by a project. BRIs can also be used in MCDA/MODA, SDM, and DMDU as measures against which tradeoffs can be illustrated, even without monetization. Finally, the causal chain diagrams (example provided in Section 4.2) that incorporate BRIs are essentially hypotheses/descriptors of the impact map or theory of change for a particular ecosystem service, which is both a key component of SROI (SROI Network 2012) and a component identified in the integrated evaluation "funnel" (see Section 3.3.2).

In the subsequent sections, an approach to developing BRIs and ecosystem function metrics is described and the USACE policy context for use of non-monetized metrics in project evaluation is reviewed. A set of potential metrics that can be used in project evaluation and that are consistent with USACE policy guidance are then provided.

[10] It should be noted that quantifying biophysical or sociocultural changes is also a prerequisite for monetization.

## 4.2 APPROACH TO DEVELOPING BENEFIT-RELEVANT INDICATORS

Figure 4-1 shows a framework to identify BRIs (Olander et al., 2018). The process begins with the action that is driving ecosystem change, such as the implementation of a NBS. The change in ecosystem function that action will drive is then identified along with indicators that quantify those impacts. Next, BRIs are identified that combine those indicators with other data to capture the change in ecosystem services. It is recommended that developed BRIs are 'SMART', that is Specific, Measurable, Achievable, Relevant, and Timeline-realistic (Table 4-1). From there, the impacts of the action in terms of social value are evaluated through a benefits assessment based on stakeholder values and preferences (e.g., the relative importance of an ecosystem service to the local community).

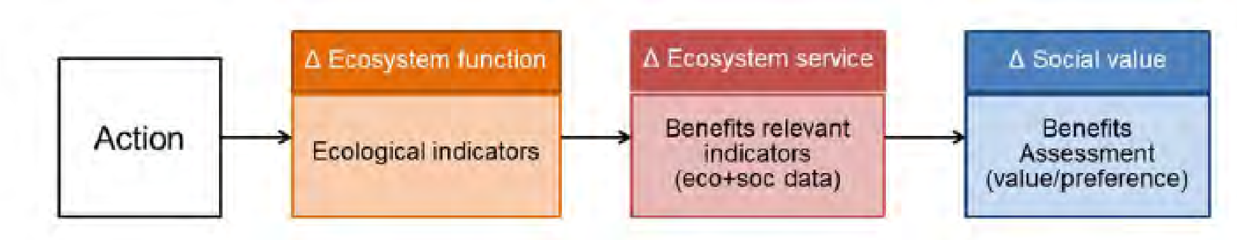


*Figure 4-1. Ecosystem service causal chain with BRIs. Adapted from Olander et al. (2018).*

*Table 4-1. Table of 'SMART' guidelines for development of most informative BRIs*

| | Status of Ecosystem Function | Change in Ecosystem Function | Change in Beneficiary Realized Ecosystem Service |
|---|---|---|---|
| **Specific** | Clear metric with established protocol | Metric still relevant before and after change has occurred | Change in ecosystem function can be directly linked to a change in availability of a related ecosystem function |
| **Measurable** | Statistically valid and known technology, method | Change reasonably expected to be large enough to measure | A rate of use can be estimated so that change in ecosystem service to beneficiary can be quantified |
| **Achievable** | Cost appropriate, effort appropriate, method available | Can be feasibly re-measured after project implemented | Beneficiary data is available to quantify use rates of ecosystem service |
| **Relevant** | Change can be linked to an ecosystem service or beneficiary | Scale of change is relevant to delivery of service to beneficiary | The beneficiary will actually increase or decrease benefits with the change in ecosystem function |
| **Timeline realistic** | Responsive at timescale of project implementation / project lifespan | Rapid response, i.e., not a very long temporal lag between project implementation and change in ecosystem function | Rate of change appropriate to reasonable expect a change in behavior to the change in ecosystem function |

The number of candidate metrics associated with the ecosystem causal chain reduces at each step while the uncertainty rises, which provides some challenges to implementation as part of integrated project evaluation. One approach is to move beyond static components and individual directional relationships of ecosystem services and beneficiaries (both positive and negative) to more comprehensive process based models and analytical frameworks (Milner-Gulland et al., 2014). These frameworks can identify key metrics that quantify the interaction between ecosystem resources and beneficiaries and how the rate of change in ecosystem services influences change (and rate of change) in beneficiary use of the ecosystem service. However, it should also be noted that metrics identified at any point in the causal chain can be used as part of multi-objective analysis (Chapter 2.0) and the project evaluation ‘funnel’ (Chapter 3.0) if relative value can be established.

Approaches have also been implemented that avoid the need for direct measures of ecosystem functions, instead surveying households as to how much they depend upon ecosystem services before and after an event, such as a natural disaster, and then calculate an index of human well-being (Yang et al., 2015). That approach assumes that the resource user will use the ecosystem function to the maximum level that can be supported. Therefore, reduction in use reflects a reduction in the quantity of the ecosystem function.

Another linkage approach is geospatial and/or contextual so that the demand for using the ecosystem service can be quantified (Olander et al., 2017b; Ouyang et al., 2016). For example, if the ecosystem function is the abundance of fish in a stream, quantifying where beneficiaries are located, how frequently is the stream fished, or how many people benefit from the ecosystem services allows quantification of the change in total benefits that could be predicted from an increase or decrease in the ecosystem function.

A third form of linkage that can be captured in a BRI is an ecological flow or process, where the change has a flow on indirect ecological effect that may ultimately impact upon the beneficiaries, these linkages are generally more difficult to quantify, due to knowledge and data gaps (Olander et al., 2017b). An example of an ecological linkage would be a pollutant (such as a heavy metal) being added to a stream from a project which is bioaccumulated and renders a human-consumed upper predator fish population unfit for consumption. There could be geospatial and temporal disconnects between the project causing the direct ecosystem change (e.g., heavy metal concentration) and the change in the ecosystem resource (e.g., edible fish).

## 4.3 USACE CONTEXT FOR BRI IMPLEMENTATION

There is an extensive history of valuation of ecosystem services through non-monetized metrics within USACE (0), going back as early as 1969 and the creation of the Institute for Water Resources to develop planning methods and analytical tools to address economic, social, institutional, and environmental needs related to water resources. Building on this foundation, USACE and the Obama Administration developed a supporting policy framework in 2013 for more comprehensive project evaluation framework that considers a broader range of societal impacts. As discussed previously in Ehrenwerth et al. (2022), the Principles, Requirements, and Guidelines for Federal Investments in Water Resources (PR&G) established a specific goal for federally implemented projects to be resilient, enduring, and sustainable with respect to social and environmental factors in addition to economics (USACE, 2020). The 2020 Appropriations Act called for USACE to finalize guidelines for interagency implementation of the PR&G (USACE, 2020), and USACE is currently pursuing this through rulemaking (*Federal Register*, 2022-11881). Overall, this provides a high-level policy environment driving towards more comprehensive project outcomes that assess economic, social, and environmental benefits and costs.

In 2019, the USACE Chief of Engineers' Environmental Advisory Board (EAB), a Subcommittee of the Army Science Board, was asked to identify approaches to more fully capture environmental benefits resulting from civil works projects to inform the project evaluation process (USACE, 2020). For water resource projects, the EAB concluded that incorporation of additional metrics in decision making for plan selection, prioritization, and budgeting, as well as design, construction, operations and maintenance, and monitoring and adaptive management, would advance USACE missions and better address the water resource needs of the nation. Where feasible, metrics would be objective and quantitative with the potential to improve decision making and communicate broader project benefits (USACE, 2020).

Specifically, the EAB recommended the following framework of five categories of metrics and four principles for the development of metrics (USACE, 2020):

1. Ecosystem goods and services (e.g., food web, nutrient cycling, supporting or provisioning services, cultural, floodwater storage);
2. Public interest (e.g., reduction and control of societal risks);
3. Restoration potential (e.g., ecosystem/environmental lift by quantity and quality);
4. Landscape and watershed (e.g., connection between watershed characteristics and aquatic ecosystems, local/regional/national/international connectivity); and
5. Sustainability/threat (e.g., natural resilience [lifespan], adaptability, self-sustaining).

The EAB-recommended principles for the development of metrics were that they should be:

1. Quantitative to semi-quantitative, or at least objectively scalable
2. Compatible with business line objectives;
3. Adaptable for consistent cross-comparison and decision-making; and

4. Capable of indicating either beneficial or adverse effects (including tradeoffs).

The study team took these recommended principles into account when identifying metrics for the case study analysis. Please see the forthcoming case study analysis report for a more detailed discussion of study-specific BRIs.

# 5.0 VALUATION OF ECOSYSTEM GOODS AND SERVICES

## 5.1 OVERVIEW

This chapter provides an introduction to ecosystem service valuation using methods for economic analysis that can be used as part of the integrated evaluation approach described in Chapter 2.0. A more comprehensive accounting of the potential ecosystem service benefits from a project will improve decision-making and illustrate the dimensions that will likely change as a result of the project. The chapter begins with an overview of the economic theory for measuring well-being, and then provides a short discussion about the challenges of estimating and monetizing biophysical changes as a result of a project or intervention. The chapter then discusses the main methods used in environmental economics to empirically estimate changes in value, as well as some alternatives that can be used when the use of these methods is infeasible. A final subsection concludes.

## 5.2 WELFARE BASIS OF NON-MARKET VALUATION

The value of a good or service, or a bundle of goods and services, is defined by what one is willing to relinquish in order to obtain it (willingness to pay, or WTP), or what one is willing to accept to forego it (willingness to accept, or WTA). This value is a measure of the (gross) benefit of the good or service to an individual and is not the same as what the individual has to pay in order to obtain it (i.e., the cost). This value measure need not be monetary (though monetization, where possible, is often a convenient normalization for purposes of analysis). However, it does require a reference good (what is being foregone or received). The sum of the values across all individuals that consume or use the good or service is a measure of the overall benefits provided by it, though this conceptualization does not account for differences in the value of the reference good across individuals.

Willingness to pay is the concept that drives the more familiar "demand curve" in economics.[11] In particular, points along a demand curve are representations of the incremental, or marginal, willingness to pay for a particular good or service. For example, suppose Figure 5-1 represented a community's demand for saltwater marsh habitat, and the amount supplied was $Q_1$ (measured in acres). The value of the $Q_1$st acre of habitat is given by the demand curve, which represents the community's willingness to pay for that acre.

If a project were to increase the amount of saltwater marsh habitat to $Q_2$, then the overall benefit to the community is the sum of their willingness to pay for the additional $Q_2$-$Q_1$ acres, as shown in the shaded area in Figure 5-1. In this case, because the demand curve is downward sloping, each additional unit of marsh habitat is valued less than the previous unit. The shaded area (known as consumer surplus) is thus a theoretically consistent measure of the benefits to the community of increasing marsh habitat that can be

[11] Strictly speaking, willingness to pay (or compensating variation) is calculated from the Hicksian demand curve that only accounts for substitution effects.

compared with the costs of providing it. If the benefits are larger than the costs, then provision of the additional marsh habitat is desirable (in terms of community welfare).[12]

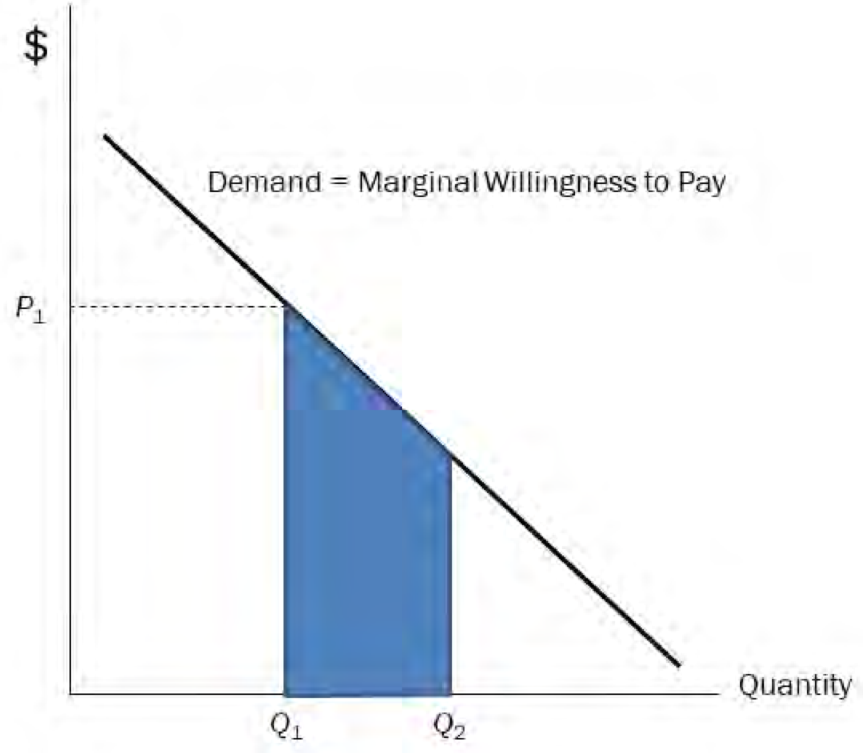


*Figure 5-1. Demand and Willingness to Pay. Source: Author's interpretation. The incremental, or marginal willingness to pay for the $Q_1^{st}$ unit is $P_1$. The total willingness to pay, or value, of moving from $Q_1$ to $Q_2$ is the shaded area (measured in dollars).*

Natural capital produces a *flow* of ecosystem services over time, just as man-made capital produces multiple periods of benefits. Each period's benefit can be measured by willingness to pay as in Figure 5-1. However, because of the opportunity costs and risks associated with future streams of benefits and costs, it is necessary to discount future flows in order to make meaningful comparisons across time. Exponential discounting provides the time-consistent means of comparing monetary values across time.[13] However, any positive discount rate necessarily implies that at some point (depending on the rate itself),

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[12] Note that this does not necessarily imply that all members of the community will experience an increase in welfare, rather that welfare increases in aggregate.

[13] Exponential discounting for discrete time periods takes the form $V_t/(1+r)^t$, where $V_t$ is the (nominal) value of a flow in time $t$, $r$ is the constant real discount rate, and the perspective is from time $t$=0. Time consistency refers to the property that only relative time increments matter when comparing values in different time periods.

future flows will be worth approximately zero; the larger the discount rate, the less the discounted benefit in any future time period.

A known challenge when comparing monetary flows across time (including evaluation of ecosystem services) is determining an appropriate discount rate. Per the Water Resources Development Act of 1974 (WRDA 1974; P.L.93-251), USACE uses a discount rate for water planning projects calculated using the (rounded) average yield on Treasury, with restrictions on annual changes from year to year (Carter & Nesbitt, 2016). The rate of treasury yields is presumed to represent the "risk-free" rate of return, as treasury bonds are backed by the full faith of the U.S. government. The executive branch, however, uses a different rate per the OMB's Circular A-94 for investments and regulations designed to represent the marginal pretax rate of return in the private sector. Certain executive branch analyses (including cost-effectiveness analysis, lease-purchase analysis, internal government investments, and asset sale analysis), however, use Treasury borrowing rates. OMB A-94 also recommends using sensitivity analysis to explore how analysis results changes with key assumptions, including the discount rate. Since the year 2000, the discount rate for USACE water projects (just under 3 percent in 2020) has been below the OMB rate (7 percent; see Ehrenwerth et al. [2022], Figure 6).

## 5.3 CHALLENGES IN NON-MARKET VALUATION

There are a number of challenges in applying this theory for valuing changes in ecosystem services or natural capital to particular interventions. Valuing environmental changes is an inherently multidisciplinary process involving estimation of both changes in ecosystems (in a physical sense) and the monetary value of those changes. Although many techniques and methods have been developed for both, predicting changes in ecosystem services tend to be complex, context-dependent (in terms of the physical environment and the subpopulations it affects), and requires a considerable amount of information. As such, feasibly predicting changes is often subject to a great deal of uncertainty, with tradeoffs between accuracy of the predictions and the costs of producing them.

Nevertheless, purposefully excluding particular benefit categories due to a lack of information necessarily underestimates the benefits of a project (by implicitly setting their value to zero), which introduces bias. For example, negative externalities associated with disrupted viewsheds or noise pollution are often not factored into the analysis. In addition, traditional infrastructure BCA may also be challenged by significant uncertainty, such as underestimated engineering and construction costs.

In the next two subsections, specific challenges in estimating biophysical changes as well as their monetization is briefly discussed.

### 5.3.1 Estimating Biophysical Effects of Complex Systems

The first step in estimating the change in ecosystem service values resulting from a project or program is to estimate the changes in natural capital and associated services as a result of the management action. While research into the biophysical relationships of various ecosystems continues to grow, there have been several challenges identified in the literature that have raised the costs of incorporating ecosystem service values into formal BCA used by federal, state, and local decisionmakers. These include (see Olander, Bagstad et al. [2017a] and citations therein):

- Available models of environmental and ecological systems tend to focus on measurable quantities or functions of ecosystems that are not necessarily the same ecosystem services directly valued by people (called “ecological endpoints”, “final goods and services”, or “benefit-relevant indicators”; Boyd & Krupnick [2013]; Ringold et al. [2013]).
- The accuracy, reliability, and uncertainty associated with models of ecological systems are not necessarily known.
- Results of structural and statistical models of ecosystems are often tied to a particular place and time, making generalization difficult.
- The costs of using ecosystem models to represent biophysical outcomes tends to be high.[14]

Even when these challenges can be overcome, the inherent uncertainty involved in estimating changes in ecosystem services is high due to a combination of factors such as lack of knowledge about the system itself, the stochasticity of nature, and model uncertainty. The fact that changes in natural capital are likely to change the flow of ecosystem goods and services into the future, coupled with the uncertainties about that future that are unrelated to any project, further exacerbate concerns about accuracy and reliability of model results.

While complexity and uncertainty is a complicating factor for non-market valuation, decision-makers rarely have full and perfect information when making any decision. Best practice to support decision making is to document the best evidence available and any maintained assumptions, as well as performing sensitivity analysis to test the implications of making alternative assumptions or using different data sources on intermediate or final results. This includes not only documenting the limitations of the chosen approach, but also the limitations of alternative approaches that could have been pursued.[15]

### 5.3.2 Monetization of Biophysical Effects

Estimating changes in biophysical outcomes that affect people’s welfare is necessary, but not sufficient, to value changes in ecosystem services. Monetization requires both estimates of individual (or household) willingness to pay for the final goods and services that change, as well as estimates of the number (and possibly type) of people affected. Monetization of the provision of various ecosystem goods and services and summing the results is a form of weighting, with weights given by the willingness to pay of the (sub)population affected by a change. This is the same as any index, except that the weights are based on welfare theory. Specific approaches to monetization are discussed in the next section.

Monetization of ecosystem services poses a number of challenges. Preferences and willingness to pay differ across individuals, and may change over time, as may the number and type distribution of people

[14] This includes both monetary and non-monetary costs.

[15] For example, while there may be uncertainty in a model of predicted biophysical changes given implementation of a project, not considering the possibility of changes at all will also bias results in a particular (usually known) direction.

affected. The lack of markets for many ecosystem goods and services precludes observation of their unit prices, unlike tradable goods and services. Other challenges related to the use of monetized values to describe changes in ecosystem services include (see Olander, Bagstad et al. [2017a] and citations therein):

- The lack of a linkage between available biophysical models with economic models and valuation studies.
- Differences in supply and demand-side conditions for ecosystem services across time and space affect willingness to pay, resulting in significant differences across contexts.
- The costs of using primary economic research techniques to value goods and services tends to be high and the results uncertain, with lower-cost methods associated with greater uncertainty or error.
- The use of non-market valuation techniques is not universally accepted among the economic and decision-making communities.

Once again, however, virtually all analytic methods have their own challenges, and best practice is to acknowledge those limitations, ensure transparency, and test the sensitivity of results to alternative assumptions.

## 5.4 PRIMARY APPROACHES FOR ECOSYSTEM GOODS AND SERVICES VALUATION

Despite the challenges noted in the previous sections, economists have developed methods to estimate willingness to pay using alternative techniques. They are generally categorized into revealed preference techniques, in which value information is inferred through data based on actual purchases or behavior, and stated preference techniques, which use hypothetical survey data to infer values. The major methods are briefly discussed below; for more information, the reader is referred to Champ, Boyle, and Brown (2017).

### 5.4.1 Averting Behavior and Cost Based Methods

*Averting Behavior* and *Cost Based Methods* do not directly estimate willingness to pay, but rather use expenditures, avoided costs, mitigation and restoration costs, or replacement costs to reflect the value of a good or service. For example, using the averting behavior method, if an individual facing an environmental risk spends money on a good or service to mitigate or avoid that risk, then these expenditures provide a lower bound on the value of the environmental good being degraded. The avoided cost method assumes that if an individual or society decides to provide a nonmarket good or service, then the relatively known costs of providing it is an estimate of the minimum value of that good or service.[16]

To use these methods, information about actual or hypothetical expenditures related to the provision or mitigation of environmental good and services is needed. Fischbach et al. (2020) provides a recent

[16] This implicitly assumes the decision-makers are rational, in that they would not provide a service in which the total costs exceed the total benefits.

example of this approach applied to water quality cobenefits from urban green stormwater infrastructure. The key advantage to these methods is that cost information is generally more readily available then information about benefits; the key disadvantage is that these methods cannot offer information on the difference between willingness to pay and what actually must be paid to provide a good or service.

### 5.4.2 Hedonic Property Method

The *Hedonic Property Method* is a revealed-preference technique based on the theory that property (or other capital stock) values are determined by the characteristics of the property itself (e.g., number of bedrooms, square footage, etc.) as well as the surrounding environment, including but not limited to neighborhood public services (such as local school quality) and environmental amenities (e.g., proximity to environmental features, air quality, etc.). The method uses statistical models to estimate the marginal values associated with an amenity or dis-amenity, and these values are used as estimates of the value of the service. These can be converted to a flow measure using an appropriate discount rate. The values obtained for this method are specific to the property owners.

To estimate a hedonic property model, users need information on property values (or rents) and their likely determinants, including but not limited to the characteristic(s) being valued, with variation across space. Local governments typically maintain records of property sales and their inherent characteristics, and GIS layers typically contain information relevant to environmental amenities (such as land use type and calculators that can be used to generate distance measures). However, some goods and services valued by individuals and incorporated into their decision-making processes about location and housing choice may be difficult to measure (e.g., viewsheds).

One of the key advantages of the hedonic property method is that property transactions are based on revealed behavior; that is, the observed market price of the sale represents an actual transaction between buyer and seller. In many cases, data on home sales is relatively easy to obtain and the data tends to be reliable. Because of this, the hedonic property method has a long history of use in the economics and other professions.

In addition to several well-known econometric issues that can lead to biased coefficient estimates,[17] the hedonic property method is based on several assumptions that may not hold in practice. First, the method assumes that property markets are competitive, in equilibrium, and that individuals face limited transaction costs when deciding where to live/what property to purchase. Second, in order to ascribe estimated values to neighborhood or environmental characteristics, it must be assumed that individual perceptions match objective reality. Finally, the method does not typically uncover the entire willingness to pay function, which is needed for valuing relatively large changes in the supply of an ecosystem

[17] Among these are the potential omission of unobserved factors that impact both a characteristic of interest and housing prices (endogeneity) and self-selection in housing sorting leading to bias in the estimated value of a characteristic (Chay & Greenstone, 2005).

service (i.e., the marginal value is not sufficient for analysis). Finally, as with all revealed preference methods, hedonics can only uncover use values.

### 5.4.3 Travel Cost Method

The *Travel Cost Method* is a revealed preference technique generally appropriate for recreational and other trip-based values. It assumes that the demand for recreation is inversely related to the costs to travel to a site and uses statistical models, travel time, and measures of the value of time to estimate the value of a recreational trip to a participant. An individual travel cost model generally involves a single site, while random utility models are often used to model visits to multiple sites.[18] Statistical methods are used to estimate the models, which show how visitation or recreation behavior changes with a change in travel costs and other site characteristics.

To estimate a single site travel cost model, users need information on trips taken (or shares of trips taken), travel time, and assumptions about the value of recreational time, with observations over subgeographies or individuals.[19] Some models include demographic information about visitors. This information is typically collected via surveys. Multiple site models require this information plus information on the set of potential substitute sites, including site-level characteristics relevant to recreational decision-making.

The travel cost/random utility method is considered one of the most accurate and effective approaches for estimating recreational values and simulating the welfare effects of site closures or additions. This is because, like the hedonic property method, it is based on revealed behavior. Relative to mail surveys, it can be comparatively inexpensive to survey visitors onsite.

However, single site models are typically ineffective for incorporating information about substitute sites and generally cannot be used to model site closures or additions. Multiple site models are more complex and require more data to estimate. Disentangling multi-purpose trips with this method is a challenge, and the theory behind the model suggests that consumers react to travel costs in a manner identical to other prices. Finally, estimated welfare effects critically depend on the value of travel or recreational time assumed, though this variable cannot be directly observed.

### 5.4.4 Stated Preference Methods

*Stated Preference Methods*, such as *Contingent Valuation* and *Choice Experiments*, use survey data to estimate willingness to pay. Researchers use choice responses (in which a respondent makes a choice over a menu of alternatives) in conjunction with an experimental design to generate data appropriate for

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[18] The random utility model is based on the idea that an option that generates the most utility is the most likely to be chosen in a given setting. It is also the modeling technique used in stated preference methods, as both involve modeling discrete choices involving multiple characteristics (OECD iLibrary | Home (oecd-ilibrary.org)).

[19] There is considerable literature on this topic of the value of recreational time; however, prevailing wisdom usually puts the value somewhere less than the wage rate. Variation in travel time/travel cost is necessary to estimate the model. For a good discussion, see Larson and Lew (2014).

estimating a random utility model. Unlike the hedonic property and travel cost methods, stated preference methods are not based on revealed behavior, but rather stated choice. Because no actual transaction takes place as a result of the stated choice, if the proper incentives are not in place to align stated and revealed behaviors, then respondents may act strategically, and welfare estimates may be biased.[20] However, stated preference methods are the only tools that can be used to estimate non-use values (those values that result from the existence of a good or service or the option to use it in the future).

Statistical methods are used to estimate a random utility model from the data, with coefficient estimates representing the marginal utilities associated with changes in each modeled characteristic. If one of these characteristics is related to income (typically, an involuntary payment vehicle such as a tax), then ratios of the coefficients on non-income characteristics relative to the income coefficients provide an estimate of the (marginal) willingness to pay for a change in the non-income good or service.

To estimate a stated preference model, users need information about stated choices over a set of options/characteristics that vary in a sufficient way to statistically identify the coefficients of interest in the random utility model. Data is typically collected via surveys. The validity of welfare estimates obtained from the model depends critically on the properties of the survey instrument and the incentives related to respondent strategic behavior.[21]

In addition to the ability to estimate non-use values, stated preference methods can be used in a flexible manner to value virtually any non-market good or service, even when an environmental change in a particular location has not been experienced (and thus no empirical evidence exists to generate data about behavioral changes as a result). These methods can also be used to estimate willingness to pay under conditions of demand or supply uncertainty.

However, issues related to incentive compatibility, hypothetical bias, and sensitivity to payment vehicle have led to concerns about validity and reliability of welfare estimates derived from stated preference methods. Despite the conclusion of an expert panel of economists exploring the validity of the contingent valuation method that these methods, when done well, can provide useful information about preferences and values, and the acceptance of use of stated preference results in the U.S. justice system, there remains a fairly robust debate about the use of stated preference methods in the literature and in policy circles

[20] Strategic behavior is said to occur when a respondent answers a question untruthfully in order to manipulate the ultimate policy outcome. Broadly, the term “hypothetical bias” is used to describe the errors resulting from a divergence between revealed and stated behavior.

[21] If these incentives are such that there are no theoretical gains to be had by misrepresenting choices, then the exercise is said to be “incentive compatible.”

(see, e.g., Diamond and Hausman [1994] and Hausman [2012] for a skeptical view, and Haab et al. [2013] and Kling, Phaneuf, and Zhao [2012] for the corresponding rejoinders).[22]

### 5.4.5 Advantages and Disadvantages of Primary Valuation Methods

The hedonic property method, travel cost models, and stated preference techniques can all be used to estimate the value of non-market goods and services, such as those provided by the environment. Each can be tailored to a particular location or (sub)population, are consistent with economic theory, and provide evidence about the willingness to pay for—and thus the benefits and costs of—changes in the provision of ecosystem services. Revealed preference techniques can provide information about use values, while stated preference methods can be used to estimate use and non-use benefits.

When applied with care and rigor, each of these methods can provide valuable information for use in BCA and other decision-support approaches. There are, however, several disadvantages that tend to preclude their widespread use, especially in federal policy analysis.[23] First, using primary studies to value changes in ecosystem services is relatively expensive and takes both time and expertise. In particular, the timeframe for information demanded during the evaluation process may not match the timeframe necessary to actually perform the analysis, and these problems may be exacerbated by issues of scale (Newbold et al., 2018).

Second, surveys conducted by USACE and other federal agencies in support of these methods are subject to the Paperwork Reduction Act and require approval from the OMB. This adds time and expense to primary valuation studies, discouraging their use.

Third, given 1) the professional disagreements over primary valuation methods, and stated preference methods in particular (as discussed above); 2) concerns about hypothetical and other forms of bias; and 3) the absence of an objective formula to assess whether a study is of high quality, it may be that primary valuation methods are perceived as too risky for use in decision analysis (OMB, 2003).

Finally, given the uncertainties associated with both biophysical changes and monetary values (including the lack of observable prices due to the non-market nature of many ecosystem services), the information content of a primary study may not be sufficient to justify the costs.

[22] The 1993 “Blue Ribbon Panel” concluded “that [contingent valuation] studies can produce estimates reliable enough to be the starting point of a judicial process of damage assessment, including lost passive-use values” (Arrow et al., 1993, p. 44).

[23] Nevertheless, primary valuation methods and protocols for benefit-cost analysis appear in guidelines such the U.S. Environmental Protection Agency’s *Guidelines for Preparing Economic Analyses* (U. S. Environmental Protection Agency, 2010) and OMB’s *Circular A-4* (OMB, 2003).

## 5.5 ALTERNATIVES TO PRIMARY VALUATION STUDIES

Partially due to these issues, and capitalizing on the growing non-market valuation and ecological production function literature, valuation techniques based on already published secondary information have been developed. With these methods, estimates of the production or values of ecosystem goods and services estimated in one context are used to estimate their value in another (generally similar) context. The primary advantage of these techniques is that they are less expensive and simpler than primary studies based on non-market valuation methods. The tradeoff, however, is that the transfer of physical or monetary information from one context to the next may introduce bias, especially if the demand or supply side characteristics at the new site are markedly different than the source studies.

This section discusses the available options for using secondary data to value changes in ecosystem goods and services. These methods are split into two broad categories based on whether they include ecological production (predictions of the flow of the biophysical services themselves rather than values alone).

### 5.5.1 Benefit Transfer Methods

*Benefit (or Value) Transfer* methods use estimates from previous valuation research in a new context in order to value nonmarket goods and services (Johnston et al., 2021). This can be as straightforward as taking one value from a study that is similar in context and using it in another, or as complex as a meta-regression that statistically models the results of research from many studies, accounting for differences across studies (Richardson et al., 2015).[24] In the latter case, the statistical model is used to obtain appropriate values for a new study.

At minimum, benefit transfer requires at least one value of the ecosystem service of interest from a context that is deemed "similar" to the one under consideration and from either a well-designed study or reputable source. Multiple estimates are required to use means over several studies (which may tend to cancel out individual biases) or to estimate a benefit function or meta-analysis equation that takes into account varying characteristics of the good or service across studies or the methodological details of the studies or estimates themselves (e.g., the valuation technique used, the sample size, etc.; Johnston & Bauer, [2020]; Richardson et al., [2015]).

Benefit transfer has been recognized as a tool for BCA in the U.S. (including the Environmental Protection Agency), Canada, Europe, and Australia (Johnston et al., 2021). Transfer values collected by organizations within the U.S. government are summarized in Table 5-1.

[24] Use of the results of a single-site study that presents an estimated value function is termed "function transfer", use of an estimated preference function from a single site is termed "structural preference-function transfer", and use of a multiple-study meta-analysis function is termed "meta-analysis function transfer" (Johnston et al., 2021).

Table 5-1. Benefit transfer tools and data sources from U.S. government agencies

| Name | Agency | Description | Source |
|---|---|---|---|
| Benefit Transfer Toolkit | USGS | Nonmarket valuation database, statistical forecasting models, and recreation activities map | https://sciencebase.usgs.gov/benefit-transfer/ |
| Recreation Use Values | USFS | Estimated recreation use values for 14 recreational categories using the Recreation Use Values Database | https://www.fs.fed.us/pnw/pubs/pnw_gtr957.pdf |
| Recreation Unit Day Values | USACE | Estimated day use values for recreation by quality tier | https://planning.erdc.dren.mil |
| Ecosystem Service Benefits | FEMA | Allowable ecosystem service values for Hazard Mitigation Assistance programs by land use type | https://www.fema.gov/sites/default/files/documents/fema_innovative-drought-flood-mitigation-projects.pdf (Table 2-2) |
| EcoService Models Library | USEPA | Library of ecological production models | https://www.epa.gov/eco-research/ecoservice-models-library |
| Value of water quality changes meta-analysis | USEPA | Meta-analysis for improvements in water quality based on 51 original studies | https://www.epa.gov/sites/default/files/2015-10/documents/steam-electric_benefit-cost-analysis_09-29-2015.pdf (Appendix H) |
| BlueValue | NOAA | Database of ecosystem service values focused on coastal areas | https://imagery2.coast.noaa.gov/digitalcoast/tools/gecoserv.html |

In addition, a variety of academic and non-government organizations have also developed relevant databases in recent years. A sample of key sources is summarized in Table 5-2 below.

Table 5-2. Selected benefit transfer tools and data sources from non-U.S. government sources

| Name | Organization | Description | Source |
|---|---|---|---|
| Ecosystem Services Value Database (ESVD) | TEEB | Database of ecosystem services values | https://www.esvd.info/ |
| Recreation Use Values Database | Oregon State University | Database of recreation use values in the U.S. and Canada | https://recvaluation.forestry.oregonstate.edu/database |
| National Ocean Economics Program Non-Market Database | Center for the Blue Economy, Middlebury Institute of International Studies at Monterey | Database of ecosystem service values related to coastal areas | https://oceaneconomics.org/nonmarket/NMsearch2.asp |
| Environmental Valuation Reference Inventory (EVRI) | Environment and Climate Change Canada (with international partners) | Database of empirical studies on the economic value of environmental assets and human health effects | https://www.evri.ca/en |

A number of authors have discussed "best practices" with respect to benefit transfer, with key points summarized in Richardson et al. (2015) and Johnston et al. (2021), among other references.[25] Original valuation studies used in any transfer exercise should be based on adequate data using well-executed theoretically sound methods with consistent measures of welfare changes (Freeman, 1984). Furthermore, the identification, evaluation, screening, and coding of data from candidate studies should be replicable and transparent (Johnston et al., 2021).

In terms of context, the commodity valued between secondary sources and the project site should be the same, with the same property rights structure and similar demand-side characteristics within the serviceshed[26] (Boyle & Bergstrom, 1992). In particular, differences in scope,[27] geographic scale and definition of the serviceshed, the availability of substitutes between contexts, and changes in preferences over time can affect values (Simpson, 2017). However, there is a balance to be struck between narrow definitions of context and the amount of information available to perform the transfer (Johnston et al., 2021). Only final ecosystem services (those which directly enter individuals' utility functions) should be valued to avoid double-counting between these and intermediate services (Boyd & Banzhaf, 2007).

The major advantage of the use of benefit transfer techniques is the generally lower cost of using pre-existing secondary data rather than primary valuation studies. The primary disadvantage comes from issues of heterogeneity across time and space; that is, between contexts, both demand-side and supply-side characteristics are almost sure to differ between studies in the literature and a given policy context. The more effort is put into minimizing differences (through search, modeling efforts, etc.), the less the advantages of using secondary methods to value non-market goods and services.[28]

### 5.5.2 Integrated Decision Support Tools

*Integrated Decision Support Tools* such as the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) and Artificial Intelligence for Ecosystem Services (ARIES) models (Tallis et al., 2009; Villa et al., 2014), generally combine (sometimes spatially explicit) collections of ecological production models with transferred monetary values at the landscape scale (see Bagstad, et al. [2013]; Redhead et al. [2016]; Sharps et al. [2017]; and Grêt-Regamey et al. [2017] for available models and recent assessments).

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[25] Interested readers can refer to these two sources and the citations therein for more detailed information about the use, validity, and credibility of benefit transfer methods.

[26] The serviceshed of a good or service defines the people to whom benefits and costs are likely to accrue, sometimes, but not always, defined geographically.

[27] If the scale or scope of changes differs between the original source and the project is different, then the incremental (marginal) values of the change may differ.

[28] For a relatively recent review of current practice and prospects, readers are referred to a special issue of Environmental and Resource Economics, Volume 69, Issue 3, March, 2018, available at https://link.springer.com/journal/10640/volumes-and-issues/69-3. Johnston, et al. (2021) provide guidelines for improving the validity and credibility of transfers.

Varying in complexity, coverage, scale, and scope, these tools aim to support decision-making through the replicable quantification of multiple ecosystem services and values using pre-programmed functions but can be customized (to an extent) to local or regional conditions through parameter and other changes. These models can be either deterministic or stochastic. Data needed to use such models varies across tools, but generally involve the use of spatial data such as land cover as model inputs.

The primary benefit of the use of integrated decision-support tools lies in their transparency and replicability. Such tools generally combine both the biophysical and monetary aspects of valuation into a single package. The challenges share many of the features of the earlier discussion, including: a) more complex models with greater coverage of multiple ecosystem services and customizability are more costly to use in terms of time, data, and expertise; b) heterogeneity across time and space can result in errors when the parameterization of the ecological production models and value estimates are derived from a different context than the area under study; c) by necessity, the *validation* of such tools is generally limited to a few case studies (through in some cases they have been widely applied).

## 5.6 SUMMARY

The non-market nature of ecosystem goods and services has resulted in the development of multiple methods to estimate the value of goods and services provided by natural capital. Primary methods, such as the hedonic property method, travel cost, and stated preference approaches, use context-specific data and statistical techniques rooted in economic theory to estimate the willingness to pay for final ecosystem goods and services that are relevant to the direct welfare of those affected by changes in their provision. When used properly, each method has been shown to generate meaningful measures of the benefits of changes in environmental goods and services, allowing for their direct incorporation into BCA. However, the cost of primary studies (in terms of labor, time, and expertise) tends to preclude their use in many circumstances.

In response to this challenge, researchers have leveraged the growing literature on ecosystem services and values to develop methods that rely on secondary data, such as benefit transfer. Others have developed "off the shelf" integrated decision support tools that combine transferred ecological production functions (which predict biophysical change) with transferred values in an effort to reduce the costs and increase the replicability of the analysis of ecosystem goods and service provision.

There is no ideal solution to solve the tension between context specificity (i.e., tailored representations of the demand- and supply-side characteristics of a particular serviceshed) and the costs of providing high-quality, context-specific estimates of value; there are only tradeoffs. When done correctly and carefully, benefit transfer may provide lower-cost opportunities for meaningful welfare measures than the use of primary valuation techniques, but such estimates will almost always introduce a positive probability of error or bias. However, decision-makers should ask themselves if it is preferable to make a known error (namely, excluding the welfare changes resulting from changes in ecosystem service provision entirely), or to use imperfect techniques to illustrate the potential welfare consequences of decisions that impact natural capital, ecosystem services, and the associated serviceshed, and acknowledge the potential flaws and biases that may result.

# 6.0 CONCLUSION

Methods for placing economic value on ecosystem goods and services have proliferated in recent decades. These approaches include advances in approaches, such as multi-objective decision analysis, that can evaluate the ecosystem services provided by NBS through non-monetized benefit metrics. These methods provide a distinct advantage over relying exclusively on benefit-cost analysis (BCA), which can exclude social benefits solely because there are insufficient data, information, or resources for monetization.

This report, the third in a series of reports intended to inform improved evaluation of NBS in USACE planning studies, draws on this evolving literature to provide methodological guidance for evaluating ecosystem services. The study team reviewed relevant valuation methodologies in recent economics and ecological management literature, leading to specific guidance for the subsequent analysis to follow. Methods and data sources identified in this review will be applied in six retrospective case studies, described in a forthcoming capstone report to this research effort.

The study team also reviewed additional methods of potential interest for USACE that are designed to support decision making across multiple objectives, explicitly account for long-term uncertainty, and/or consider a range of societal values or differing priorities among different communities. Given that many decision-relevant social and environmental metrics may be difficult to value in dollar terms, these approaches can incorporate both monetary and non-monetary metrics for assessing project impacts and necessitate developing meaningful and interpretable non-monetary metrics to capture other types of benefits and costs.

Key findings and recommendations from the entire study effort will be detailed in the capstone report. However, the study team identified several preliminary conclusions from this methodology review:

- **Integrated analysis provides a more complete evaluation of project benefits.** Exclusively focusing on economic analysis for project evaluation can exclude relevant benefits, but methods exist for more fully capturing the monetized and non-monetized benefits of water resource projects. An integrated analysis that simultaneously considers economic benefits alongside environmental and social benefits can provide a more complete valuation of project costs and benefits. This approach has already been applied for select USACE projects, suggesting that USACE planning practice can evolve to apply integrated analysis and more fully consider a wide range of economic, social, and environmental benefits and costs.
- **Leveraging of multiple methodologies can allow benefits and costs to be evaluated across the diverse range of USACE projects.** The methods reviewed in this report provide a menu of options for the subsequent case study analysis and for USACE planners to draw upon in future feasibility studies. The diversity of available methods for environmental and social benefit estimation can enable location-specific data – which will vary in availability by project and location – to be used in conjunction with other approaches to inform ecosystem services analysis across the range of USACE mission areas and projects.

- **Enabling flexibility in evaluation methods and providing support in their application can support more widespread use within USACE.** Given the range of available methods for more integrated and comprehensive analysis, USACE practitioners need flexibility to employ methods appropriate for individual projects and may benefit from additional resources to successfully employ these methods at scale. Additional resources could include expanded guidance, technical expertise, and programmatic funding to engage with and gather data from communities of interest in advance of a Congressionally authorized study process. Added flexibility would allow districts to select the best available methods and evaluation approach given the water resources challenge, local geography and community context, and non-federal sponsor interest and capacity.

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# APPENDIX A. ADDITIONAL BACKGROUND ON ECOSYSTEM SERVICE FRAMEWORKS

The frameworks applied to understanding and valuing ecosystem services have evolved substantially in recent decades. In the last several years, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) reviewed this progress and re-defined these services under a new umbrella term "nature's contributions to people" (NCP; Figure 7-1). Within this report, the term "ecosystem services" is used as it remains the more common terminology and provides a simpler framework for applying to USACE. However, it is used in an inclusive way that makes it interchangeable with the IPBES use of the term 'nature's contributions to people'.

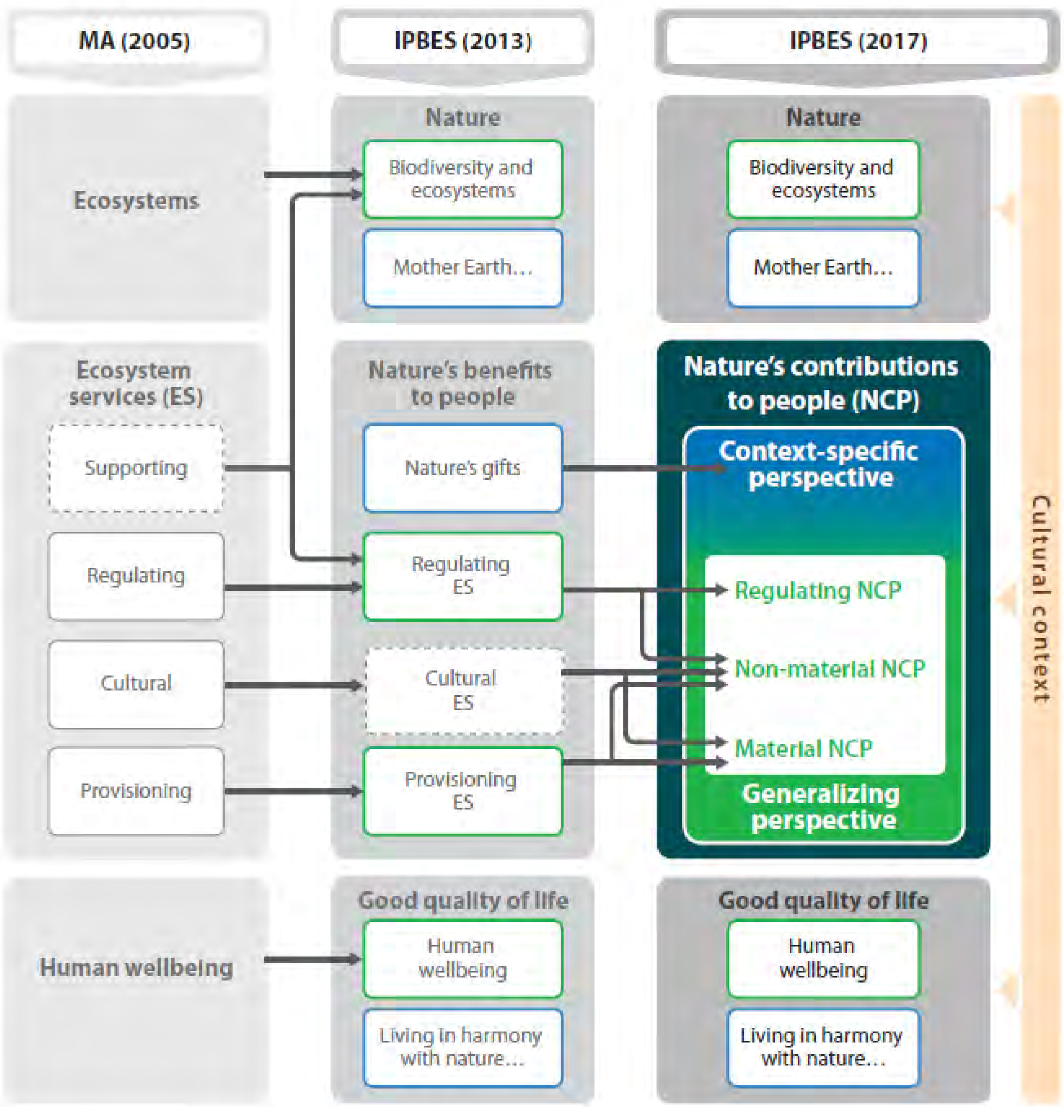


*Figure A-1. Comparison of ecosystem services and Nature's Contribution to People frameworks. Source: Pascual et al. (2017).*

During the 1960s and 1970s, increasing awareness that humans have the potential for far reaching impacts on ecosystem resources resulted in a disconnect between the need for better global ecosystem management and practical economic policymaking. Rachel Carson outlined the impact of bioaccumulating chemicals used as pesticides and initiated three decades of intense research to better understand global ecosystems and human impacts (Carson et al., 1962). A major response to this realization was the establishment of the nature conservation movement. The focus of nature conservation was based largely on the notion that areas with ecological functions (natural capital) were “inherently valuable” as justified by scientific and ethical considerations (Turner & Daily, 2008).

While successfully developing many protected areas globally, this perspective was ultimately limited in improving ecosystem management. The early 1970s publication of *Limits to Growth* formally presented the exponential rise in human population and increasing resource depletion (Meadows et al., 1972). The resulting movement focused on human and ecosystem collapse with unlimited growth and recommended reducing human birth rates. From a policy and management perspective, mandating birth rates was not a realistic or actionable policy recommendation in the western world. So, progress towards sustainable ecosystem management was further seen as unachievable within current policy frameworks and also considered incompatible with sustainable economies. These two examples show the multiple challenges that need to be overcome to include natural capital more fully in project evaluation. Beyond the technical challenge of quantifying the benefits and assigning them a monetary value, the thinking during the latter half of the 20th century by proponents of the value of natural capital actively drove the conversation away from monetization of these benefits.

Reflecting increased and accelerating ecosystem changes resulting from increasing human population (up from 2 billion in 1927 to 6 billion in 1999 [U.N., 2019]), the intergovernmental panel ecosystem assessment concluded that the human-induced ecosystem changes from altering human uses of land and sea, direct exploitation of biota, climate change, pollution, and alien invasive species, are rapid enough since 1970 to be measured at less than a decadal time scale (Diaz et al., 2019). Humans have altered some of the fundamental global ecosystem cycles. Introduction of nitrogen fertilizer to support expansion of increasingly centralized agricultural practices has permanently altered the global carbon and nitrogen cycles (Ding & Feng, 2017; Jenkinson, 2001). To address these large-scale changes, in the first decade of the 2000s the MEA focused on the linkages between ecosystems and human wellbeing (MEA, 2005). This led to a policy platform that more closely linked “ecosystem conservation” projects with “human community sustainability” linked through direct human use of ecosystems, or “ecosystem services” (Tallis et al., 2009). The increasing recognition that sustainable development and conservation of biodiversity and wildlife resources are both served when intact ecosystems are maintained increasingly replaced the historical notion of a simple tradeoff between societal and ecosystem needs (Tallis et al., 2009).

Most recently, IPBES conducted a critical assessment of the status and trends of NCP, the implications of these trends for society, direct and indirect causes, and actions for a better future condition of the natural world and societies (Diaz et al., 2019). IPBES went much further than the MEA in recognizing nature’s contribution to people and society, using data and knowledge from natural and social sciences as well as practitioners, Indigenous, and local communities. NCP is considered to better support transformative practices that move toward sustainable futures recognizing the diversity of uses of nature by people across the world (Diaz et al., 2019; Pascual et al., 2017). The IPBES framework additionally acknowledged the need to understand linkages and flow on effects upon multiple sectors including health, education, and energy (Ferrier et al., 2016).

Even with the increasing acceptance that human wellbeing is inextricably linked with sustainable and intact ecosystems, evaluating NBS is a complex challenge given that ecological processes are non-linear, change is not necessarily directly reversible, and interactions of ecosystem components do not necessarily respond to change in predictable ways (Duarte et al., 2009; Ehrlich & Ehrlich, 2008; Li et al., 2021). In addition, processes such as tropical and extratropical storms, droughts, sea level rise, and other environmental factors drive short- and long-term variability and change in ecosystems; because these factors have inherent uncertainty that cannot be fully reduced, management decisions must be robust to many possible futures and/or readily adaptable as conditions change (Landis et al., 2013). The same factors can impact infrastructure such as levees and seawalls, however, and in some cases NBS may have capacity for natural adaptability (e.g., post-storm recovery) that can provide longer-term benefits or reduced costs than engineering approaches. This complexity suggests that flexible evaluation methodologies are needed that go beyond exclusive reliance on BCA to more accurately capture the benefits and costs of NBS.

One other need is to progress beyond the outdated view of ecosystems as stable and unchanging. Recognizing that the influence of humanity has now impacted all fundamental aspects of the global ecosystem, managing that ecosystem is now a series of “best management decisions” at all spatial scales from local to global. For these reasons, developing more sophisticated approaches to quantify and monetize ecosystem services, and tailoring them to more local socio-ecological circumstances, is needed so that projects can be identified that maximize human sustainability and well-being (Turner & Daily, 2008).

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