Report on Economic Mapping and Assessment Methods for Ecosystem Services

Deliverable D3.2

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Enhancing ecosystem services mapping for policy and decision making

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Table of contents

Preface

Mapping and assessment of ecosystems and their services (ES) are core to the EU Biodiversity (BD) Strategy. They are essential if we are to make informed decisions. Action 5 sets the requirement for an EU-wide knowledge base designed to be: a primary data source for developing Europe's green infrastructure; resource to identify areas for ecosystem restoration; and, a baseline against which the goal of 'no net loss of BD and ES' can be evaluated.

In response to these requirements, ESMERALDA (Enhancing ecoSysteM sERvices mApping for poLicy and Decision mAking) aims to deliver a flexible methodology to provide the building blocks for pan-European and regional assessments. The work will ensure the timely delivery of EU member states in relation to Action 5 of the BD Strategy, supporting the needs of assessments in relation to the requirements for planning, agriculture, climate, water and nature policy. This methodology will build on existing ES projects and databases (e.g. MAES, OpenNESS, OPERAs, national studies), the Millennium Assessment (MA) and The Economics of Ecosystems and Biodiversity (TEEB). ESMERALDA will identify relevant stakeholders and take stock of their requirements at EU, national and regional levels.

The objective of ESMERALDA is to share experience through an active process of dialogue and knowledge co-creation that will enable participants to achieve the Action 5 aims. The mapping approach proposed will integrate biophysical, sociocultural and economic assessment techniques.

The six work packages of ESMERALDA are organised through four strands (see Figure 1), namely policy, research, application and networking, which reflect the main objectives of EMSERALDA.

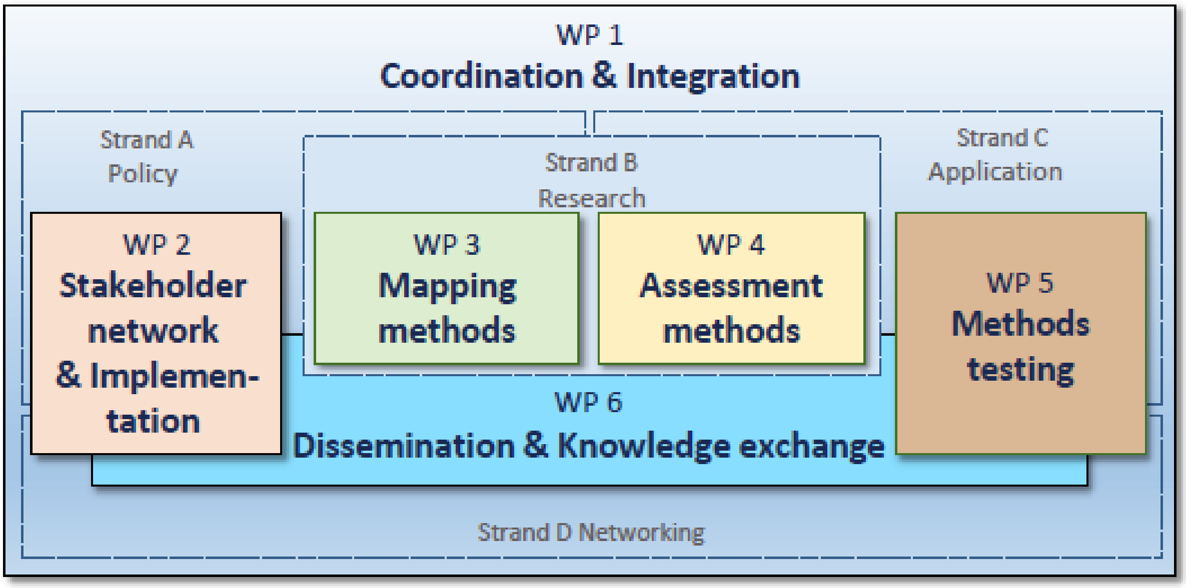


Figure 1: ESMERALDA components and their interrelations and integration within the four ESMERALDA strands.

This report sits within work packages WP3 "Mapping methods" and WP4 "Assessment Methods".

When making the proposal, the original idea was to investigate similarities and differences when using methods for the mapping and/or assessment of ecosystem services; as a result the effort was split across two different work packages, namely WP 3 "Mapping methods", and WP4 "Assessment Methods". A draft deliverable (Del. 4.2) on *"Economic assessment methods and application*" was submitted in month 24, approximately than half way through the project. The work found that it was very difficult to make a clear distinction between economic methods for mapping and/or assessment; there was also potential duplication of material between the two elements. A discussion with the task leaders for the sociocultural, economic and biophysical method work streams, as well as the wider project community, led to the decision to merge deliverables on methods for mapping (WP3) and assessment (WP4) of ecosystem services. The submission time for the three deliverables was harmonized to month 36. The merger of deliverables was submitted through amendment No 22 and was accepted by the European Commission on 22nd January 2017.

This report therefore provides an overview of the main economic methods for mapping *and* assessment of ecosystem services. This report also highlights the need to link and integrate information from sociocultural and biophysical methods. ESMERALDA reports D3.1 and D3.3 specifically provide guidance on sociocultural and biophysical methods for mapping and assessment of ecosystem services. ESMERALDA report D3.4 provides guidance on how biophysical, socio-cultural and economic methods can be linked within an ecosystem service assessment and on methods for integrating information outputs across disciplinary domains; and report D4.8 provides guidance on integrated assessment of ecosystem services. All these deliverables address the challenge of improving the applicability of these approaches with specific examples, particularly with respect to the MAES process and the ESMERALDA case studies.

Summary

This report provides an overview of the main economic methods for mapping and assessment of ecosystem services. Here we provide a brief summary of the main points of information addressed in this report.

- **Economic mapping** of ecosystem services involves the measurement of their economic value accounting for spatial variation in supply and demand. **Economic assessment** of ecosystem services involves the structuring and integration of value information into decision making and the design of policy instruments.
- **Economic value of ecosystem services is a measure of the human welfare** derived from the use or consumption of ecosystem services. Economic valuation is one way to quantify and communicate the importance of ecosystem services to decision makers, and can be used in combination with other forms of information (e.g. socio-cultural values or biophysical indicators).
- The concept of **Total Economic Value** (**TEV**) of an ecosystem is a framework for identifying the comprehensive set of utilitarian values derived from that ecosystem. The word "total" in Total Economic Value refers to the inclusion of different sources of value; TEV does not imply the calculation of an aggregate value of a resource. The classification of different sources of economic value within the concept of TEV is complementary to the classification of ecosystem services.
- The System of National Accounts (SNA) used to produce conventional macro-economic statistics (e.g. GDP) uses a non-welfare based concept of economic value termed **exchange value**. For the purposes of producing ecosystem service accounts that are consistent with the SNA, it is necessary to use estimates of ecosystem services values that are quantified as exchange values.
- The economic value of an ecosystem service is determined by its supply and demand. The supply side of an ecosystem service is largely determined by ecological processes and characteristics that

may be influenced by human activities, either deliberately or inadvertently. The demand side is largely determined by the characteristics of human beneficiaries of the ecosystem service. The determinants of both the supply and demand of ecosystem services are spatially variable, which makes the estimation of ecosystem service values inherently spatial. Value mapping addresses this spatial dimension of ecosystem service valuation.

- Ecosystem services are often not traded in markets and so a number of **"primary" non-market valuation methods** have been developed to estimate their economic values. These include the use of replacement costs, avoided damage costs, production functions, revealed preferences (e.g. hedonic pricing, travel costs), and stated preferences (e.g. contingent valuation, choice experiments).
- **Value transfer** (benefit 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 methods are a relatively expeditious and inexpensive means of obtaining estimates of ecosystem service values and can be applied at geographic scales that are not feasible for primary valuation applications. The accuracy of value transfer is dependent on the similarities of characteristics across study sites and policy sites and the extent to which differences are controlled for.
- Economic methods for the assessment of ecosystem services are frameworks for generating and structuring economic information to support decision-making regarding ecosystem services. These assessment methods include: cost-effectiveness analysis, cost-benefit analysis, multi-criteria analysis, ecosystem service accounting and corporate ecosystem service reviews.
- 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, establish protected areas, or target financial incentives to change the behaviour of land 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.
- The choice of which economic mapping or assessment method to use is largely determined by the ecosystem service(s) under consideration, the type of decision problem and the availability of information. To understand the differences between economic mapping and assessment methods, we describe the procedural steps of each approach, provide brief example applications and discuss the strengths and weaknesses of each approach. Each method is assigned to one of the three defined tiers to reflect the precision of its output and the resources required for its application.
- The application of economic mapping and assessment methods will often require inputs from socio-cultural and biophysical methods (and vice versa). In addition, the production of policy relevant information may require the integration of separate outputs from biophysical, economic and socio-cultural mapping and assessment applications.

1. Introduction to economic mapping and assessment methods

This report provides an overview of economic methods for mapping and assessment of ecosystem services. In this context, the term “mapping” is used to mean the description and representation of spatial variation. Mapping therefore includes both the representation of data on maps and/or the process of accounting for spatial variation in the phenomena under consideration. Economic methods for mapping of ecosystem services examine spatial variation in the economic value of ecosystem services.

The process of mapping ecosystem service values falls within the broader process of ecosystem service assessment. The term “assessment” is defined in the ESMERALDA project as “the analysis and review of information derived from research for the purpose of helping someone in a position of responsibility to evaluate possible actions or think about a problem. Assessment means assembling, summarising, organising, interpreting, and possibly reconciling pieces of existing knowledge and communicating them so that they are relevant and helpful to an intelligent but inexpert decision-maker”. Assessment therefore focuses on how information on ecosystem services can be structured to support decision-making.

Economic methods for mapping and assessing ecosystem services principally involve measuring the economic value of ecosystem services, including its spatial variation, and structuring this information to support decision making and the design of policy instruments. As such, economic methods operate on the right side of the ecosystem services cascade model to quantify the benefits to humans (Potschin and Haines-Young, 2011). Any economic mapping or assessment therefore fundamentally relies on biophysical data and methods to quantify the capacity of ecosystems to supply ecosystem services (i.e. the left side of the cascade model). Economic methods can also be used in combination with socio-cultural methods to gain a broader understanding of the importance of ecosystem services to society.

Economic methods for measuring and *mapping* ecosystem services include primary valuation and value transfer methods. The process of representing economic values on maps necessarily involves some form of spatial extrapolation or transfer of value information. The principal primary and transfer methods are explained in this report together with an evaluation of their strengths, weaknesses and applicability to different ecosystem services.

Economic methods for *assessing* ecosystem services include cost-effectiveness analysis, cost-benefit analysis, multi-criteria analysis, ecosystem service assessments, ecosystem service accounting, and corporate ecosystem service review. These methods are explained in this report together with an evaluation of their strengths, weaknesses and applicability to ecosystem service assessment. Where relevant, the potential role of ecosystem service maps as input into economic assessments is highlighted.

2. Framework for economic methods and key concepts

2.1. What is economic value?

Economic value of ecosystem services is a measure of the human welfare derived from the use or appreciation of ecosystem services (Pascual et al., 2010). Economic valuation is one way to quantify and communicate the importance of ecosystem services to decision makers, and can best be used in combination with other forms of information to provide a complete picture of how human welfare depends on natural capital (e.g. socio-cultural values or bio-physical indicators – see ESMERALDA reports D3.1 and D3.3 respectively). The comparative advantage of economic valuation is that it conveys the importance of ecosystem services directly in terms of human welfare and uses a common unit of account (i.e. money) so that values can be directly compared across ecosystem services and across other goods and services in the economy.

Here we provide definitions of the various concepts of economic value that may be encountered when valuing and mapping ecosystem services.

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 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'. The demand curve is 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). The supply curve is represented as an upward sloping line since marginal costs of production are generally expected to increase with quantity (as low cost inputs become exhausted).

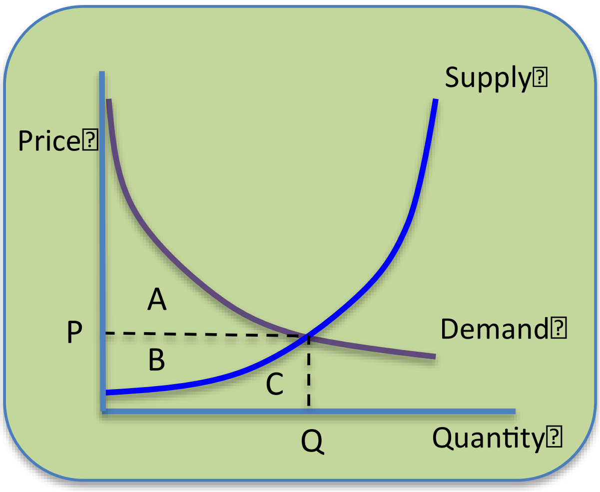


Figure 1: Demand and supply curves for a conventional good or service traded in a market

In Figure 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 ecosystem services are not traded in a market, the interpretation of the welfare derived from their provision can also be represented in terms of surplus. Figure 2 represents the supply and demand of a non-marketed ecosystem service. In this case, the ecosystem 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 ecosystem service that is 'supplied' is not determined through a market at all but by other decisions regarding ecosystem protection, land use, management, access etc. The quantity of ecosystem service supplied is therefore independent of its value. This is represented in Figure 2 as a vertical line. For the most part, biophysical indicators of ecosystem services measure the quantity supplied but not the welfare obtained. The demand curve for non-marketed ecosystem services is still represented as a downward sloping line since marginal benefits are expected to decline with quantity. In this case, consumers do not 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 picture in mind when considering the measurement of ecosystem service supply and the welfare people derive from it.

Note that the demand for ecosystem services that are inputs into the production of marketed goods (e.g., pollination and erosion control are generally uncompensated inputs in agricultural production) is derived from the demand for the good or service that is finally consumed (e.g. food). Figure 2 also represents the conceptual value of ecosystem services that have a 'derived demand'.

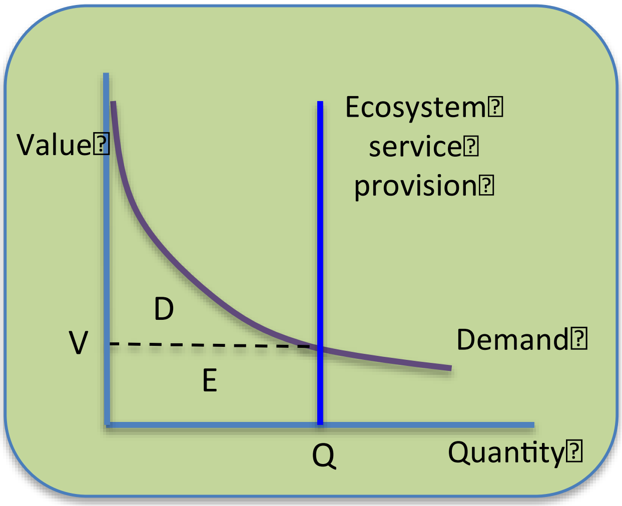


Figure 2: Demand and supply for a non-marketed ecosystem service

The **marginal value** of an ecosystem service is the contribution to wellbeing of one additional unit of the service (V in Figure 2). It is equivalent to the price of the service in a perfectly functioning market (P in Figure 1). Small changes in ecosystem service provision should be valued using marginal values. The **average value** of an ecosystem service can be calculated as its total value divided by the total quantity of the service provided and consumed. From Figure 2, average value can be calculated as (D+E)/Q. Average values may be useful for comparing the aggregate value of an ecosystem service relative to the scale of provision (defined in terms of units of provision, area of ecosystem, or number of beneficiaries).

2.2. 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 (Pearce and Turner, 1990). 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. fuel wood) or non-consumptive activities (e.g. recreation). **Indirect use values** are derived from off-site services that are related to the resource (e.g. downstream flood control, climate regulation). **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 3. It is important to understand that the "total" in Total Economic Value refers to the inclusion of different sources of value rather than the sum of all value derived from a resource. TEV is a measure of total value as opposed to partial 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.

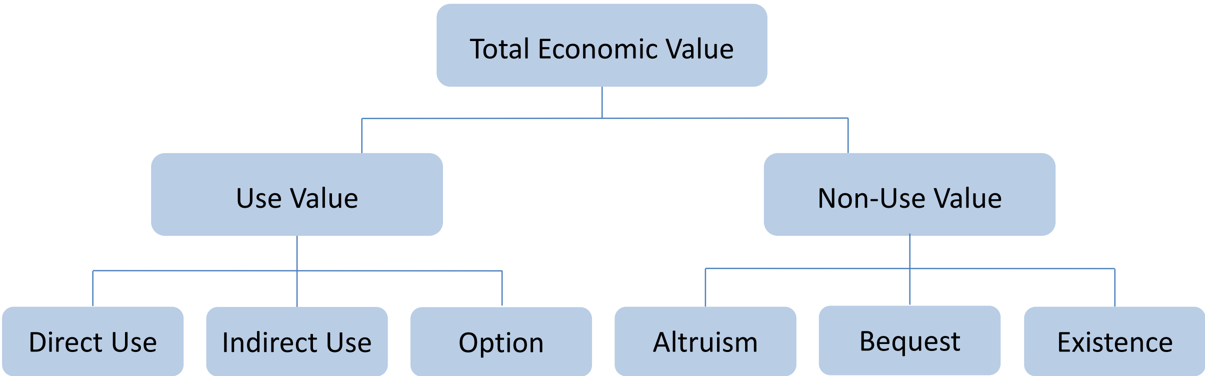


Figure 3: The components of Total Economic Value (Pearce and Turner, 1990)

The classification of different types of economic value within the concept of TEV is complementary to the classification of ecosystem services. Table 1 sets out the correspondence between categories of ecosystem service and components of TEV.

Table 1: Correspondence between ecosystem services and components of Total Economic Value

	Total Economic Value			
Ecosystem service	Direct use	Indirect use	Option value	Non-use
Provisioning	X		X	
Regulation and maintenance		X	X	
Cultural	X		X	X

2.3. Exchange value

The concept of welfare value is used in most economic 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 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.[1]

3. Methods for estimating economic values for ecosystem services

A variety of methods have been developed for estimating the economic value of ecosystem services that are designed to span the range of valuation challenges raised by the application of economic analyses to the complexity of the natural environment. Figure 4 provides a representation of the available economic methods for valuing ecosystem services. A key distinction is between methods that produce new or original information generally using primary data (primary valuation methods) and those that use existing information in new policy contexts (value transfer methods). Primary valuation methods are described in section 3.1 and value transfer methods are described in section 3.2.

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

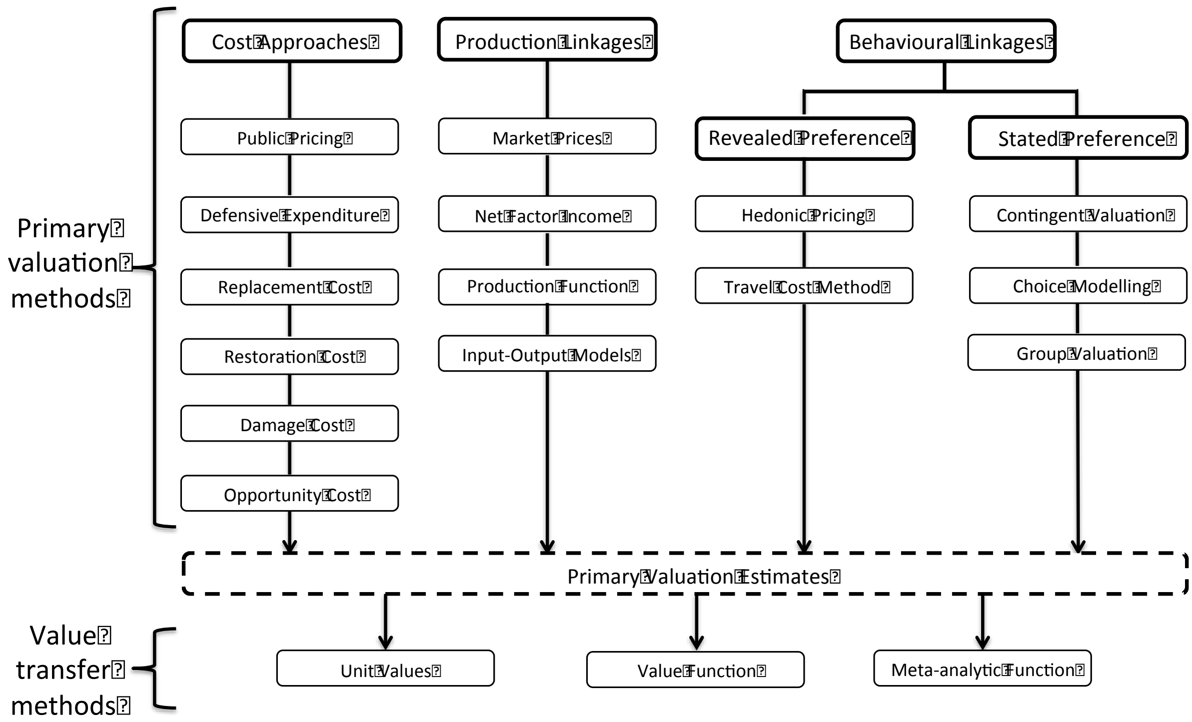


Figure 4: Overview of primary valuation and value transfer methods

3.1. Primary valuation methods

Primary valuation methods can be divided into three categories: 1. Cost-based approaches that use some measure of the costs associated with an ecosystem service as a proxy for the value of the service; 2. Methods that estimate the value of ecosystem services as inputs into production; and 3. Methods that use consumer behaviour to measure the value of ecosystem services. This third category can be further usefully divided 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.

Table 2 provides an overview of primary valuation methods, typical applications, limitations and indicates which primary valuation methods can be used to value which ecosystem service. It should be noted that different valuation methods produce different measures of economic value that are not equivalent and cannot necessarily 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 is listed in Annex 1.

Table 2: Primary valuation methods, applicability to ecosystem services, examples and limitations (adapted from Table A2, Brander 2013)

Valuation method	Approach	Application to ecosystem services	Example ecosystem service	Limitations	Tier[2]
Market prices	Prices for ES that are directly observed in markets	ES that are traded directly in markets	Timber and fuel wood from forests; clean water from wetlands	Market prices can be distorted e.g. by subsidies. Most ES are not traded in markets	1
Public pricing	Public expenditure or monetary incentives (taxes/subsidies) for ES as an indicator of value	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	1
Defensive expenditure	Expenditure on protection of ES	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	1
Replacement cost	Estimate the cost of replacing an ES with a man-made service	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.	1
Restoration cost	Estimate cost of restoring degraded ecosystems to ensure provision of ES	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.	1

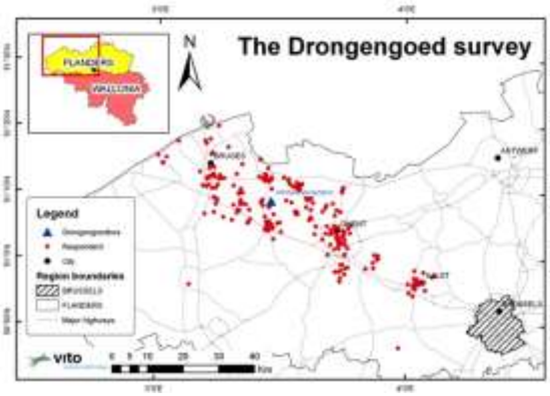
[2] Each method is assigned to a tier (1-3) to broadly reflect the accuracy, detail, technical capacity and data requirements of the method. See section 7 for a full explanation of the approach used to assign methods to tiers.

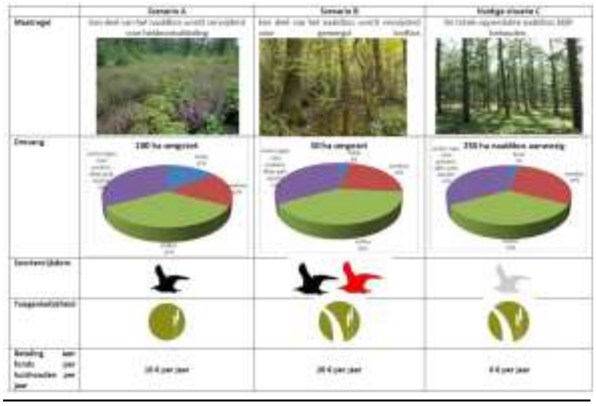
Valuation method	**Approach**	**Application to ecosystem services**	**Example ecosystem service**	**Limitations**	**Tier**[2]
Damage cost avoided	Estimate damage avoided due to ecosystem service	Ecosystems that provide storm, flood or landslide protection to houses or other assets	Coastal protection by dunes; river flow control by wetlands; landslide protection by forests	Difficult to quantify changes in risk of damage to changes in ecosystem quality.	2
Social cost of carbon	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.	Carbon storage and sequestration	Carbon sequestered and stored by protected or restored forests	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.	1
Opportunity cost	The next highest valued use of the resources used to produce an ecosystem service.	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 agricultural instead of conserved in a natural state.	Measures the cost of providing ecosystem services instead of the benefit.	1
Net factor income (residual value)	Revenue from sales of ecosystem-related good minus cost of other inputs	Ecosystems that provide an input in the production of a marketed good	Filtration of water by wetlands; commercial fisheries supported by coastal wetlands	Tendency to over-estimate values since all normal profit is attributed to the ES	2
Production function	Statistical estimation of production function for a marketed good including an ES input	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	3

Valuation method	**Approach**	**Application to ecosystem services**	**Example ecosystem service**	**Limitations**	**Tier[2]**
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.	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	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.	3
Travel cost	Estimate demand for ecosystem recreation sites using data on travel costs and visit rates	Recreational use of ecosystems	Recreational use of national parks	Technically difficult. High data requirements. Limited to valuation of recreation. Complicated for trips with multiple purposes or to multiple sites.	3
Contingent valuation	Ask people to state their willingness to pay for an ES through surveys	All ecosystem services	Biodiversity; recreation; landscape aesthetics; flood risk attenuation	Expensive and technically difficult to implement. Risk of biases in design and analysis	3
Choice modelling (choice experiment)	Ask people to make trade-offs between ES and other goods to elicit willingness to pay	All ecosystem services	Biodiversity; recreation; landscape aesthetics; flood risk attenuation	Expensive and technically difficult to implement. Risk of biases in design and analysis	3
Group / participatory valuation	Ask groups of stakeholders to state their willingness to pay for an ES through group discussion	All ecosystem services	Biodiversity; recreation; landscape aesthetics; flood risk attenuation	Risk of biases due to group dynamics	3

Box 1. Choice experiment valuation of nature restoration in Drongengoed, Belgium (De Valck et al., 2014)

To ensure the long-term survival of its most valuable and threatened habitats, the European Union (EU) is committing its Member States to develop a network of protected areas. Flanders (northern Belgium) is a highly urbanised region, where natural environments are scarce. Policy-makers are converting existing forest plantations (mostly former coniferous plantations) into natural areas to comply with the EU requirements about nature restoration and satisfy the growing demand for recreation and amenity spaces. The conversion of forest plantations into higher value nature, however, sometimes meets public opposition because it often involves clearcuts and landscape modification. Regional planning authorities are looking for case studies demonstrating which type of nature restoration is valued and thus supported by citizens. Past valuation studies show that personal, site-specific and spatial characteristics influence preferences. However, little is known about the relative importance of such factors. We conduct a discrete choice experiment to investigate preferences for nature restoration scenarios that involve forest conversion.





A mixed logit and a latent class model are estimated and the influence of socio-demographic characteristics is explored. Willingness-to-pay (WTP) estimates are elicited. Though people generally prefer the forest habitat type, our results suggest that public support exists for converting forest plantations if this contributes to increasing landscape diversity and species richness (coniferous forest plantation into deciduous forest and heathland). People in Flanders prefer also large variation in a landscape so a preference in changing the total area are less preferred than changing smaller parts. Based on our findings, we recommend small scale cuts. This in order to gently open the landscape, assist the natural regeneration process and help current species adapt to that landscape modification. The willingness to pay estimates are also transferred through a value function transfer to similar restoration projects in order to estimate the benefits of this restoration.

3.2. 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 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") (Johnston et al., 2015). 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, and indeed the MAES (Mapping and Assessment of Ecosystems and their Services[3]) process, is often needed at a regional or 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 (see Annex 2 for an overview of databases that compile existing valuation studies). 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 (e.g. the UK NEA, 2011; Hussain et al., 2011), value mapping applications (see Schaegner et al., 2013) and policy appraisals (e.g. World Bank, 2002). The use of value transfer is widespread but requires careful application. The alternative methods of conducting value transfer are described here.

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).

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.

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 (see Brander et al., 2012, for a detailed explanation of this method).

[3] http://biodiversity.europa.eu/maes

These three principal methods for transferring ecosystem service values are summarised in Table 3 together with their respective strengths and weaknesses, and the tier to which they are assigned. 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). See Brander (2013) for further guidance on value transfer methods.

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

	Approach	Strengths	Weaknesses	Tier
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	1
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)	2
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	3

4. Methods for mapping economic values for ecosystem services

The economic value of an ecosystem service is determined by its supply and demand. The supply side of an ecosystem service is largely determined by ecological processes and characteristics that may be influenced by human activities, either deliberately or inadvertently. The demand side is largely determined by the characteristics of human beneficiaries of the ecosystem services. The determinants of both the supply and demand of ecosystem services are spatially variable, which makes the estimation of ecosystem service values inherently spatial. Value mapping addresses this spatial dimension of ecosystem service valuation. Economic value mapping can be defined as the valuation of ecosystem services in monetary terms across a relatively large geographic area that includes the examination of how demand and supply vary across space. It therefore includes not only studies that produce graphical value maps but also studies that explicitly address spatial variability in values.

The estimation of accurate values for ecosystem services requires that account is taken of spatial heterogeneity in biophysical and socioeconomic conditions. Spatial factors that affect the supply of ecosystem services include among others: ecosystem area (possibly characterised by a non-linear relationships and thresholds), networks, resilience, biodiversity, fragmentation, disturbance, and accessibility. Spatial factors that affect demand for ecosystem services include: the number of beneficiaries, culture and preferences, ecosystem area, distance to the ecosystem, and the availability of substitutes and complements.[4]

Value mapping thereby reveals additional information as compared to conventional valuation, which is potentially useful for designing effective policies and institutions for maintaining ecosystem service supply. Besides communication and visualisation, value mapping makes site-specific ecosystem service values available on a large spatial scale. Spatially explicit ecosystem service value maps have specific advantages for several policy applications including ecosystem service accounting, land use policy evaluation, conservation planning, targeting land restoration activities and designing payments for ecosystem services. It allows decision makers to extract estimated values from a map or database for the locations or areas of policy interest in order to evaluate potential policy measures.

4.1. Representing economic values on maps

The representation of economic values on maps involves estimating variable combinations of supply and demand across spatial units and plotting the resulting values. The spatial units used in a value map can be land parcels (e.g. polygons representing ownership), ecosystem patches (e.g. polygons representing distinct ecosystems of different type), ecosystem units (e.g. raster grids of ecosystem type), grid cells (e.g. raster grids with land use/land cover), or beneficiaries (e.g. people plotted using residential or activity location). In most cases, spatial units are used to represent the ecosystem that supplies the ecosystem service, but mapping values by the location of beneficiaries can be useful in some decision making contexts (e.g. for representing the distributional consequences of changes in ecosystem service provision across communities; or for designing payment mechanisms for ecosystem services).

Figure 5 provides a conceptual representation of spatially variable combinations of supply and demand across nine spatial units within a mapped study area. In order to map ecosystem service values, each spatial unit is treated as a separate sub-market for the ecosystem service; variation in both supply and demand results in variation in economic value. Spatial unit 2 is characterised by both high demand and supply, and consequently high value for the ES; whereas spatial unit 3 represents the case of high demand but low supply, and consequently lower value. Spatial unit 5 represents a location with high supply but low demand, and consequently low value. Spatial unit 6 represents a

[4] See Bateman et al. (2002), Hein et al. (2006), and Schaafsma (2015) for more detailed discussions of spatial determinants of ecosystem service demand and supply.

location with demand for the ES but zero supply, and consequently zero value. Conversely, spatial unit 9 represents a situation with ES supply but zero demand, and again zero value.

Methods for mapping ecosystem service values can focus on spatial variations in supply, demand, or ideally the combination of both determinants. In general terms, bio-physical methods are used to estimate the spatially variable quantities of ecosystem services supplied (e.g. probability of flood damage, quantity of clean water, area of recreational space, tonnes of carbon stored) and economic methods are used to estimate spatially variable marginal values per unit of ecosystem service used or consumed. Mapping economic values therefore necessarily involves linking biophysical ecosystem supply maps with economic valuation methods.

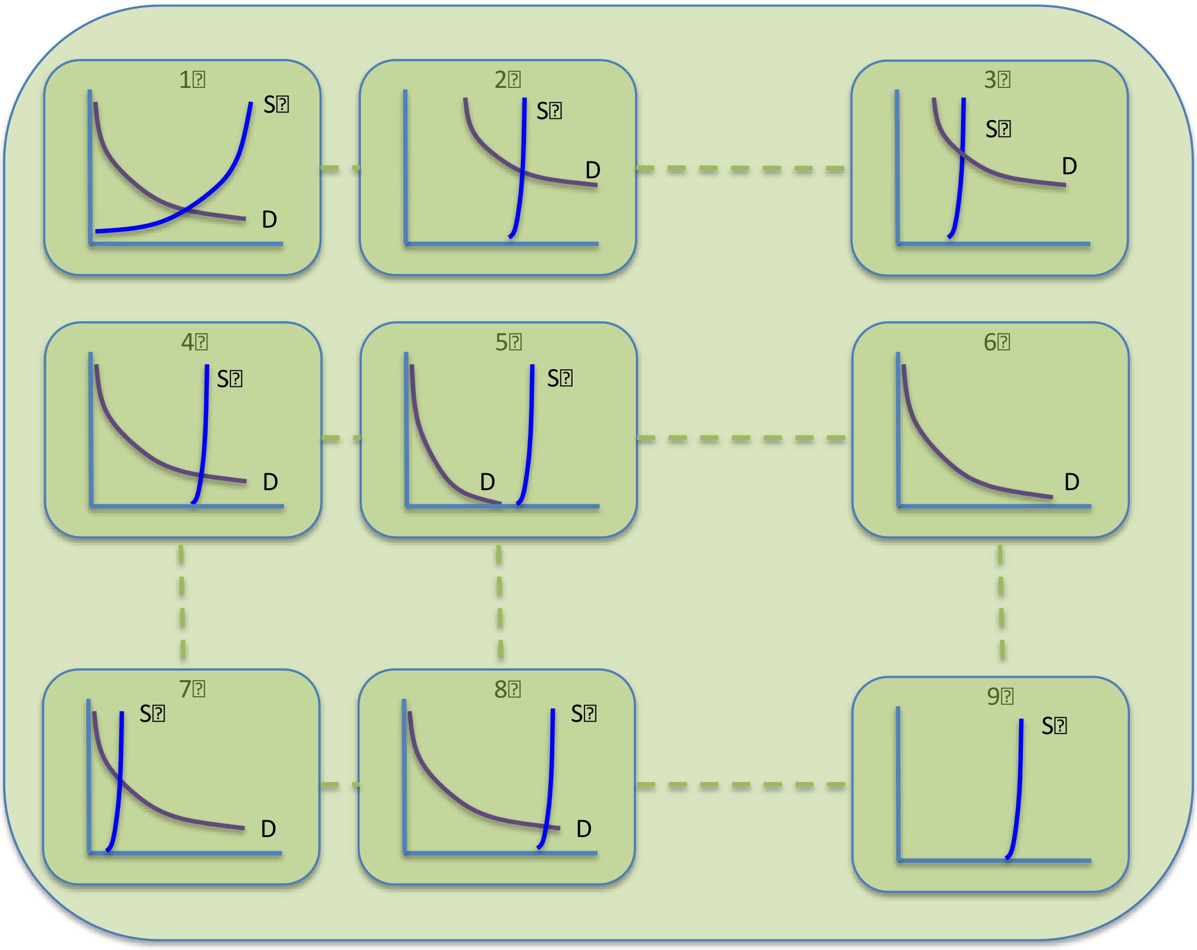


Figure 5: Conceptual representation of variation in supply and demand for an ecosystem service across spatial units within a mapped study area. Each spatial unit is a separate sub-market in which the value of an ecosystem service can vary substantially depending on location specific demand and supply.

Methodologies for biophysical mapping of ecosystem service supply are addressed in ESMERALDA report D3.3. These methods are be summarised in the following categories: 1. Spatial proxy methods; 2. Phenomenological models; 3. Macro-ecological models; 4. Trait-based models; 5. Process-based models; 6. Statistical models; 7. Ecological connectivity models; 8. State and transition models; 9. Conceptual models; 10. Integrated modelling frameworks; 11. Direct measurements.

Economic methods for estimating spatially variable ecosystem service demand, or marginal values per unit of ecosystem service supplied, are described in section 3. The process of representing these values on maps mirrors that of conducting value transfers. Following Schaegner et al. (2013), these methods can be placed into four categories:

1. **Unit value** approach: a constant value per unit of ecosystem service is applied to estimates of supply (or a constant value per unit area of ecosystem is applied to the area of ecosystem as a proxy of supply). Thus, variations in ecosystem service value across space result only from variations in supply. Unit values can be obtained from existing applications of the valuation methods reviewed in section 2. The unit value approach has been the predominant methodology used for valuing ecosystem services within the value-mapping literature (Schaegner et al., 2013).
2. **Adjusted unit values** approach: adjusts values per unit of ecosystem service across spatial units using simple variables in order to account for spatial variations in value. Typically, such variables are population density, income levels or price levels. Thereby, such adjustments respectively account for the number of beneficiaries of an ecosystem service, the effect of income levels on willingness to pay, and differences in price levels.
3. **Value function** approach: estimates spatially variable unit values across the study area using a value function, which may contain multiple spatial variables (e.g. income, household size, distance to ecosystem). A value function is typically estimated from a single primary valuation study, which may be conducted within the mapped study area (subsequent use for mapping involves spatial extrapolation of results) or outside of the mapped study (in which case the mapping involves value transfer in a strict sense). Parameter values for each spatial unit in the study area are plugged into the value function to estimate unit values vary across spatial units. Value functions can be obtained from a number of primary valuation methods including hedonic pricing, travel cost, production function, avoided damage cost, contingent valuation and choice experiment methods.
4. **Meta-analytic value function transfer** approach: also enables the estimation of unit values that vary across spatial units within the study area by applying a value function containing multiple spatial variables. In this case, however, the function is estimated from the results of multiple primary valuation studies, which increases the scope for including additional spatial variables that might not be feasible within a single primary valuation study (e.g. crowdedness, accessibility, fragmentation, scarcity).

4.2. Scaling up economic values

The currently available information on the value of ecosystem services is mostly for relatively small spatial scales (e.g. distinct individual ecosystems). Assessments of changes in ecosystem service provision at larger geographic scales, e.g. national level reporting of ecosystem services, require the “scaling-up” of value information. The term “scaling up” is used to describe the transfer and aggregation of values that have been estimated for localised changes in individual ecosystem sites to assess the value of simultaneous changes in multiple ecosystem sites within a large geographic area (e.g. country or region). Scaling-up ecosystem service values is therefore highly relevant for the MAES process and ecosystem service accounting.

At the level of individual ecosystem sites, marginal unit values for ecosystem services are likely to vary with the characteristics of the ecosystem site (area, integrity, and type of ecosystem), beneficiaries (number, income, preferences), and context (availability of substitute and complementary sites and services). The estimation of the value of ecosystem services from individual ecosystem sites therefore needs to account for these characteristics. Localised changes in the extent of an individual ecosystem may be adequately evaluated in isolation from the rest of the stock of the resource, which is effectively assumed to be constant.

When valuing simultaneous changes in multiple ecosystem sites within a region, however, it is not sufficient to estimate the value of individual ecosystem sites and aggregate them without accounting for the changes that are occurring across the stock of the resource. As an environmental resource becomes scarce, its marginal value will tend to increase. This means that ignoring the scarcity effect on the marginal values of individual ecosystem sites, as is often done in scaling up exercises, is likely to underestimate the value of the change. Valuations of changes in the stock of ecosystems across large geographic areas, such as for national ecosystem assessments or ecosystem service accounts, therefore need to account for the effects of scarcity on marginal values. Brander et al. (2009; 2012) propose an extension of the meta-analytic function transfer methodology to specifically address the challenge of scaling up ecosystem service values. The steps in this method for scaling up values for changes in across the stock of ecosystems within a region are summarised here:

1. Construct a database of primary valuation estimates for the ecosystem of interest. Standardise value estimates in terms of monetary units per unit area of ecosystem per year (e.g. € per hectare per year).
2. Estimate a meta-analytic value function for the ecosystem in question. The dependent variable in the value function is the standardised value (defined in step 1). The estimated value function should include explanatory variables that capture study site characteristics (i.e. size, services provided); context characteristics (i.e. abundance of the ecosystem in the region); socio-economic characteristics of beneficiaries (i.e. size of relevant population, income); and study characteristics (i.e. valuation method used to produce each primary value estimate included in the meta-analysis).
3. Construct a database of "policy sites" (ecosystems) in the region of interest using a Geographic Information System (GIS) to include information on the variables in the meta-analytic value function (i.e. size of each ecosystem site, abundance of the ecosystem within the vicinity of each site, population in the vicinity of each site, and income level of the population). The database should contain values for each of these variables at pre- and post-change states for each policy site (e.g. at two different points in time or for two different policy scenarios).
4. For each policy site, estimate the marginal value per hectare at the pre-change and post-change levels. This is done by substituting in pre- and post-change variable values into the meta-analytic value function. Calculate the average of the pre-change and post-change marginal values per hectare for each site in order to approximate the average per hectare value of the area that is lost.
5. Multiply the value per hectare for each site by the change in area for each site. This gives an estimate of the value of the change in size of each ecosystem site.
6. Aggregate the estimated changes in value of individual policy sites to the regional or national level. This gives the annual value of the change in ecosystem service provision at that scale.

4.3. Example value mapping applications

In this section, we provide a selection of value mapping applications that illustrate alternative approaches to mapping ecosystem service values. Box 2 presents the mapped economic values of five provisioning and regulating ecosystem services provided by the Serra de S. Mamede national park in

Portugal; unit values for each ecosystem service are estimated using market prices and value transfer methods. Box 3 presents a national map of the value of agricultural production in Spain (estimated using market prices) and its spatial correspondence with ecological values (estimated using socio-cultural valuation methods). Box 4 presents some of the results of a spatially explicit scenario analysis for the UK National Ecosystem Assessment, which mapped a wide range of market and non-market ecosystem services. Box 5 presents mapped values for changes in landslide risk in Adjara autonomous republic of Georgia; avoided damage costs from landslides under alternative scenarios are estimated and mapped for individual villages. Box 6 summarises the results of a study that maps changes in the value of coral reef recreation in Southeast Asia using a predictive model of recreational visits and a meta-analytic value transfer approach to estimate spatially variable values per visit. Box 7 presents a global map of changes in human welfare resulting from projected changes in freshwater quality; spatially variable values are estimated using a meta-analytic value function for water quality.

Box 2. Mapping the economic value of ecosystem services provided by the Natural Park of Serra de S. Mamede (PNSSM), Portugal (Marta-Pedroso et al., 2014).

Based on land-use and cover, a mapping approach for economic value was implemented (Table 1) and the outcomes presented in Figure 1.

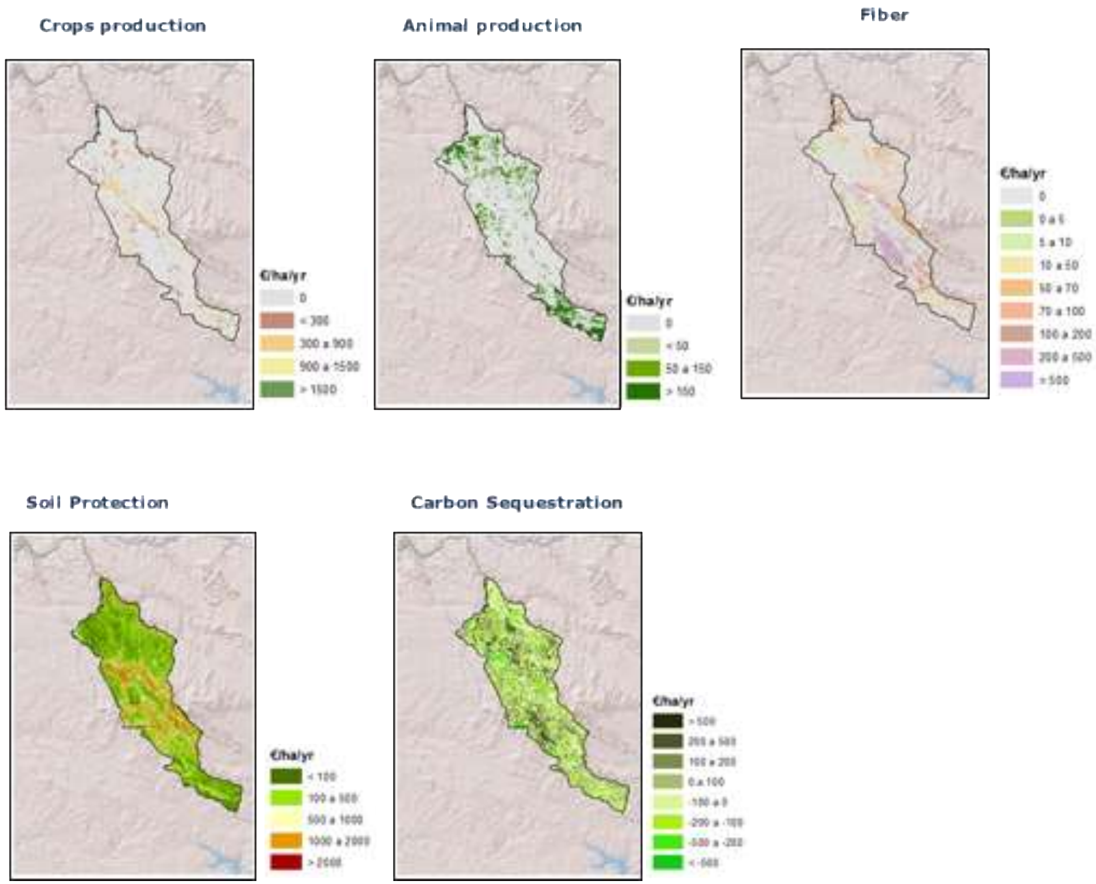


Figure 1 Economic value of ES considered in the PNSSM.

Table 1 Overview of selected ES, ES indicators and value mapping approaches (Tier 1) used to estimate the economic value of the PNSSM.

Biophysical Mapping	Economic Value Mapping (€.ha^{-1}.yr^{-1})

Section	Division	Group	Class	ES Specification	Indicator unit	Valuation Method	Description
Provisioning	Nutrition	Biomass	Crops	**Crops production**	$ton.ha^{-1}.yr^{-1}$	Market Price	Standard Gross margin (SGM) of each crop. SGM for each land use class was estimated as $SGM_{LUC\ j} = \Sigma A_i SGM_i / \Sigma A_i$,, where Ai represents the area of crop $_i$ in the land use class (LUC) $_j$. SGM_i and A_i were obtained from official statistics.
			Livestock	**Stocking rate**	$LU.ha^{-1}.yr^{-1}$	Market Price	Standard Gross margin (SGM) of pastures typologies. An average LU (livestock unit) for each different type of pasture was considered.
	Materials	Biotic materials	Non-food vegetal fibers	**Timber production**	$m^3.ha^{-1}.yr^{-1}$	Market Price	ANPV (Annualized Net Present Value) of timber given the Investment Return Analysis for the species of interest provided by Machado & Louro (2009). For mixed LULC classes (i.e., when more than one forest species was present), the value was weighted according to an estimated cover percentage per species.
Regulating	Regulation of physico-chemical environment	Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations	**Carbon sequestration/ emission**	$tonCO_2.ha^{-1}.yr^{-1}$	Value transfer	Unit Value: 79,5€/ton based on Stern (2006) social cost of carbon estimations. Amount of carbon sequestered/emitted estimated in each pixel by considering the land use transitions observed between 1990-2006) was multiplied by the unit value.
	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates	**Avoided erosion**	$ton.ha^{-1}.yr-^{-1}$	Value Transfer	Unit value: (4.75 €. $ha^{-1}.yr^{-1}$) based on Marta-Pedroso et al. (2007). The avoided erosion value estimated in each pixel was multiplied by the unit value.

Notes: Economic values adjusted using consumer price index when appropriate; ES Classification accordingly to CICES 4.3; Machado, H.& LOURO, G. – Análise de Rentabilidade das Áreas Submetidas a Regime Florestal. In Actas do 6.º Congresso Florestal Nacional. Ponta Delgada: Sociedade Portuguesa de Ciências Florestais, 2009. Pp. 883-889; Stern, N., 2006. Executive summary (full). Stern Review Report on the Economics of Climate Change. http://webarchive.nationalarchives.gov.uk/20130129110402/http://www.hm-treasury.gov.uk/d/Executive_Summary.pdf; Marta-Pedroso et al. (2007). Incorporating the benefits supplied by soil in agri-environmental policy efficiency analysis: the case of the Zonal Program of Castro Verde (Portugal). Soil & Tillage Research 97: 79–90

Box 3. Integrating economic and ecological values of agricultural ecosystem services in Spain (Santos-Martín F. et al., 2016)

This study present a nationwide study of Spanish agro-ecosystems in which the spatial distribution of food provisioning services indicators has been mapped. First we quantify and mapped the value of Spanish agricultural provisioning services expressed in biophysical (t/ha/yr) and monetary (€/ha/yr) units. Secondly we mapped "High Nature Value farming areas" in Spain, with the aim of identify important and valuable habitats for species with a high ecological value. Finally we explore the spatial correlations between the economic and ecological value with the objective to identify those areas with high values on both dimensions that should be considered as priority for landscape management intervention. These results show how current land-use management in Spain is creating a landscape dichotomy between areas with agricultural practices with a high ecological value and other land-uses managed under intensification practices that is creating a clear negative effect on the maintenance of essential functions to maintain the good condition of majority of Spanish agro-ecosystems.

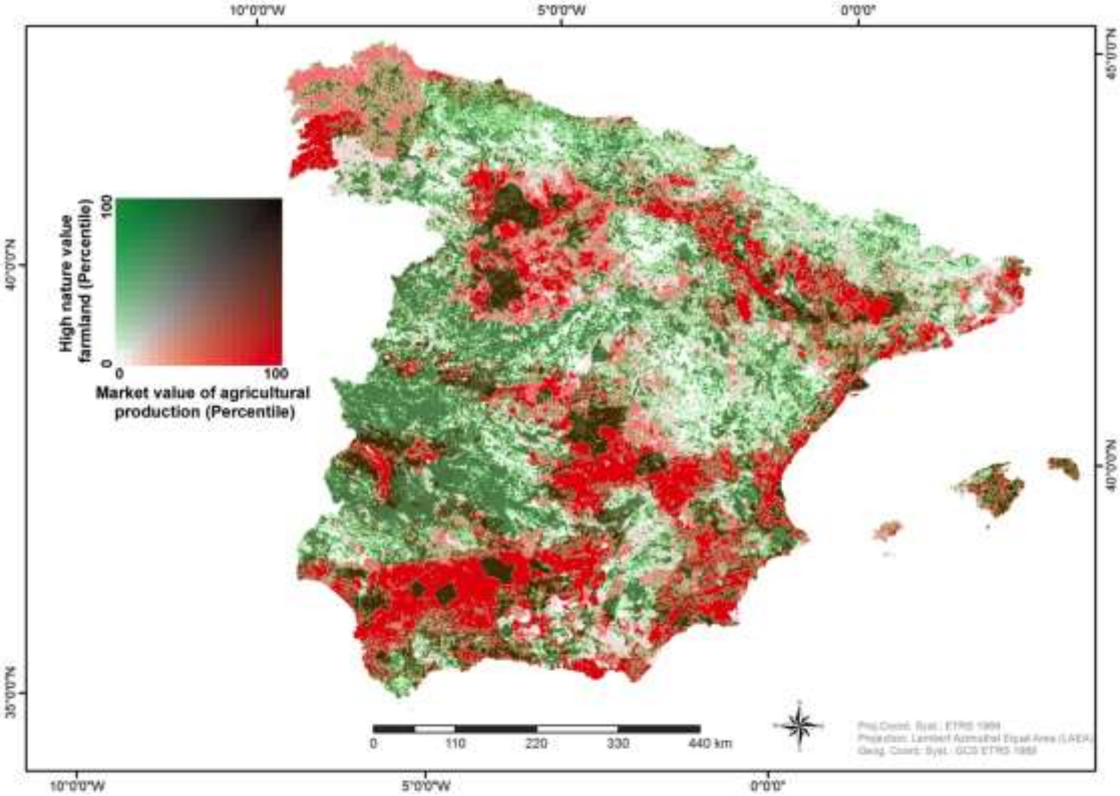


Figure 1: Superimposition of the spatial representation of the economic value of the agricultural production of Farming Areas with High Nature Value in Spain. In red: areas with a high economic value and a low ecological value; In green areas with a high ecological value and a low economic value; in black: areas with a high economic value and a high ecological value.

Box 4. Scenario, mapping and valuing ecosystem services in the UK National Ecosystem Assessment

The UK National Ecosystem Assessment (NEA) provides a good example of bringing together the development of plausible future scenarios using different socially developed story lines, modelling the impacts of these to understand biophysical changes and then building on this to understand changes in associated monetary and non-monetary values. The scenarios used had similar foundations e.g. aging populations and declining global resource availability, but were augmented with different emphases on development choices ranging from environmental awareness and ecological sustainability to national self-sufficiency and pursuit of economic growth irrespective of the wider implications. To compare the outcomes, a range of market and non-market ecosystem service benefits were valued. However, the authors felt it was inappropriate to value biodiversity, citing particularly controversies around the robustness of valuing non-use existence values, and hence the analysis across the scenarios was presented in a range of ways. This included, for example, ranking scenarios in terms of their economic value, but excluding scenarios which led to a decline in biodiversity; and also by presenting maps of the market and non-market values alongside the estimated impacts of the chosen indicator of biodiversity (bird diversity) as illustrated in the maps below. The links between the economic value and the biophysical underpinning is clear for example in the maps of changes in urban green space value (fourth from left), which are focused around major cities. Being unable to value biodiversity means that cost benefit analysis (CBA) alone cannot be used to judge between scenarios, but as the UK NEA itself points out CBA is simply an informational input to the decision making process, using more of the supporting information by comparing across maps of values (monetised or not) more of the trade-offs and complementarities become visible.

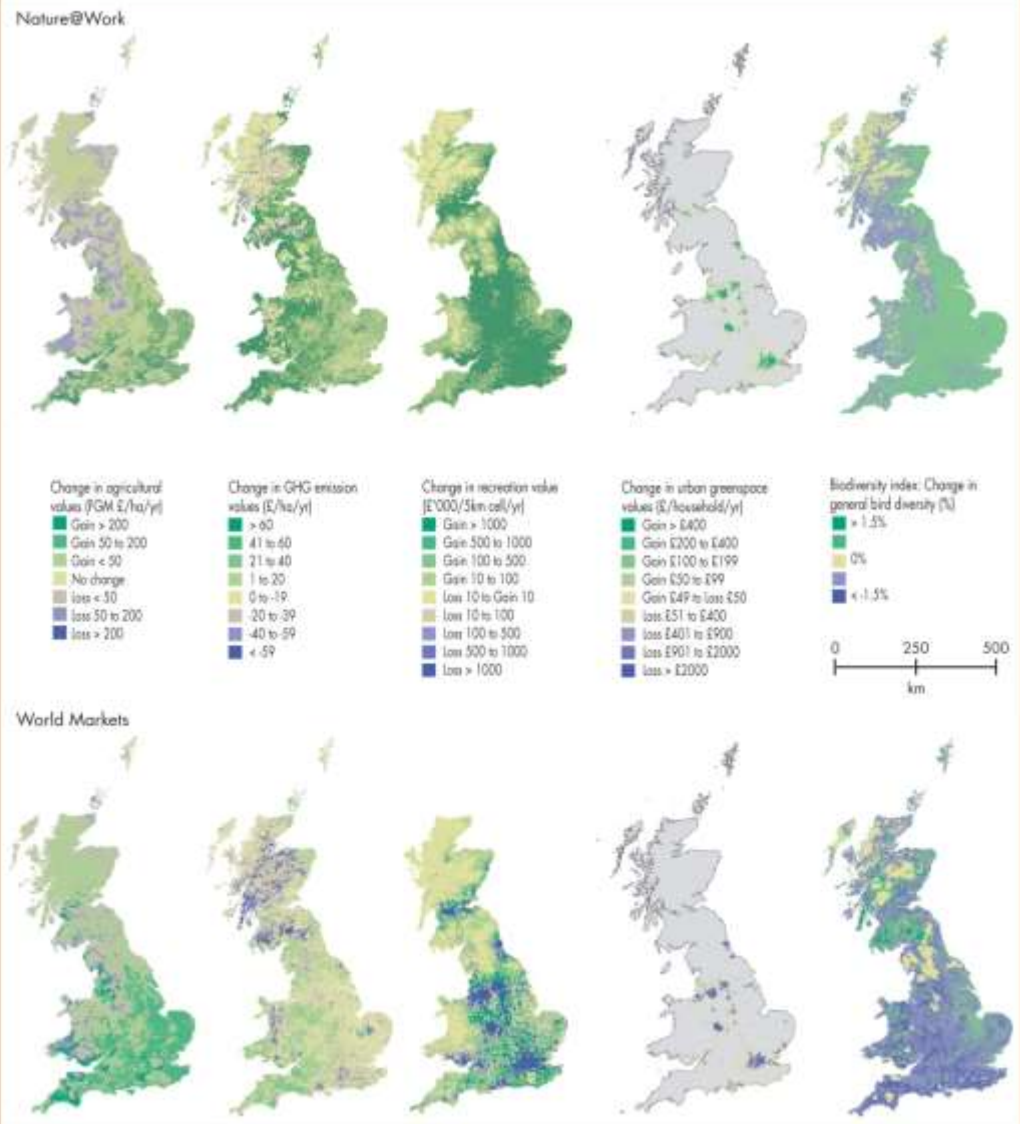


Figure 22 Spatial distribution of changes from the baseline in five ecosystem service related goods (agricultural production (FGM: Farm Gross Margin); greenhouse gas (GHG) emissions; recreation; urban greenspace; biodiversity) under the *Nature@Work* scenario (upper row) and the *World Markets* scenario (lower row) for Great Britain.

Box 5. Mapping the economic value of landslide regulation by forests in Georgia (Brander et al., 2017)

This study develops a methodology for mapping the value of forests in mitigating landslide risks and applies it at a regional scale for the Adjara Autonomous Republic, Georgia. Like the rest of Georgia, Adjara is mostly mountainous and its steep slopes are prone to landslides. By mapping the value of landslide regulation by forests, the study aims to deliver information to support political and administrative decision-making regarding long term forestry management.

The general methodological framework for quantifying the economic value of landslide regulation as an ecosystem service provided by forests is represented in Figure 1. The approach involves first developing land cover maps for a baseline scenario and alternative policy scenarios. Spatial data on land cover is then combined with a bio-physical model of sediment retention and export to estimate spatially variable rates of sediment export as a proxy measure of landslide susceptibility. In the case study application we use the InVEST model to quantify changes in sediment export resulting from changes in land cover. The data on sediment export is combined with spatially referenced historic data on the frequency of landslide damage to houses and used to estimate a predictive function for landslide damage. To model changes in the frequency of landslide damages under alternative policy scenarios, spatial data on sediment export under each future scenario is fed into this function to predict changes in landslide damage frequency. The costs of predicted damages are estimated using data on compensation payments to impacted households.

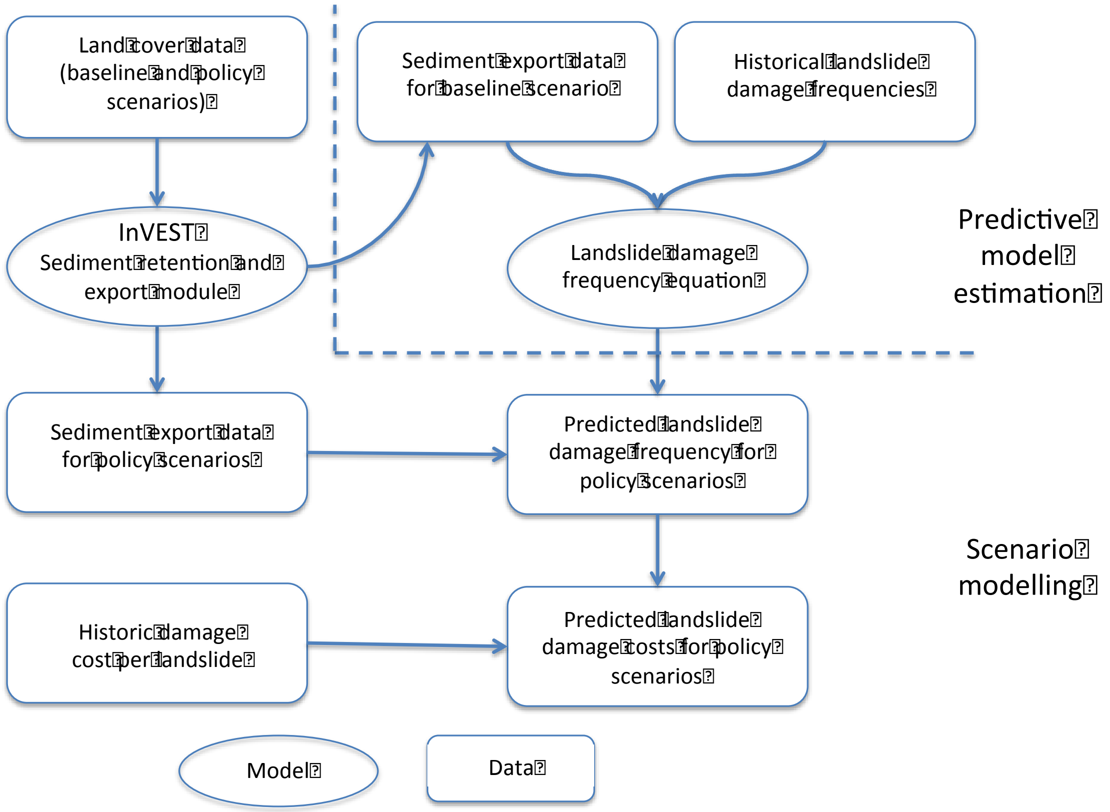


Figure 1. Methodological framework for the valuation of landslide damages

The case study application for Adjara developed a baseline land cover scenario for the period 2015-2035 and two alternative future scenarios representing increased forest degradation and restoration storylines respectively. Land cover changes under each scenario are modelled in a GIS and the resulting changes in sediment export are modelled using the InVEST tool. Changes under each scenario are assessed at two points in time (2020 and 2035) in order to enable the evaluation of short term and long term impacts on landslide damages. The mapped changes (relative to the baseline) in landslide damage under each alternative scenario and time step are represented in Figure 2.

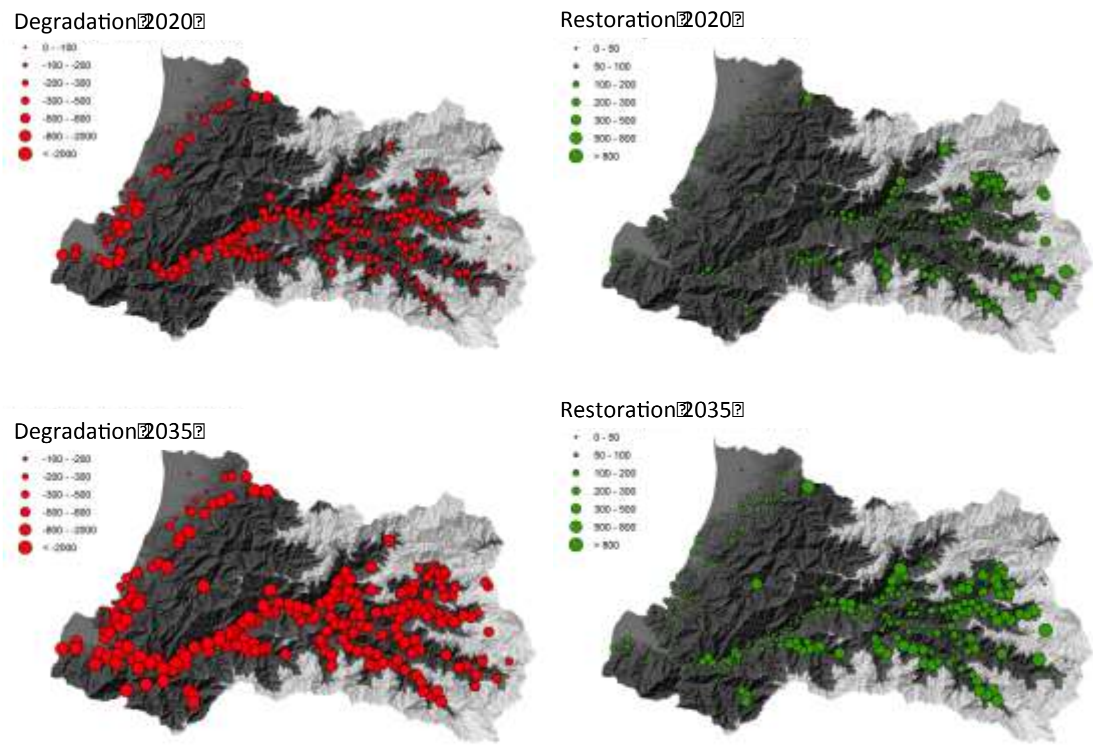


Figure 2. Spatial distribution of annual change in landslide damages (US$/year)

Box 6. Coral reef recreation values in Southeast Asia (Brander et al., 2015)

This study illustrates the process of mapping ecosystem service values with an application to coral reef recreational values in Southeast Asia. The case study provides an estimate of the value of reef-related recreation foregone due to the decline in coral reef area under a baseline scenario for the period 2000-2050. This value is estimated by combining a visitor model, meta-analytic value function and spatial data on individual coral reef ecosystems to produce site-specific values. Following Sen et al. (2014), the selected methodology uses a combination of a validated model for visits to coral reefs and a meta-analytic value function to estimate the value per visit. The methodology involves the following steps:

1. Estimate a model of recreational visits to individual coral reef sites. The visitor model relates the number of visits per day to the site and context characteristics of each coral reef ecosystem such as degree of siltation or fishing damage.
2. Estimate a value function for coral reef recreation through a meta-analysis of existing monetary estimates. The value function relates the value per visitor day to the characteristics of the ecosystem and its surroundings.
3. Develop a database of coral reef ecosystems in Southeast Asia containing information on the variables included in the visitor model and value function estimated in steps 1 and 2.
4. Develop a baseline scenario for the change in the quality and spatial extent of coral reef ecosystems in Southeast Asia for the period 2000-2050. This baseline scenario is spatially variable to reflect variation in location-specific pressures on coral reef ecosystems.
5. Combine the models and data generated in steps 1 through 4 to produce estimates of the value of the loss in coral reef-related recreation under the baseline scenario. This

approach allows the estimation of spatially variable, site-specific values that reflect the characteristics and context (e.g. pressure or threat) of each coral reef.

Values are mapped in order to communicate the spatial variability in the value of coral reef degradation (see Figure 5). Although the aggregated change in the value of reef-related recreation due to ecosystem degradation is not high, there is substantial spatial variation in welfare losses, which is potentially useful information for targeting conservation efforts.

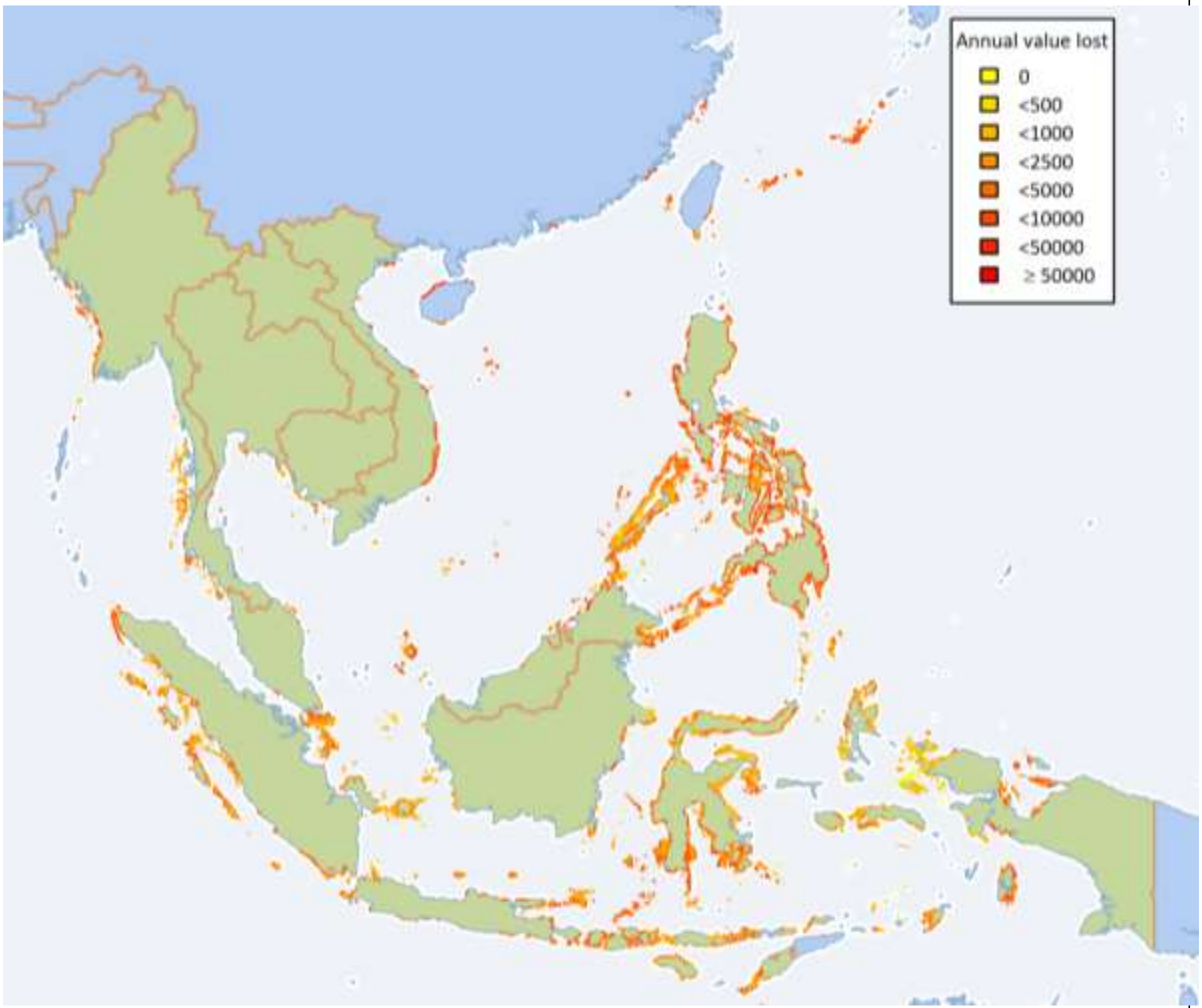


Figure 1: Loss in the annual value of coral reef-related recreation in 2050 due to business-as-usual coral reef degradation.

Box 7. Global value of changes in water quality 2000-2050 (Hussain et al., 2011)

This study combines output data from a validated model (IMAGE-GLOBIO) with a meta-analytic value function to estimate the economic value of global changes in water quality under a business-as-usual scenario for the period 2000-2050. The analysis is performed at the resolution of 50km grid cells. The supply of ecosystem services from water bodies (rivers and lakes) is implicitly modelled within the meta-analytic value function. The results of this value transfer application are mapped in order to communicate the spatial distribution of benefits (losses) derived from improvements (declines) in water quality (see Figure 8). In this application, the spatial units used to map changes in value are beneficiaries (households aggregated within 50km grid cells) rather than the rivers or lakes providing the ecosystem services.

Figure 1: Value map for changes in water quality 2000-2050 (Annual willingness to pay; USD; 2007 price levels)

5. Economic assessment methods

5.1. Introduction to economic assessment methods

Economic assessment methods are used for structuring information on the value of ecosystem services into decision making, often in combination with other forms of information. It is important to recognise that the economic assessment methods reviewed in this report are each applicable to different decision contexts. The choice of which assessment method to use will largely be determined by the type of decision problem and the availability of relevant information. Table 5 provides a summary of each economic assessment method with a description of its application, strengths, weaknesses and an indication of the tier to which it is assigned.

Table 5. Summary of economic assessment methods for ecosystem services

Economic assessment method	Application	Strengths	Weaknesses	Tier
Cost-Effectiveness Analysis	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	1
Cost-Benefit Analysis	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	3
Multi-Criteria Analysis	Used to rank alternative investments and policies	Allows the inclusion of effects that cannot be expressed in monetary terms	Heavily reliant on the subjective judgement of the analytical team	2

Economic assessment method	Application	Strengths	Weaknesses	Tier
Ecosystem Service Accounting	Provides a structured way of measuring the economic significance of ecosystem services that is consistent with existing macro-economic accounts	Consistent accounting rules enable the direct comparison of ES economic contribution over time and with other parts of the economy	Methodological challenges to include highly important ecosystem services within the existing accounting framework (e.g. cultural services)	3
Corporate Ecosystem Service Review	Supports private sector decision-makers to manage business risks and opportunities arising from their company's dependence and impact on ecosystems	Flexible methodology allows firms to tailor assessments to needs	Challenge to integrate ecosystem service assessments into core business decision making	2

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 protected area might involve costs in terms of the purchase of land, compensation of local communities, and on-going maintenance and enforcement costs; and benefits in terms of biodiversity conservation, recreational use and improved watershed services. 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. Economic methods for assessment, evaluation or appraisal of complex decision contexts provide systems for structuring the information and factors that are relevant to a decision.

There are a number of economic assessment 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. To understand the differences between economic assessment methods, we describe the procedural steps of each approach, which are often comparable yet differ in subtle ways.

- For decisions that involve selecting between options to achieve a single specific goal (e.g. meeting a specified ecological standard, supplying a specified quantity of clean water, sequestering a targeted quantity of carbon) and where all costs can be expressed in monetary terms, the **cost-effectiveness analysis** (CEA) method can be used. This approach therefore does not involve any assessment of what the benefits are of meeting the objective but only compares alternative options in terms of their costs.
- When all the impacts of alternative options can be quantified in monetary terms, the most common economic assessment method is **cost-benefit analysis** (CBA). This assessment method involves summing up the value of the costs and benefits of each option and comparing options in terms of their net benefits (i.e. the extent to which benefits exceed costs).
- In the situation that the relevant criteria (costs and benefits) to the decision cannot be expressed in monetary values, but can only be expressed in other units or in qualitative terms

(i.e. impacts can be ranked in order of importance), **multi-criteria analysis** (MCA) is a useful assessment method.

- **Ecosystem service accounting** is a structured way of measuring the economic significance of nature that is consistent with existing macro-economic accounts. The general aim of ecosystem service accounting is also to highlight and quantify the importance of ecosystem services to society and enable direct comparisons with other parts of the economy.
- **Corporate ecosystem service review** is a structured methodology that helps private sector decision-makers to proactively develop strategies to manage business risks and opportunities arising from their company's dependence and impact on ecosystems.

It should be noted that CEA, CBA and MCA are general economic assessment methods (i.e. not ecosystem service specific) that can be applied to help select between alternative investments, projects and policies. In this report, the focus is on supporting decision-making regarding ecosystem services. Although the main steps in the assessment methods remain relevant, 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. EU Biodiversity Strategy Target 2 to restore at least 15% of degraded ecosystems[5]), establish protected areas, or target financial incentives to change the behaviour of land 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.

5.2. Cost-Effectiveness Analysis

Cost-effectiveness analysis (CEA) involves identifying the lowest cost option to achieve a given objective.[6] 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 clean water, or sequestering a targeted quantity of carbon) and for which all costs can be measured in monetary terms. The use of CEA is required in river basin management plans under the EU Water Framework Directive[7].

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.

[5] http://ec.europa.eu/environment/nature/biodiversity/comm2006/2020.htm

[6] 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.

[7] http://ec.europa.eu/environment/water/water-framework/index_en.html

This approach therefore 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). CEA of policy targets for ecosystem service provision are likely to require mapped inputs given the underlying spatial variation in determinants of both ecosystem services and the costs of supply (e.g. the opportunity costs of alternative land uses).

In practice, this economic assessment method is not frequently used in the context of managing ecosystem services due to the complex and multifunctional nature of their 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. Crossman and Bryan (2009) provide an example of how a cost-effectiveness analysis of meeting a specified planning target for ecological restoration requires a spatial analysis of both the costs and benefits resulting from alternative land use allocations. The multiple benefits from ecological restoration and the many relevant secondary policy targets mean that it is not applicable to address the land allocation decision in terms of meeting a single target at minimum cost. The assessment therefore extends beyond a CEA and assesses both costs and benefits.

5.3. Cost-Benefit Analysis

Cost-benefit analysis (CBA) is the most commonly used economic assessment method for evaluating and comparing investments, projects and policies. There is a call for the use of CBA in the appraisal of investments under the Cohesion Policy 2014-2020 (European Commission 2014).

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 1. These steps are described below:

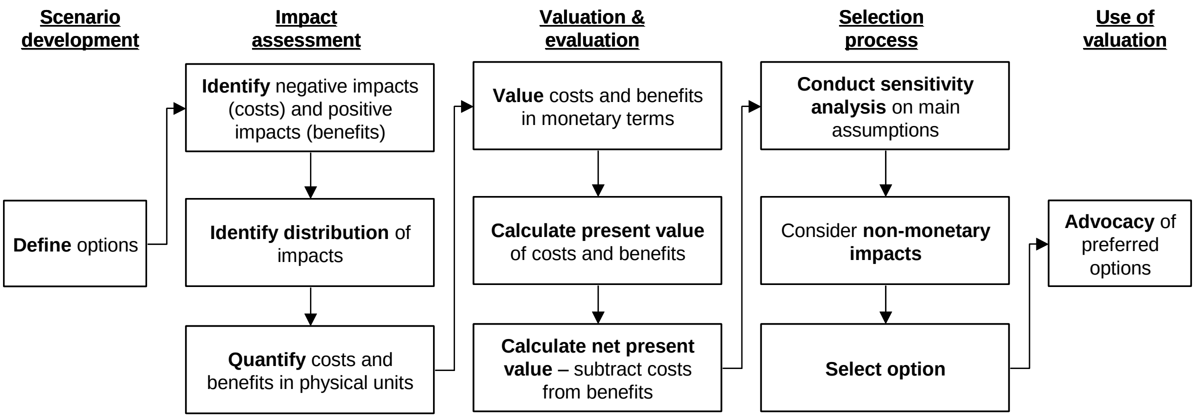


Figure 6: Methodological steps in cost-benefit analysis (source: Brander and van Beukering, 2015)

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. For performing this last important step, the Esmeralda project develops a multi-tiered flexible method for mapping and quantifying the impact on ecosystem services in biophysical units.

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. A summary of these methods is provided in Annex 1. After estimating annual values, the time-series of costs and benefits are converted to present values (PV), which involves discounting and summing values that occur in future years.

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.

Box 8. Cost-Benefit Analysis of expanding marine protected areas (Brander et al., 2015)

This study provides an example application of a spatial CBA that estimates the net benefits of expanding global marine protected areas (MPAs) to 10% and 30% coverage of total marine area. The study developed a set of six mapped scenarios for the global expansion of MPAs (see Figure 2). The scenarios vary along two dimensions: 1. the coverage of MPAs as a proportion of total marine area; 2. the characteristics of target locations for MPAs in terms of biodiversity and degree of human impact.

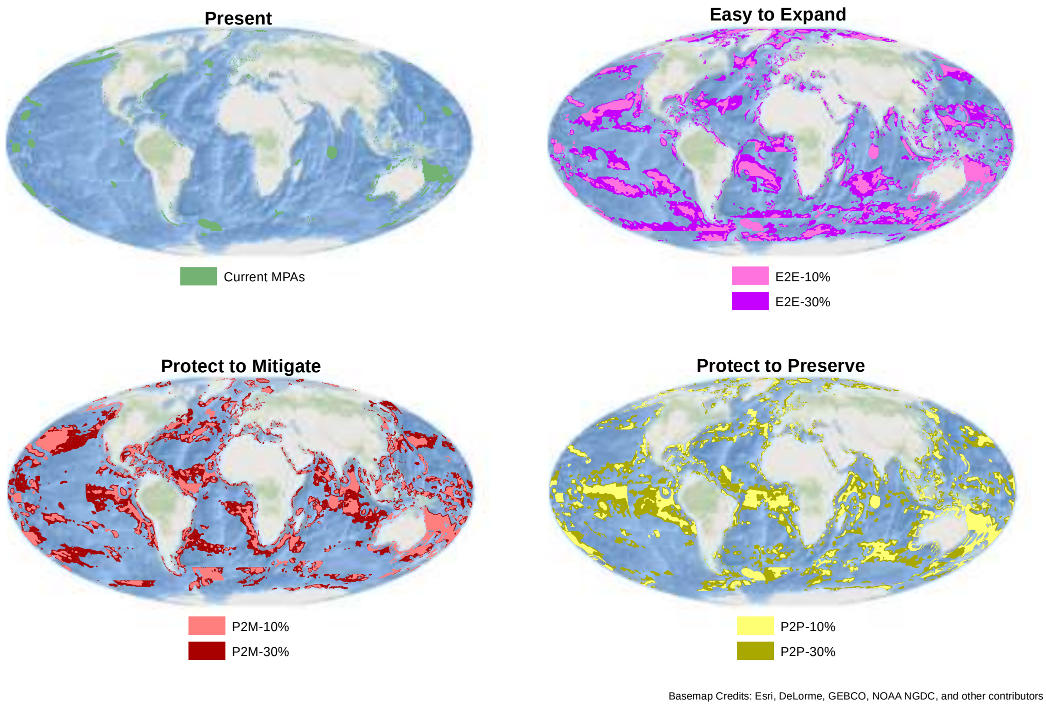


Figure 1. Current and future global distributions of marine protected areas

The methodological framework for the CBA follows that of Balmford et al. (2011), Bateman et al. (2011), Hussain et al. (2011) and Brander et al. (2012) and is represented in Figure 1. In particular it incorporates spatially explicit estimations of bio-physical effects, benefits and costs.

The results of the cost-benefit analysis show that all six scenarios for expanding MPAs to 10% and 30% coverage are economically advisable. The ratios of benefits to costs are in the range 3.17 – 19.77. In the case of the scenario that achieves 10% coverage of total marine area and targets areas with high biodiversity and low human impact, each dollar invested yields a return of around 20 dollars in benefits.

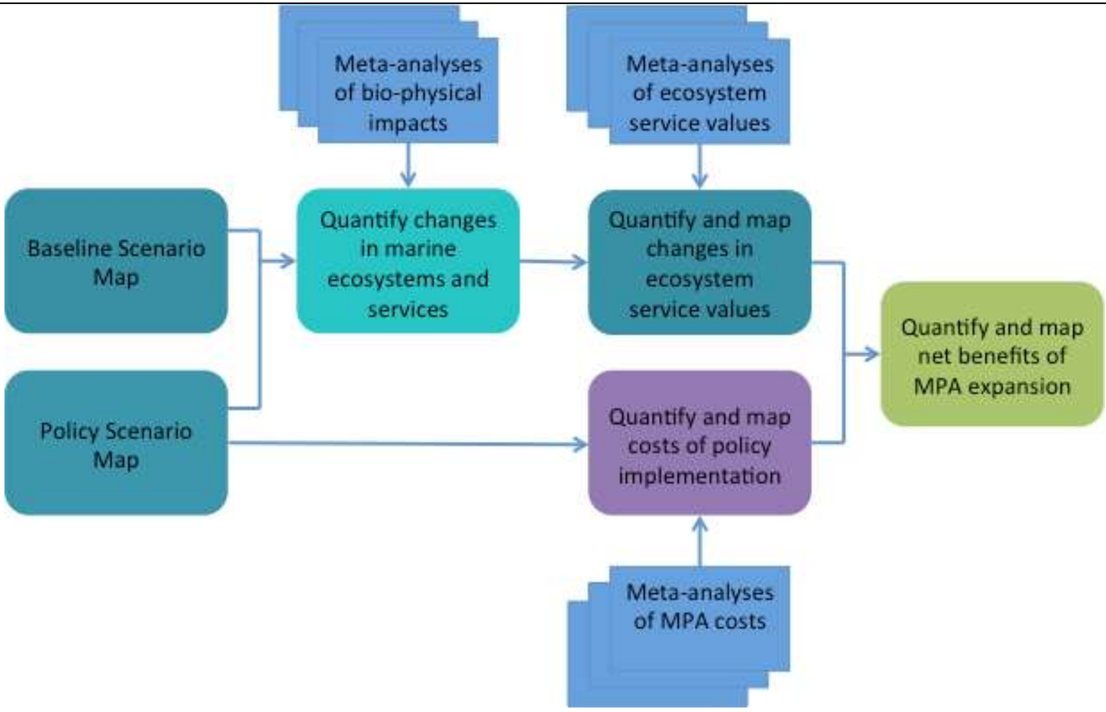


Figure 2. Methodological framework for assessing the net benefits of expanding marine protected areas. Adapted from Figure 2, Balmford et al. (2011); and Figure 2, Hussain et al. (2011).

Box 9. Example application of a spatial Cost-Benefit Analysis, River Schelde, Belgium (Broekx et al. 2011)

Major infrastructure works were planned in the Scheldt estuary, flowing from Belgium into the Netherlands, including the deepening of the fairway to the harbour of Antwerp and complementary measures to protect the land from storm floods coming from the North Sea.

A Cost-Benefit Analysis was carried out, taking into account the value of ecosystem services using a number of valuation methods. In addition to technical measures such as a storm surge barrier and dikes, two types of floodplains were considered: a system where the existing land use is maintained (mostly agriculture) and a system with controlled reduced tide that delivers a large number of ES.

Regulating services were quantified through the OMES-model. This process based ecosystem model was developed for the Scheldt estuary in order to study the possible impact of different water management strategies on the ecosystem. This model was based on a monitoring program for all major groups (plankton, benthos, avifauna, fish, and littoral vegetation), carried out by different universities and institutes, and simulated major ecosystem processes, such as the C, N and P cycles. The OMES-model makes distinctions between the impact of riverine wetlands in the fresh water, brackish and salt zone of the river. The value was estimated through replacement costs and avoided costs.

The flood control service was quantified by a large hydrodynamic model. Based on land use data, damage factors and replacement values for houses, household furniture, roads, industry, crops and other damage categories the flood damages in the inundated area were estimated. A contingent valuation study was performed to value the recreational value of new floodplains.

Results of the cost benefit analysis show that an intelligent combination of dikes and floodplains can offer similar safety benefits, but far more co-benefits at lower costs compared to more drastic measures such as a storm surge barrier near Antwerp. The hydrodynamic modelling also indicated that floodplains are necessary to ensure safety levels in the longer term in the Scheldt basin. Merely dike heightening mainly causes a shift in flooded areas but does not suffice to importantly reduce future flood risk. Additionally results showed that the benefits of the controlled reduced tidal areas (RTA) mostly exceed the benefits of the controlled inundation area (CIA) with agricultural use.

The Dutch and Flemish government approved an integrated plan consisting of the restoration of approximately 2500 ha of intertidal and 3000 ha of non-tidal flooding areas, the reinforcements of dikes and dredging to improve the fairway to Antwerp.

Table 1: Alternative options for flood protection in the Cost-Benefit Analysis (phase 1: different measures, phase 2 optimisation)

	Phase 1					Phase 2
	Storm surge barrier	Over-schelde	Dykes (*340km*)	Floodplains (*CIA, 1800 ha*)	Floodplains (*RTA, 1800 ha)*	Floodplains (*1325 ha) +* dykes (*24 km*)
Investment and maintenance costs	387	1.597	241	140	151	132
Loss of agriculture				16	19	12
Flood protection benefits	727	759	691	648	648	737
Ecological benefits				8	56	9
Other impacts:						
- shipping	-1					
- visual intrusion				-3	-3	-5
Total net benefits	339	-837	451	498	530	596
Payback period (years)	**41**	**/**	**27**	**17**	**14**	**14**

All figures are net present values in million Euro 2004, based on central estimates for economic growth and discounting (4%). Non-use values for nature development are not included in the figures.

5.4. 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 4.

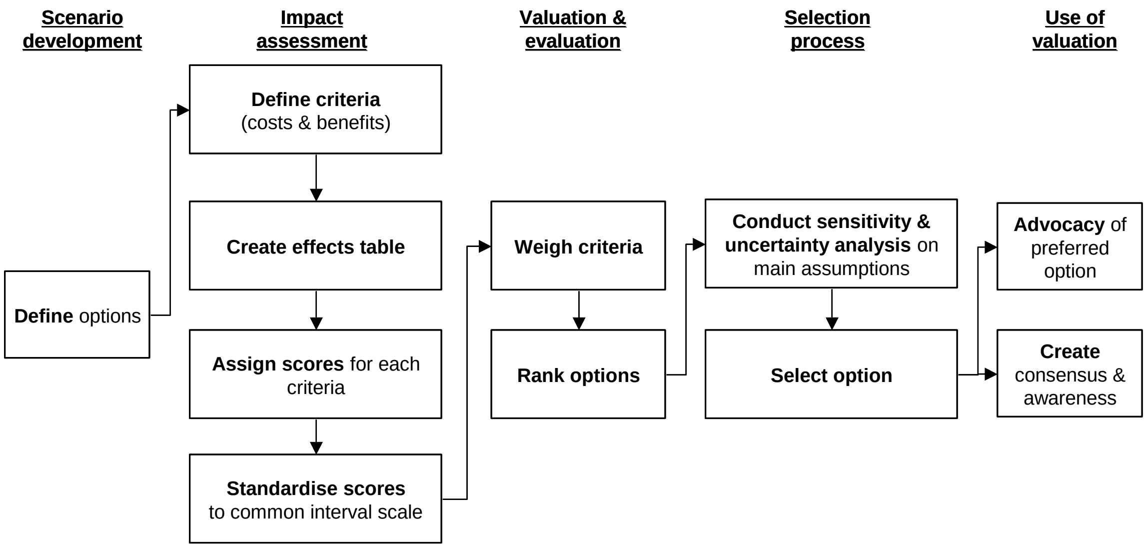


Figure 7: Methodological steps in multi-criteria analysis (source: Brander and van Beukering, 2015)

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.[8]

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

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

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.

Box 10. Spatial Multi-Criteria Analysis of habitat restoration in the River Frome catchment, Dorset, England (Newton et al., 2012)

This study provides an example of a spatial MCA of ecosystem restoration options for potential landscape-scale habitat restoration in the catchment of the River Frome in Dorset, England. The analysis involved mapping 8 ecosystem services, four of which were quantified in monetary terms using market prices (carbon storage, arable crops, livestock and timber) and four that were qualitatively assessed using a survey of stakeholders and a ranking approach (flood risk mitigation, aesthetic, recreational and cultural value). Maps were produced for each ecosystem service and habitat restoration scenario by estimating values according to land cover type. The costs of restoration were estimated as capital cost of habitat establishment and a maintenance cost per hectare; and the opportunity cost of ecosystem services negatively affected by restoration (i.e. arable crops and timber). A 10-m grid cell raster map was generated for each criterion (ecosystem service) for each of the scenarios, and all criterion maps were combined in a spatial MCA using a weighted-sum method. The results if the MCA consistently ranked restoration scenarios above a non-restoration comparator, reflecting the increased provision of multiple ecosystem services. However, restoration costs consistently exceeded the market value of ecosystem services.

5.5. Ecosystem Service Accounting

5.5.1. Introduction to ecosystem service accounting frameworks

Ecosystem service accounting frameworks aim to provide a structured way of measuring the economic significance of nature that is consistent with existing macro-economic accounts. They can help to identify trends and drivers of ecosystem change within the wider economy and society. By linking to the System of National Accounts (SNA) they can provide comprehensive, integrated and consistent data sets to support national decision-making. Ecosystem service accounting is part of Action 5 of the EU Biodiversity Strategy, requiring Member States to “promote the integration of these [ecosystem service] values into accounting and reporting systems at EU and national level by 2020”.

This section provides a review of selected ecosystem services accounting initiatives in Europe and elsewhere. The descriptions of the initiatives focus on operational details (agency, timeframe, ecosystems, ecosystem services, goals) and valuation methods (general methodology, specific method per ecosystem service).

There are a number of on-going initiatives that aim to develop recommendations for integrated natural capital accounting and the incorporation of ecosystem service values in national accounts. These initiatives are at various stages of development and closely linked to already existing satellite accounting systems around the core SNA in several countries, focusing primarily on provisioning services such as timber, and natural capital such as subsoil minerals. An important question is to what extent ES can be fully integrated into the core SNA or included as satellite accounts around the SNA, either in physical or monetary terms. The approach taken will (or should) ultimately depend on the question one would like to see answered. The SEEA guidance on ecosystem accounting encompasses a broad description of the conceptual framework, which includes discussion on the scope and purpose of the accounts along with the proposed accounts, the classification of ecosystem services, the definition and measurement for the ecosystem accounting units and the valuation and recording methods of physical and monetary flows and stocks (United Nations Statistical Division, 2012).

An important issue for accounting is the distinction between ecosystem services whose values are already implicitly accounted for in conventional SNA (e.g., pollinators to agricultural production) and those services whose values are not (e.g., free access recreation in nature areas). In the former case, the challenge is mainly attribution: what fraction of value added of a sector or the economy should be attributed to what ecosystem services? In the latter case, conventional GDP will be augmented by hitherto unpriced goods and services (e.g., carbon storage or flood protection). This involves extending both the production boundary (i.e. the flows / transactions) and the asset boundary (i.e. the assets that are recorded in balance sheets) of the SNA (Edens and Hein, 2013; Pettini et al., 2013).

For the ecosystem services within the production boundaries of SNA (that are implicitly accounted for), market prices can be used to calculate their values. In theory, however, one would need to use empirically estimated production function approaches (e.g. bio-economic modelling) to assess the marginal value of the ecosystem service involved. For other ecosystem services, where such market prices do not exist, it is necessary to “conduct valuations at a scale which is feasible, credible and policy relevant. In order for these valuations to be consistent with the SNA, they will need to approximate prices, and not to attempt to represent a holistic or social identity of value” (United Nations Statistical Division, 2011, p.9).

There are different views on what valuation methods are “feasible, credible and policy relevant”. Weber (2011) for example, asserts that “compatibility with SNA excludes some methods frequently used in cost-benefit analysis (typically contingent valuation)..,” and proposes to use “remediation costs” to value the degradation of ecosystems. In contrast, the UK National Ecosystem Assessment, has, for reasons of consistency with economic theory, “excluded the use of restoration or replacement costs as a proxy for the value of ecosystem services”(UKNEA, 2011, p. 1072). Glenn-Marie Lange of the WAVES project summarizes this issue as follows: valuation techniques must stay within the SNA concept of value, that is: market-based/marginal. Cost-based, remediation, approaches are “third-best” (Lange, 2011).

5.5.2. System of Environmental-Economic Accounts (SEEA)

The System of Environmental-Economic Accounting (SEEA) provides detailed methodological guidance on how to prepare environmental-economic accounts.[9] The SEEA includes three volumes: the Central Framework, Experimental Ecosystem Accounts, and Applications and Extensions.

The SEEA 'Central Framework' (SEEA-CF) was adopted as an international statistical standard for environmental-economic accounting by the United Nations Statistical Commission at its 43rd session in 2012. It has been prepared jointly by the United Nations, the European Commission, FAO, IMF, OECD and the World Bank. It provides an accounting framework that is consistent and can be integrated with the structure, classifications, definitions and accounting rules of the System of National Accounts (SNA), thereby enabling the analysis of the changes in natural capital, its contribution to the economy and the impacts of economic activities on it. SEEA-CF focuses on the stock of natural resources and the flows that cross the interface between the economy and the environment.

The SEEA 'Experimental Ecosystem Accounting' (SEEA-EEA) has been published as a white cover publication in 2013.[10] It aims to measure ecosystem conditions (with a particular focus on carbon and biodiversity) and the flows of ecosystem services into the economy and other human activities. SEEA-EEA offers a synthesis of the current knowledge of ecosystem accounting and serves as a platform for its development at national and sub-national levels. It provides a common set of terms, concepts, accounting principles and classifications, and an integrated accounting structure for ecosystem services and characteristics of ecosystem condition, in both physical and monetary terms. It also includes a chapter on the main challenges and methodological options for the monetary valuation of ecosystems and ecosystem services.

The SEEA 'Applications and Extensions' is currently under development. It will provide compilers and users of SEEA-based environmental-economic accounts with examples showing how the collected information can be used in decision-making, policy review and design, analysis and research.

Furthermore, the TEEB Secretariat at UNEP and the UN Statistics Division, in collaboration with the CBD Secretariat, have been implementing a project entitled, "Advancing SEEA-EEA in pilot countries", funded by the Norwegian Government, which aims at supporting selected Governments in initiating the testing of SEEA-EEA. The national level activities focus on the assessment of policy priorities, data availability and tools used for ecosystem accounting, stakeholder meetings, the preparation of reports outlining national programmes of work on the advancement of the testing of the SEEA-EEA, as well as relevant national stakeholders to be engaged in the processes. In addition to these national level activities, the project also focuses on facilitating a forum of experts in ecosystem accounting, the preparation of guidance training material and a global strategy for testing the SEEA-EEA, as well as outreach and communication.

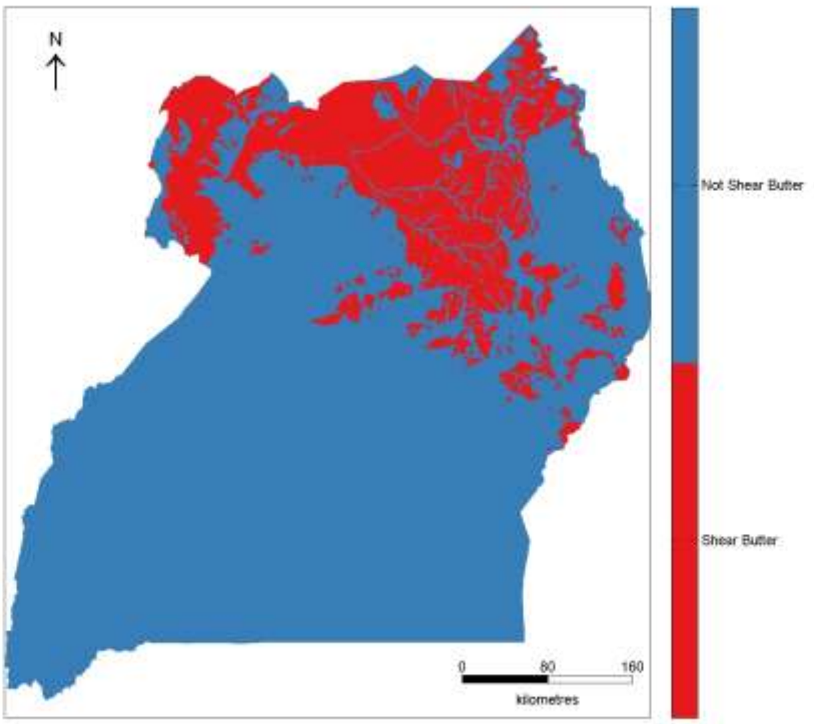
[9] See http://unstats.un.org/unsd/envaccounting/seea.asp

[10] http://unstats.un.org/unsd/envaccounting/eea_project/default.asp

Box 11. Experimental Ecosystem Accounts for Uganda

The Government of Uganda, UN Environment World Conservation Monitoring Centre and Institute for the Development of Environmental-Economic Accounting have recently worked together to develop experimental ecosystem accounts for Uganda. The report compiled is a first attempt to develop the required underlying spatial-data infrastructure and the compilation of key ecosystem and biodiversity related accounts using the System of Environmental-Economic Accounting – Experimental Ecosystem Accounting (SEEA-EEA) framework. The accounts compiled for Uganda concern land cover, ecosystem extent, three non-timber forest products (Gum Arabic, Shea butter tree nuts and Prunus africana) and two flagship mammals (Chimpanzees and Elephants) species. Collectively, these accounts provide significant insights into the state and trends in ecosystems and biodiversity for Uganda.

The map below is one of the inputs to those accounts. It shows the natural areas which have potential for Shea Butter tree nut harvesting (Red). The Shea tree is slow growing native African tree that occurs natural in dry savannah. Based on a combination of land cover mapping over time (derived from remote sensing) and understanding of original ecosystem cover (22 main vegetation types) it shows the potential area suitable for supporting Shea trees and therefore producing Shea Butter tree nuts (for which there is a strong international market not currently exploited by Uganda).



produced using EnSym

The accounts generated present information by Sub-Region of the country over the period from 1990 to 2015 and show that the change in potential Shea butter tree nut provisioning services that have occurred. The spatial mapping also allows understanding of where there may and may not be conflicts with Protected Areas and can highlight where further sustainable harvesting may be possible.

https://www.unep-wcmc.org/system/dataset_file_fields/files/000/000/445/original/Ecosystem_Accounting_in_Uganda_Report_FINAL.pdf?1494865089

5.5.3. Wealth Accounting and Valuation of Ecosystem Services (WAVES)

WAVES is an initiative of the World Bank to implement green accounting in a critical mass of countries, both developed and developing. The project was launched in October 2010 at the CBD meeting in Nagoya and will last five years. The first two years are the preparation phase to establish the global partnership, to establish a Policy and Technical Experts Committee, and conduct feasibility and planning studies in pilot countries. The implementation phase of the project is from 2012 through 2015. Partner countries currently include: Botswana, Colombia, Costa Rica, Madagascar, the Philippines, Australia, Canada, Japan, Norway, and the United Kingdom. Mauritius will join with funding provided directly by France.

"The partners want to take natural capital accounting beyond the SEEA-approved material resources, such as timber and minerals, to include ecosystem services and other natural resources that are not traded or marketed and are therefore harder to measure. That includes the "regulating" services of ecosystems, such as *forests for pollination and wetlands for reducing the impact of floods. A Policy and Technical Experts Committee, working closely with the processes set up by the UN Statistical Commission, has been established to take this forward." (http://www.wavespartnership.org /waves/natural-capital-accounting?active=2)*

The country plans are driven by the countries' needs and preferences. Each partner country is developing a road map to take the initiative further. For Botswana and Madagascar the road map includes developing and implementing macro-indicators such as the Adjusted Net National Income and the Adjusted Net Savings. In addition, the focus in Botswana is on energy resources and energy use, ecosystem-based tourism, and water accounts. In Madagascar, the additional focus is on mining, river basins, ecotourism, coastal zone management, and fishery accounts. The other countries have also presented progress reports on the recent second WAVES partnership meeting Washington D.C.: http://go.worldbank.org/O3A2TJSP30

The approach towards the valuation of non-marketed goods and services is spatially-explicit and demand-based. The challenge to use spatially-specific and demand-based value estimates for national accounting is best described by the World Bank:

"The power of the national accounting approach is to provide an economy-wide picture of the value of ecosystem services. There are many challenges to incorporating natural capital in a national accounting framework, due to the unique characteristics of natural capital. Many case studies of ecosystem services have been done, but there remain many gaps where services are not covered. In some cases, these gaps can be filled by scaling out or borrowing values from other studies. But the value of many ecosystem services is highly site-specific, which makes gap filling and scaling out a potentially complex undertaking. To address this, country implementation teams will be encouraged to seek and use values from local or sub-national case studies for ecosystem services, and identify reasonable methods for scaling up local value to fill data gaps. Technical advice will also be provided to draw on meta-data analyses, and ecosystem models such as InVEST from the Natural Capital project, ARIES or local models to do this." (World Bank, 2011).

It is also one of the tasks of the Policy and Technical Experts Committee to think about how case study value data can be aggregated, scaled-up and reported in National Accounts (Lange, 2011b).

5.5.4. Integrated system for Natural Capital and ecosystem services Accounting (INCA)

The European Commission has launched an internal initiative on natural capital accounting (Knowledge Innovation Project: Integrated system for Natural Capital and ecosystem services

Accounting – KIP-INCA[11]), in line with the objectives of the 7th Environment Action Programme (EAP) and the EU Biodiversity Strategy. The project aims to design and implement an integrated accounting system for ecosystems and their services in the EU by connecting relevant existing projects and data collection exercises to build up a shared platform of geo-referenced information on ecosystems and their services. This system will be used to derive indicators and assess the economic importance and value of ecosystems and their services, in a manner that is consistent with UN standards on environmental accounting (SEEA-EEA). An innovative outcome of the project is that biophysical and economic data related to the extent and condition of ecosystems can be integrated in a systematic way, so that they can be aggregated and disaggregated at the required scale, including at national level, to complement figures of economic performance.

The project is structured in two main phases, a feasibility and design phase which lasted until May 2016 and a follow-up implementation phase, running until 2020. The project focuses on establishing an accounting system for the EU level, primarily using EU-wide data sources, thereby contributing the EU layer to the MAES initiative. The main project partners are Eurostat, the European Environment Agency, DG Environment, the Joint Research Centre and DG Research and Innovation.

KIP INCA will work in line with the UN System of Environmental-Economic Accounting- Experimental Ecosystem Accounts (SEEA-EEA) and will make proposals for improving approaches to accounting based on experience in the EU. With respect to ecosystem services accounts, the general approach of KIP INCA is to quantify supply and use tables and link these tables to the tables which describe the extent and condition of ecosystem assets on the one hand and tables which describe the benefits from ecosystem services on the other hand. This approach differs to some extent with the SEEA-EEA but the resulting accounting tables are fully compliant with the technical recommendations.

A first technical INCA report (La Notte et al. 2017a) outlines initial proposals for the ecosystem service supply and use tables that will be produced by KIP INCA. So far, supply and use tables at EU scale are available for three ecosystem services: recreation and pollination including a description of the models used to quantify the accounts are presented in Vallecillo et al. (2018); water purification accounts are methodologically described in La Notte et al. (2016) whereas the supply and use tables for water purification can be consulted in La Notte et al. (2017b).

The KIP will connect relevant existing projects (in particular ESMERALDA) and data collection exercises (such as LUCAS – land use/cover statistics[12]) to enable them to contribute more information about the ecosystem components of natural capital. JRC will be responsible for feeding outputs of ESMERALDA into the KIP. In particular tier-3 physical and economic mapping approaches of ecosystems, ecosystem condition and ecosystem services would be relevant input of ESMERALDA to INCA.

Box 12. How to read ecosystem services supply and use tables?

The main purpose of supply and use tables for ecosystem services is to show the origin of the actual flow of the service and which economic actor is using it. Figure 1 (taken from La Notte et al. 2017a) presents a graphical simplification of a supply and use table.

The supply table can show the physical or monetary flows of ecosystem services from ecosystems (assets) into the economy (actual flows in figure 1).

The use table records the use of ecosystem services by types of economic units as input to further production or as final consumption. The use table also recognises the possibility of recording the use of ecosystem services by other ecosystem types, i.e. intermediate ecosystem services.

[11] http://ec.europa.eu/environment/nature/capital_accounting/index_en.htm
[12] http://ec.europa.eu/eurostat/web/lucas/overview

The supply and use tables can also record flows of economic products to which ecosystem services contribute.

In an accounting framework, total supply of ecosystem services equals total use. La Notte et al. (2017) propose extensions to the supply use table to also include quantities such as the capacity of ecosystems to deliver ecosystem services, the potential or the sustainable supply.

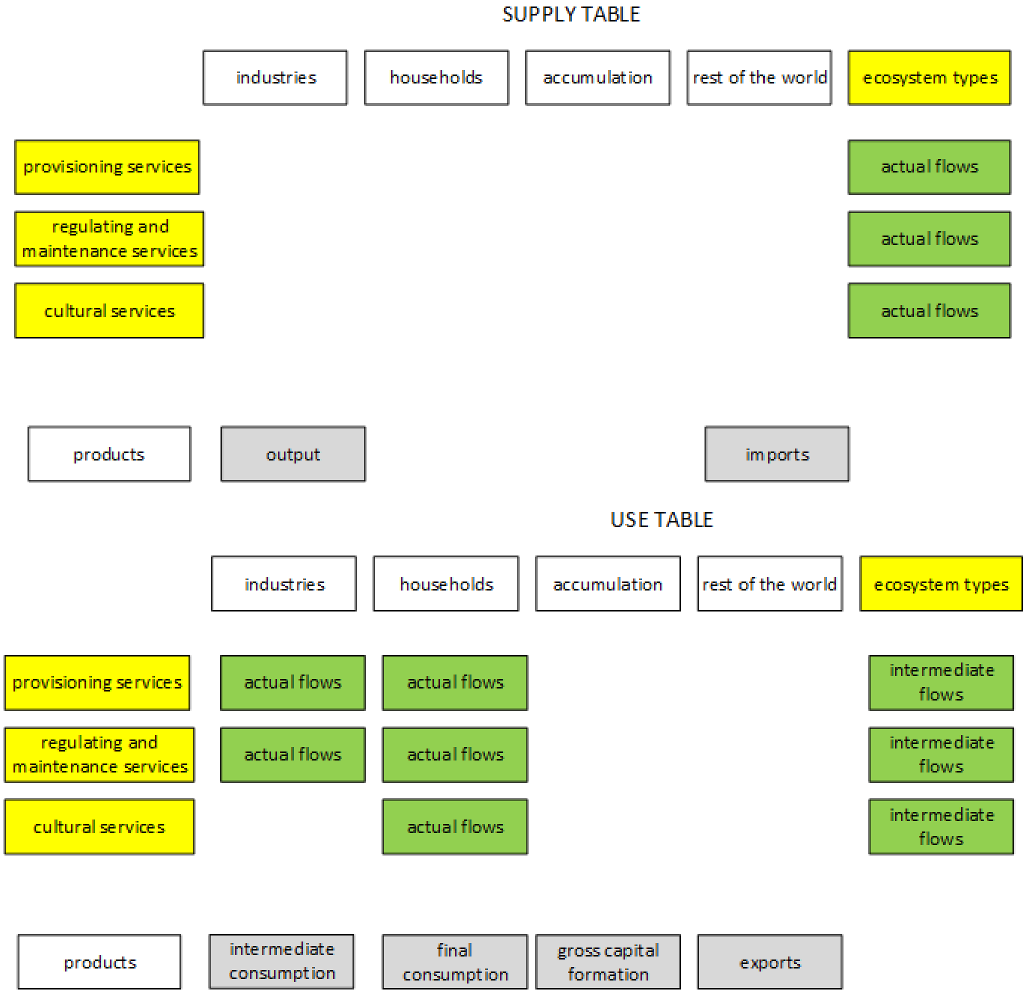


Figure 1. Simplified model of a supply use table to report ecosystem services in a natural capital account (taken from La Notte et al. 2017a)

5.5.5. National/Regional accounting initiatives in the EU

UK Office of National Statistics

The UK Office for National Statistics (ONS), working closely with the UK Department for Environment Food and Rural Affairs (DEFRA), engages in the international developments on experimental ecosystem accounts; and works closely with experts and users in the UK to inform the development of a roadmap for further improvements up to 2020.

In December 2012, the ONS published a Roadmap "Accounting for the value of nature in the UK", which set out a strategy to incorporate natural capital into UK Environmental accounts by 2020.[13] The

[13] The Roadmap and related documents on natural capital accounting can be found at:

Roadmap includes the development of a number of ecosystem accounts based around the eight broad habitats set out in the UK National Ecosystem Assessment. The ONS has also published a set of basic principles to be followed when developing ecosystems accounts (see ONS DEFRA, 2014).

In May 2014, the ONS published "UK Natural Capital - initial and partial monetary estimates", which sets out some experimental methods to estimate the value of a selected number of natural capital assets (see Kahn et al., 2014). The ecosystem services included in these accounts are timber, fisheries, water abstracted for public water supply, outdoor recreation and net greenhouse gas sequestration. These estimates provide an initial overview of the possible value of certain components of natural capital but they also highlight the importance of developing physical accounts, and more detailed and spatially disaggregated ecosystem-based accounts.

Statistics Netherlands

Statistics Netherlands has a long history in developing and implementing integrated environmental-economic accounting. In the beginning of the 1990s, parallel to the publication of the UN's first handbook on integrated environmental and economic accounting (SEEA), Statistics Netherlands extended the National Accounting Matrix (NAM) with a 'satellite account', which includes the environmental pressures related to the production of goods and services and the consumption of households. This resulted in the National Accounting Matrix including Environmental Accounts (NAMEA) (de Haan et al., 1993; de Haan and Keuning, 1996). The NAMEA provided the basis for a Dutch Government commissioned comprehensive macro-economic modelling exercise using an applied general equilibrium model by Gerlagh et al. (2002) to estimate a sustainable national income measure for the Netherlands based on the macro-economic adjustments needed to meet ecological threshold values, which were considered crucial to sustainable environmental development.

Based on the NAMEA and linked to the implementation and reporting requirements of the EU Water Framework Directive (WFD), an integrated water accounting system was developed in 2004, called National Accounting Matrix including Water Accounts for River Basins NAMWARiB (Brouwer et al., 2005). Physical water and pollution flows are linked in this system of integrated accounts to the core System of National Accounts, and disaggregated to the different river basins in the Netherlands using GIS. Time series linking financial transactions in economic sectors to water abstraction, wastewater discharge, corresponding pollution loads of close to 100 chemical substances (including nutrients, heavy metals and other chemical compounds which are systematically monitored in Dutch water bodies), and wastewater treatment are available since 1996. Annual financial flows related to the water services as defined in Article 2 of the WFD (about which MS have to report cost recovery rates to the European Commission) are distinguished explicitly in NAMWARiB. This integrated water accounting system was the basis for another macro-economic modelling exercise using an updated version of the existing applied general equilibrium model for the Dutch economy to estimate the macro-economic and sector impacts of different WFD implementation scenarios (Brouwer et al., 2008; Dellink et al., 2012).

Spanish Agro-forestry Accounts System

The Spanish accounting system for agro-forestry ecosystem services has been developed and tested (Campos and Caparrós, 2006; Caparrós et al., 2003). The accounting unit is a forest ecosystem, e.g. the Mediterranean Monfragüe cork oak forest or the Guadarrama pine forest. Services accounted for are timber, cork, firewood, grazing, hunting, wild mushrooms collected, public recreation, and

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http://www.ons.gov.uk/ons/guide-method/user-guidance/natural-capital/index.html conservation (existence) value. It also includes a value category called "owner's self-consumption of environmental services".

The innovation of the Agro-forestry Accounts System (AAS) is the way in which shadow prices for non-marketed good and services (e.g. mushrooms, public recreation) are estimated. Standard benefits estimates would measure consumer surplus over a change in the level of provision of service. Consumer surplus is not consistent with the concept of exchange values used in the SNA. Therefore the AAS estimates the income that would be earned in a hypothetical market in which ecosystem services would be bought and sold. They estimate hypothetical demand and supply curves for the ecosystem services and make further assumptions on the price that would be charged by a profit maximizing resource owner under alternative market structures (monopoly, competition). Campos et al. (2003) call this the Simulated Exchange Value approach. The hypothetical income of the resource owner thus derived is consistent with the general valuation approach of the SNA.

Another difference is that Campos et al. (2003) include government expenditure in the forests as a cost rather than as output (as is standard in SNA) because, as they argue, the lion's share of government expenditure in forest in Spain is fire fighting and this has a direct impact on commercial timber output. The fire fighting service is therefore already (to a certain extent) valued by the 'saved' timber output. To avoid double counting, government expenditures are therefore only recorded on the cost side.

5.6. Corporate Ecosystem Service Review

The majority of economic methods for assessing ecosystem services focus on decision-making in the public domain. Private sector decision-making may also apply the CBA and MCA frameworks using a private perspective of relevant impacts. The private sector, however, often fails to make the link between ecosystem health and business performance. Many companies are not aware of the extent of their dependence and impact on ecosystems and the possible consequences. As a consequence, corporate environmental management rarely takes into account the risks and opportunities arising from the degradation and use of ecosystem services. Most companies consider 'traditional' issues of pollution and natural resource consumption and therefore focus on environmental impacts, not dependence. Furthermore, they typically address corporate risks, not business opportunities. As a result, companies may be caught unprepared or miss new sources of revenue associated with ecosystem change.

Although the interest of the overall business community in ecosystem services may still be relatively small, there is a growing number of firms that recognise the importance of healthy ecosystems to their operations. This growing recognition is supported by international initiatives and organisations such as The Economics of Ecosystems and Biodiversity (TEEB), the World Business Council for Sustainable Development (WBCSD), The Natural Capital Coalition, and the World Resources Institute (WRI) who have developed assessment tools that aim at integrating natural capital in business and investor decision-making. Ecosystem services approaches for business, among others, focus on various corporate interests such as strategic planning, management of supply chains, procurement, corporate reporting/disclosure and assessing new markets.

One of the challenges in the uptake of ecosystem services approaches by business is the lack of a harmonized approach to clarify why and how the concept of ecosystem services can be practically used in business and finance sector applications. For example, in 2013 the WBCSD published an overview of ecosystem services and biodiversity tools to support business decision-making, containing more than 30 examples of business applications (WBCSD, 2013). To illustrate this rapidly emerging field, we describe the Ecosystem Services Review (ESR), which is one of the most prominent and popular ecosystem services tools in business (Hanson et al. 2012).[14] The ESR consists of a structured methodology that helps managers proactively develop strategies to manage business risks and opportunities arising from their company's dependence and impact on ecosystems. It is a tool for strategy development, not just for environmental assessment. Businesses can either conduct an Ecosystem Services Review as a stand-alone process or integrate it into their existing environmental management systems. In both cases, the methodology can complement and augment the environmental due diligence tools companies already use.

The ESR involves five steps, shown in Figure 8. The first step involves selecting the scope or boundary of the ESR assessment by specifying the stage of the value chain (e.g. suppliers, company, customer) while focussing on strategic, timely and supported business aspects. Candidates of scope include a business unit, product, market, corporate landholdings, infrastructure project, major supplier, or major customer segment, among others.

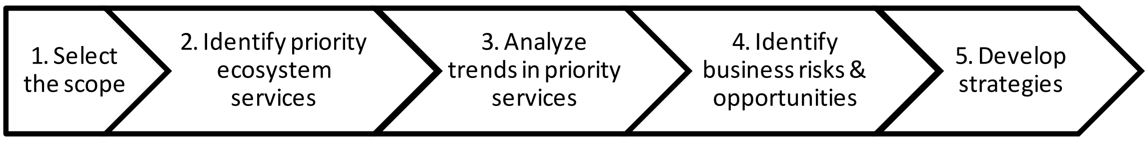

Figure 8. Steps in a Corporate Ecosystem Services Review (Hanson et al. 2012, p.11).

In step 2, priority ecosystem services are identified through a systematic evaluation of the company's dependence and impact on more than 20 ecosystem services as defined by the MA (2005). The priority services are the ones most relevant to corporate performance. A company *depends* on an ecosystem service if that service functions as an input or if it enables, enhances, or influences environmental conditions required for successful corporate performance. What is also important is that, if indeed the ecosystem service serves as a crucial input or enhances conditions for successful performance, whether this ecosystem service has cost-effective substitutes. If there is no such substitute, then the company is considered to be highly dependent upon that service. A company *impacts* an ecosystem service if it affects the quantity or quality of that service. The degree to which a company impacts an ecosystem service in a manner that might pose a business risk or opportunity for itself is a function of whether or not the impact limits or enhances the ability of others to benefit from the service.

Step 3 involves the analysis of the conditions and trends in the priority services, as well as the drivers of these trends. The purpose of this assessment is to provide managers with sufficient relevant information so that they can later identify business risks and opportunities that may arise from these trends. This involves the identification of the present and expected future supply and demand for the services which can be affected by a range of influences such as changes in land use and land cover, over-consumption, climate change, discharge of pollution and overuse of fertilizers, introduction of invasive non-native species. The methodologies developed in ESMERALDA may help companies to map current and expected future supply and demand for priority services.

The fourth step is to evaluate the implications for the company of the trends in the priority ecosystem services. The purpose of this step is to identify the business risks and opportunities that might arise due to these trends. Types of risks and opportunities include (a) operational, (b) regulatory and legal, (c) reputational, (d) market and product, and (e) financing, which are summarised in Table 4.

[14] WRI developed the ESR in collaboration with the Meridian Institute and the World Business Council for Sustainable Development (WBCSD). Since 2008, an estimated 300 companies have used the Ecosystem Services Review.

Table 4. Risks and opportunities arising from trends in ecosystem services

Type	Risks	Opportunity
Operational	• Increased scarcity or cost of inputs • Reduced output or productivity • Disruption to business operations	• Increased efficiency • Low-impact industrial processes
Regulatory and legal	• Extraction moratoria • Lower quotas • Fines • User fees • Permit or license suspension • Permit denial • Lawsuits	• Formal license to expand operations • New products to meet new regulations • Opportunity to shape government policy
Reputational	• Damage to brand or image • Challenge to social 'license to operate'	• Improved or differentiated brand
Market and product	• Changes in customer preferences (public sector, private sector)	• New products or services • Markets for certified products • Markets for ecosystem services • New revenue streams from company-owned or managed ecosystems
Financing	• Higher cost of capital • More rigorous lending requirements	• Increased investment by progressive lenders and socially responsible investment funds

Source: Hanson *et al*. (2012) p.24

The fifth step is to develop and prioritize strategies for minimizing the risks and maximizing the opportunities identified in the previous step. Strategies for responding to ecosystem service-related risk and opportunities fall into three broad categories: (a) internal changes in the company through, for example, changes in operations and product/market strategies; (b) partnering with industry peers, collaborating with other sectors, or structuring transactions with partners through sector and/or stakeholder engagement; and (c) engage policy makers and voice support for incentives or effective government rules for sustainable management of ecosystem services.

After the identification and prioritization of strategies to address ecosystem service risks and opportunities, companies can implement a number of follow-up activities. Building on the ESR experience in one part of the company, managers can extend the methodology to additional divisions, markets, customers, suppliers, or other aspects of their business. Managers can also incorporate the ESR—or elements of it—into their existing environmental management and due diligence systems or into their corporate strategy development processes in order to augment them.

6. Distributional considerations

6.1. 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 assessment 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. It is often the case with the establishment of protected areas that attempting 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 each policy option can provide important insights into the incentives that different groups have to support or oppose each project. This approach can thus provide useful information in the design of appropriate responses and increase success in implementing projects/plans.

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 of alternative options is to construct a distributional matrix, which displays the costs and benefits of a policy option, and shows how they are distributed among different socio-economic groups.

Information on the distribution of the impacts of alternative options may be included directly in a MCA as an additional criterion in the analysis, which then contributes to the overall weighted standardised score of each option. It is technically more challenging to include distributional considerations directly in a CBA. Generally the distributional consequences of alternative options can be provided alongside the outputs of the analysis as additional information for decision-makers to consider. To this end, ESMERALDA deliverable 3.1 provides an overview and guidance on socio-cultural mapping and assessment methods.

Including the consideration of distributional consequences in the ESMERALDA case studies will enhance the real use of assessment results since decision-making, for a large part, is based on the stakeholders involved and their reactions to proposed projects.

6.2. Spatially distributed impacts

As noted earlier, the decision-making context regarding the management of ecosystem services is often one of spatial targeting. Decisions are being made about where to invest in ecosystem restoration, establish of protected areas, or target financial incentives to change the behaviour of land users. In this case, the spatial correspondence of costs and benefits relevant to the decision is of crucial importance and mapping these inputs is necessary.

The spatial distribution of impacts from alternative policy options may also be of interest to decision makers, particularly where different user groups are located in different areas. The analysis of the spatial distribution of impacts 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. 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.

6.3. Temporally distributed impacts

Most policy options will result in impacts not only in the year in which they are implemented but also over a number of years into the future. Both the costs and benefits of a project will therefore have a temporal distribution. It is often the case that projects involve initial investment costs followed by a stream of benefits received over several years in the future. It is important to account for this distribution of costs and benefits over time because people tend to value a benefit or cost in the future less than a benefit or cost now. 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 Euro's worth of resources now will generate more than a Euro's worth of goods and services in the future. Therefore, an entrepreneur is willing-to-pay more than one Euro in the future to acquire one Euro'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 temporal effects in the analysis is to apply a discount rate to future impacts. Suppose an annual value of an ecosystem service X $ will occur over a period of T years, and a discount rate of r per cent is applied, then the present value of the total damage over time 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 damage 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.[15] 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.[16]

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.

[15] 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/Rmanual2/8.3.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.

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

7. A tiered approach to economic mapping and assessment methods

Based on the conceptualisation of a tiered approach for classifying ecosystem service mapping and assessment methods developed by Grêt-Regamey et al. (2017), we adapt that framework to provide guidance on the selection of economic mapping and assessment methods. In order to provide practical guidance, the intention is to assign each method to one of three tiers reflecting the accuracy, detail, technical capacity and data requirements. For example, methods that produce information with a high level of accuracy and detail but have high technical and data requirements are assigned to tier 3. Table 6 provides a definition of each tier. The assignment of economic mapping and assessment methods to a specific tier, however, is not straightforward since each method can be applied with varying degrees of complexity to produce information with varying degrees of accuracy and detail, largely dependent on the availability of data and resources for conducting the analysis. Nevertheless, we have attempted to make generalisations regarding the accuracy and complexity of each method.

Table 6. Definition of tiers for economic mapping and assessment methods

	Accuracy	Detail	Technical Expertise	Data
Tier 1	(Usually) lower accuracy and robustness of results (suitable for awareness raising)	Lower level of detail and spatial specificity	Requires some technical expertise	Uses readily available data
Tier 2	Moderate accuracy and robustness of results (suitable for informing broad policy direction)	Moderate level of detail and spatial specificity	Requires some technical expertise across multiple disciplines	Requires processing existing data from multiple sources
Tier 3	Higher accuracy and robustness of results (suitable for informing the selection of investments)	Higher level of detail and spatial specificity	Requires high levels of technical expertise across multiple disciplines	Requires collection of detailed new data from multiple sources

Figures 9, 10, and 11 provide representations of guiding questions for selecting economic valuation methods, value transfer methods and economic assessment methods respectively. The tiers of each specific economic mapping and assessment method addressed in this report are reported in Tables 2, 3 and 5.

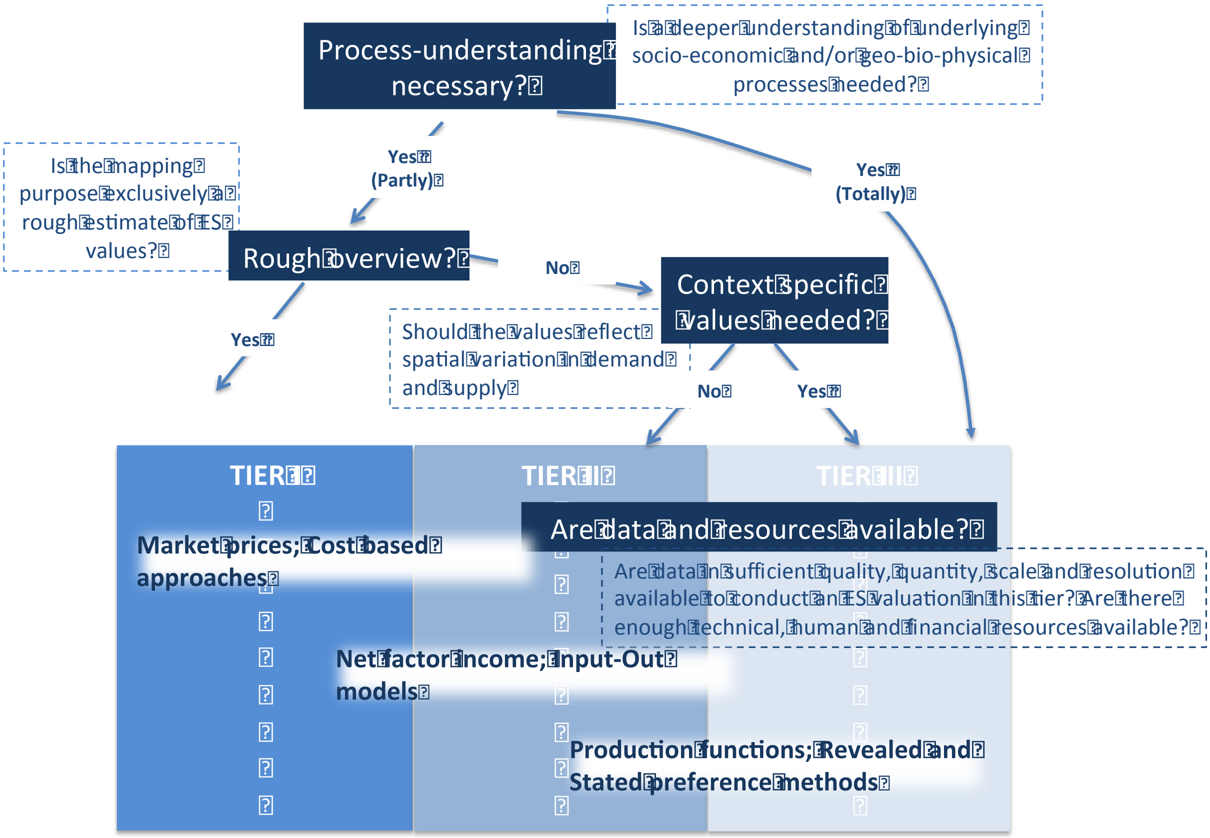

Figure 9. Tiered approach to selecting appropriate primary economic valuation methods. Adapted from Grêt-Regamey et al. (2017)

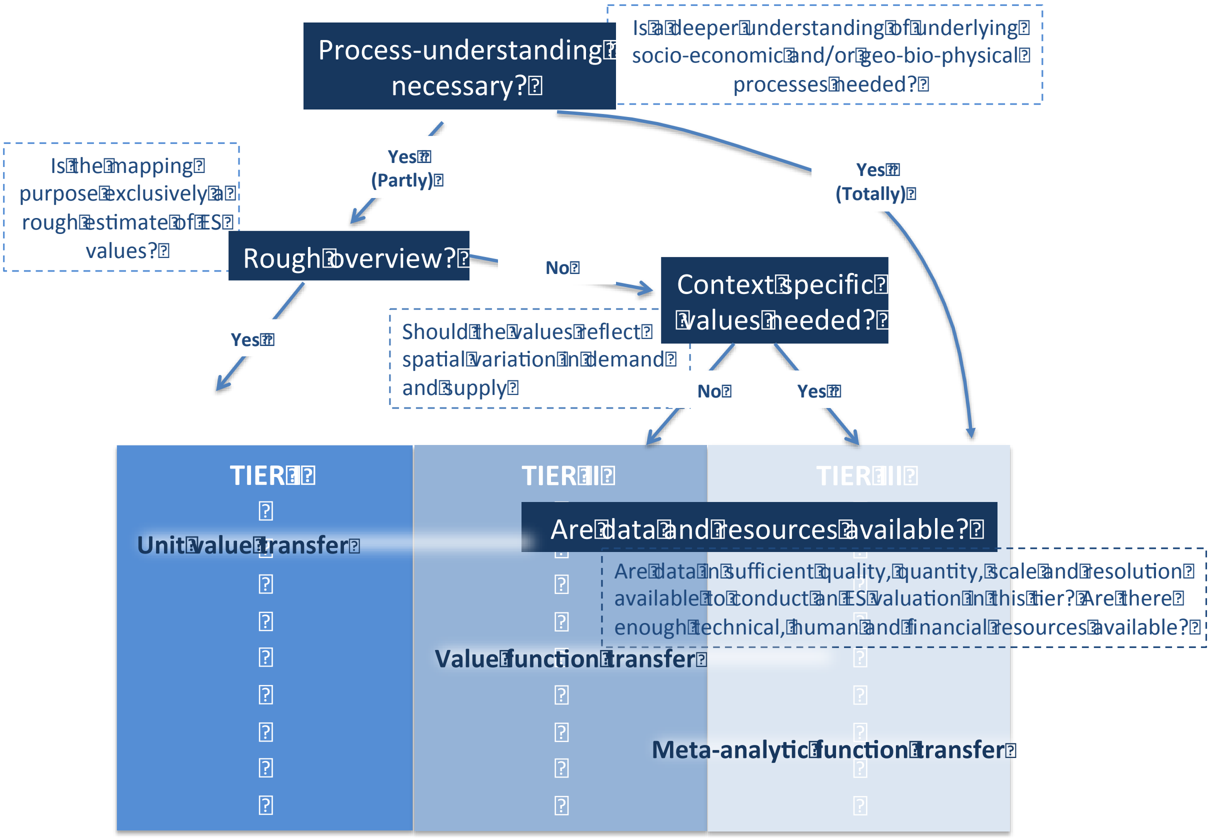

Figure 10. Tiered approach to selecting appropriate value transfer methods. Adapted from Grêt-Regamey et al. (2017)

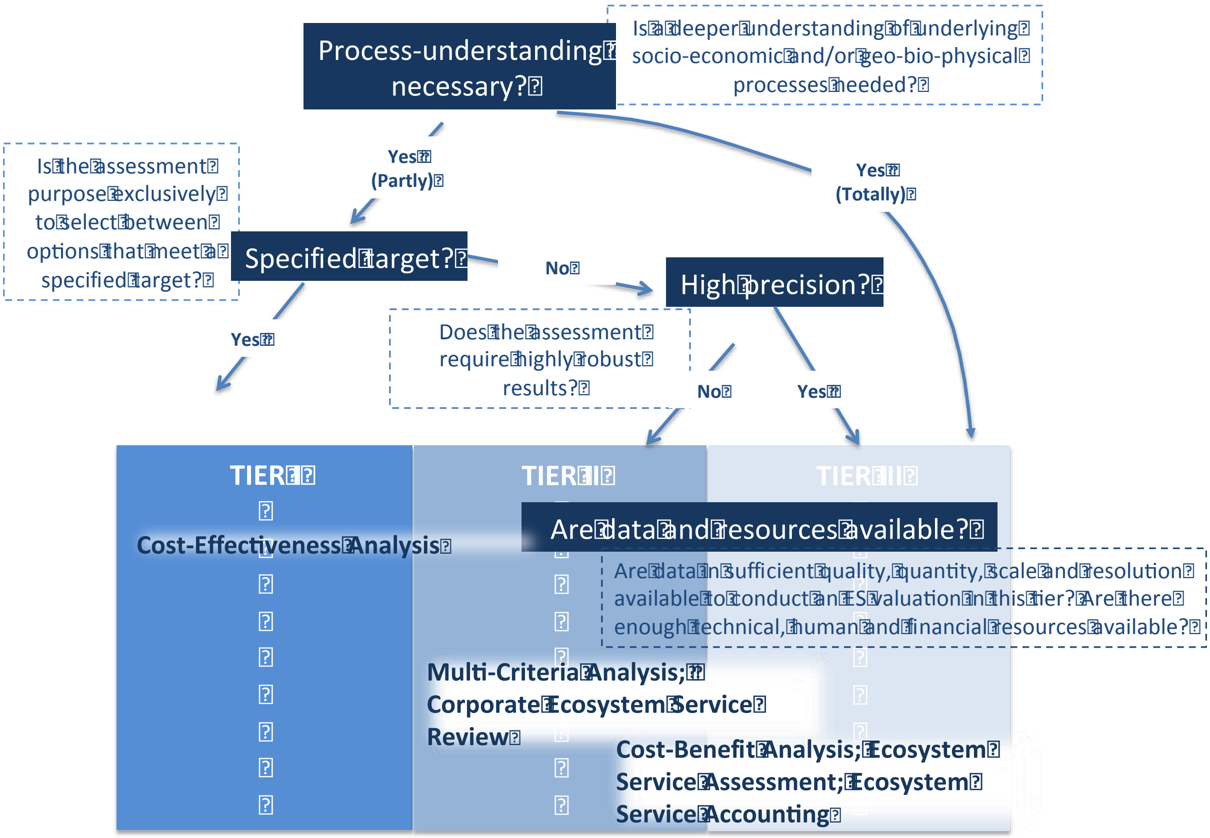

Figure 11. Tiered approach to selecting appropriate economic assessment methods. Adapted from Grêt-Regamey et al. (2017)

8. Integration of economic methods with socio-cultural and bio-physical mapping and assessment methods

Economic methods for mapping and assessing ecosystem services primarily focus on measuring changes in human welfare following changes in the availability of ecosystem services, often driven by biophysical changes in ecosystem extent, condition and functioning. As discussed in Section 4.1, any economic mapping and assessment of ecosystem services therefore fundamentally relies on inputs from biophysical measurement or modeling of changes in ecosystem service availability.

It is also the case that economic mapping and assessment methods use inputs from socio-cultural methods, for example to define the scope of an assessment (e.g. participatory GIS, narrative assessment, Q-methodology) or develop scenario storylines (e.g. participatory scenario planning).

The flow of information from one set of methods to another can also travel in the other direction, with results from economic methods used as inputs in biophysical and socio-cultural mapping and assessment applications. The reality is that methods defined by disciplinary boundaries are to a large extent complements rather than substitutes in providing information on the importance of ecosystem services in decision making.

Mapping and assessment studies will generally require the linking of biophysical, economic and socio-cultural methods. By *linking* we mean that the outputs of one method are used as inputs into another method. A mapping or assessment application may involve several linked steps using multiple methods to produce a final map or other information that is presented to decision makers. ESMERALDA deliverable D3.4 provides specific guidance on how to link methods for mapping and assessing ecosystem services.

In addition to linking methods in a knowledge production process to produce policy-relevant information, there may be a need to integrate separate outputs from biophysical, economic and socio-cultural mapping and assessment applications. By *integration* we mean the combination of complementary pieces of information that address different aspects of an ecosystem service (e.g. sustainability, value and distribution) to support decision making. ESMERALDA deliverable D3.4 also provides guidance on how to integrate information produced by biophysical, economic and socio-cultural methods.

9. Conclusions

Here we provide a brief summary of the main points of information addressed in this report.

- **Economic mapping** of ecosystem services involves the measurement of their economic value accounting for spatial variation in supply and demand. **Economic assessment** of ecosystem services involves the structuring and integration of value information into decision making and the design of policy instruments.
- **Economic value of ecosystem services is a measure of the human welfare** derived from the use or consumption of ecosystem services. Economic valuation is one way to quantify and communicate the importance of ecosystem services to decision makers, and can be used in combination with other forms of information (e.g. socio-cultural values or biophysical indicators).
- The concept of **Total Economic Value** (**TEV**) of an ecosystem is a framework for identifying the comprehensive set of utilitarian values derived from that ecosystem. The word “total” in Total Economic Value refers to the inclusion of different sources of value; TEV does not imply the calculation of an aggregate value of a resource. The classification of different sources of economic value within the concept of TEV is complementary to the classification of ecosystem services.

- The System of National Accounts (SNA) used to produce conventional macro-economic statistics (e.g. GDP) uses a non-welfare based concept of economic value termed **exchange value**. For the purposes of producing ecosystem service accounts that are consistent with the SNA, it is necessary to use estimates of ecosystem services values that are quantified as exchange values.
- The economic value of an ecosystem service is determined by its supply and demand. The supply side of an ecosystem service is largely determined by ecological processes and characteristics that may be influenced by human activities, either deliberately or inadvertently. The demand side is largely determined by the characteristics of human beneficiaries of the ecosystem service. The determinants of both the supply and demand of ecosystem services are spatially variable, which makes the estimation of ecosystem service values inherently spatial. Value mapping addresses this spatial dimension of ecosystem service valuation.
- Ecosystem services are often not traded in markets and so a number of **"primary" non-market valuation methods** have been developed to estimate their economic values. These include the use of replacement costs, avoided damage costs, production functions, revealed preferences (e.g. hedonic pricing, travel costs), and stated preferences (e.g. contingent valuation, choice experiments).
- **Value transfer** (benefit 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 methods are a relatively expeditious and inexpensive means of obtaining estimates of ecosystem service values and can be applied at geographic scales that are not feasible for primary valuation applications. The accuracy of value transfer is dependent on the similarities of characteristics across study sites and policy sites and the extent to which differences are controlled for.
- Economic methods for the assessment of ecosystem services are frameworks for generating and structuring economic information to support decision-making regarding ecosystem services. These assessment methods include: cost-effectiveness analysis, cost-benefit analysis, multi-criteria analysis, ecosystem service accounting and corporate ecosystem service reviews.
- 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, establish protected areas, or target financial incentives to change the behaviour of land 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.
- The choice of which economic mapping or assessment method to use is largely determined by the ecosystem service(s) under consideration, the type of decision problem and the availability of information. To understand the differences between economic mapping and assessment methods, we describe the procedural steps of each approach, provide brief example applications and discuss the strengths and weaknesses of each approach. Each method is assigned to a tier to reflect the precision of its output and the resources required for its application.
- The application of economic mapping and assessment methods will often require inputs from socio-cultural and biophysical methods (and vice versa). In addition, the production of policy relevant information may require the integration of separate outputs from biophysical, economic and socio-cultural mapping and assessment applications.

10. Acknowledgements

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12. Annex 1: Guidelines on valuation methods

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13. Annex 2: Ecosystem service value databases

There are a number of good databases of primary valuation studies available online. These are listed in the table below together with an indication of the region that they cover and the web address.

Table 4: Databases of primary valuation estimates

Database	Region	Website
ASEAN TEEB Valuation Database	Southeast Asia	http://lukebrander.com/
CaseBase	All	http://www.fsd.nl/naturevaluation/73766/5/0/30
ConsValMap	All	http://www.consvalmap.org
Ecosystem Service Valuation Database (ESVD)	All	http://www.es-partnership.org/esp/80763/5/0/50
Ecosystem Services Project Database	All	http://www.naturalcapitalproject.org/database.html
Ecosystem Valuation Toolkit	All	http://www.esvaluation.org/gap_analysis.php
Envalue	US and Australia	http://www.environment.nsw.gov.au/envalueapp/
Environmental Valuation Reference Inventory (EVRI)	All	https://www.evri.ca/Global/Splash.aspx
Marine Ecosystem Services Partnership Library	All	http://www.marineecosystemservices.org/explore
National Ocean Economics Program (NOEP)	All	http://www.oceaneconomics.org/nonmarket/NMsearch2.asp
Non-market Valuation Database	New Zealand	http://www2.lincoln.ac.nz/nonmarketvaluation/
ValueBaseSwe	Sweden	http://www.beijer.kva.se/valuebase.htm