Guidelines on Methodologies for the Valuation of Coastal & Marine Ecosystems

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1. Background to the development of the guidelines

The Contracting Parties to the Nairobi Convention have received funding from the Global Environment Facility (GEF) to implement a Programme entitled 'Implementation of the Strategic Action Programme for the protection of the Western Indian Ocean from land-based sources and activities' (WIOSAP). The Project is intended to reduce impacts from land-based sources and activities and sustainably manage critical coastal and marine ecosystems through the implementation of the agreed WIOSAP priorities with the support of partnerships at national and regional levels. The Project is implemented and executed through a 'Partnerships Approach' with the Nairobi Convention Secretariat being the Executing Agency. The participating countries include Comoros, Madagascar, Mauritius, Seychelles, Mozambique, Kenya, Tanzania, Somalia and South Africa. It is anticipated that the sum of the learning derived from activities implemented by the Project will be significantly enhanced if they share standardised approaches to their design, implementation, analysis and reporting of results. To facilitate this a set of resources are being made available to project implementation teams. One of these resources will be the WIO Guidelines on Methodologies for the Valuation of Coastal and Marine Ecosystems (VCME). Although a number of excellent documents providing advice and guidelines for VCME have been developed in recent years and many approaches and tools for VCME are fairly universal in their potential application, the particular relevance, utility or practicality of one versus another is determined by the specific local context. For example, countries of the WIO differ in the availability of data or in terms of access to the capacity required for VCME. Cultural influence on the philosophy and potential use of economic valuation also varies across the WIO. The objective of preparing WIO specific guidelines on VCME is therefore to help users in the region to focus on what is most likely to work for them, to assist them to better match the vast array of available tools and approaches to their particular situation.

2. Introduction

2.1 The Western Indian Ocean Context

This section provides a brief description of the geographic, ecological, socio-economic and governance characteristics of the Western Indian Ocean (WIO) region. The text is largely drawn from the Transboundary Diagnostic Assessment (TDA) for the WIO region (ASCLME/SWIOFP (2012).

The WIO region from covers approximately 22.3 million square kilometres and is floored by deep abyssal plains and bounded to the west by non-volcanic continental shelves. The width of the continental shelf of the WIO region tends to increase southwards from the Somali coast in the north and extends to 150m depth on average. Approximately 700 seamounts have been identified in the WIO region and are known to be hotspots of biodiversity and marine biomass in the pelagic ecosystem. The coastline of the WIO countries, including mainland and island states, is over 15,000 km in length. It includes a wide diversity of coastal habitats including rocky shores, sandy beaches, coral reefs, mangrove systems, seagrass beds and estuaries which, in combination, supply a wealth of ecosystem services to the human populations along the coast. Coastal habitats, however, tend to be at high risk because of their proximity to land and marine based impacts and because they are typically easily accessible from land and vulnerable to overexploitation if not managed properly.

In terms of fisheries, the WIO generates a catch of more than 4 million tonnes of fish per year, produced by fisheries ranging from traditional subsistence and artisanal activities using a wide variety of different gears, to large-scale industrial operations fishing mainly with longlines, purse seines and trawling. The best available estimates are that one third of stocks in the region are now either overfished or depleted.

A high diversity of species and communities exists in the WIO region. By 2005, 11,257 marine species had been recorded from the WIO region but this is estimated to be less than 50% of the marine species that are actually present. The biodiversity includes a total of 37 marine mammal species, five species of sea turtle, and eleven seabird families occur as breeding species within the geographical scope of the WIO.

Coral reefs, mangroves and seagrass beds are critically important tropical habitats in the region. They provide habitat for coastal species and for coastal human populations which depend on them for food, livelihoods and other ecosystem services. These habitats are under threat from a range of human impacts including pollution, sedimentation, physical removal, human settlement and the damaging effects of fishing.

Over 160 million people reside in the WIO countries and approximately 55 million of them live on the coast. Although variable from place to place, there is a high reliance by these people on coastal and marine resources for food security and livelihoods in general. Due to their high dependence and limited resilience or adaptive capacity, environmental variability and extreme events have a disproportionately severe effect on the communities. Further, coastal cities and settlements are growing and developing at a rapid rate. Tourism, fisheries, coastal agriculture, mining, aquaculture, and ports and coastal transport provide the main coastal livelihoods in the WIO region.

In terms of economy, ASCLME/SWIOFP (2012) estimates that almost USD 22.4 billion a year is derived from the coastal and marine resources of the WIO. Coastal tourism makes the

largest contribution to GDP at over USD 11 billion a year, equivalent to 40% of the total from marine and coastal resources. Agriculture and forestry are of second highest importance at 20%, followed by mining and energy at 15% and fisheries at 11%. The fisheries of the WIO region generate a resource rent estimated at approximately USD 68 million per year, of which about USD 59 million are generated by WIO countries and the remainder by countries outside of the region.

Regarding management, policy and governance of marine and coastal resources in the region, there are substantial differences between countries in terms of their systems of governmental organisation, processes and priorities, the levels of economic development, scientific capacity and incorporation of science into policy processes, patterns of social organization, culture and values, and in their political relations. Similarly, there are differences in the governance of the major sectors related to sustainable use of marine and coastal resources. Inadequacies and gaps exist in the application of the existing legislation to ecosystem-based management, also varying from country to country. A number of regional agreements and bodies are in place in the WIO region including, for example, the Nairobi Convention (Convention on the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region) and the South Western Indian Ocean Fisheries Commission (SWIOFC). There are also the wider regional agreements such as the Indian Ocean Tuna Commission (IOTC) and several Regional Economic Commissions and the African Union's New Economic Partnership for African Development (NEPAD) also play a role in the region. The countries are also parties to the most important relevant international agreements including the 1982 United Nations Convention on the Law of the Sea (UNCLOS). Notwithstanding this existing framework, there is currently no single mechanism that could implement an integrated region-wide approach to the governance of marine and coastal resources in the WIO region.

2.2 What are we seeking to value, why and how?

The natural environment provides a wide range of goods and services that contribute to the wellbeing of people. For short, the various benefits that ecosystems provide to people are termed "ecosystem services" and their importance to human wellbeing is termed "value" (see Sections 3 and 4 for a detailed introduction to ecosystem services and systems for classifying them). The values of ecosystem services depend on the quantities that are supplied by ecosystems and demanded by people, and will therefore vary greatly across locations with different environmental conditions and populations of beneficiaries.

Many ecosystem services are openly accessible to all and are not traded in markets, for example, climate regulation by mangroves, water filtration by wetlands, and storm protection by coral reefs. Such ecosystem services are important contributors to human wellbeing in the WIO region but this value is often hidden or ignored by resource owners who do not necessarily gain any benefit themselves (or can gain more by putting their land or resources to other uses than the provision of ecosystem services). Highlighting the full value of ecosystem services to society can be useful information in making decisions about how to manage resources. Section 4 expands on the case for valuation of marine and coastal ecosystem services.

A wide range of valuation methods have been developed to measure the importance of ecosystem services to human wellbeing (see Section 6). These guidelines set out the rationale and methods for valuation of ecosystem services to help improve the management of coastal and marine resources in the Western Indian Ocean.

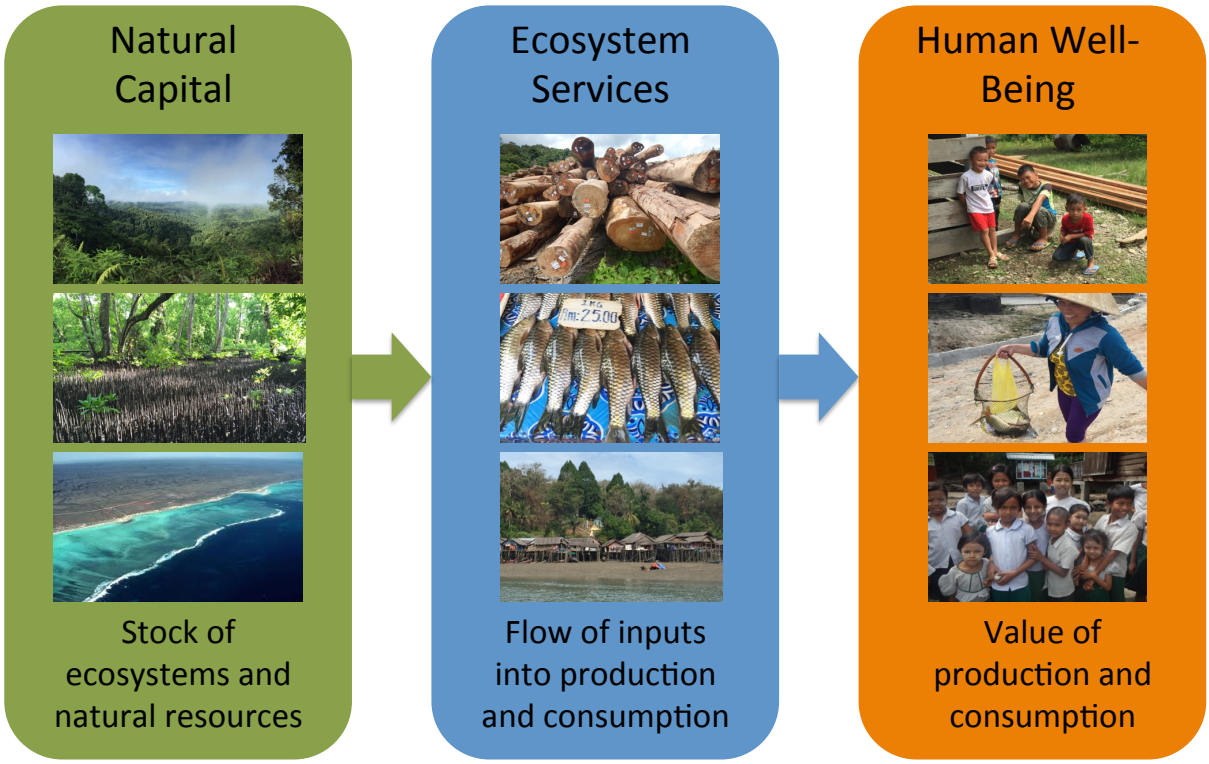


Figure 1. The contribution of natural capital and ecosystem services to human well-being

2.3 Who are these guidelines for?

The purpose of these guidelines is to explain how methods for valuing coastal and marine ecosystems can be used to produce information to support decision-making in the Western Indian Ocean context. Specifically, it is designed to help a broad audience of practitioners, managers, government officials, private sector managers, NGOs, local community-based organisations and statisticians to understand the available methods for valuing ecosystem services and how the information generated can be used to inform the decisions that they make. There are a wide range of decision contexts in which such information can be used, including integrated coastal management (ICM), marine spatial planning (MSP), development of marine protected areas (MPA), evaluation of ecosystem-based adaptation to climate change, and generally on the importance of ecosystem services to the Blue Economy. The guidelines have been written with the regional context in mind and could potentially be further tailored to suit the context of each WIO country.

The broad objective of these guidelines is to provide an understanding of how valuation methods can be used to support decision-making in the WIO context. To this end, the guidelines provide:

- An introduction to the main frameworks for identifying values for coastal and marine ecosystems.
- Non-technical explanations of valuation methods and their applicability to different ecosystem services;
- An explanation of the strengths and limitations of each valuation method;

- Links to available resources and manuals for conducting valuation of coastal and marine ecosystems;
- Illustrative applications of the use and impact of valuation in the WIO region.

2.4 How to use the guidelines

The aim is to provide a practical set of guidelines on the use of valuation methods in the context of the WIO region. To be able to use these guidelines, a basic understanding and experience of applied environmental economics is useful but not necessary. For users that are unfamiliar with environmental economics or need a refresher, a brief introduction to relevant basic principles is provided in Annex 2.

Each section of the guidelines describes a distinct step in the process of delivering information on the value of coastal and marine ecosystems to support decision making. Users can go directly to the sections that are relevant to their needs. Links between steps are highlighted so that users can navigate between sections to suit their purposes. The guidelines provide an introduction to each valuation method, guidance on what information it can be used to produce, and its strengths and limitations. It does not provide step-by-step technical instructions on how to conduct each method since many of the methods require separate dedicated manuals to themselves. Throughout the guidelines, references are made to other useful resources. These guidelines can and should be used alongside these other resources.

It should be noted that the production of information on the value coastal and marine ecosystems requires input from multiple fields of expertise. A general description of a decision process that involves impacts on the environment includes the following steps: problem identification, identification of impacts, bio-physical assessment, valuation, determination of policy or investment options, policy evaluation and decision support. These steps require inputs from many fields of expertise (e.g. climate scientists, marine biologists, coastal geographers, hydrologists, ecologists, economists, policy analysts, and experts in decision support). Although the emphasis of these guidelines is on economic valuation methods, these other crucial elements in the assessment process should not be ignored.

3. Ecosystem Functions and Services

3.1 The ecosystem services framework

The concept of ecosystem services provides a useful framework to identify the importance of the natural environment to humans. The term "ecosystem services" has been defined in a number of different ways (see summary of definitions in Box 1) but put most simply, they are the variety of benefits that we obtain from the environment.

Ecosystems contribute to human wellbeing in a wide variety of ways and the processes by which ecosystems provide benefits to people has been described as an "ecosystem services cascade" in which bio-physical structures and processes ("ecosystem functions") can deliver inputs (ecosystem services) to the production of goods and services that are consumed by people (see Figure 2).

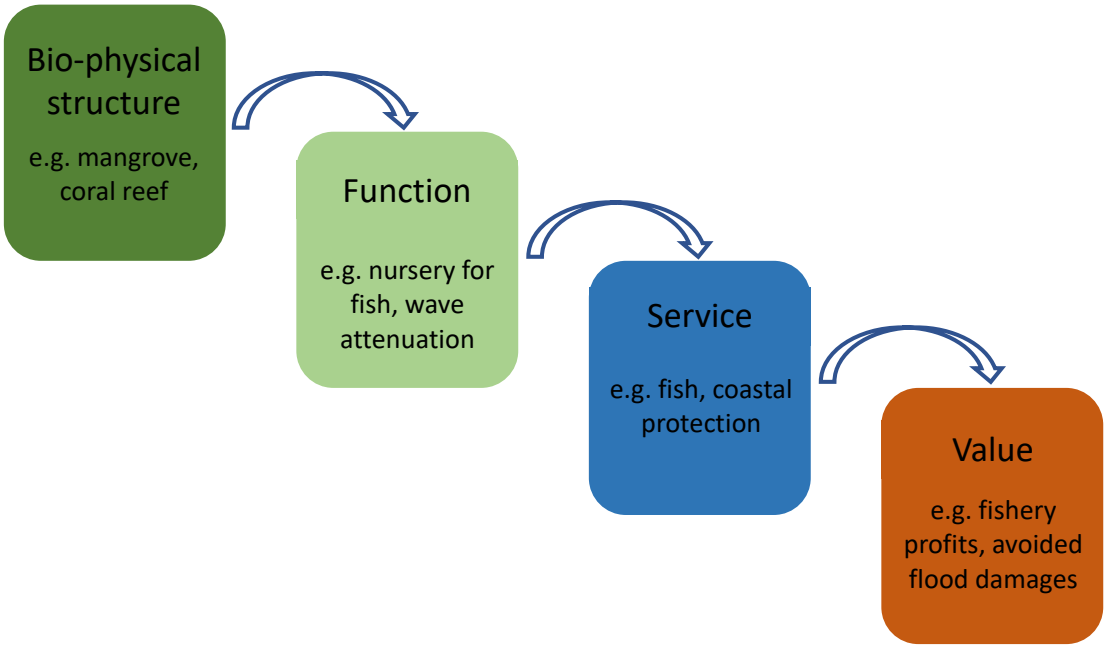


Figure 2. Ecosystem services "cascade". Adapted from Haines-Young and Potschin (2010)

Ecosystem services can also be viewed as the flow of benefits received from "ecosystem capital" – see Figure 3. Ecosystem capital is a component of natural capital, which can be defined as the stock of natural assets that provide society with renewable and non-renewable resources and a flow of ecosystem services. Natural capital includes abiotic assets (e.g. fossil fuels, minerals, metals) and biotic assets (ecosystems that provide a flow of ecosystem services). The biotic component of natural capital is termed ecosystem capital. Natural capital is analogous to built capital (e.g. transport infrastructure), human capital (e.g. a skilled and educated work force) or social capital (e.g. rules, norms and trust) as an input to the production of goods and services that we consume. Natural capital may be both a complement to other forms of capital (i.e. used in combination with them to produce goods and services) or a substitute (used instead of other forms of capital).

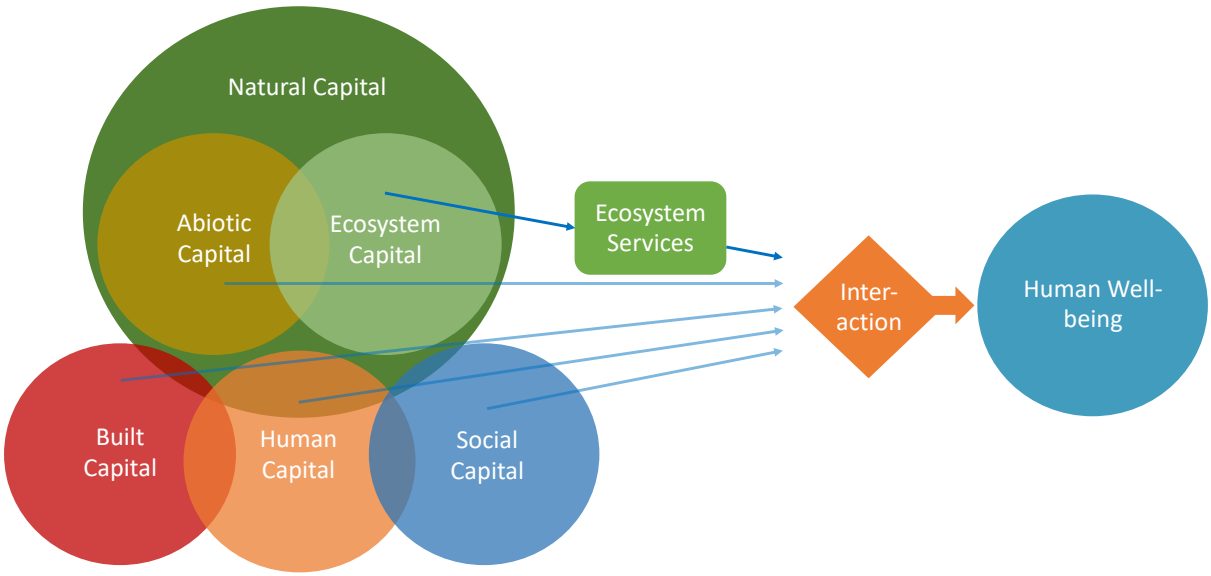


Figure 3. Interactions between natural, abiotic, ecosystem, built, human and social capital to contribute to human well-being. Adapted from Costanza et al. (2014)

Box 1. Defining ecosystem services

The conceptualisation and understanding of ecosystem services has gradually been refined over the past 20+ years and a number of different definitions have been provided by different initiatives. These include:

- Ecosystem services are the benefits that ecosystems provide for people (Millennium Ecosystem Assessment – MA, 2005).
- Ecosystem services are the direct and indirect contributions of ecosystems to human well-being (The Economics of Ecosystems and Biodiversity – TEEB, 2010)
- Ecosystem services refer to those contributions of the natural world that are used to produce goods which people value (UK National Ecosystem Assessment – UKNEA, 2011).
- Ecosystem services are the contributions that ecosystems make to human well-being (Common International Classification of Ecosystem Services – CICES, 2012).
- The US Environmental Protection Agency (US EPA) use the term "final ecosystem goods and services" (FEGS) to mean "components of nature, directly enjoyed, consumed or used to yield human well- being" (Landers and Nahlik, 2013).
- The EU Mapping and Assessment of Ecosystems and their Services (MAES) working group defines ecosystem services as "the contributions of ecosystem structure and function (in combination with other inputs) to human well-being" (Burkhard and Maes, 2017)
- The International Panel of Biodiversity and Ecosystem Services (IPBES) introduced an additional term for ecosystem services – "nature's contributions to people" (NCP) – to describe the contributions, both positive and negative, of living nature (diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to people's quality of life (Diaz et al., 2018).

3.2 Classification of ecosystem services

A number of classification systems for ecosystem services have been developed by different initiatives to describe a complete and consistent set of the benefits people derive from ecosystems. This section provides a brief introduction to five of the most prominent classifications: Millennium Ecosystem Assessment (MA), The Economics of Ecosystems and Biodiversity (TEEB), Common International Classification of Ecosystem Services (CICES), Final Ecosystem Goods and Services (FEGS), and Nature's Contributions to People (NCP). Annex 1 provides a cross-tabulation of these classifications to indicate correspondence across typologies.

The Millennium Ecosystem Assessment (MA) classification of ecosystem services introduced the following four categories of services.

- Provisioning services are the "products obtained from ecosystems". Examples include food, timber and fuel.
- Regulating services are the "benefits obtained from the regulation of ecosystem processes". Examples include water flow regulation, carbon sequestration and protection from storms.
- Cultural services are the "non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences".
- Supporting services "are necessary for the production of all other ecosystem services". Examples include nutrient cycling, soil formation and primary production.

The distinction between supporting services and other ecosystem services can also be described as the difference between "intermediate" and "final" ecosystem services. Final ecosystem services are the last item in the chain of natural processes that provide inputs to the generation of products (goods and services) that are used by humans. Some final ecosystem services are used as inputs in the production of manufactured products (e.g. mangrove trees used to make charcoal) whereas others are consumed directly (e.g. a beach used for recreation). Intermediate ecosystem services are natural processes that contribute to final ecosystem services, but do not directly input into the production of goods and services consumed by humans. The inclusion of both final and intermediate services in the classification can potentially lead to the double counting of services in an ecosystem assessment. In recognition of this, some classification systems have attempted to focus on final ecosystem services.

The Economics of Ecosystems and Biodiversity (TEEB) typology of ecosystem services modified the MA classification by keeping the provisioning, regulating and cultural services categories, leaving out the supporting services category but introducing a category for habitat services, which partially includes intermediate services. See de Groot et al. (2010) and http://www.teebweb.org/resources/ecosystem-services/ for an explanation of the TEEB classification.

The Common International Classification of Ecosystem Services (CICES) includes only final ecosystem services and therefore omits the supporting services category. CICES is designed to be compatible with the System of Environmental Economic Accounting (SEEA) developed by the UN Statistics Division and is intended to be suitable for ecosystem accounting purposes. The classification is hierarchically organised. At the highest level, services are grouped into three "Sections" that relate to whether the contributions to human well-being

support: a) the provisioning of material and energy needs; b) regulation and maintenance of the environment for humans; or c) the non- material characteristics of ecosystems that affect physical and mental states of people. These sections broadly correspond to the provisioning, regulating and cultural services categories in the MA and TEEB classifications. Sections are then split further into 'divisions', 'groups' and 'classes', allowing users to select the appropriate level of detail required for their application and to consistently aggregate results if necessary. See https://cices.eu/ for full documentation and guidance on CICES.

The Final Ecosystem Goods and Services (FEGS) classification system developed by the US Environment Protection Agency also focuses on final ecosystem services. It does not use the categories of provisioning, regulating and cultural services. The FEGS classification takes an orthogonal approach to identify services by environmental class (e.g. terrestrial, aquatic, atmospheric) and sub-class (e.g. grassland, wetlands, rivers and streams etc.), beneficiary class (e.g. agricultural, recreational etc.) and sub-class (e.g. farmers, livestock grazers etc.), and by final good or service (e.g. land, flora, fauna etc.). The full combination of sub-classes results in 589 categories of ecosystem service. See https://www.epa.gov/eco-research/final-ecosystem-goods-and-services-classification-system-fegs-cs for full documentation and guidance on the FEGS classification.

The Nature's Contributions to People (NCP) classification developed for the International Platform on Biodiversity and Ecosystem Services (IPBES) uses three categories termed "material", "non-material" and "regulating". These categories broadly correspond to the provisioning, cultural and regulating categories used in the MA, TEEB and CICES classifications. Within but slightly extending across these three categories, the NCP classification identifies 18 distinct NCPs, which to a large extent correspond with the MA and TEEB ecosystem services. The "maintenance of options" NCP partially corresponds with supporting services. The NCP classification explicitly allows flexibility in the conceptualisation and definition of ecosystem services to reflect specific local conditions and cultures ("context-specific perspective") although this also negates to role of the classification system in enabling consistent comparisons across different assessments. The NCP classification also includes a number of overlapping definitions. For example, NCPs 3, 4 and 5 on regulation of air quality, climate and ocean acidification all include the regulation of carbon dioxide in the atmosphere; similarly, NCPs 7 and 9 on regulation of freshwater and extreme events both include the regulation of flooding. See Diaz et al., (2018) for an explanation and of the NCP classification.

The choice of the most appropriate ecosystem services classification system depends on the objective of the study and the decision-making context. The strengths and weaknesses of each classification system are summarised in Table 2. However, the use of different classification systems across valuation studies in the Western Indian Ocean (and indeed the rest of the world) reduces the comparability and transferability of results. On balance, CICES arguably offers the best combination of flexibility, consistency and wide applicability.

Table 2. Strengths and weaknesses of ecosystem service classification systems

	Strengths	Weaknesses
MA	Widely used; Straightforward	Inclusion of supporting (intermediate) services and final services can lead to double-counting if used incautiously
TEEB	Widely used; Straightforward	Category of "habitat services" is not an improvement on "supporting services"
CICES	Widely used; Optional inclusion of abiotic resources; Four level hierarchical structure (section, division, group, class) adds flexibility on level of detail	Four level hierarchical structure (section, division, group, class) adds complexity; Unnecessary distinction between terrestrial and aquatic ecosystems to define services; Some final services implicitly include other capital inputs (e.g. cultivated plants and reared animals)
FEGS	Explicit identification of beneficiaries	Missing some regulating services (e.g. flood mitigation, climate regulation); Inclusion of abiotic resources (e.g. weather); Overlapping categories (e.g. weather and wind); FEGS are not self-explanatory and require further information; Orthogonal approach produces an over-whelming number of sub-categories
NCP	Used by the International Panel on Biodiversity and Ecosystem Services (IPBES); Explicitly includes negative contributions of nature; "context-specific" perspective allows flexibility	Distinction between "generalising" and "context-specific" perspectives is confusing and negates comparisons across studies; Risk of double counting overlapping categories (e.g. regulation of freshwater and hazardous events); The "supporting identities" category lumps together many cultural services

4. The case for valuation of ecosystem services

4.1 The case for valuation of ecosystem services

The rationale for valuation of ecosystem services to support decision making is as follows. Ecosystem services contribute substantially to human welfare and in some cases are fundamental to sustaining life (e.g. climate regulation, nutrient recycling). The natural capital from which these services flow is, however, finite and cannot necessarily be regenerated or replaced. With growing human populations and consumption per capita increasing over time, it is highly likely that human use of natural resources will outstrip their availability (i.e. human use of the environment will be unsustainable). These simple realities of resource limitation mean that choices have to be made between alternative uses of available resources; and every time a decision is made to do one thing, this is also a decision not to do another. In other words, we are implicitly placing values on each option. This valuation is unavoidable and is the essence of decision making. So if valuation of alternative resource uses is unavoidable in making decisions, it is arguably better to make these values explicit and ensure that they are well informed in order to aid decision making. The valuation of ecosystem services attempts to do this.

4.2 The case for economic valuation

Economic value is simply a means to describe how important the things we use are to us, including our use of the natural world or "natural capital". In the case of ecosystem services from the coastal and marine environment, there are often no prices that reflect their value, since the services that are provided are not traded in markets (e.g. climate regulation, coastal protection, biodiversity). As a result, we tend not to take the value of ecosystem services into consideration when we make decisions that affect the marine and coastal environment. When we investigate the consequences of environmental change (e.g. climate change, development, marine accidents) we need to fully understand the effects on ecosystem services and human wellbeing. Economic valuation tries to measure the importance of environmental change, usually in monetary terms, in order to communicate the scale of impacts to human wellbeing. Such information can be used to raise awareness of the economic importance of marine ecosystems, set fees for the use of marine ecosystem services, or determine compensation payments for environmental damage.

Economic valuation of ecosystem services involves identifying and quantifying the contribution of environmental resources to human wellbeing; and incorporating this information into decision-making and the design of financing mechanisms and policy instruments.

Economic valuation methods do not stand alone but are often used in combination with other methods for assessing environmental change and the provision of ecosystem services. The added value of using economic valuation methods is that the importance of ecosystem services is expressed in terms of human welfare and measured in common units (i.e. money), allowing values to be aggregated across ecosystem services and directly compared with the values of other goods and services in the economy.

4.3 Decision-making contexts that potentially use information on ecosystem service values

There are many decision-making contexts in which information on the value of coastal and marine ecosystems may be useful, including to:

- Raise ***awareness*** of the value of the marine environment. Estimates of the value of ecosystems can highlight its importance to the public and to policy makers;
- Design effective ***policy instruments*** for environmental management. Resource use and polluting activities affecting coastal and marine ecosystems can be managed using policy instruments such as taxes, transferable quotas, certification and labelling, and trade restrictions;
- Design mechanisms for ***sustainable financing***, including setting appropriate *fees* for use of ecosystem services. This is relevant to sustain financing for resource management after initial project funding ends;
- **Compare *costs and benefits*** of alternative uses of the environment. This may be done, for example, in the context of Marine Spatial Planning to evaluate the net benefits from alternative activities;
- Reveal the ***distribution of costs and benefits*** of management decisions among different stakeholders. Transparently measuring who incurs the costs and who receives the benefits of resource management provides key information for decision makers;
- Include ecosystem service values in ***green accounts*** with the aim of measuring the importance natural capital to the economy and identify whether the exploitation of resources is unsustainable;
- **Set *compensation for environmental damage***. Information on the full costs of marine accidents (e.g. oil spills, ship groundings) can be used to determine the level of compensation that needs to be paid.

4.4 Limitations and criticisms of ecosystem valuation

The concept of ecosystem services provides a useful framework for identifying and quantifying the benefits that humans derive from nature. There are, however, a number of limitations to the effective implementation of this framework and criticisms of attempting to value ecosystem services.

The limitations or barriers to implementing the ecosystem service approach include:

- Lack of knowledge and understanding of the underlying state and functioning of ecosystems. The bio-physical relationships between ecosystem functioning and the provision of ecosystem services are often not well understood and are characterised by high uncertainties. Similarly, the understanding of long-run impacts, sustainability, positive and negative feedbacks and thresholds effects is limited. An understanding of such relationships, however, is fundamental to determining how policy decisions that affect natural capital stocks and ecosystem functioning will filter through to changes in the flow and value of ecosystem services.
- A related challenge in assessing ecosystem services is due to the complexity of trade-offs between different ecosystem services. In many cases, the level of sustainable activity for one ecosystem service may not be compatible with the sustainable level of another. For

example, trade-offs have been observed between fisheries and tourism sectors in which restricting one, benefits the other. Such trade-offs introduce further complexity to any analysis since it becomes necessary to consider how the one use of a marine resource affects other potential uses. This, however, can also be seen as a strength of the ecosystem service framework in that is enables these trade-offs to be explicitly analysed.

- Ecosystem service assessments are resource intensive and time consuming. The physical and social scientific methods applied to assess ecosystem services are sophisticated, time consuming and often expensive to implement. Assessment methods generally require extensive data, which may not be available especially for small scale studies. Moreover, the necessary technical expertise to conduct valuation studies is often lacking in the agencies that are responsible for environmental protection and resource management.

The criticisms and potential risks of the ecosystem service approach include:

- Quantification and valuation of ecosystem services may lead to their commodification and sale. Many ecosystem services are public goods that beneficiaries enjoy without any charge for their use. There is concern that the process of quantifying the value of such services is a step towards setting prices for them and requiring beneficiaries to pay. Such a development potentially represents a transfer of wealth from beneficiaries to resource owners.
- The explicit identification of resource owners, custodians, users, and beneficiaries can raise questions of property rights, tenure and conflict. The tenure or property rights to many natural resources remains unassigned. For society, this can be both a positive characteristic from the perspective that such resources are open to all, or a negative characteristic from the perspective that such resources tend to be over-exploited. A potential risk in applying an ecosystem service approach is that issues of resource ownership become sources of conflict between different stakeholders.
- Valuation of ecosystem services can lead to changes in the management of natural resources to favour the highest value uses, to the detriment of lower valued uses. A potential result of an ecosystem assessment is the recommendation to manage a resource (e.g. a coral reef) to increase the availability of high value ecosystem services (e.g. tourism) as the expense of relatively low value ecosystem services (e.g. subsistence fisheries). Without sufficient and appropriate compensation, this can have major distributional consequences across stakeholder groups.
- The framing of ecosystem services as nature's contributions to people is contrary to traditional understanding of the relationship between humans and the environment in some cultures and can disrupt traditional approaches to managing common natural resources. The concept of humans as recipients of benefits from nature, as opposed to part of the natural system, might be at odds with some indigenous and traditional systems of managing natural resources even to the point that it alters the effectiveness of such systems.
- The ecosystem service approach narrows the conception of the value of nature to anthropocentric or utilitarian values. The concept of nature having intrinsic value irrespective of any benefits it contributes to people does not fit in the ecosystem services framework.

- The ecosystem services approach has not resulted in substantial changes in environmental policy or human behaviour to address the serious environmental challenges that the planet faces. The required scale of change in humanity's use of natural resources to avoid major environmental disasters (e.g. climate change, massive loss of biodiversity) is not taking place. The ecosystem services approach has only resulted in small incremental changes in environmental and development policies and not the fundamental changes in economic systems that are necessary.

5. Ecosystem valuation methods

A variety of methods have been developed for quantifying the importance of ecosystem services. These valuation methods are designed to span the range of complex interactions between the natural environment and people. Here we make a distinction between bio-physical, economic and social approaches to measuring the value of ecosystem services. Table 3 summarises the broad strengths and weaknesses of each set of methods.

Bio-physical approaches use data and models to assess the physical provision of ecosystem services. They generally focus on the righthand side of the ecosystem services cascade (Figure 2) and measure the supply of services. Ecosystem service values are expressed in physical units or indicators (e.g. stocks of fish, extent and condition of mangroves/coral reefs, etc.).

Economic approaches use data on activities and expenditure by beneficiaries or collect data through surveys to elicit people's preferences and willingness to pay for ecosystem services. Ecosystem service values are usually expressed in monetary units but can be expressed in other units that households value and allocate to alternative uses (e.g. time).

Social approaches use surveys, interviews, dialogues, workshops and other participatory approaches to collect data on people's perceptions and preferences for ecosystem services. Social methods express the value of ecosystem services in a variety of qualitative and quantitative terms.

It is important to note that bio-physical, economic and social valuation methods are not necessary substitutes but more often provide complementary information on the importance of ecosystem services. Individual methods within each of these categories are introduced in the following sub-sections, with a more in-depth focus on economic valuation methods.

Table 3. Summary of bio-physical, economic and social valuation methods

Method	Approach	Strengths	Weaknesses
Bio-physical	Quantifies physical variables that determine or indicate the provision of ecosystem services	+ Uses a variety of objective, data-based, scientific methods to quantify ecosystem services in physical units; + Generalised models for most ecosystem services are available	- High technical and data requirements; - Focuses on supply side without reflecting demand for ecosystem services; - Some ecosystem services are difficult to value in physical units (e.g. aesthetic enjoyment); - Physical units for different ecosystem services cannot be easily aggregated or compared
Economic	Estimates the contribution of ecosystem services to human wellbeing, usually measured in monetary units	+ Money as a unit of measurement can be aggregated and directly compared to other values that are relevant to decision makers	- Some methods have high technical and data requirements; - Some ecosystem services are difficult to value in monetary terms (e.g. biodiversity existence);

Method	Approach	Strengths	Weaknesses
			- Use of money as a unit of measurement is repugnant to some stakeholders - Elicited values can be constrained by monetary income/wealth and therefore aggregate results can disproportionately reflect the preferences of wealthy beneficiaries
Social	Measures the perception of ecosystem services and contribution to human wellbeing in non-monetary (qualitative and quantitative) units	+ Participatory methods aid learning and knowledge production; + Enables measurement of multiple value concepts; + Elicited values are not constrained by monetary income/wealth	- Different units of measurement and value concepts cannot be easily aggregated or compared; - Methods can be manipulated and subject to bias

5.1 Bio-physical valuation methods

Biophysical methods for measuring the importance of ecosystem services are based on quantification of different parameters of biotic and abiotic structures that determine the provision of ecosystem services. Biophysical quantification is built on spatial and temporal measures of ecosystem processes. Following Vihervaara et al. (2018), biophysical methods are grouped into three categories distinguished by the character of the measurements and

how the information is extracted: 1. Direct measurements; 2. Indirect measurements; 3. Modelling (see Figure 4).

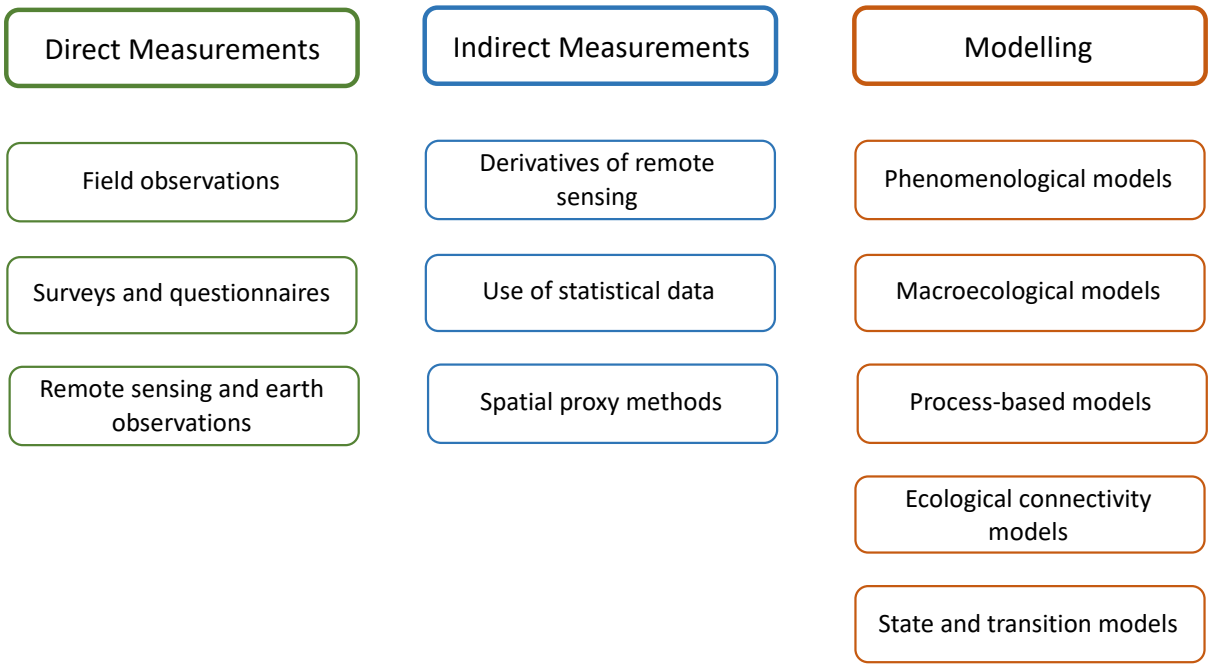


Figure 4: Overview of bio-physical valuation methods

Direct measurement methods of ecosystem services are the measurements of a state, a quantity, or a process from ecosystem observations, monitoring, surveys, questionnaires, or data from remote sensing and earth observations, which cover the entire study area in a representative manner. Direct measurements deliver a biophysical value of an ecosystem service in physical units that correspond to the units of the indicator, and quantify or measure a stock or a flow value. Direct measurements are also used as primary data to other methods, as they are one of the most accurate ways to quantify ecosystem services. However, they are often impractical and expensive beyond the site level, and therefore are usually used as an input for other biophysical mapping methods or to validate certain mapping and assessment elements. In many cases, direct measurements are simply not available for all ecosystem services.

Indirect measurement methods rely on the use of different data sources to measure biophysical values in physical units, but such values need further interpretation, assumptions, or data processing before they can be used. Indirect measurements can be based on remote sensing and earth observation derivatives such as land cover, Normalised Difference Vegetation Index (NDVI), surface temperature, or soil moisture, which are extracted from the original sources by specific procedures. For example, land cover can be derived from remote sensing images through visual interpretation or automated classification, whereas NDVI is derived by measuring the difference of particular spectral bands.

Modelling methods include several groups of modelling approaches from ecology (phenomenological, macro-ecological, trait-based), statistics, or other earth sciences fields such as hydrology, climatology, soil science etc. Conceptual models and integrated modelling frameworks are also included in this group.

5.2 Economic valuation methods

Economic valuation methods include a wide range of approaches for estimating the contribution of ecosystem services to human wellbeing. As such, they focus on the right-hand side of the ecosystem services cascade (see Figure 2). The intention of the present guidelines is to provide an understanding of which valuation method can be used to value each ecosystem service, and explain the key strengths, limitations and data requirements of each method. For selected methods, the main steps in conducting a valuation are explained in Methods Boxes. It is beyond the scope of these guidelines, however, to provide a complete manual on how to apply each valuation method. Indeed, some methods are highly technical, require advanced expertise and have lengthy manuals devoted to explaining their application. Useful resources and manuals for detailed instructions are listed in Annex 3.

Figure 5 provides a representation of the available economic methods for valuing ecosystem services. A first categorisation of methods is into "primary valuation methods" and "value transfer methods". Primary valuation methods produce new or original information generally using primary data whereas value transfer methods use existing information from primary valuation studies in new policy contexts.

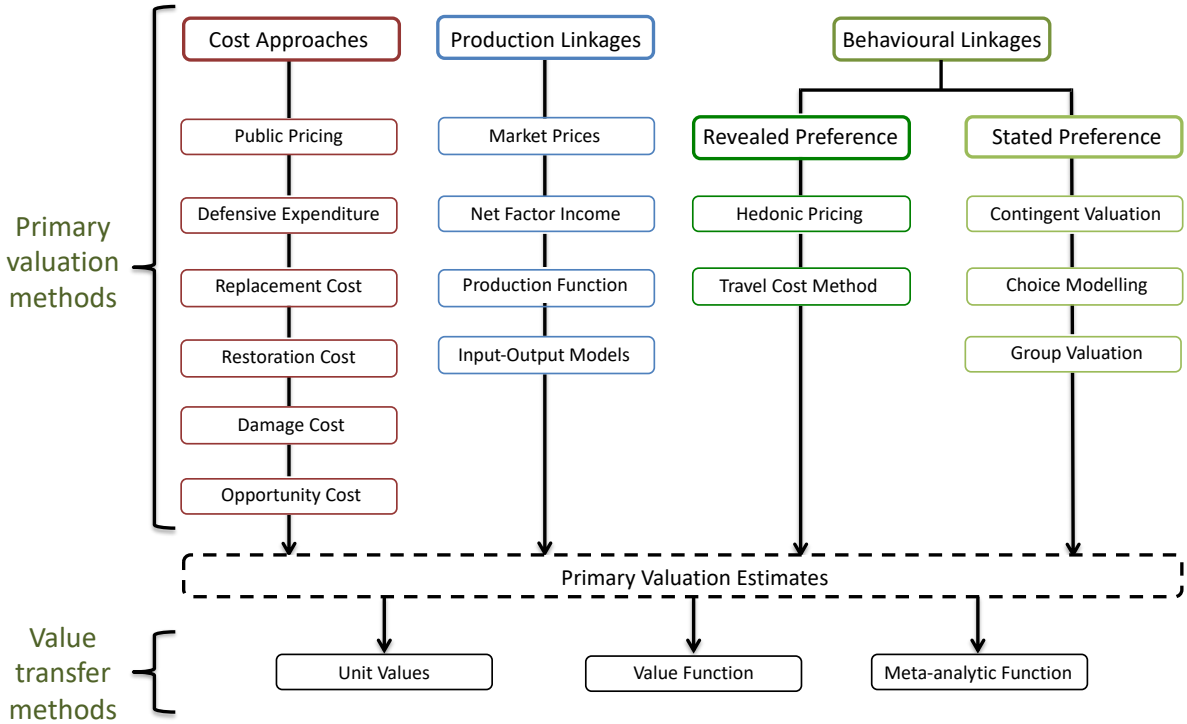


Figure 5: Overview of economic valuation methods

Primary valuation methods

Primary valuation methods are those that produce new or original value information generally using primary data. Table 4 provides an overview of primary valuation methods, typical applications, limitations and indicates which primary valuation methods can be used to value which ecosystem service.

An important distinction between primary valuation methods is the difference between revealed preference methods (those that observe actual behaviour of the use of ecosystem services to elicit values) and stated preference methods (those that use public surveys to ask

beneficiaries to state their preferences for, generally hypothetical, changes in the provision of ecosystem services). Revealed preference methods may be favoured since they reflect actual behaviour but are limited in their applicability to some ecosystem services. Stated preference methods on the other hand rely on responses recorded in surveys or experiments but are more flexible in their application and can in principle be used to value any ecosystem service.

It should be noted that different valuation methods produce different measures of economic value that are not necessarily equivalent and cannot be directly compared. The valuation method, and the measure of economic value that it estimates, will have a substantial bearing on the magnitude of the value estimated. It is therefore important to understand what each measure is and to select a measure that is relevant to the case in hand. There are numerous existing publications that provide guidance on the use of primary valuation methods. A selection of these are listed in Annex 3.

The choice of which valuation method to use is determined to a large extent by what is being valued. The applicability of some valuation methods is limited to specific ecosystem services. Figure 6 illustrates this by drawing linkages between a set of ecosystem services and the valuation methods that are most applicable to value them.

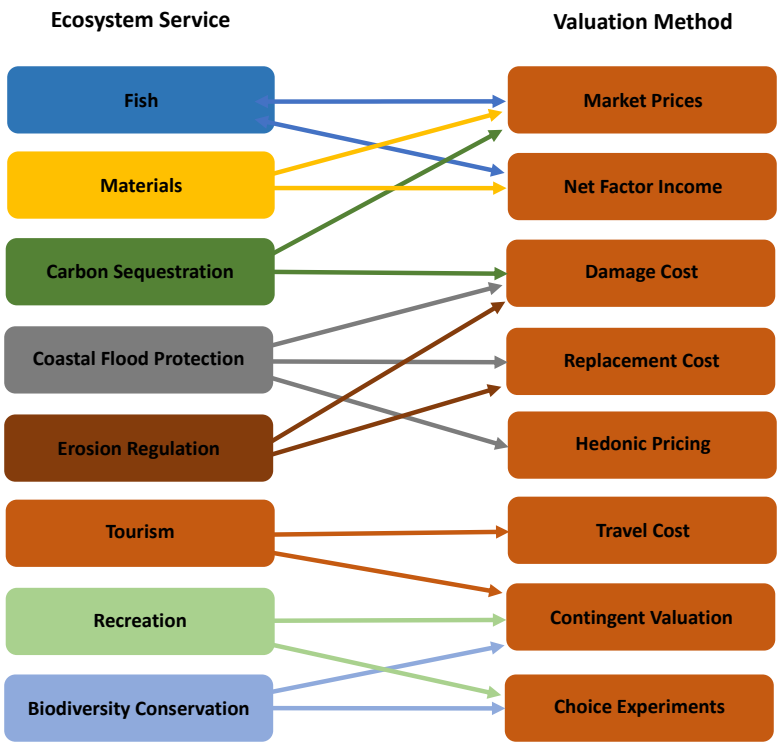


Figure 6: Linkages between ecosystem services and relevant primary economic valuation methods

Table 4: Primary valuation methods, applicability to ecosystem services, examples and limitations (adapted from Table A2, Brander 2013)

Valuation method	Approach	Data requirements and sources	Application to ecosystem services	Example ecosystem service	Limitations
Market prices	Prices for ES that are directly observed in markets	Prices of some ES can be obtained from markets or surveys of businesses and households	ES that are traded directly in markets	Timber and fuel wood from mangroves; fish; ecotourism	Market prices can be distorted e.g. by subsidies. Most ES are not traded in markets
Public pricing	Public expenditure or monetary incentives (taxes/subsidies) for ES as an indicator of value	Data on public expenditures on the provision of ES obtained from government reports or key informants	ES for which there are public expenditures	Watershed protection to provide drinking water; Purchase of land for protected area	No direct link to preferences of beneficiaries
Defensive expenditure	Expenditure on protection of ES	Data on public or private expenditure obtained from government reports, key informants, or surveys of businesses and households	ES for which there is public or private expenditure for its protection	Recreation and aesthetic values from protected areas	Only applicable where direct expenditures are made for environmental protection related to provision on an ES. Provides lower bound estimate of ES benefit
Replacement cost	Estimate the cost of replacing an ES with a man-made service	Estimates of construction costs can be obtained from experts or based on past investments	ES that have man-made equivalents	Coastal protection by dunes (replaced my seawalls); water storage and filtration by wetlands (replaced by reservation and filtration plant)	No direct relation to ES benefits. Over-estimates value if society is not prepared to pay for man-made replacement. Under-estimates value if man-made replacement does not provide all of the benefits of the original ecosystem.
Restoration cost	Estimate cost of restoring degraded ecosystems to ensure provision of ES	Estimates of restoration costs can be obtained from experts or based on past investments	Any ES that can be provided by restored ecosystems	Coastal protection by dunes; water storage and filtration by wetlands	No direct relation to ES benefits. Over-estimates value if society is not prepared to pay for restoration. Under-estimates value if restoration does not provide all of the benefits of the original ecosystem.

Valuation method	Approach	Data requirements and sources	Application to ecosystem services	Example ecosystem service	Limitations
Damage cost avoided	Estimate damage avoided due to ecosystem service	Data on past damage costs and frequencies can be obtained from government reports and household surveys	Ecosystems that provide storm, flood or landslide protection to houses or other assets	Coastal protection by dunes, mangroves and reefs; river flow control by wetlands; landslide protection by forests	Difficult to quantify changes in risk of damage to changes in ecosystem condition.
Social cost of carbon (SCC)	The monetary value of damages caused by emitting one tonne of CO_2 in a given year. The social cost of carbon (SCC) therefore also represents the value of damages avoided for a one tonne reduction in emissions.	Estimates of the SCC can be obtained from Integrated Assessment Models of climate-economy impacts and published summaries of model results	Carbon storage and sequestration	Carbon sequestered and stored by protected or restored mangroves	SCC is a specific application of the "damage cost avoided" method. SCC is characterised by high modeling uncertainties and partial coverage of climate change impacts.
Opportunity cost	The next highest valued use of the resources used to produce an ecosystem service.	Data on the value of alternative land uses (e.g. agriculture, aquaculture, housing, hotels etc.) can be obtained from markets and surveys of businesses and households	All ecosystem services	The opportunity cost of ecosystem services from a natural ecosystem might be the value of agricultural output if the land is converted to agriculture instead of conserved in a natural state.	Measures the cost of providing ecosystem services instead of the benefit.
Net factor income (residual value)	Revenue from sales of a marketed good with an ES input minus the cost of other inputs	Revenues can be obtained from markets; costs can be obtained from business surveys	Ecosystems that provide an input in the production of a marketed good	Filtration of water by wetlands; commercial fisheries supported by mangroves and reefs	Tendency to over-estimate values since all normal profit is attributed to the ES
Production function	Statistical estimation of production function for a marketed good with an ES input	Data on production, inputs, costs and revenues can be obtained from business surveys	Ecosystems that provide an input in the production of a marketed good	Soil quality or water quality as an input to agricultural production	Technically difficult. High data requirements

Valuation method	Approach	Data requirements and sources	Application to ecosystem services	Example ecosystem service	Limitations
Input-Output Models	Quantifies the interdependencies between economic sectors in order to measure the impacts of changes in one sector to other sectors in the economy. Ecosystems can be incorporated as distinct sectors.	Data on production inputs, outputs and prices for multiple economic sectors can be obtained from government statistics. Data on ecosystem inputs and outputs can be observed or modelled using bio-physical methods	Ecosystem services with direct and indirect use values, particularly inputs into production	Ecosystem inputs into agriculture; or into the tourism sector	Requires substantial data on ecosystem-economy linkages to parameterise connections between sectors
Hedonic pricing	Estimate influence of environmental characteristics on price of marketed goods (usually residential property)	Data on house prices and characteristics can be obtained from estate agents or public records. Data on environmental characteristics can be observed or modelled using bio-physical methods	Environmental characteristics that vary across goods (usually houses)	Urban green open space; air quality moderated by ecosystems	Technically difficult. High data requirements. Limited to ES that are spatially related to property locations.
Travel cost	Estimate demand for ecosystem recreation sites using data on travel costs and visit rates	Data on travel costs and visit rates can be obtained through visitor surveys	Recreational use of ecosystems	Recreational use of beaches, reefs, national parks etc.	Technically difficult. High data requirements. Limited to valuation of recreation. Complicated for trips with multiple purposes or to multiple sites.
Contingent valuation	Ask people to state their willingness to pay for an ES through surveys	Data collected through public surveys	All ecosystem services	Biodiversity; recreation; landscape aesthetics; flood risk attenuation	Expensive and technically difficult to implement. Risk of biases in design and analysis
Choice modelling (choice experiment)	Ask people to make trade-offs between ES and other goods to elicit willingness to pay	Data collected through public surveys	All ecosystem services	Biodiversity; recreation; landscape aesthetics; flood risk attenuation	Expensive and technically difficult to implement. Risk of biases in design and analysis

Valuation method	Approach	Data requirements and sources	Application to ecosystem services	Example ecosystem service	Limitations
Group / participatory valuation	Ask groups of stakeholders to state their willingness to pay for an ES through group discussion	Data collected in workshop settings	All ecosystem services	Biodiversity; recreation; landscape aesthetics; flood risk attenuation	Risk of biases due to group dynamics

Methods Box: Avoided damage costs

The avoided damage costs method measures the value of protection by coastal ecosystems as the cost of expected damages to homes, businesses, agriculture, or public infrastructure that will be avoided because of the presence of coastal ecosystems. The damage costs method requires (i) determination of the extent of protection provided by natural ecosystems, (ii) the population, property, and human infrastructure at risk from erosion or flood damage, and (iii) the probability of damages given the estimated frequency of flood or erosion events.

The main steps in applying the damage cost method are:

Step 1. Identify areas at risk from coastal flooding events. Areas at risk to coastal flooding are identified as sites that have a lower elevation than the maximum wave height at high tide and up to 1 km inland.

Step 2. Assess the level of coastal protection for each segment of coast using a Coastal Protection Index (CPI). A CPI is a qualitative assessment of the multiple factors determining the overall level of coastal protection.

Step 3. Identify assets at risk. Assets at risk to flooding can include people, buildings, transport and communication infrastructure, vehicles, livestock and crops.

Step 4. Estimate damage costs in the event of a flood. Damage costs to buildings in the event of coastal flooding can be calculated as a fixed percentage of total construction costs. This approximates the cost of repairing a building in the case that it is damaged by flood.

Step 5. Estimate the probability of a storm surge. The probability of a storm surge occurring is used to annualize the value of damage costs. The spatial and temporal occurrence and severity of past events can be used to predict potential tropical cyclones that may affect the study zone in the future.

Step 6. Compute the expected annual damage due to coastal flooding. The expected annual damage is a function of the frequency of storm surges, the level of coastal protection (represented by the CPI) and the value of assets at risk. A general equation for expected annual damage costs due to coastal flooding is:

$$D_t = P_t * (1\text{-}CPI) * (A*C*DF)$$

where:

D_t = expected flood damage in year t

P_t = probability of storm surge in year t

CPI = coastal protection index

A = assets at risk (e.g. numbers of houses)

C = construction costs (e.g. cost per house)

DF = damage factor (flood damage as a % of construction cost)

Step 7. Compute the avoided damage due to the presence of coastal ecosystems. Re-calculate the expected damage costs of flooding assuming the absence of coral reef (i.e. set the coastal protection scores for the two factors describing coral reefs in the CPI to the

lowest level). The difference between the expected damage with and without reefs is the avoided damage due to the presence of reefs.

Source: Adapted from Salcone et al. (2016). Guidance manual on economic valuation of marine and coastal ecosystem services in the Pacific.

Methods Box: Social cost of carbon

The social cost of carbon (SCC) is the monetary value of damages caused by emitting one additional tonne of CO_2 in a given year. The SCC therefore also represents the value of damages avoided for a small reduction in emissions, in other words, the benefit of a CO_2 reduction. The SCC is intended to be a comprehensive estimate of climate change damages but due to current limitations in the integrated assessment models and data used to estimate SCC, it does not include all important damages and is likely to under-estimate the full damages from CO_2 emissions. The estimated SCC used by the US EPA and other US agencies for appraisal of emissions reductions in 2015 is US$ 56/tonne CO_2.

Mangroves, salt marshes, seagrasses, and algae (pelagic or benthic) all remove carbon dioxide from the atmosphere and store it in their fibres, in the soil, and/or in the ocean substrate. The amount of carbon that is captured from the atmosphere by different plant species can be quantified in terms of a rate of sequestration. If a tree or plant is destroyed or damaged, the carbon stored in the plant's cells is released as the biomass decays or burns. Carbon stored in the soil/substrate may be released over time if left un-vegetated, or released quickly if the substrate is disturbed. Both the rate at which carbon is added to biomass/substrate (*sequestration rate*) and any release of stored carbon are important and can be used together to calculate the net change in atmospheric carbon dioxide, in a given time period. Data on the rates of carbon sequestration by different ecosystems and the extent of those ecosystems can be used to estimate annual quantities of carbon sequestration; data on the quantity of stored carbon in different ecosystems and reductions in extent of those ecosystems can be used to estimate the annual quantity of carbon prevented from release or decay into the atmosphere.

By convention, quantities of carbon are often expressed in terms of tonnes of CO_2-equivalent in order to allow comparison with other greenhouse gases. The conversion rate between carbon and CO_2 is 1 tC = 3.67 t CO_2.

The main steps in applying the social cost of carbon to value sequestration and storage by coastal and marine ecosystems are:

1. Estimate the quantity of carbon added to the stock of carbon stored in coastal ecosystems during the current year.
    1.1. Obtain data on the current spatial extent of mangroves and seagrass beds.
    1.2. Compute the quantity of carbon sequestered in the current year (i.e. the addition to the stored stock of carbon in that single year). Multiply the area of each ecosystem by estimates of the annual sequestration rate of each ecosystem. Where available, use estimates that reflect local species and conditions. The Blue Carbon Initiative summarized global coastal carbon data and report an average sequestration rate for mangroves of 6.3 tCO_2/ha/yr.
2. Estimate the (potentially avoided) quantity of carbon released due to reductions in area of coastal ecosystems.

2.1. Identify current rates of change in areas of coastal ecosystems.

2.2. Compute the change in area of each ecosystem in the current year (total area of ecosystem multiplied by percentage change).

2.3. Compute the quantity of stored carbon released to the atmosphere. Here it is necessary to make an assumption regarding the rate at which stored carbon is released following a change in land use from coastal ecosystem to some other land use, such as agriculture or commercial/industrial development.

2.3.1. Compute the quantity of carbon stored in living biomass using available estimates. For mangroves, average biomass carbon ranges between 237 t CO_2-eq/ ha - 563 t CO_2-eq/ha. Regarding the rate at which biomass carbon is released, it can be assumed that if the mangrove is burned, 75% of biomass carbon for mangroves is released immediately and that the remaining 25% decays with a half-life of 15 years (i.e. a further 12.5% is released within 15 years, a further 6.25% is released within 15 years after that, etc.) (Murray et al., 2011).

2.3.2. Compute the total quantity of carbon stored in soil that is released following removal of the ecosystem using available estimates. The average amount of carbon stored in the top meter of soil beneath mangroves is 1060 t CO_2-eq /ha for estuarine mangroves and approximately 1800 t CO_2-eq /ha for oceanic mangroves (Murray et al. 2011). Regarding the rate at which this is released, it can be assumed that mangrove soil organic carbon has a half-life of 7.5 years (i.e. 50% of the stored carbon is released in the first 7.5 years, 25% in the following 7.5 years, etc.) (Murray et al., (2011).[1]

3. Value the flow of carbon

3.1. For additions to the stocks of carbon stored in each ecosystem, multiply the annual quantity of sequestered carbon in step 1.2 (tonnes CO_2-eq) by the social cost of carbon.

3.2. For the market value of (potentially avoided) carbon release, the "benefit" is the sale of carbon credits for avoided emissions. In this case, multiply the total quantity of (potentially avoided) carbon emissions (tonnes CO_2-eq) estimated in step 2.31 and 2.32 by the market price.[2] If relevant cost data is available, subtract the costs of managing and crediting emissions reductions to estimate producer surplus.

Note that the observed price in carbon markets reflects the value to the resource owners (i.e. what price can they sell their carbon for), whereas the social cost of carbon represents the global benefits of sequestering and storing carbon. The problem with using market prices for carbon is that such prices are largely artefacts of the set up and regulation of the market and do not reflect the benefits of carbon sequestration. It is therefore advisable to

[1] An alternative assumption, also from Murray et al. (2011), is that oceanic mangroves release 82 t CO_2e/ha/yr and estuarine mangroves release 59tCo2e/ha/yr for 25 years following clearance of the mangrove trees.

[2] This calculation is made with the assumption that avoided emissions that will occur in the future (i.e. as biomass and soil carbon is released over time) can be credited and sold in the current year. If this is not the case, it would be necessary to estimate the quantity of carbon released in each year following the land use change and then compute a present value of the stream of credits.

use SCC for assessments of the global value of carbon sequestration by ecosystems. The use of carbon market prices should, however, be used in financial assessments of carbon sequestration projects in order to reflect potential revenues for the project. An indicative estimate of the price of carbon credits on the voluntary market is provided by Forest Trends (2014), which reports an average price in 2013 of US$ 4.90 t CO_2-eq.

Source: Adapted from Salcone et al. (2016) Guidance manual on economic valuation of marine and coastal ecosystem services in the Pacific; and Brander (2018) Toolkit on environmental economics for marine ecosystem management.

Methods Box: Contingent valuation

Contingent valuation is a stated preference method and involves directly asking people, in a survey, how much they would be willing to pay for specific changes in the provision of ecosystem services. The underlying idea behind this method is that a hypothetical, yet realistic, market for buying or selling the use and/or conservation of an ecosystem service can be described in detail to an individual, who then participates in the hypothetical market by responding to a series of questions. These questions relate to a proposed change in the provision or quality of the ecosystem service.

Contingent valuation may be a useful valuation method in the WIO context given its flexibility for valuing the full range of ecosystem services but it can involve complex data analysis and relatively expensive data collection.

The main steps in applying the contingent valuation method are:

Step 1. Define the policy issue in terms of the ecosystem services that need to be valued and the relevant population of beneficiaries.

Step 2. Design the survey. This involves a number of steps including deciding what type of survey will be used (main, telephone, face-to-face, internet) and developing the sampling strategy.

Step 3. Develop the questionnaire. This involves deciding on question formats, description of the ecosystem service to be valued, payment vehicle and willingness-to-pay question. Test the questionnaire on focus groups and/or small samples and adjust if necessary.

Step 4. Survey implementation. This includes recruiting and training enumerators, pilot testing the questionnaire and adjusting if necessary, full sampling, and data entry.

Step 5. Analysing the results. This includes cleaning the data and dealing with non-responses and protest bids. Mean WTP for the sample of respondents can be calculated and extrapolated to the relevant population of beneficiaries to estimate a total value for the ecosystem service.

Source: Adapted from van Beukering et al. (2007)

Case Study Box: Use of contingent valuation to estimate willingness to pay for marine tourism in Ponto do Ouro Partial Marine Reserve, Mozambique

The Ponto do Ouro Partial Marine Reserve (PPMR) stretches for 86 km along the coast from the border with South Africa to its northernmost point past Inhaca Island and into Maputo Bay. PPMR protects 678 km^2 of marine area, of which 6% is a no-take zone.

The reserve encompasses the habitats of many vulnerable marine species, including two species of sea turtle, dugongs, migratory birds and at least 10 species of shark, as well as a diverse Indo-Pacific fish community. The reefs within the reserve draw marine-based tourists to the area for various recreational activities including boating, dolphin swims, angling, jet-skiing, kite-boarding, SCUBA diving, snorkelling, spear-fishing, surfing and swimming.

A valuation study by Daly et al. (2015) applied the contingent valuation method to estimate the willingness-to-pay (WTP) of user groups for access to the reserve and to investigate the potential for the reserve to increase revenues for conservation through the implementation of a user fee for selected marine-based activities. The study interviewed 120 visitors and asked them how much they would be willing to pay per day as a user fee for their main recreational activity at the reserve. A payment card with a range between 1-200 South African Rand was used to elicit WTP amounts.

The results were used to estimate average WTP for three user groups:

	SCUBA divers	Dolphin tourists	Anglers
WTP (Rand/visit)	45.20	41.70	39.70

Regarding the potential revenue from introducing a user fee, the study estimates that a user fee of R 43.75 (the average WTP across user groups) charged to 36,800 visitors (the average number of visitors per year in 2011-2012) would bring in R 1.65 million per year.

The study noted that respondents voiced concerns about misuse of funds and inadequate management of the PPMR. This suggests that a lack of transparency, management and/or enforcement could lead to opposition to a user fee. Therefore, the implementation and collection of a user fee must be unambiguous and the use of revenues from a user fee must be well documented and conspicuous.

Source: Daly, C. A. K., Fraser, G., and Snowball, J. D. (2015). Willingness to pay for marine-based tourism in the Ponta do Ouro Partial Marine Reserve, Mozambique. African journal of marine science, 37(1), 33-40.

Methods Box: Choice modelling

Choice modelling or choice experiment is a stated preference method in which a public survey is used to elicit the preferences or values of respondents for specified changes in a good or service. Choice modelling is widely used in market research and economics to obtain information on public preferences that are otherwise not observable in consumer behaviour.

In practical terms, a choice model valuation involves asking survey respondents to make repeated choices between alternative multi-attribute descriptions of a good or service. By observing the trade-offs that are made between attributes, it is possible to estimate their relative values. By including one attribute that represents a monetary payment on the part of the respondent it is possible to compute the willingness to pay (WTP) for changes in the other attributes.

The main steps in conducting a choice modelling valuation are:

Step 1. Define the policy issue in terms of the ecosystem services that need to be valued and the relevant population of beneficiaries.

Step 3. Designing the choice cards. This involves selecting the ecosystem service attributes to be valued and the payment vehicle attribute, defining the levels used to describe each attribute, generating a statistical or experimental design that determines the combinations of attribute levels shown in each option on each choice card, and building the choice cards that will be shown to respondents.

Step 3. Develop the questionnaire. This involves drafting questions regarding use/knowledge of the ecosystem services to be valued, background socio-economic characteristics of respondents, and follow-up questions on the choice process. Test the questionnaire, including the choice cards, on focus groups and/or small samples and adjust if necessary.

Step 4. Survey implementation. This involves a number of steps including deciding what type of survey will be used (main, telephone, face-to-face, internet), developing the sampling strategy, recruiting and training enumerators, pilot testing the questionnaire and adjusting if necessary, full sampling, and data entry.

Step 5. Analysing the results. This includes cleaning the data and dealing with non-responses and protest bids. Choice data is generally analysed using multinomial logit regressions to estimate marginal utilities for each ecosystem service attribute and the payment vehicle. Mean WTP per respondent can be calculated by computing the ratio of the marginal utility of ecosystem service to the marginal utility of money – and extrapolated to the relevant population of beneficiaries to estimate a total value for the ecosystem service.

Source: Adapted from van Beukering et al. (2007)

Value transfer methods

Decision-making often requires information quickly and at low cost. New 'primary' valuation research, however, is generally time consuming and expensive. For this reason, there is interest in using information from existing primary valuation studies to inform decisions regarding impacts on ecosystems that are of current interest. This

transfer of value information from one context to another is called value or benefit transfer.

Value transfer is the use of research results from existing primary studies at one or more sites or policy contexts ("study sites") to predict welfare estimates or related information for other sites or policy contexts ("policy sites"). Value transfer is also known as benefit transfer but since the values that are transferred may be costs as well as benefits, the term value transfer is more generally applicable.

In addition to the need for expeditious and inexpensive information, there is often a need for information on the value of ecosystem services at a different geographic scale from that at which primary valuation studies have been conducted. So even in cases where some primary valuation research is available for the ecosystem of interest, it is often necessary to extrapolate or scale-up this information to a larger area or to multiple ecosystems in the region or country. Primary valuation studies tend to be conducted for specific ecosystems at a local scale whereas the information required for decision-making is often needed at a regional or multi-national scale. Value transfer therefore provides a means to obtain information for the scale that is required.

The number of primary studies on the value of ecosystem services is substantial and growing rapidly. This means that there is a growing body of evidence to draw on for the purposes of transferring values to inform decision-making. With an expanding information base, the potential for using value transfer is improved.

Value transfer can potentially be used to estimate values for any ecosystem service, provided that there are primary valuations of that ecosystem service from which to transfer values. Value transfer methods have been employed widely in national and global ecosystem assessments, value mapping applications and policy appraisals. The use of value transfer is widespread but requires careful application. The alternative methods of conducting value transfer are described here:

1. Unit value transfer uses values for ecosystem services at a study site, expressed as a value per unit (usually per unit of area or per beneficiary), combined with information on the quantity of units at the policy site to estimate policy site values. Unit values from the study site are multiplied by the number of units at the policy site. Unit values can be adjusted to reflect differences between the study and policy sites (e.g. income and price levels).
2. Value function transfer uses a value function estimated for an individual study site in conjunction with information on parameter values for the policy site to calculate the value of an ecosystem service at the policy site. A value function is an equation that relates the value of an ecosystem service to the characteristics of the ecosystem and the beneficiaries of the ecosystem service. Value functions can be estimated from a number of primary valuation methods including hedonic pricing, travel cost, production function, contingent valuation and choice experiments.
3. Meta-analytic function transfer uses a value function estimated from the results of multiple primary studies representing multiple study sites in conjunction with information on parameter values for the policy site to calculate the value of an

ecosystem service at the policy site. A value function is an equation that relates the value of an ecosystem service to the characteristics of the ecosystem and the beneficiaries of the ecosystem service. Since the value function is estimated from the results of multiple studies it is able to represent and control for greater variation in the characteristics of ecosystems, beneficiaries and other contextual characteristics. This feature of meta-analytic function transfer provides a means to account for simultaneous changes in the stock of ecosystems when estimating economic values for ecosystem services (i.e. the "scaling up problem"). By including an explanatory variable in the data describing each "study site" that measures the scarcity of other ecosystems in the vicinity of the "study site", it is possible to estimate a quantified relationship between scarcity and ecosystem service value. This parameter can then be used to account for changes in ecosystem scarcity when conducting value transfers at large geographic scales.

These three principal methods for transferring ecosystem service values are summarised in Table 5 together with their respective strengths and weaknesses. The choice of which value transfer method to use to provide information for a specific policy context is largely dependent on the availability of primary valuation estimates and the degree of similarity between the study and policy sites. In cases where value information is available for a highly similar study site, unit value transfer may provide the most straightforward and reliable means of conducting value transfer. On the other hand, when study sites and policy sites are different, value function or meta-analytic function transfer offers a means to systematically adjust transferred values to reflect those differences. Similarly, in the case that value information is required for multiple different policy sites, value function or meta-analytic function transfer may be a more accurate and practical means for transferring values. Using meta-analytic functions that include a parameter for ecosystem scarcity provides a means to account for simultaneous changes in the stock of ecosystem on the value of all ecosystem services (i.e. more accurately "scale-up" ecosystem service values).

Table 5: Value transfer methods, strengths, weaknesses (adapted from Table 3, Brander 2013)

Method	Approach	Strengths	Weaknesses
Unit value transfer	Select appropriate values from existing primary valuation studies for similar ecosystems and socio-economic contexts. Adjust unit values to reflect differences between study and policy sites (usually for income and price levels)	Simple	Unlikely to be able to account for all factors that determine differences in values between study and policy sites. Value information for highly similar sites is rarely available
Value function transfer	Use a value function derived from a primary valuation study to estimate ES values at policy site(s)	Allows differences between study and policy sites to be controlled for (e.g. differences in population characteristics)	Requires detailed information on the characteristics of policy site(s)

Method	Approach	Strengths	Weaknesses
Meta-analytic function transfer	Use a value function estimated from the results of multiple primary studies to estimate ES values at policy site(s)	Allows differences between study and policy sites to be controlled for (e.g. differences in population characteristics, area of ecosystem, abundance of substitutes etc.). Practical for consistently valuing large numbers of policy sites.	Requires detailed information on the characteristics of policy site(s). Analytically complex

Methods Box: Unit value transfer

Unit value transfer uses primary valuation estimates for ecosystem services at a study site, expressed as a value per unit (usually per unit of area or per beneficiary), combined with information on the change in quantity of units at the policy site to estimate policy site values. Value per unit at the study site is multiplied by the relevant number of units at the policy site. The main steps in conducting a unit value transfer are:

Step 1. Conduct a literature search to identify primary valuation studies for study sites that are as similar as possible to the policy site in terms of ecosystem type and condition, level of ecosystem provision, and beneficiary population and characteristics.

Step 2. From the selected study site valuation results, obtain or compute the value per unit (e.g. US$ per household, US$ per visit, US$ per hectare, US$ per cubic meter water). The unit value may be from a single study site valuation or the average unit value from multiple study sites.

Step 2. Where necessary and feasible, adjust the study site unit value to reflect any identified differences between the study site(s) and the policy site. Common adjustments are for differences in incomes or price levels between the study and policy sites.

Step 3. For the policy site, quantify the change in ecosystem service provision in the units in which the transfer is being made (e.g. visits, hectares, cubic meters of water).

Step 4. Multiply the unit value by the change in units at the policy site to estimate the aggregate change in ecosystem service value.

Source: Adapted from Brander (2013) Guidance manual on value transfer methods for ecosystem services (http://wedocs.unep.org/handle/20.500.11822/8434)

Methods Box: Meta-analytic function transfer

Meta-analytic function transfer uses a value function estimated from the results of multiple primary studies representing multiple study sites in conjunction with information on parameter values for the policy site(s) to calculate the value(s) of ecosystem services at the policy site(s). The main steps in conducting a meta-analytic function transfer are:

Step 1. Obtain or estimate a meta-analytic value function for the ecosystem service of interest. There are numerous published meta-analyses in the economic valuation literature for different ecosystems and ecosystem services from which value functions can be obtained. Alternatively, a new meta-analysis for the ecosystem service of interest can be conducted. The main steps in conducting a meta-analysis of primary valuation results in order to estimate a value function are:

a) From the available primary valuation studies, construct a database containing information on the value of the ecosystem service of interest.

b) Value information presented in the primary valuation literature may be reported in different physical and temporal units. Values need to be standardised into the same set of units (e.g. US$ per household per month, US$ per hectare per year) so that they can be directly compared and analysed. Similarly, value estimates are likely to be reported in different currencies and for different years and price levels. Values should therefore be standardised to the same currency, year of value/price level. In addition, value estimates produced using different primary valuation methods may estimate different concepts of value and may therefore not be directly comparable. If there is a sufficiently large number of primary value estimates available, it is preferable to only use estimates produced by the same primary valuation method. If this is not possible, variables should be included in the meta-analysis regression model to control for methodological differences between value estimates.

c) For each primary value estimate included in the database, include information on the valuation method used, type of ecosystem service valued, base level of provision, change in provision, characteristics of the ecosystem (e.g., size, condition), and the characteristics of beneficiaries (e.g., number, household size, income, age).

d) In addition to information obtained directly from each primary study, information on each study site can be added using secondary data sources including spatially defined data using GIS. Examples of such additional data include population density, income, abundance of other ecosystems in the vicinity of the study site, landscape fragmentation, and distance to population centres.

e) Estimate a multiple regression equation with the standardised value as the dependent variable and measures of study, ecosystem and beneficiary characteristics as explanatory variables.

Step 2. Collect information for the policy site(s) on each of the parameters (explanatory variables) in the meta-analytic value function and for the quantity of units in which the dependent variable is defined (e.g. number of households, hectares of ecosystem).

Step 3. Input the policy site parameter values into the meta-analytic value function to estimate the unit value(s) of the ecosystem service at the policy site(s).

Step 4. Multiply the estimated unit value(s) by the number of units at the policy site(s) to compute the value of the ecosystem service at the policy site(s).

Source: Adapted from Brander (2013) Guidance manual on value transfer methods for ecosystem services (http://wedocs.unep.org/handle/20.500.11822/8434)

5.3 Social valuation methods

Social valuation methods attempt to measure the relative importance of ecosystem services to people. As such, they also focus on the right-hand side of the ecosystem services cascade (see Figure 2). Social methods are distinct from economic methods in that they measure value in non-monetary units and enable a multi-dimensional conceptualisation of human well-being.

Social methods necessarily involve people in the valuation process and can be grouped into three broad categories in relation to how they engage stakeholders and elicit their perceptions and values (Santos-Martin et al., 2018): 1. Observation methods; 2. Consultation methods; 3. Engagement methods (see Figure 7)

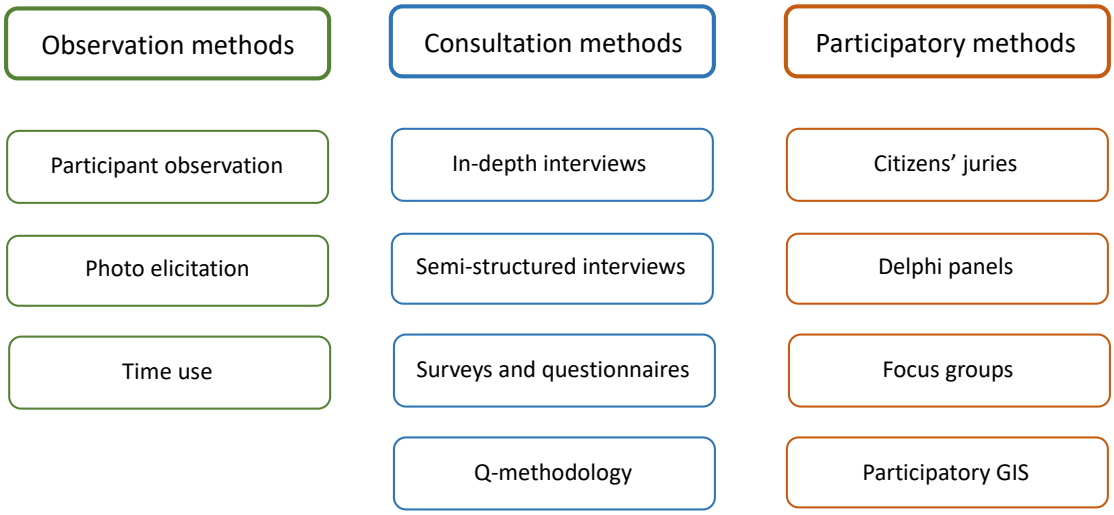


Figure 7: Overview of social valuation methods

Observation methods involve the monitoring of stakeholders' behaviour by researchers and the analysis of social preferences and values. Observation methods generally yield quantitative measurement of values for ecosystem services. Examples of observation methods include the use photographs posted to social media to infer preferences for landscape features; or measurement of time allocated to different activities to measure relative recreational values.

Consultation methods are based on qualitative data that are usually obtained through an interactive process involving stakeholders and researchers. These methods make use of in-depth and semi-structured interviews that allow participants to express their motivations and diverse values for ecosystem services through their own stories and direct actions. These types of methods are usually applied to understand and describe the variety of motivations behind the social value that different stakeholders attribute to nature. Other examples of engagement methods are ranking and rating exercises, in which participants are asked to first rank ecosystem services in order of priority and then rate their relative importance by assigning a fixed number of units (e.g. 20 pebbles or beans) across the services (see Olseson et al., 2015 for an example application in Madagascar); and "photo and speech" approaches, in which stakeholders are given a camera and asked to photograph ecosystems and locations that are of importance to them, which are then shown and discussed with the researcher.

Engagement methods gather both qualitative and quantitative data through interactive processes involving stakeholders and researchers. These methods use participatory and deliberative tools such as focus groups, citizens' juries, Delphi panels, and participatory GIS. Often these methods involve co-learning and knowledge co-production as they foster discussion between different stakeholder groups regarding trade-offs among different ecosystem services.

5.4 Important considerations: distribution of values, discounting, double-counting and uncertainty

Distribution of impacts across stakeholders

The distribution of costs and benefits across different groups in society is usually an important criterion in public decision-making and needs to be addressed as part of the valuation process. The allocation of the benefits and costs among different groups within society may well determine the political acceptability of alternative options.

The uneven distribution of costs and benefits has both practical and ethical consequences. In practical terms, it is important to assess the burden of costs and benefits received by local stakeholders, as they often have a strong influence on how successful project implementation will be. For example, the establishment of protected areas that attempt to exclude local stakeholders from accessing an environmental resource will not be successful without sharing the benefits of conservation with them. Understanding who gains and who loses from environmental change or management can provide important insights into the incentives that different groups have to support or oppose policy options.

In terms of ethical considerations, the analysis of the distribution of costs and benefits is important to ensure that conservation interventions do not harm vulnerable groups within society. Identifying and estimating the distribution of costs and benefits across different groups is the first step in designing measures to avoid disproportionate or undesirable allocation of impacts, compensation mechanisms, or payment schemes between gainers and losers. A general approach to identifying which groups will be affected by alternative options is through stakeholder analysis. One way of displaying the distributional effects is to construct a distributional matrix, which displays the impacts of environmental change, and indicates how they are distributed among different socio-economic groups.

Case Study Box: Using economic valuation of fisheries to assess ecosystem dependence in Madagascar

This study provides an estimate of the socioeconomic contribution of small-scale fisheries within a locally marine managed area in Velondriake, Madagascar. The study site covers twenty-four villages that collaboratively manage a complex array of islands, mangroves and coastal ecosystems spanning more than 1000 km^2.

Small-scale fisheries make key contributions to food security, sustainable livelihoods and poverty reduction. The study uses data from a household survey to measure post-landing trends of small-scale fisheries resources and estimate their total economic value, including both commercial and subsistence values. The

analysis measures gear and habitat use, post- landing trends, fishing revenue, total market value, costs and net income, profitability, employment and dependence on small-scale fisheries.

The results show that the small-scale fisheries sector employs 87% of the adult population, generates an average of 82% of all household income, and provides the sole protein source in 99% of all household meals with protein. The majority (83%) of total catch was sold commercially, generating revenues of nearly $6 million. When accounting for subsistence catch, total annual landings had an estimated value of $6.9 million. The results demonstrate the importance of small-scale fisheries for food security, livelihoods, and wealth generation for coastal communities, and highlight the need for long-term management strategies that aim to enhance their ecological and economic sustainability.

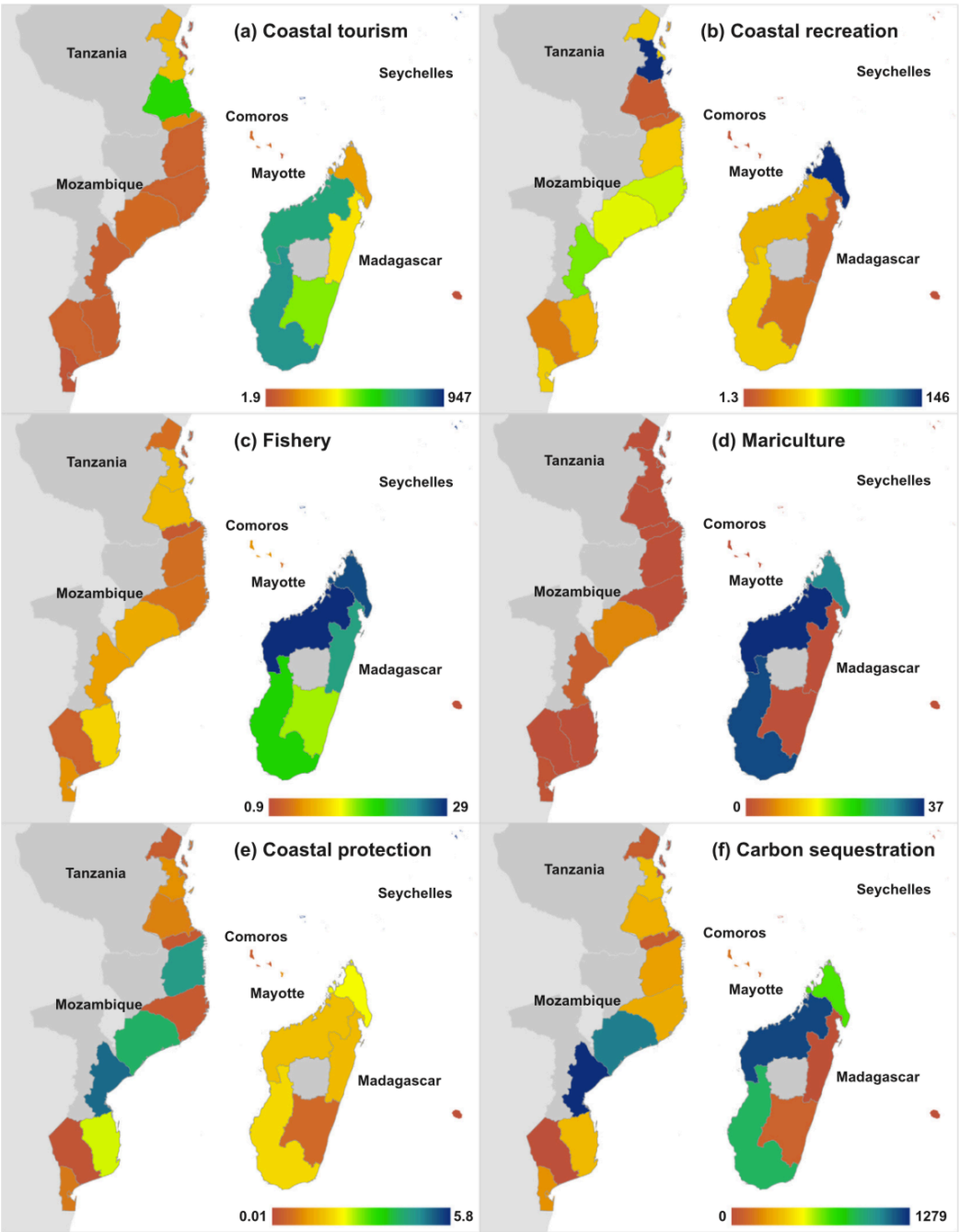
Source: Barnes-Mauthe, M., Oleson, K. L., and Zafindrasilivonona, B. (2013). The total economic value of small-scale fisheries with a characterization of post-landing trends: An application in Madagascar with global relevance. Fisheries Research, 147, 175-185.

Spatially distributed impacts

The management of ecosystem services is often one of spatial targeting. Decisions are being made about where to invest in ecosystem restoration, establish protected areas, or target financial incentives to change the behaviour of land users. In such cases, the spatial distribution of ecosystem services is relevant to the decision and mapping this information is necessary. Alternative policy options will generally result, not only in different aggregate costs and benefits, but also in the spatial distribution of impacts. If these differences in spatial distribution are considered of importance, they also need to be represented to decision makers. The analysis of the spatial distribution of values may be seen as an extension of the distributional analysis described in the previous section and may be a useful approach to identifying different societal groups that are impacted by a project. For example, projects that address water management at a river basin level are likely to affect upstream and downstream stakeholders differently – and this should be identified through spatial analysis.

Case Study Box: Mapping coastal and marine ecosystem service values in the Northern Mozambique Channel

This study estimates and maps the value of six coastal and marine ecosystem services for six countries in the Northern Mozambique Channel. The analysis combines economic valuation, using market prices and value transfer methods, and geospatial analysis to map ecosystem service values at the administrative level (values are in millions of USD/year). This enables an assessment of not just absolute country performance with respect to the provision of ecosystem services but relative performance across the region.



Source: Ghermandi, A., Obura, D., Knudsen, C., & Nunes, P. A. (2019). Marine ecosystem services in the Northern Mozambique Channel: A geospatial and socio-economic analysis for policy support. Ecosystem Services, 35, 1-12.

Temporally distributed impacts

Changes in the coastal and marine ecosystems will result in impacts not only in the year in which the change takes place but also over a number of years into the future. Resulting changes in the flow of ecosystem service values will therefore have a temporal distribution. It is important to account for this distribution of ecosystem services over time because people tend to place higher importance on values received in the present compared to values received in the future. The practice of accounting

for this time preference is called discounting and involves putting a higher weight on current values.

There are two motivations for this higher weighting of current values. The first is that people are impatient and simply prefer to have things now rather than wait to have them in the future. The second reason is that, since capital is productive, a shilling's worth of resources now will generate more than a shilling's worth of goods and services in the future. Therefore, an entrepreneur is willing-to-pay more than one shilling in the future to acquire one shilling's worth of these resources now. In most cases, the discount rate is therefore based on the opportunity cost of capital – the prevailing rate of return on investments elsewhere in the economy, i.e. the interest rate.

The usual way to deal with temporally distributed values is to apply a discount rate to future values so that they can be compared as "present values". Suppose that an annual value X of an ecosystem service will occur over a period of T years, and a discount rate of r per cent is applied, then the present value of the ecosystem service is:

$$\sum_{t=0}^{T} X / (1+r)^t$$

The present value of the value X in any given year with t>0, $X/(1+r)^t$, is smaller than the value X in year t=0. From the equation it can be seen that the higher the discount rate r and the higher the number of years (t), the lower the discounted value of future benefits in any given year.

The choice of the appropriate discount rate remains a contentious issue because it often has a significant impact on the outcome of the analysis.[3] Various respected organisations provide advice on the discount rate to be used. For example, the UK Treasury guidelines recommend a discount rate of 6% for public sector projects while for most environmental and social impact studies 3.5% is recommended.[4]

There is evidence to suggest that people discount the future differently for different goods. If people have lower rates of time preference for environmental goods than for money, a lower discount rate than the interest rate should be used. It is also possible that rates of time preference diminish over time, i.e. that the discount rate declines for impacts in the far future. The choice of discount rate can have a large impact on the findings of an evaluation or valuation study, and should therefore be varied in a sensitivity analysis to check how it influences the results.

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[3] For a comprehensive discussion about the discount rate in environmental assessments, visit the website of the US Environmental Protection Agency (EPA): http://www.epa.gov/ttnecas1/econdata/Rtoolkit2/8.3.html. See also Pearce, D. (2003) Valuing the future: Recent advances in social discounting. World Economic, 4 (2); and Kahn and Greene (2013) Selecting discount rates for natural capital accounting, ONS-DEFRA.

[4] See The Green Book https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/220541/green_book_complete.pdf

Assessing and communicating uncertainty

The magnitude of uncertainty regarding estimated values needs to be quantified and communicated in order to provide an understanding of the robustness of the value information provided. Decision makers can then assess whether the information is sufficiently precise to be considered in making the decision. A balance has to be struck between presenting too little information on the level of uncertainty (e.g. giving the impression of high certainty for an estimate value) and too much information that cannot be taken in (e.g. a table of results for an extensive sensitivity analysis).

Alternative ways to quantify and communicate uncertainties in estimated values include:

1. Ranges of values. In cases where multiple primary value estimates are available for the ecosystem service under consideration, the range of values can be presented to give an impression of the variability of unit value estimates.
2. Distribution of values. In order to give a more complete picture of the distribution of value estimates, information on the average, median and standard error of the average value can be presented (in addition to information on the range of values). Minimum and maximum values may be 'outliers' and not necessarily representative of the likely values of the ecosystem service.
3. Confidence intervals. A confidence interval is an estimated range of values which is likely to include the actual value. The estimated range is calculated from the set of sample data on the ecosystem service value under consideration. Confidence intervals are usually expressed as a range of values within which the actual value lies with a given confidence level or probability.
4. Sensitivity analysis. A sensitivity analysis can be used to show how estimated ecosystem service values change as value function parameters, data inputs and assumptions change. A sensitivity analysis involves systematically varying (within plausible ranges) the uncertain inputs to a model to assess how sensitive the results are to those changes. Joint sensitivity analysis (varying more than one parameter at a time) is sometimes also useful if possible changes in parameters are not independent of each other. In this case, scenarios can be developed that describe how multiple parameters might change in combination.

It is evident that in almost all cases, the value of ecosystem services will not be estimated with complete certainty. The question therefore becomes, how much uncertainty is too much? Assessments of the 'size' of uncertainty are important but require careful interpretation and are not comparable across contexts. Arguably the simplest and most general answer to this question is that the degree of uncertainty becomes unacceptable when a valuation estimate no longer provides information that enables better decisions to be made. For example, if the level of uncertainty is such that the analyst or decision maker can still tell whether, say, benefits (with uncertainty) are still clearly greater or less than costs, then that information helps the decision and the level of uncertainty is acceptable.

Different decision-making contexts may require different levels of certainty regarding the information that they use. For example, the use of value information for raising general awareness of the importance of ecosystem services arguably does not need to be as accurate as valuation information used in litigation for compensation of damages to ecosystems. A general ordering of decision contexts with respect to their required level of accuracy for value information is represented in Figure 8.

The level of uncertainty and the accuracy requirement of each decision-making context should be assessed to determine whether the estimated values can provide sufficiently accurate information. In the case that estimated values are judged to be insufficiently accurate, it is advisable to conduct robust valuations using more reliable methods, if resources (data, time, expertise, knowledge) are available.

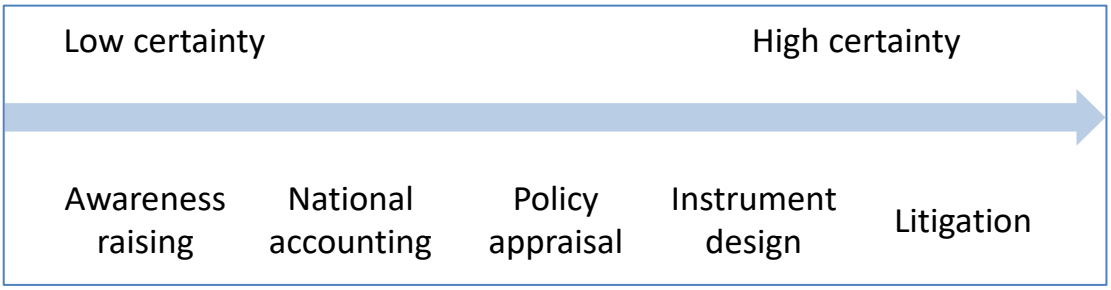


Figure 8. Value certainty requirements for different applications of value information (adapted from Bateman et al., 2009)

6. Undertaking the valuation

Undertaking a valuation study can be viewed as comprising two phases: firstly preparing the valuation study; and secondly conducting the valuation study. Preparing for a valuation of ecosystem services involves the following steps: 1. Understanding what information is needed through stakeholder engagement; 2. Identifying which ecosystem services need to be valued and who the beneficiaries are; 3. Reviewing existing studies and data in order to build on the information that is already available; 4. Defining the objectives of the valuation study and selecting the methods; Building a team that can complete the valuation and disseminate the results. Depending on the selected methods, conducting the valuation involves: 1. Data collection, surveying and sampling; 2. Participatory input and analysis; 3. Developing scenarios for future provision of ecosystem services. These steps are elaborated on in the following sections. Note that the ordering of some of these steps can be flexible. For example, the study preparation might start with a joint exercise of identifying the key ecosystem services and beneficiaries together with the identification of threats and what information is then needed to improve decision making.

6.1 Stakeholder engagement to define mandate and the demand for a valuation

Stakeholder engagement plays an important role throughout the valuation process, including identifying the questions that the study aims to answer, defining the scope, providing input to the valuation itself, and applying the results. For a valuation study to provide useful input to improve the management of coastal and marine ecosystems, it is essential that stakeholders are engaged. The valuation process itself offers the opportunity to raise awareness of environmental challenges, to get diverse stakeholders involved, to understand their concerns and address them, encourage ownership of the results, and ensure that results are subsequently used in decision-making.

A stakeholder is a person, group or organisation with direct or indirect interests in the coastal or marine ecosystem that is the subject of the valuation. Potential stakeholders for a coastal and marine ecosystem may include, but not be limited to, scientists/experts from different disciplines, government departments and agencies (e.g. environment, treasury, water, agriculture, protected areas) and levels (communal, sub-national and national), non-governmental organisations, businesses, local communities, and the media. It is necessary to involve a balanced involvement of stakeholder groups to ensure representativeness and inclusivity.

The first role for stakeholders is to identify the policy, management, or investment question the valuation will help to address is. What is the issue or challenge that needs to be addressed? What is the value information going to be used for? There are multiple possible uses and section 5.3 provides a list of the main potential applications.

The format of stakeholder engagement can take many forms including surveys, interviews, focus groups and workshops through which stakeholders can be asked to identify the key issues or threats facing coastal and marine ecosystems and the

potential management or policy solutions. Participatory formats can be useful to allow discussion and reach consensus but care needs to be taken regarding dominant relationships between stakeholders and restrictions on the expression of different opinions.

6.2 Stakeholder engagement to identify key ecosystem services and beneficiaries

The next step is to identify the key ecosystem services provided by the coastal or marine ecosystem and the relevant beneficiaries. This is also often most effectively undertaken through stakeholder engagement. Again, this can take several forms including surveys, interviews, focus groups and workshops. One of the most effective ways to gather information on ecosystem services is through participatory mapping, in which workshop participants collectively discuss and describe the location of ecosystems and the services they derive from them. Participatory mapping or participatory GIS is a process in which multiple stakeholders (e.g. local community members, fishermen, business owners) jointly create an ecosystem services map, identify 'hotspots' of importance for ecosystem service supply and use. The process helps to integrate stakeholder perceptions and knowledge in maps of ecosystem services.

One potential challenge in obtaining information from local stakeholders on indigenous and traditional knowledge of ecosystem services is an observed reluctance to share such information. Such reluctance might be motivated by various factors including deference to formal scientific knowledge and experts, or unwillingness to speak in unfamiliar public settings. It is necessary to be aware of this challenge and design the engagement process to address it.

Another challenge in obtaining stakeholder input to identify ecosystem services is related to gender roles and participation. In many societies, including in the WIO, men and women have different roles regarding natural resource and to some extent constitute different beneficiary groups. In can also be the case that women are not well represented in stakeholder engagement processes and so their use and dependence on ecosystem services is poorly identified. Again it is necessary to be aware of this challenge and design the engagement process to address it.

6.3 Review of existing studies, information and data

Reviewing relevant valuation studies can provide guidance on applicable methods, identify gaps in existing knowledge, potentially be a useful source of data, and can also provide lessons in terms of challenges to address or avoid. There is now a wealth of existing ecosystem valuation studies on coastal and marine ecosystems and they are an excellent starting point for a new valuation. Annex 4 provides a list of existing ecosystem valuation studies for the WIO region.

6.4 Defining scope, objectives and selecting methods

Based on the preceding three steps, the scope of the valuation study can be defined in terms of geographic area, ecosystems, ecosystem services to be valued and their

beneficiaries. The specific objectives should be defined in terms of type of value information that is required to answer the policy/management question. Ideally the key stakeholders should be consulted again to confirm that the scope and objectives meet their needs.

The selection on relevant valuation methods is largely driven by the ecosystem services that are to be valued and the type of value information that is need. Additional factors are the available resources for conducting the valuation and the availability of data since some methods have greater time and data requirements than others. Section 6 provides details on the applicability, requirements, strengths and weaknesses of available methods.

6.5 Building a team with all required competencies

The mix of expertise within the research team conducting the valuation will depend on the ecosystems, services and methods that have been identified. In general, an ecosystem service valuation will involve an understanding of the bio-physical processes and functions of the ecosystem that underlie the delivery of services, the estimation of preferences and values received by beneficiaries, and knowledge of the management or policy process into which the value information feeds. The research team therefore needs to include expertise from bio-physical and social science disciplines. Often it is also useful to include GIS expertise for the purposes of obtaining and extracting spatial data, modelling ecosystem service flows, and mapping results to highlight their spatial distribution.

6.6 Data collection, survey and sampling methods

Data for valuation studies include multiple types including both secondary (existing data sets, maps and statistics) and primary data (collected first-hand for the purposes of the study).

Primary data for a valuation study can be collected in a number of ways including direct observations and measurements (e.g. fish catch, number of visitors), stakeholder interviews, participatory workshops, and surveys of beneficiaries.

Economic and social valuation methods often use surveys of beneficiaries to collect data on their perception, use and value of ecosystem services. A survey is a process of collecting information from a target population by recording answers to a set of questions (a questionnaire). Basic principles for developing a questionnaire include:

- Use simple language and avoid technical terms and jargon
- Use understandable and locally relevant units of measurement
- Use short questions and limit number of questions to avoid respondents losing attention and not completing the questionnaire
- Ask precise questions that obtain one piece of information at a time so that the interpretation of responses is clear
- Collect socio-economic and demographic information on the respondent in order to analyse the influence of such factors on responses

- Test the questionnaire for clarity of questions and answer options through focus groups or small samples and revise if necessary. The usefulness of testing the questionnaire cannot be over-emphasised.

Methods for conducting a survey include face-to-face interviews, telephone interviews, postal/mail delivery and return, and internet surveys. The costs and effectiveness of each method varies. In general, conducting face-to-face interviews is the most expensive approach but also observed to obtain more reliable responses. Internet surveys are low cost and can obtain large sample sizes but face difficulties in communicating complex information and questions. The combination of face-to-face interviews with web-based tools (online response forms administered using mobile devices) offer a promising approach. Web-based tools, however, may not be feasible in some locations in the WIO region.

Generally, it is not feasible or affordable to interview the entire target population of a survey and so it is necessary to collect information from a subset or sample of that population. Ideally the sample should be representative of the population so that the collected information can be interpreted as reflecting the values of the population. Representativeness of the sample should be monitored by comparing key characteristics (e.g. gender, age, income) of the sample and target population during the survey implementation. At this stage, if specific characteristics are under-represented, they can be targeted in subsequent sampling.

Two general approaches to identifying individuals or households to invite to answer a questionnaire are through random or convenience sampling. Random sampling selects members of the target population randomly but requires data or a list of the target population to select from, which is not always available. Convenience sampling involves selecting respondents that are easily accessible (e.g. on the street) but is likely to be biased towards certain types of people and non-representative of the target population.

6.7 Options for participatory analyses

The potential role for stakeholder participation in ecosystem valuation is extensive. While the use of surveys of ecosystem service beneficiaries described in the preceding section is participatory in the sense that stakeholders provide direct input into the valuation process, there are several other options for participatory valuation that go beyond collection of data and promote knowledge development and sharing.

Deliberative methods is an umbrella term for various tools and techniques engaging and empowering stakeholders in the valuation process. These methods ask stakeholders to share and form their preferences for ecosystem services in a transparent way through an open and structured discourse. It may combine different social techniques, including focus groups and citizens' juries, to flexibly adapt to local contextual factors and stakeholder needs. Deliberative assessment allows consideration of ethical beliefs, moral commitments, social norms and multiple conceptions of value. Furthermore, deliberative assessment enables the inclusion of marginalized stakeholder groups and can shed light on social conflicts associated with

ecosystem service trade-offs. Ideally the deliberative assessment process produces results that are collectively accepted. Open discourse, generated by deliberative techniques, is able to unfold relational values and reflect the social context of valuation. Therefore, deliberative methods are also proposed to account for social equity issues in valuation. Deliberative assessment is particularly suited for understanding the motivations for the values they express.

Deliberative monetary valuation (DMV) is a combination of stated preference valuation with deliberative (group discussion) elements. As opposed to conventional stated preference surveys, deliberative approaches place emphasis on learning, education, knowledge and information exchange. This approach also allows time for reflection to form preferences before stated preference valuation questions are answered. In addition, the process of group discussion can help to build trust, legitimacy, accountability, acceptance or conflict resolution between stakeholders. See Schaafsma et al. (2018) for guidance on the use of deliberative monetary valuation.

6.8 Developing scenarios

Scenarios can be used to explore how ecosystem services might change in the future and how these changes can influence human well-being. Depending on the purpose of the ecosystem valuation, it is often useful to inform decision making by estimating the value of ecosystem services under a set of future scenarios representing alternative development paths or management options.

A scenario is a description of the future that might potentially arise under certain assumptions and conditions. Scenarios can be defined in terms of a set of key variables (e.g. ecosystem extent, condition, provision of services, number of beneficiaries etc.) and the values that these variables take over time. Note that a scenario does not only describe the state of an ecosystem for a single year in the future but for the entire time profile between the start (usually the current year) and end of the period of analysis. Scenarios should be plausible and internally consistent. The development of scenarios can use a number of different approaches:

- Predictive – predict what future ecosystem conditions, service provision and use will be under likely assumptions and driving factors.
- Explorative – describe future conditions etc. under possible assumptions and potential policy directions. Explorative scenarios ask “what if ….”
- Backcasting – identify desired future ecosystem conditions etc. and work backwards to describe courses of action that would achieve that future outcome.

The process of developing scenarios can usefully involve stakeholder consultation to obtain inputs on plausible futures and management options. Participatory scenario planning applies various tools and techniques (e.g. brainstorming or visioning exercises) to develop descriptions of alternative future options. Assumptions about future events or trends are questioned, and uncertainties are made explicit. Participatory scenario planning typically takes place in a workshop setting, where

participants explore current trends, drivers of change and key uncertainties, and how these factors might interact to influence the future

In the context of the WIO, it might be relevant to consider existing scenarios for the region developed under Nairobi Convention, including the WIO SOC report and WIO/NMC scenarios.

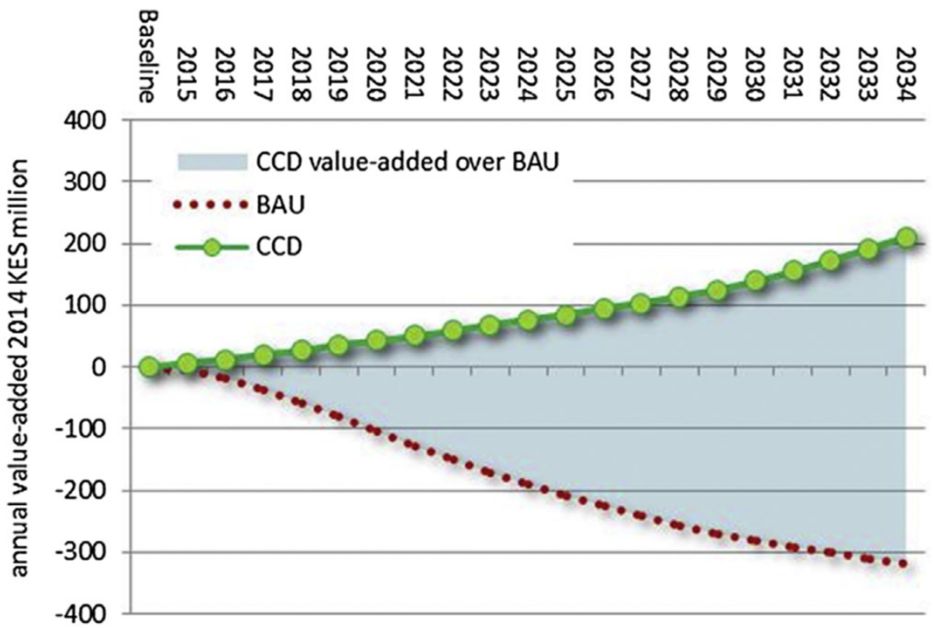
Case Study Box: Using scenarios and valuation to make the case for climate compatible development in a case study of Kenya's mangrove forests.

This study develops alternative development scenarios for a coastal area of Kenya and estimates the implications for the value of ecosystem services. The study concerns the South Coast of Kenya, lying to the south of Mombasa and covering the coastline of Kwale county (comprising the four sites of Funzi, Gazi, Mwache and Vanga).

Local stakeholders were engaged in developing two plausible storyline scenarios for business as usual (BAU) and climate-compatible development (CCD) over the next 20 years. In brief, BAU was assumed to entail gradual loss of mangrove coverage and degradation of remaining forests, loss of fisheries resources, increasing coastal vulnerability and increasing poverty, while CCD emphasised ecosystem conservation and sustainable management, resulting in healthy mangroves supporting improved local livelihoods and enhanced resilience.

Ten ecosystem services were identified as being of key economic importance to the coastal communities, and were selected for valuation: honey, fuelwood, timber, protection against shoreline erosion, protection against extreme weather, mitigation of climate change, breeding and nursery habitat for fisheries, tourism, research, cultural practices and knowledge.

A variety of methods were used to value these ecosystem services, including market prices, net factor income, defensive expenditure and value transfer. The value of ecosystem services under each scenario is represented in the Figure below. The study estimates that the net present value of services under the CCD scenario is more than USD 18 million higher than under the BAU scenario.



Source 1: Emerton, L. (2014): Valuing & investing in ecosystems as development infrastructure: economic analysis of options for climate-compatible development in coastal zones of Kenya and Sri Lanka. LTS International, Penicuik.

Source 2: Huxham, M., Emerton, L., Kairo, J., Munyi, F., Abdirizak, H., Muriuki, T., Nunan, F. and Briers, R. A. (2015). Applying Climate Compatible Development and economic valuation to coastal management: A case study of Kenya's mangrove forests, Journal of Environmental Management 157:168-181.

7. Employing the Outputs of Ecosystem Valuations

7.1 Design of valuation products and communication strategies

Making the results of an ecosystem valuation accessible to the different stakeholders that use the information requires different types of communications strategies, different messages and different ways of presenting information. The main steps that should be part of a communication plan are:

Step 1. Identify the *audience*(s) for the information produced and determine what is of interest to them. What is the "hook" for the audience? The stakeholder engagement process is useful for this. Different audiences may have different interests and be receptive to different messages and information formats/channels.

Step 2. Identify and formulate the *main message* that you want to convey. Keep it simple and do not try to be too comprehensive in reporting all results from an analysis. It is better to get across one message that sticks than five messages that are lost. It is important to recognise that the main message is generally in the interpretation or implication of the valuation, not the value estimate itself. Value estimates are the evidence that help support a particular case or argument.

Step 3. Select the *tools* to communicate the message to the audience. This includes the choice of statistics, indicators and visual representation (e.g. maps, illustrations, diagrams, pictures, charts, graphs, tables) and the materials for communication (e.g., reports, policy briefs, films, media coverage). It is generally useful to communicate results through multiple avenues. Technical reports are necessary to ensure that all sources and analyses are well documented but are generally not widely read. Executive summaries, synthesis reports and policy briefs that distil the main results and message into one or two pages are more accessible and effective in disseminating information. Other media, such as short videos or animations can also be more engaging for a wider audience. For such communication materials it is important to use suitable language and visual information, and to avoid using jargon of technical terminology.

7.2 Influencing policies and strategies

An ecosystem valuation is intended to be policy relevant, not policy prescriptive. It will provide evidence on the impact of changes in coastal and marine ecosystems and the management responses that were identified in the initial stakeholder engagement. This, however, is just part of the policy development process, which typically requires time and involves many other considerations. The results of a valuation study should therefore not be expected to lead to major policy or strategy changes overnight. In recognition of this, it is advisable for valuation studies to be set up to include time and budget for monitoring and engaging in and the policy assessment process that follows it. This is crucial to ensure that the study and momentum survives beyond the production of results and remains relevant and useful. This can include several elements:

- Recommendations on policy responses including subsequent appraisal of options (see section 9.4 on evaluation methods)

- Organize events open to external audiences to present the results or present at events organised by others (locally, nationally and internationally)
- Organize meetings at which stakeholders can report on progress towards improved coastal and marine management
- Publicly report the progress of any further work on ecosystem valuation and, if relevant, keep the study website up to date
- Provide training for stakeholders that are likely to take the results forward
- Geographical expansion of the valuation study. If, for example, the original study was for a specific ecosystem, there might be interest and funding for scaling up the analysis to the regional or national level.
- Content expansion of the valuation study. If, for example, the original study was for a limited set of ecosystem services, there might be interest and funding for extending the analysis to other relevant services.

7.3 Evaluation of policies and investments

Ecosystem service valuations can be used in *ex-ante* or *ex-post* appraisals of a given set of policy options by weighing the ecological and economic consequences of those options. It is a way to incorporate the present and future values of negative and positive policy impacts with a common metric and provide decision rules regarding the social desirability of each option

Making decisions between alternative investments, projects or policies that affect the provision of ecosystem services often involves weighing up and comparing multiple costs and benefits that are measured in different metrics and are incurred at different locations and points in time. For example, the establishment of a new marine protected area might involve costs in terms of the purchase of coastal land, compensation of local communities, and on-going maintenance and enforcement costs; and benefits in terms of biodiversity conservation, recreational use and enhanced fish stocks. These costs and benefits are likely to be measured in different units, be incurred at different locations by different groups of stakeholders, and have different time profiles. Organising, comparing and aggregating information on such a complexity of impacts; and subsequently choosing between alternative options with different impact profiles requires a structured approach. Decision support tools provide systems for structuring the information and factors that are relevant to a decision.

There are a number of evaluation methods available to help decision makers to structure the information and factors that are relevant to a decision and to select between alternative investments, projects or policies. The choice of which assessment method to use will largely be determined by the type of decision problem and the availability and nature of information related to each potential option.

Here the three main economic evaluation methods are introduced: Cost-Effectiveness Analysis (CEA), Cost-Benefit Analysis (CBA) and Multi-Criteria Analysis (MCA). These are general decision support tools that can be applied to help select between

alternative investments, projects and policies. The nature of ecosystem-related decisions may require emphasis on specific types of input, particularly spatial analysis. The decision-making context regarding the management of ecosystem services is often one of spatial targeting or optimisation. Decisions are being made about where to invest in ecosystem restoration (e.g. CBD Aichi Target 11 that 10 per cent of coastal and marine areas are conserved through protected areas and other effective area-based conservation measures), or target financial incentives to change the behaviour of resource users. In such cases, the spatial correspondence of costs and benefits relevant to the decision is of crucial importance and mapping these inputs is a necessary step in the assessment process.

Table 6. Economic evaluation methods, strengths and weaknesses

Evaluation method	Application	Strengths	Weaknesses
Cost-Effectiveness Analysis (CEA)	Used for identifying lowest cost policy options to achieve a given objective	Does not require assessment of benefits and is analytically relatively straightforward	Limited applicability to ecosystem services given complex and multi-functional nature of ES provision; and the absence of single quantified policy targets
Cost-Benefit Analysis (CBA)	Used to estimate the economic performance of investments and policies	Provides a measure of how much an investment or policy contributes to societal wellbeing	Requires that all costs and benefits are quantified in monetary terms; can result in omission of important effects
Multi-Criteria Analysis (MCA)	Used to rank alternative investments and policies	Allows the inclusion of effects that cannot be expressed in monetary terms	Heavily reliant on the subjective judgment of the analytical team

Cost-Effectiveness Analysis

Cost-effectiveness analysis (CEA) involves identifying the lowest cost option to achieve a given objective.[5] CEA is an applicable assessment method for decisions that involve selecting between alternative measures or technologies to achieve a single specific goal (e.g. meeting a specified ecological standard, supplying a specified quantity of

[5] Note that the term "cost-effective" is often used to describe investment or policy options that result in a gain in efficiency or, equivalently, for which benefits exceed costs. A "cost-effectiveness analysis", however, only involves ranking options that achieve a given target in order of their cost.

clean water, or sequestering a targeted quantity of carbon) and for which all costs can be measured in monetary terms.

The steps in conducting a CEA are take the following sequence, but there may be feedback loops between steps during the process:

Step 1: Identify the environmental objective(s) involved (target situation).

Step 2: Determine the extent to which the environmental objective(s) is (are) met.

Step 3: Identify sources of pollution, pressures and impacts now and in the future over the appropriate time horizon and geographical scale (baseline situation).

Step 4: Identify measures to bridge the gap between the reference (baseline) and target situation (environmental objective(s)).

Step 5: Assess the effectiveness of these measures in reaching the environmental objective(s).

Step 6: Assess the direct (and if relevant indirect) costs of these measures.

Step 7: Rank measures in terms of increasing unit costs.

Step 8: Determine the least cost way to reach the environmental objective(s) based on the ranking of measures.

Note that CEA does not involve any assessment of the benefits of meeting the policy target but only compares alternative options in terms of their costs. As such, CEA is a relatively straightforward assessment method to apply and is relevant to decision contexts in which a specific policy target has been set. It does not, however, provide an indication of the magnitude of changes in societal welfare resulting from implementing policy options (i.e. whether society is better or worse off as a result of the decision).

In practice, this economic assessment method is not frequently used in the context of managing ecosystems due to the complex and multifunctional nature of ecosystem service provision. It is generally not the case that a single specific goal for ecosystem service provision can be set and it becomes necessary to consider the multiplicity and variability of benefits derived from alternative options.

Cost-Benefit Analysis

Cost-benefit analysis (CBA) is the most commonly used economic assessment method for evaluating and comparing investments, projects and policies.

It is important to recognise the difference between a CBA that is carried out from the perspective of society as a whole and CBA that is conducted from the perspective of an individual, group, or firm. If applied from this latter perspective, CBA is generally used to determine the financial return of private investments. This private application is commonly known as a 'financial CBA'. Alternatively, government departments apply CBA as the standard tool for evaluating investments, projects and policies from the perspective of society as a whole. This so-called 'extended CBA' is used as a method in which the societal costs and benefits of alternative options are expressed and compared in monetary terms. The extended CBA provides an indication of how much a prospective project or investment contributes to social welfare by calculating the

extent to which the benefits of the project exceed the costs – essentially society's 'profit' from a project. In this application, the CBA provides a framework into which monetised ecosystem service values can be integrated.

The main steps in performing a CBA are presented in Figure 9.

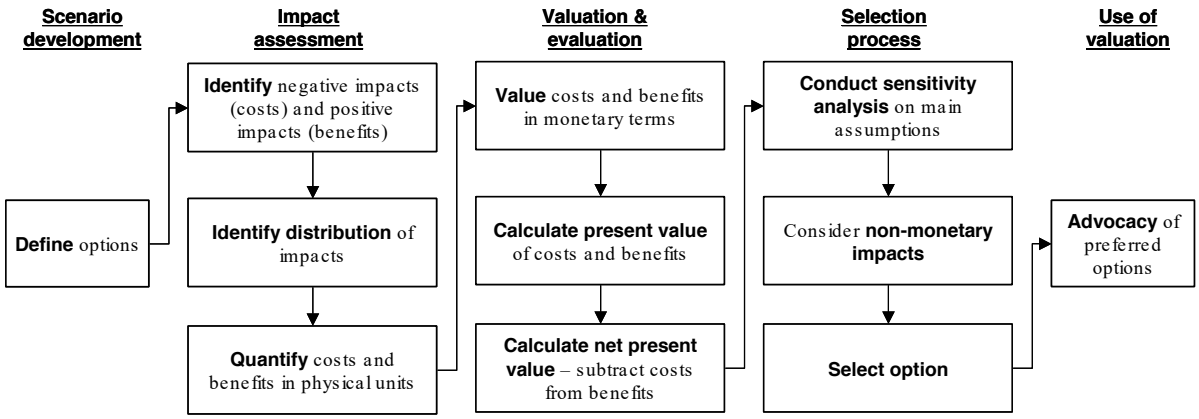


Figure 9: Methodological steps in cost-benefit analysis (source: Brander and van Beukering, 2015)

The first step in a CBA is to identify the alternative options or alternatives to be considered. The options under consideration will generally be specific to the particular problem and context, but may include investments, projects, policies, development plans etc.

The impact assessment in a CBA starts with the identification of the complete set of negative impacts (costs) and positive impacts (benefits) related to the policy or intervention options under consideration. This includes costs and benefits accruing to all affected groups and individuals (not just those involved in the project development) and costs and benefits that are incurred in the future. It is important to describe the geographical and temporal boundaries of the analysis. This is especially crucial for ecosystem services impacts since effects emerging from ecosystem change often show major variations in time and space. The final step in the impact assessment phase is to quantify each cost and benefit in relevant physical units for each year in which it occurs. Estimating changes in ecosystem services requires specific expertise and models on ecological, hydrological and climatic processes.

To conduct a CBA, all of the quantified positive and negative effects need to be expressed in monetary units. In cases where costs and benefits are not directly observable in monetary terms in well-functioning markets (as is the case for many ecosystem services), estimates need to be generated using non-market valuation methods or value transfer (see Chapter 4).

The economic performance of each alternative option can be calculated in three different ways:

1. The net present value (NPV) of each option is calculated by subtracting the present value costs from present value benefits. A positive NPV indicates that

implementing a project will improve social welfare. The NPVs of alternative investments can be compared in order to identify the most beneficial project;

2. The benefit cost ratio (BCR) is the ratio of discounted total benefits and costs, and shows the extent to which project benefits exceed costs. A BCR greater than 1 indicates that the benefits of a project exceed the costs;
3. The internal rate of return (IRR) is the discount rate at which a project's NPV becomes zero. If the IRR exceeds the discount rate used in the analysis, the project generates returns in excess of other investments in the economy, and can be considered worthwhile.

A final step in a CBA is to conduct sensitivity analysis to check the robustness of the conclusions to the assumptions made. Another element is to estimate whether or not the omission of certain costs and benefits that cannot be monetised affects the decision result.

An important drawback of CBA is the requirement that all costs and benefits need to be expressed in monetary terms. Although a range of economic valuation methods are available to estimate values for marketed and non-marketed ecosystem services, there are still considerable limitations to the accuracy of estimated value in some cases. Furthermore, the application of non-market valuation techniques can be expensive and time consuming. For these reasons it may not be possible to estimate monetary values for some costs and benefits and they cannot be entered into a CBA. In some cases the omitted impacts can be significant and therefore alternative evaluation methods are needed.

Case Study Box: Cost-Benefit Analysis of management options for coastal ecosystems in the Seychelles

This study evaluates the socio-economic impacts of coastal degradation in the Seychelles with a focus on the following two issues: (i) the value of ecosystem resources and services, (ii) the costs and benefits of an adaptation strategy regarding coral bleaching.

In the Seychelles there is general concern that the coastal and marine resources are being overexploited and that their value is declining as a result. In order to investigate how these trends can be reversed, management scenarios are developed for six study sites to assess whether management efforts would be justified in economic terms, i.e. the costs of management is outweighed by the benefits of such interventions. Interventions include expenditures on services, enforcement/compliance, education/awareness, monitoring and infrastructure. For each of the sites, assumptions are made as to the most appropriate investments and their associated costs. The benefits of management are estimated as the difference in the total value of a range of ecosystem services under a "with active management" scenario versus a "without active management" scenario – see Figure 1.

Table 1 summarizes the costs and benefits of the management measures proposed for the six sites. From a societal perspective, the measures are desirable if the ratio of benefits to costs is greater than 1. As can be seen in the bottom row of the table, all the proposed investments are economically feasible. The net benefits widely exceed the net-costs to

society.

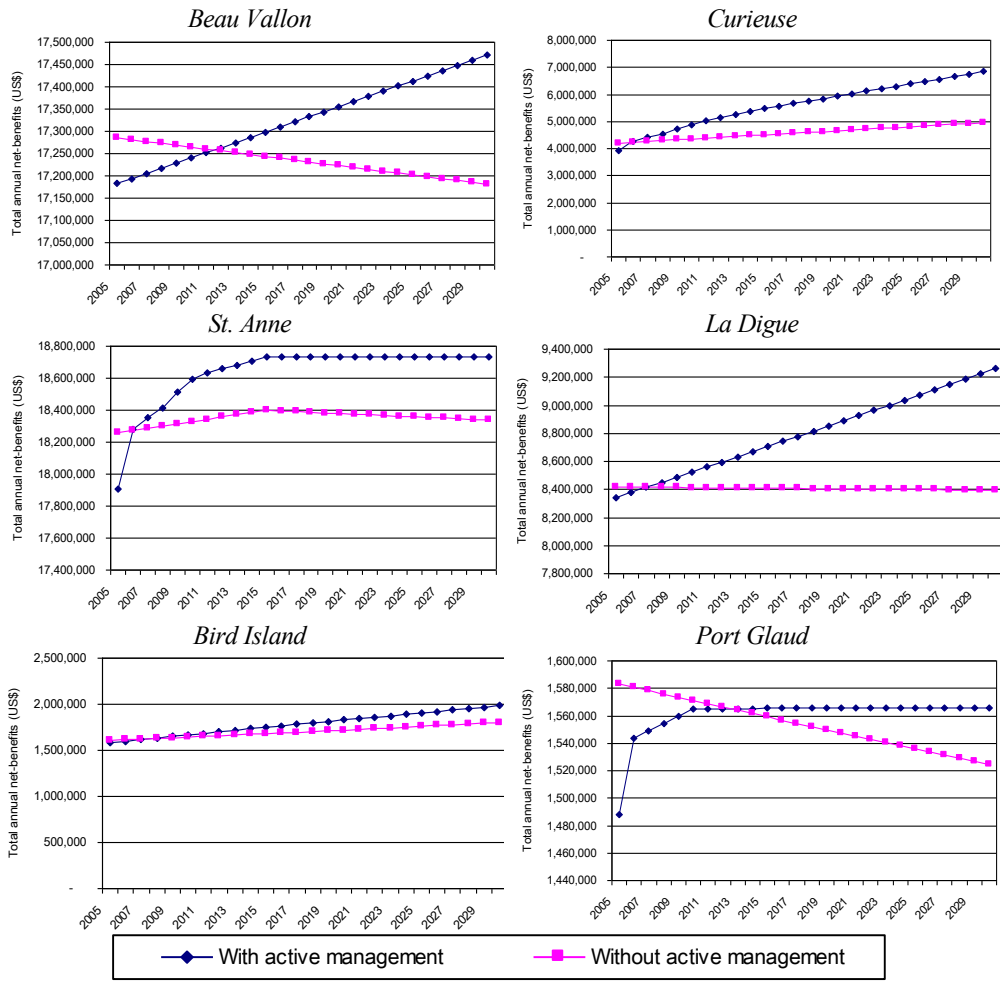


Figure 1. The benefits of marine management (2005-2030)

Table 1. Present value benefits and costs (million US$) and the benefit-cost ratio (BCR); * discount rate 3%; time horizon 25 years

	Beau Vallon	La Digue	St Anne	Curieuse	Bird Island	Port Glaud
Benefits	3.77	8.18	10.15	20.8	1.82	1.06
Costs	1.38	1.38	2.18	2.7	0.46	0.88
BCR	2.73	5.93	4.65	7.7	3.95	1.21

Source: Cesar et al. (2004). Economic analysis of threats to coastal ecosystems in the Seychelles: Costs and benefits of management options. Ministry of Environment, Victoria Seychelles.

Multi-Criteria Analysis

Multi-criteria analysis (MCA) has become a well-established tool for decision-making that involves conflicting or multiple objectives. MCA can be used to establish preferences between alternative options by reference to a set of measurable criteria that the decision-making body has defined. Unlike in a CBA, criteria do not need to be quantified in a common metric (i.e. money). Instead MCA provides a number of alternative ways of aggregating the data on individual criteria to provide indicators of the overall performance of options. This allows the inclusion in the analysis of effects that cannot be expressed in monetary terms. The basic idea behind MCA is to allow the integration of different objectives (or criteria) without assigning monetary values to all of them. In short, MCA provides a systematic method for comparing these criteria, some of which may be expressed in monetary terms and some of which are expressed in other units. The main steps in performing a MCA are presented in Figure 10.

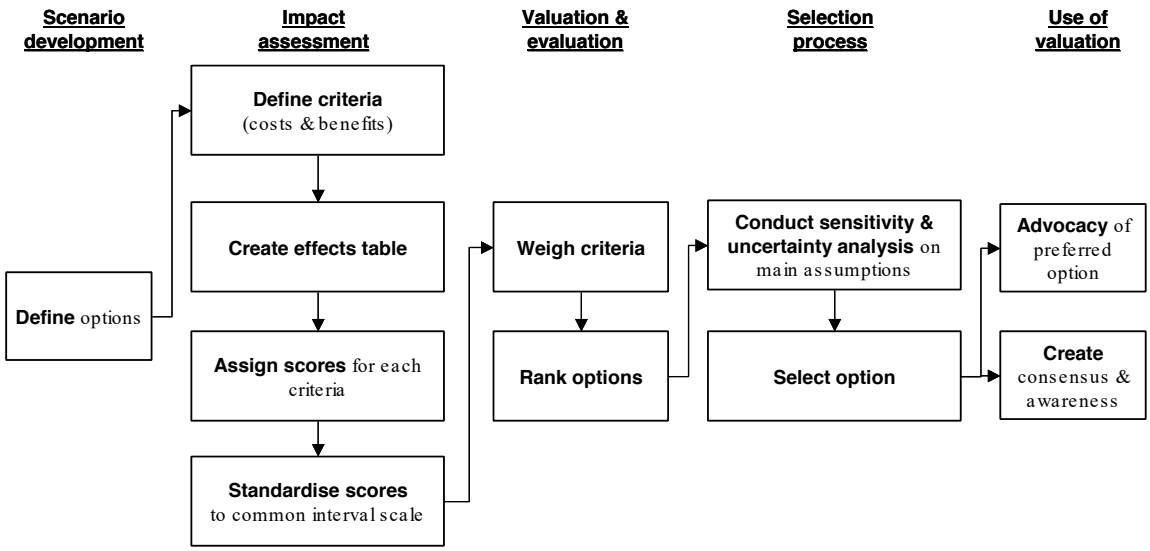


Figure 10: Methodological steps in multi-criteria analysis (source: Brander and van Beukering, 2015)

Impact assessment in a MCA involves identifying and defining all criteria that are relevant to the decision problem. These include all important categories of negative and positive effects resulting from the options under consideration. In a MCA it is possible to include criteria that are difficult to quantify and can perhaps only be assessed in qualitative terms such as political sensitivity, equity and irreversibility. The quantification of the different effects is summarised in an “effects table”, which is a matrix with the alternative options listed in the columns and the criteria listed in the rows. The effects table is completed by assigning scores to each criterion for each alternative. Information on the magnitude of each criterion can be expressed in monetary units, physical units, or simply on a qualitative scale. Data on impacts can be collected from surveys, existing data, experts, or stakeholders. In cases in which the spatial distribution of impacts is important to the decision, the data on impacts

can be represented on maps. To enable the direct comparison of different criteria, standardisation of scores for each criterion to a common interval scale is conducted (usually to values between 0-100 or 0-1). There are several software packages available that can be used to help with the computations in MCA.[6]

MCA does not explicitly value the criteria in monetary terms but instead applies weighting of criteria to quantify the relative importance of each criterion in the decision process. Weights can be derived from existing information or from stakeholders by asking them to state their preferences for the various criteria. By combining the standardised scores and weights of the criteria, the alternative options can be ranked, usually through a weighted summation of criteria scores for each alternative. Similar to CBA, MCA applies sensitivity and uncertainty analysis to assess the robustness of the ranking result to changes in weights and scores. Finally, based on the ranking of options and the sensitivity of the results, a decision maker can select the most preferred option.

A key strength of MCA is that it is not necessary to quantify all impacts in monetary terms. This means that complex and time-consuming valuation studies of all environmental impacts can be avoided, and that qualitative criteria such as political sensitivity can be included in the decision framework. MCA can therefore provide a degree of structure, analysis, and openness to decision problems that lie beyond the practical reach of CBA.

MCA is, however, heavily reliant on the judgement of the analytical team for defining alternatives and criteria, estimating the relative importance of criteria and, to some extent, in calculating and inputting data into the effects table. The subjectivity that pervades these processes can be a matter of concern. The involvement of stakeholders in defining criteria and setting weights can also be time consuming process if conducted using surveys, interviews or deliberative methods. Another important limitation of MCA is that the results do not necessarily show whether alternative options produce welfare gains or losses. Unlike CBA, there is no decision rule (such as a positive NPV, a BCR greater than 1, or an IRR greater than the market interest rate) that indicates that benefits exceed costs. In MCA, as is also the case with CEA, the analysis can only produce a ranking of alternative options and does not indicate whether the options result in a welfare improvement. It is, however, often possible to include a business-as-usual alternative in the set of options, and this can be used as a reference point to indicate whether the other options are better or worse than undertaking no action.

7.4 Policy instruments

Information on the value of ecosystem services can be used in the design a range of different policy instruments have been developed and employed to manage the sustainable use of natural resources. Policy instruments are the mechanisms through which policy objectives are pursued. They include direct intervention and regulation by public bodies as well as the promotion of changes in the activities and behaviour

[6] A number of software packages are available to structure and process information in an MCA, including: DEFINITE, HIVIEW, MACBETH, VISA and ILWIS.

of other relevant actors. Annex 5 provides an overview of incentive and market-based instruments relevant to coastal and marine management including:

- Taxes and subsidies
- Tradeable permits and quotas
- Area based user rights
- Certification and labelling
- Sustainable financing
- Conservation trust funds
- Debt for nature swaps
- Payments for ecosystem services
- Biodiversity offsets, banking and trading

8. Conclusion

The valuation methods introduced in these guidelines can be applied to inform the use and management of marine and coastal ecosystems in a wide array of policy contexts, including: advocacy and raising public awareness; appraisal of projects, investments and policies; impact assessment; sustainable financing; and setting compensation for environmental damage.

The purpose of applying valuation methods is ultimately to provide relevant, credible and actionable information to support better use and management of coastal and marine resources. This primary aim should be kept firmly in mind when applying methods and presenting results; and any application should be designed to provide information that is directly useful and understandable to the decision makers involved. Adhering to the following conditions/principles can help ensure that the information produced by a valuation study achieves this aim: access to and partnership with the decision-makers using the information; identify clear policy questions or information demands to be addressed; high transparency regarding the methods, data and analysis to ensure trust and credibility. The valuation of ecosystems is not an end in itself, but a means to better informed decision-making that results in sustainable use of coastal and marine ecosystems.

9. Glossary of terms

Avoided (damage) cost valuation method: A cost-based valuation technique that estimates the value of the role an ecosystem plays in regulating natural hazards (e.g. floods and landslides) by calculating the damage that is avoided due to the ecosystem service.

Choice modelling: Choice modelling attempts to model the decision process of an individual in a particular context. Choice modelling may be used to estimate non-market environmental benefits and costs. It involves asking individuals to make hypothetical trade-offs between different ecosystem services.

Consumer surplus: The difference between what consumers are willing to pay for a good and its price. Consumer surplus is a measure of the bene t that consumers derive from the consumption of a good or service over and above the price they have paid for it.

Contingent valuation: Contingent valuation is a survey-based economic technique for the valuation of non-market resources, such as environmental preservation or the impact of contamination. It involves determining the value of an ecosystem service by asking what individuals would be willing to pay for its presence or maintenance.

Cost-Benefit Analysis: An evaluation method that assesses the economic efficiency of policies, projects or investments by comparing their costs and benefits in present value terms. This type of analysis may include both market and non- market values and accounts for opportunity costs.

Demand: The amount of a good or service consumed or used at a given price; consumers will demand a good or service if the bene t is at least as high as the price they pay.

Direct use value: The value derived from direct use of an ecosystem, including provisioning and recreational ecosystem services. Use can be consumptive (e.g. fish for food) or non-consumptive (e.g. viewing reef fish).

Discount rate: The rate used to determine the present value of a stream of future costs and benefits. The discount rate reflects individuals' or society's time preference and/or the productive use of capital.

Discounting: The process of calculating the present value of a stream of future values (benefits or costs). Discounting reflects individuals' or society's time preference and/or the productive use of capital. The formula for discounting or calculating present value is: present value = future value/(1+r)n, where r is the discount rate and n is the number of years in the future in which the cost or benefit occurs.

Economic activity: The production and consumption of goods and services. Economic activity is conventionally measured in monetary terms as the amount of money spent or earned and may include 'multiplier effects' of input costs and wages

Economic benefit: the net increase in social welfare. Economic benefits include both market and non-market values, producer and consumer benefits. Economic bene t refers to a positive change in human wellbeing.

Economic contribution: The gross change in economic activity associated with an industry, event, or policy in an existing regional economy.

Economic cost: A negative change in human wellbeing.

Economic impact: The net changes in new economic activity associated with an industry, event, or policy in an existing regional economy. It may be positive or negative.

Economic value: i) The wellbeing or utility associated with the production and consumption of goods and services, including ecosystem services. Economic value is comprised of producer and consumer surplus and is usually described in monetary terms; or ii) The contribution of an action or object to human wellbeing (social welfare).

Ecosystem functions: The biological, geochemical and physical processes and components that take place or occur within an ecosystem.

Ecosystem service approach: A framework for analysing how human welfare is affected by the condition of the natural environment.

Ecosystem service valuation: Calculation, scientific and mathematic, of the net human benefits of an ecosystem service, usually in monetary units.

Ecosystem services: The benefits that ecosystems provide to people. This includes goods (e.g. fish, timber, water) and services (e.g. water filtration, coastal protection, recreational opportunities).

Ecosystem: A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.

Evaluate: To assess the overall effect of a policy or investment.

Evaluation: The assessment of the overall impact of a policy or investment. Evaluations can be conducted before (*ex ante*) or after (*ex post*) implementation of a policy or investment.

Existence value: The value that people attach to the continued existence of an ecosystem good or service, unrelated to any current or potential future use.

Factor cost: Total cost of all factors of production consumed or used in producing a good or service.

Financial benefit: A receipt of money to a government, firm, household or individual.

Financial cost: A debit of money from a government firm, household or individual.

Future value: A value that occurs in future time periods. See also present value.

Green accounting: The inclusion of information on environmental goods and services and/or natural capital in national, sectoral or business accounts.

Gross revenue: Money income that a firm receives from the sale of goods or services without deduction of the costs of producing those goods or services. Gross revenue from the sale of a good or service is computed as the price of the good (or service) multiplied by the quantity sold.

Gross value: The total amount made as a result of an activity.

Hedonic pricing method: A revealed preference method for valuing environmental quality or resources that are attributes of a marketed good or service.

Indirect use value: The contribution of a resource to human welfare without direct contact between the beneficiaries and the resource. In general, indirect use values are obtained from regulating services such as carbon storage, coastal protection and flood regulation.

Instrumental value: The importance of something as a means to providing something else that is of value. For example, a coral reef may have instrumental value in reducing risk to human life from extreme storm events.

Intermediate costs: The costs of inputs or intermediate goods that are used in the production of final consumption goods. For example, the cost of fishing gear used to catch fish is an intermediate cost to the harvest and sale of fish.

Intrinsic value: The value of something in and for itself, irrespective of its utility to something or someone else. Not related to human interests and therefore cannot be measured with economic methods.

Marginal value: The incremental change in value of an ecosystem service resulting from an incremental change (one additional unit) in the quantity produced or consumed.

Market value: The amount for which a good or service can be sold in a given market.

Negative externality: A lost in welfare of one economic agent caused (unintentionally) by the consumption or production behaviour of another economic agent. An example of a negative externality is the health impacts from air pollution caused by the use of petrol vehicles.

Net revenue: Monetary income (revenue) that a firm receives from the sale of goods and services with deduction of the costs of producing those goods and services. Net revenue from the sale of a good is computed as the price of the good multiplied by the quantity sold, minus the cost of production.

Net value: The value remaining after all deductions have been made.

Nominal: The term 'nominal' indicates that a reported value includes the effect of inflation. Prices, values, revenues etc. reported in 'nominal' terms cannot be compared directly across different time periods. See also real and constant prices.

Non-use value: The welfare that people gain from an ecosystem that is not based on the direct or indirect use of the resource. Non-use values may include existence values, bequest values and altruistic values.

Opportunity cost: The value to the economy of a good, service or resource in its next best alternative use.

Option value: The premium placed on maintaining environmental or natural resources for possible future uses, over and above the direct or indirect value of these uses.

Present value: A value that occurs in the present time period. Present values for costs and benefits that occur in the future can be computed through the process of discounting (see discount rate). Expressing all values (present and future) in present value terms allows them to be directly compared by accounting for society's time preferences.

Producer surplus: The amount that producers bene t by selling at a market price that is higher than the minimum price that they would be willing to sell for. Producer surplus is computed as the difference between the cost of production and the market price. Value-added, pro t, and producer surplus are similar measures of the net bene t to producers. Although they differ slightly, the terms are used synonymously for this report to represent economic value.

Profit: The difference between the revenue received by a firm and the costs incurred in the production of goods and services. Value-added, profit and producer surplus are similar measures of the net bene t to producers. Although they differ slightly, the terms are used synonymously for this report to represent economic value.

Purchasing power parity adjusted exchange rate: An exchange rate that equalises the purchasing power of two currencies in their home countries for a given basket of goods.

Purchasing power parity: An indicator of price level differences across countries. Figures represented in purchasing power parity represent the relative purchasing power of money in the given country, accounting for variance in the price of goods. Typically presented relative to the purchasing power of US dollars in the United States.

Real: The term 'real' indicates that a reported value excludes or controls for the effect of inflation (synonymous with constant prices). Reporting prices, values, revenues etc. in 'real' terms allows them to be compared directly across different time periods. See also nominal and constant prices.

Regulating services: A category of ecosystem services that refers to the benefits obtained from the regulation of ecosystem processes. Examples include water ow regulation, carbon sequestration and nutrient cycling.

Rent: Any payment for a factor of production in excess of the amount needed to bring that factor into production (see also producer surplus and resource rent).

Replacement cost method: A valuation technique that estimates the value of an ecosystem service by calculating the cost of human-constructed infrastructure that would provide same or similar service to the natural ecosystem. Common examples are sea walls and wastewater treatment plants that provide similar services to reefs, mangroves, and wetland ecosystems.

Resource rent: The difference between the total revenue generated from the extraction of a natural resource and all costs incurred during the extraction process (see also producer surplus). Refers to pro t obtained by individuals or firms because they have unique access to a natural resource.

Revenue: Money income that a firm receives from the sale of goods and services (often used synonymously with gross revenue).

Social cost of carbon (SCC): The social cost of carbon is an estimate of the economic damages associated with a small increase in carbon dioxide (CO_2) emissions, conventionally one tonne, in a given year. This dollar figure also represents the value of damages avoided for a small emission reduction (i.e. the benefit of a CO_2 reduction).

Stated preference method: A survey method for valuation of non-market resources in which respondents are asked how much they would be willing to pay (or willing to accept) to maintain the existence of (or be compensated for the loss of) an environmental feature such as biodiversity.

Supply: The quantity of a good or service that producers will supply at a given price; producers will supply goods and services if they at least cover their costs.

Supporting services: A category of ecosystem services that are necessary for the production of all other ecosystem services. Examples include nutrient cycling, soil formation and primary production (photosynthesis).

Total economic value: All marketed and non-marketed benefits derived from a resource, including direct, indirect, option and non-use values.

Use value: Economic value derived from the human use of an ecosystem. It is the sum of direct use, indirect use and option values.

User cost: The cost incurred over a period of time by the owner of a xed asset as a consequence of using it to provide a ow of capital or consumption services; the implications of current consumption decisions on future opportunity. User cost is the depreciation on the asset resulting from its use.

Utilitarian value/Utility: A measure of human welfare or satisfaction. Synonymous with economic value.

Valuation: The process or practice of estimating human benefits of ecosystem services or costs of damages to ecosystem services, represented in monetary units.

Value: The contribution of an action or object to human wellbeing (social welfare).

Value-added: The difference between cost of inputs and the price of the produced good or service. Value-added can be computed for intermediate and final goods and services. Value-added, profit, and producer surplus are similar measures of the net benefit to producers. Although they differ slightly, the terms are used synonymously for this report to represent economic value.

Welfare: An individual's satisfaction of their wants and needs. The human satisfaction or utility generated from a good or service.

Willingness-to-accept: The minimum amount of money an individual requires as compensation in order to forego a good or service.

Willingness-to-pay: The maximum amount of money an individual would pay in order to obtain a good, service, or avoid a change in condition.

10. Acronyms

ABMJ	Areas Beyond National Jurisdiction
BCR	Benefit Cost Ratio
CBA	Cost Benefit Analysis
CBD	Convention on Biological Diversity
CEA	Cost Effectiveness Analysis
CICES	Common International Classification for Ecosystem Services
CITES	Convention on International Trade in Endangered Species
CTF	Conservation Trust Fund
DNS	Debt for Nature Swaps
EbA	Ecosystem based Adaptation
EBA	Ecosystem Based Approach
EBM	Ecosystem Based Management
EEA	Experimental Ecosystem Accounting
EIA	Environmental Impact Assessment
ERA	Ecosystem Risk Assessment
ES	Ecosystem Services
FAO	Food and Agriculture Organisation
GDP	Gross Domestic Product
GEF	Global Environmental Facility
IAM	Integrated Assessment Model
ICM	Integrated Coastal Management
IEEFM	Integrated Ecological-Economic Fisheries Models
IMF	International Monetary Fund
INCA	Integrated system for Natural Capital and ecosystem services Accounting
IOC	International Oceanographic Commission
IRR	Internal Rate of Return
IUU	Illegal, Unreported, and Unregulated Fishing
LME	Large Marine Ecosystem
MCA	Multi Criteria Analysis
MPA	Marine Protected Area
MSP	Marine Spatial Planning
NGO	Non-Governmental Organisation
NPV	Net Present Value
PES	Payments for Ecosystem Services

SCC	Social Cost of Carbon
SEEA	System of Environmental-Economic Accounting
SIA	Sustainability Impact Assessment
SNA	System of National Accounts
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TURF	Territorial Use Rights for Fishing
UNDP	United Nations Development Programme
UNEP	United Nations Environmental Programme
VCME	Valuation of Coastal & Marine Ecosystems
WAVES	Wealth Accounting and Valuation of Ecosystem Services
WIO	Western Indian Ocean
WIOSAP	Strategic Action Programme for the protection of the Western Indian Ocean

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13. Annex 1. Cross tabulation of ecosystem service classification systems

Table A1. Cross tabulation of ecosystem service classifications. Services included in the same row are considered to be partially or wholly equivalent. The four categories of supporting, provisioning, regulating and cultural services used by Millennium Ecosystem Assessment are included in the left-hand column for reference.

	MA	TEEB	FEGS	CICES	NCP
Supporting Services	Soil formation	Maintenance of soil fertility	Soil	Regulation of soil quality	Formation, protection and decontamination of soils and sediments
	Nutrient cycling	Maintenance of soil fertility			
	Primary production				
		Maintenance of life cycles of migratory species		Lifecycle maintenance, habitat and gene pool protection	Habitat creation and maintenance
		Maintenance of genetic diversity			
					Maintenance of options
Provisioning Services	Food	Food	Flora; Fauna; Fish; Fungi; Open space; Presence of the environment	Cultivated terrestrial and aquatic plants for nutrition, materials or energy	Food and feed

MA	TEEB	FEGS	CICES	NCP
			Reared animals for nutrition, materials or energy	
			Wild plants (terrestrial and aquatic) for nutrition, materials or energy	
			Wild animals (terrestrial and aquatic) for nutrition, materials or energy	
Fresh water	Water	Water	Surface water used for nutrition, materials or energy	Regulation of freshwater quantity, location and timing
			Ground water for used for nutrition, materials or energy	
Fuelwood	Raw materials	Flora	Cultivated terrestrial and aquatic plants for nutrition, materials or energy	Energy

MA	TEEB	FEGS	CICES	NCP
Fiber	Raw materials	Fiber; Natural materials	Cultivated terrestrial and aquatic plants for nutrition, materials or energy	Materials, companionship and labour
Biochemicals	Medicinal resources		Flora; Fauna	Medicinal, biochemical and genetic resources
Genetic resources	Genetic resources		Genetic material from plants, algae or fungi	
			Genetic material from animals	
			Genetic material from organisms	
	Ornamental resources	Flora; Fauna; Fungi		
		Air for transportation		
		Substrate		
		Timber		
		Water for transportation		
		Open space for transportation, infrastructure and waste		

	MA	TEEB	FEGS	CICES	NCP
Regulating Services	Climate regulation	Climate regulation		Atmospheric composition and conditions	Regulation of climate
					Regulation of ocean acidification
					Regulation of air quality
	Disease regulation	Biological control	Depredators and (pest) predators	Pest and disease control	Regulation of detrimental organisms and biological processes
	Water regulation	Regulation of water flows			Regulation of freshwater quantity, location and timing
		Moderation of extreme events		Regulation of baseline flows and extreme events	Regulation of hazards and extreme events
	Water purification	Waste treatment		Water conditions	
				Mediation of wastes or toxic substances of anthropogenic origin by living processes	

	MA	TEEB	FEGS	CICES	NCP
				Mediation of nuisances of anthropogenic origin	
	Pollination	Pollination Air quality regulation Erosion prevention	Pollinators		Pollination
Cultural Services	Spiritual and religious	Spiritual experience	Presence of the environment	Spiritual, symbolic and other interactions with natural environment	
	Recreation and ecotourism	Opportunities for recreation and tourism	Sounds and scents; Fauna; Fish; Wind	Physical and experiential interactions with natural environment	Physical and psychological experiences
	Aesthetic	Aesthetic information	Atmospheric phenomena; Viewscapes;	Physical and experiential interactions with natural environment	
	Inspirational	Inspiration for culture, art and design	Presence of the environment; Sounds and scents	Intellectual and representative interactions with natural environment	Learning and inspiration

MA	TEEB	FEGS	CICES	NCP
Educational	Information for cognitive development	Open space; Presence of the environment	Intellectual and representative interactions with natural environment	Learning and inspiration
Sense of place		Presence of the environment		Supporting identities
Cultural heritage				Supporting identities
			Other biotic characteristics that have a non-use value	Supporting identities
				Materials, companionship and labour

14. Annex 2. Economic value

Economic value is a measure of the human welfare derived from the use or consumption of goods and services. Economic valuation is one way to quantify and communicate the importance of something (e.g. environmental damage, changes in resource availability, ecosystem services etc.) to decision makers, and can be used in combination with other forms of information (e.g. bio-physical indicators and social impacts). The comparative advantage of economic valuation is that it conveys the importance of environmental change directly in terms of human welfare and uses a common unit of account (i.e. money) so that values can be directly compared across other goods, services, investments and impacts in the economy.

Here we provide definitions of the various concepts of economic value that are relevant to the assessment coastal and marine ecosystems.

In neo-classical welfare economics, the economic value of a good or service is the monetary measure of the wellbeing associated with its production and consumption. In a perfectly functioning market, the economic value of a good or service is determined by the demand for and supply of that good or service. Demand for a good or service is determined by the benefit, utility or welfare that consumers derive from it. Supply of a good or service is determined by the cost to producers of producing it. Figure A1 Panel 1 provides a simplified representation of demand (marginal benefit) and supply (marginal cost) for a good traded in a market at quantity 'Q' and price 'P'.

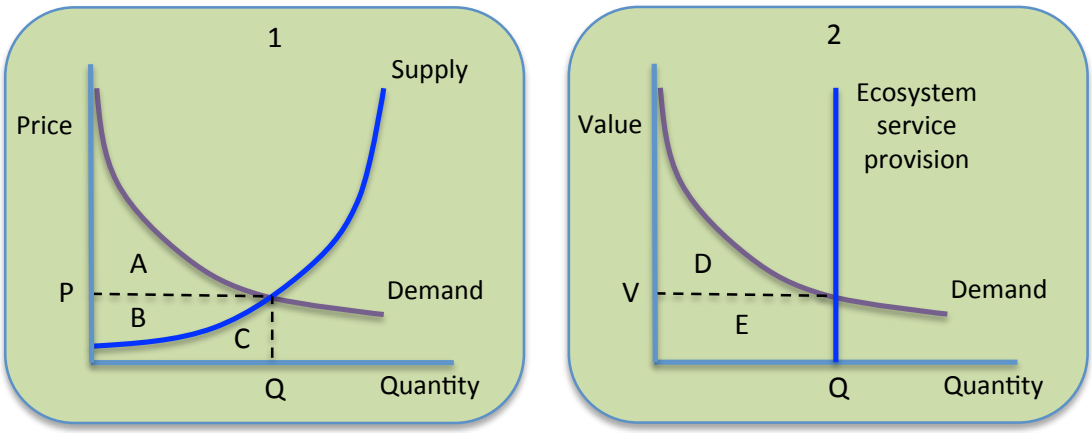

Figure A1: Demand and supply

In Figure A1 Panel 1, area 'A' represents the **consumer surplus**, which is the gain obtained by consumers because they are able to purchase a product at a market price that is less than the highest price they would be willing to pay (which is related to their benefit from consumption and represented by the demand curve). The **producer surplus**, depicted by 'B', is the amount that producers benefit by selling at a market price that is higher than the lowest price that they would be willing to sell for (which is related to their production costs and represented by the supply curve). The area 'C' represents production costs, which differ among producers and/or over the scale of production. The sum of areas A and B is labelled the 'surplus', and is interpreted as

the net economic gain or welfare resulting from production and consumption with a quantity of Q at price P.

In the case that goods and services are not traded in a market (as is the case for many ecosystem services such as climate regulation, coastal protection and biodiversity), the interpretation of the welfare derived from their provision can also be represented in terms of surplus. Figure A1 Panel 2 represents the supply and demand of a non-marketed service. In this case, the service does not have a supply curve in the conventional sense that it represents the quantity of the service that producers are willing to supply at each price. The quantity of the service that is 'supplied' is not determined through a market at all but by other decisions regarding protection status, land use, management, access etc. The quantity of the service supplied is therefore independent of its value. This is represented in Figure A1 Panel 2 as a vertical line. The demand curve for non-marketed services is still represented as a downward sloping line since marginal benefits are expected to decline with quantity (the more that we have of a service, the lower the additional welfare of consuming more). In this case, consumers don't pay a price for the quantity (Q) that is available to them and the entire area under the demand curve (D+E) represents their consumer surplus. It is useful to keep this Figure in mind when considering the measurement of service supply from an coastal or marine ecosystem and the welfare people derive from it.

Note that the demand for goods and services that are used as inputs into the production of marketed goods and services (e.g., the habitat and nursery service provided to fisheries by mangroves and coastal wetlands are generally uncompensated inputs into fisheries production) is derived from the demand for the good or service that is finally consumed (e.g. fish).

The **marginal value** of a good or service is the contribution to wellbeing of one additional unit. It is equivalent to the price of the service in a perfectly functioning market (P in Figure 5). Small changes in ecosystem service provision should be valued using marginal values. The **average value** of a good or service can be calculated as its total value divided by the total quantity of the service provided and consumed. From Figure 5 Panel 2, average value can be calculated as (D+E)/Q. Average values may be useful for comparing the aggregate value of a good or service relative to the scale of provision (defined in terms of units of provision, area of ecosystem or number of beneficiaries).

Total Economic Value (TEV)

The concept of **Total Economic Value** (**TEV**) of an ecosystem is used to describe the comprehensive set of utilitarian values derived from that ecosystem. This concept is useful for identifying the different types of value that may be derived from an ecosystem. TEV comprises of **use values** and **non-use values**. Use values are the benefits that are derived from some physical use of the resource. **Direct use values** may derive from on-site extraction of resources (e.g. fisheries) or non-consumptive activities (e.g. recreation). **Indirect use values** are derived from off-site services that are related to the resource (e.g. climate regulation, coastal protection). **Option value** is the value that people place on maintaining the option to use an ecosystem resource in the future. Non-use values are derived from the knowledge that an ecosystem is

maintained without regard to any current or future personal use. **Non-use values** may be related to altruism (maintaining an ecosystem for others), bequest (for future generations) and existence (preservation unrelated to any use) motivations. The constituent values of TEV are represented in Figure A2. It is important to understand that the "total" in Total Economic Value refers to the identification of all components of value rather than the sum of all value derived from a resource. TEV is a comprehensive measure, as opposed to a partial measure, of value. Accordingly, many estimates of TEV are for marginal changes in the provision of ecosystem services but "total" in the sense that they take a comprehensive view of sources of value.

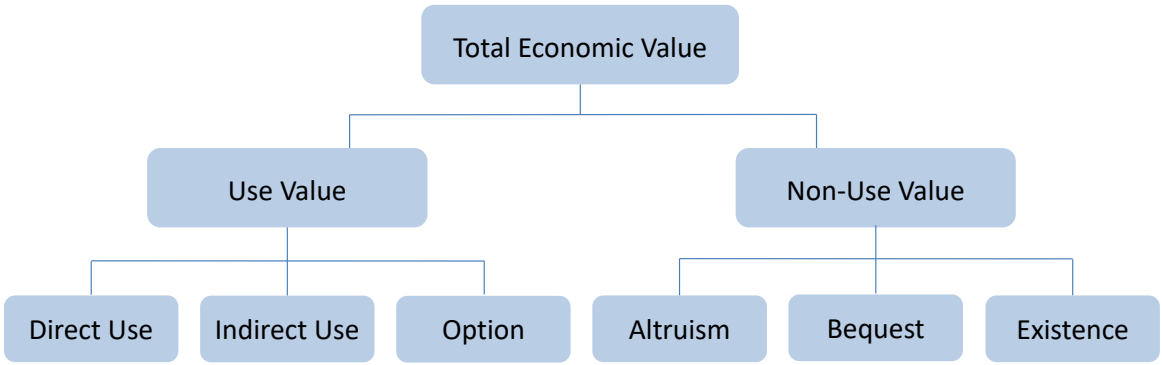

Figure A2: The components of Total Economic Value

The classification of different types of economic value within the concept of TEV is complementary to the classification of ecosystem services. Table 2 sets out the correspondence between categories of ecosystem service and components of TEV.

Table A1: Correspondence between ecosystem services and components of Total Economic Value

	Total Economic Value			
Ecosystem service	Direct use	Indirect use	Option value	Non-use
Provisioning	E.g. fish		Option to use Provisioning service	
Regulation and maintenance		E.g. climate regulation	Option to use Regulating service	
Cultural	E.g. recreation		Option to use Cultural service	E.g. bequest value

Exchange value

The concept of welfare value is used in most assessments of ecosystem services but it is not used in the System of National Accounts (SNA) that is used to calculate Gross Domestic Product (GDP) and other economic statistics. The SNA uses the concept

exchange value, which is a measure of producer surplus plus the costs of production. In Figure A1 Panel 1 this is represented by areas B and C, or equivalently P times Q. Under the concept of exchange value, the total outlays by consumers and the total revenue of the producers are equal. For national accounting purposes, this approach to valuation enables a consistent and convenient recording of transactions between economic units since the values for supply and use of products are the same. In the context of comparing the values of ecosystem services with values in the system of national accounts, it is therefore necessary to value the total quantity of ecosystem services at the market prices that would have occurred if the services had been freely traded and exchanged. In other words, it is necessary to measure exchange value and not welfare value.

The differences between the concepts of welfare value and exchange value are the inclusion of consumer surplus (A) in the former and the inclusion of production costs in the latter (C). The concept of welfare value corresponds to a theoretically valid measure of welfare in the sense that a change in value represents a change in welfare for the producers and/or consumers of the goods and services under consideration. The concept of exchange value does not correspond to a theoretically valid measure of welfare and a change in exchange value does not necessarily represent a change in welfare for either producers or consumers.[7]

[7] See Day (2013) for a more detailed explanation of welfare and exchange values.

15. Annex 3: Other useful manuals and guidelines

Manual / guidelines	Link
DEFRA (2007). An introductory guide to valuing ecosystem services (2007). Department for Environment, Food and Rural Affairs.	https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/69192/pb12852-eco-valuing-071205.pdf
CGIAR (2016). Guidelines for wetlands ecosystems valuation in the Nile basin	https://wle.cgiar.org/guidelines-wetlands-ecosystems-valuation-nile-basin
DEFRA (2013). Guidance for policy and decision makers on using an ecosystems approach and valuing ecosystem services. Department for Environment, Food & Rural Affairs	https://www.gov.uk/ecosystems-services
Ecosystem Services Valuation Database (ESVD)	https://www.es-partnership.org/services/data-knowledge-sharing/ecosystem-service-valuation-database/
Ecosystem Valuation	http://www.ecosystemvaluation.org/uses.htm
Emerton, L. and G. Howard (2008), A Toolkit for the Economic Analysis of Invasive Species. Global Invasive Species Programme, Nairobi	https://portals.iucn.org/library/node/9248
Freeman, A.M.I. (2003). The Measurement of Environmental and Resource Values. Resources for the Future, Washington D.C.	
GEF IW:LEARN (2018). GEF Guidance Documents to Economic Valuation of Ecosystem Services in IW Projects	https://iwlearn.net/documents/28952
GEF LME:LEARN (2018). Environmental Economics for Marine Ecosystem Management Toolkit	https://iwlearn.net/manuals/environmental-economics-for-marine-ecosystem-management-toolkit
Joint Nature Conservation Committee - JNCC (2007) Valuation toolkit for small islands	http://jncc.defra.gov.uk/page-4065
Johnston, R.J., Boyle, K.J., Adamowicz, W., Bennett, J., Brouwer, R., Cameron, T.A., Hanemann, W.M., Hanley, N., Ryan, M., Scarpa, R. and Tourangeau, R., (2017). Contemporary guidance for stated preference studies. *Journal of the Association of Environmental and Resource Economists*, *4*(2), 319-405.	

Manual / guidelines	Link
IPBES (2016) Guidance on Diverse Values and Valuation	https://www.ipbes.net/diverse-values-valuation
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Salcone J., Brander, L. and Seidl, A. (2016). Guidance manual on economic valuation of marine and coastal ecosystem services in the Pacific. Report to the MACBIO Project	http://macbio-pacific.info/Resources/marine-ecosystem-services-valuation-in-the-pacific/
Schaafsma, M., Bartkowski, B., and Lienhoop, N. (2018). Guidance for Deliberative Monetary Valuation Studies. International Review of Environmental and Resource Economics, 12(2-3), 267-323.	
Swedish Environmental Protection Agency (2006). An instrument for assessing the quality of environmental valuation studies	http://www.naturvardsverket.se/Documents/publikationer/620-1252-5.pdf
Tallis, H. (2011). *Natural capital: theory and practice of mapping ecosystem services*. Oxford University Press.	

Manual / guidelines	Link
TEEB Guidance manual for country studies	http://www.teebweb.org/resources/guidance-manual-for-teeb-country-studies/
UNEP Guidance Manual on Valuation and Accounting of Ecosystem Services for Small Island Developing States	https://www.cbd.int/financial/monterreytradetech/unep-valuation-sids.pdf
UNEP Guidance manual on value transfer methods for ecosystem services	http://wedocs.unep.org/handle/20.500.11822/8434
UNEP Guidance toolkit for the valuation of regulating services	http://www.ecosystemassessments.net/resources/guidance-manual-for-the-valuation-of-regulating-services.pdf
UNEP (2015). Success Stories in Mainstreaming Ecosystem Services into Macro-economic Policy and Land Use Planning: Evidence from Chile, Trinidad and Tobago, South Africa and Viet Nam	
USAID (2014) Guidelines for the rapid economic valuation of biodiversity and ecosystem services	https://www.climatelinks.org/sites/default/files/asset/document/2014_USAID-PREPARED_Guidelines-for-Rapid-Economic-Valuation-Biodiversity-Ecosystems.pdf
USAID (2018) Best practice and lessons learned How economic valuation, management plans and conservation plans can support conservation in East Africa	https://www.climatelinks.org/sites/default/files/asset/document/2018_USAID-PREPARED_Economic-Valuation-Management-Plans-Conservation-Investments.pdf
VALUES Methods for integrating ecosystem services into policy, planning, and practice	http://www.aboutvalues.net/about_values/
van Beukering, P., Brander, L., Tompkins, E., & Mackenzie, E. (2007). Valuing the environment in small islands: An environmental economics toolkit.	http://jncc.defra.gov.uk/page-4065
WRI (2012) guidance toolkit on coastal capital	http://www.wri.org/our-work/project/coastal-capital-economic-valuation-coastal-ecosystems-caribbean

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17. Annex 5. Incentive and market-based policy instruments

Taxes

Taxes are charges that are paid on inputs, outputs or emissions from production or on the consumption of products. Taxes can have two functions: 1. To raise revenue to fund public expenditure; 2. To regulate economic activities by increasing their cost. Environmental taxes are levied on inputs or products that have negative environmental impacts, thereby providing incentives for producers or consumers to reduce the use, production or consumption of taxed items. In other words, environmental taxes work by internalising external costs so that the generator of that cost takes it into account in their decision making.

Environmental taxes can potentially deliver a “double dividend” in that they can produce two positive outcomes: 1. internalise external costs to disincentivise environmentally harmful production and consumption; 2. generate revenue that can be spent of environmental protection or potentially used to reduce other forms of tax that may distort positive economic behaviour (e.g. income tax).

In the context of coastal and marine ecosystems, there may be a case for advocating taxes on resource use and polluting activities to both partially restrict such activities and to generate revenue that can be used for ecosystem management.

Subsidies

A subsidy is a payment from the government to an economic agent to promote a particular activity. Environmental subsidies can be used to promote production or consumption that reduces negative impacts to the environment. Environmental

subsidies can take a number of different forms: direct subsidies for environmental improvements; production subsidies with environmental pre-conditions; tax breaks such as capital allowances for environmental investments; tax rebates, grants or loans for environmental investments; financial support for advice services or voluntary initiatives; tax credits that reduce a person's liability to pay an environmental tax if they have funded an approved environmental project.

Subsidies are widely used to promote economic development and support specific sectors. This includes many industries operating in or impacting on the marine environment (e.g. fisheries, energy, agriculture). In the case of fisheries, subsidies can take the form of reduced prices for fuel and equipment, or the provision of infrastructure (e.g. ports) and processing facilities. While this may be benefit the operators within the targeted sector, it is widely recognised that subsidized activities can have unintended environmental consequences through negative externalities. The design of subsidies therefore needs to include an assessment of their wider impacts and measures to mitigate negative external costs. It is also advisable that subsidies are time limited to avoid permanent dependence on government support, for example to cover periods of transition to new technologies or regulations.

Tradable permits and quotas

Tradable permits or quotas are used to control the overall level of a particular activity, type of pollution or the use of a resource but allow individual agents to buy or sell permits in order to ensure that they are allocated to the highest valued use. This form of policy instrument is also referred to as "cap-and-trade". In the context of fisheries, the term "individual transferable quota" is used. The steps in designing a tradable permit system are:

1. Set limit for total emissions or use of resource equal to an optimal/ sustainable level (for each sector, region, period)
2. Make initial allocation of permits to polluters/resource users
3. Allow permits to be traded between polluters/resource users
4. Monitor actual emissions/resource use by each polluter/resource user
5. Impose penalties if emissions/resource use exceed the amount of permits held

The advantages of using tradable permits to manage the use of environmental resources are: 1. Setting a total limit ensures that use does not exceed safe or sustainable levels (this is not guaranteed when using environmental taxes); 2. Allowing permits to be traded ensures that they are allocated to the users that gain the highest value from their use (i.e. ensures efficiency).

Challenges in designing a tradable permit system are: 1. The initial allocation of permits can determine the distribution of returns across participants within the sector. An auction of permits may be seen as a fair or neutral initial allocation but does not reflect historical use of the resource. 2. Permits may affect competition within the sector by enabling a small number of firms to gain market (monopoly) power or to behave strategically (e.g. prevent the entry of new firms). 3. In the case that the specific location and timing of emissions/resource use is relevant to its impact or sustainability, it may be necessary to introduce restrictions on where and when permits can be redeemed.

In the case of fisheries management, individual transferable quota (ITQ) programs allocate shares of a total allowable catch (TAC) to individual fishers, entities or vessels. Such systems allow more individual flexibility in decisions regarding when to fish and what technology to use (in comparison to fishing effort restrictions).

A useful resource for designing ITQ programs is the Environmental Defence Fund (EDF) catch share design manual:

http://catchshares.edf.org/catch-share-basics/manuals-and-guides?_ga=2.167034358.959366235.1518509902-1436448972.1518509902

Area based user rights

An alternative approach to quota based user rights such as individual transferable quotas (ITQs) is to allocate a share of the harvestable area for a resource to each user. In the case of fisheries management this is termed "territorial use rights for fishing" (TURF).

TURFs allocate exclusive rights to harvest one or more target species in a specified area to groups or individuals. They are usually managed by an organized cooperative of fishermen and set appropriate controls on fishing activities. The use of TURFs is most applicable for target species that are not highly mobile and remain within the specified areas.

Environmental Defence Fund (EDF) catch share design manual also provide guidance on designing TURFs and a number of example applications:

http://catchshares.edf.org/catch-share-basics/manuals-and-guides?_ga=2.167034358.959366235.1518509902-1436448972.1518509902

Certification and labelling

Product labelling can be used to indicate the environmental and social characteristics of goods and services. Certification and labelling of goods and services with positive environmental characteristics enables markets to develop for such products, in which consumers can fulfill their preferences for environmental sustainability and producers can gain a price premium or market share. Certification and labelling addresses one of the difficulties in establishing markets for environmentally friendly products, namely that consumers are otherwise unaware of (or cannot trust claimed) differences between the production processes of products. In other words, it addresses a market failure due to imperfect and asymmetric information held by producers and consumers.

The main elements in a certification and labelling system are: 1. Setting environmental and social criteria for the production process; 2. A system of third party verification of compliance with the criteria; 3. Ensuring credibility and trust of the process and label; 4. Promoting consumer recognition of the label.

Examples of certification and labelling systems that are relevant to coastal and marine ecosystems are:

Marine Stewardship Council (MSC): https://www.msc.org/

Aquaculture Stewardship Council (AFC): https://www.asc-aqua.org/

Conservation Trust Funds (CTF)

Conservation Trust Funds (CTF) or Environmental Trust Funds are generally designed as independent grant-making institutions that mobilize and manage financial resources for environmental purposes, such as biodiversity conservation, climate adaptation and mitigation.

CTFs can be structured as endowment funds (allocating a share of the income generated by the "endowment", which is usually composed of stocks or other revenue generating assets), sinking funds (disburses a share of its capital each year over a defined period of time, until it sinks to zero) or revolving funds (replenished or augmented on a regular basis, usually through government contributions).

The UNDP provides a useful detailed explanation of CTFs, guidance and case studies:

http://www.undp.org/content/sdfinance/en/home/solutions/environmental-trust-funds.html

The Conservation Finance Alliance has produced a set of practice standards for CTFs:

https://static1.squarespace.com/static/57e1f17b37c58156a98f1ee4/t/5953eae486e6c0fb1c81cb93/1498671896001/CFA_Standards_full-compressed.pdf

Debt for nature swaps (DNS)

Debt-for-nature swaps (DNS) involve debtor governments committing to invest in conservation and/or climate change adaptation or mitigation in return for a reduction or cancellation of debt on the part of creditors. Such arrangements have also been established in the form that creditors agree to sell the debt they hold to a third party (e.g. a conservation organisation) for a discounted price.

The UNDP provides a useful detailed explanation of DNS, guidance and case studies:

http://www.undp.org/content/sdfinance/en/home/solutions/debt-for-nature-swaps.html

A further resource on sustainable finance mechanisms is the OECD report on scaling up finance mechanisms for biodiversity:

https://bluesolutions.info/images/OECD-Scaling-up-Finance-Mechanisms-for-Biodiversity-2013.pdf

Payments for Ecosystem Services

Payment for Ecosystem Services (PES) is a relatively new policy instrument in resource conservation that establishes a mechanism through which ecosystem service beneficiaries can compensate service providers (Kumar and Thiaw, 2013). PES

schemes are based on the principle that people located in ecosystems that provide the services (providers) should be compensated for the continuous provision of such services, while the people who benefit (beneficiaries) from ecosystem services should pay for the protection of such ecosystems (Macandog, 2014).

The term "payments for ecosystem services" (PES) covers a broad set of mechanisms through which incentives for the provision of ecosystem services are established. In a PES scheme, providers of an ecosystem service (e.g. upstream farmers who conserve forests that control water flow) are incentivized to provide that service through some form of payment or compensation, which may be paid by the beneficiaries of the service (e.g. downstream farmers that benefit from lower exposure to flooding). PES schemes attempt to provide incentives for the continued or enhanced provision of services and address the commonly observed problem that markets do not exist for ecosystem services (Wunder, 2014 Pagiola et al., 2005; Engel et al., 2008). It is the creation of incentives that is crucially important since the provider of an ecosystem service may otherwise be better off using the ecosystem resource in another way (e.g. an upstream farmer might convert forest area to agricultural land).

A broad definition of PES is "a simple transfer of resources between social actors, which aims to create incentives to align individual or collective land use decisions with social interest in the management of natural resources" (Muradian et al, 2010). This broad definition allows for the inclusion of compulsory transactions, payments in kind rather than money (e.g. infrastructure development, education, technical assistance, tenure security), and relaxes the technically challenging condition that the ES provider is shown to ensure the provision of the service.

One of the main attractions of PES as a policy instrument is that it can in principle be self-financing in the case that payments by beneficiaries cover all associated costs (transaction costs as well as opportunity costs of the provider of ecosystem services). A further attraction of this policy instrument is that it can in principle result in an efficient allocation of resources. In theory, payments for ecosystem services will continue up until the point at which the marginal costs of providing them (i.e. the value of the foregone use of those resources to the provider) equals the marginal benefit (i.e. to value of the ecosystem service to the beneficiary).

The observed disadvantages of this policy instrument are the high transaction costs involved in establishing and operating a PES scheme, the institutional requirements for setting, collecting and disbursing payments, and the information requirements to monitor the activities of participants. Few PES schemes have proven to be financially sustainable in the long term after initial funding, often from international donors, has ended.

Biodiversity offsets, banking and trading

Biodiversity offsets are conservation actions intended to compensate for the unavoidable harm to biodiversity caused by development projects, so as to ensure no net loss of biodiversity (ten Kate et al., 2004). Such conservation actions can be on-site or off-site, and include the restoration or creation of areas of similar habitat to that which is harmed or destroyed.

Biodiversity offsetting is predominantly used by planning authorities and developers to prevent biodiversity loss and, in some cases, produce gains in biodiversity. Biodiversity offsetting involves using qualitative and quantitative measures to determine the amount, type and quality of habitat that is likely to be affected by a proposed project. Biodiversity offsetting can also involve estimating the cost of replacing damaged biodiversity by calculating the cost of creating the same amount, type and quality of habitat at other locations.

Biodiversity banking, also known as biodiversity trading or conservation banking, is a process through which gains in habitat and biodiversity can be reliably measured and traded for the purpose of offsetting losses in biodiversity elsewhere. The term "banking" is used to refer to the way offsets are created and approved prior to development and biodiversity loss taking place. The resulting conservation benefits are "banked" with the regulator and later sold as offsets to future development projects. One of the aims of banking is to avoid any temporal loss of ecosystem benefits.

Biodiversity offsets are generally viewed as a secondary conservation measure to be used in cases where direct protection of biodiversity is not feasible. In other words, developers are expected to first seek to avoid and minimise harm to biodiversity before they contemplate the use of offsets. One of the challenges involved in using biodiversity offsets as an effective conservation policy instrument is ensuring that the habitat that is created or restored is genuinely equivalent in terms of biodiversity to that which is destroyed. Further challenges are associated with monitoring and the permanence of restored habitats.

It is advisable that biodiversity offsetting is only used after all other options in the mitigation hierarchy have been considered and no alternatives are available. Avoidance of biodiversity loss is the first and most important step in the mitigation hierarchy. Biodiversity offsets should not be used to circumvent responsibilities to avoid and minimise damage to biodiversity, or to justify projects that would otherwise not happen.