Economic Assessment Methods and Applications (Draft version)

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Preface

This report provides an overview of economic methods for the assessment of ecosystem services. As a starting point, the term "assessment" is understood broadly as a social process used for decision-making.[1] Assessments therefore encompass a variety of processes in which information is generated and used to support decision-making. The process of mapping[2] ecosystem services falls within this broad definition of assessment but in order to make a practical, albeit somewhat forced, distinction between methods for mapping (ESMERALDA Work Package 3) and assessment (ESMERALDA Work Package 4), here we focus only on economic frameworks to support decision-making. These economic *assessment* methods may use ecosystem service *maps* as inputs, but the relevant mapping methods are described in ESMERALDA reports D3.1-D3.3, dealing with social, economic and biophysical mapping methods respectively. In this report "assessment" includes, and indeed focuses on, how information on ecosystem services can be structured to support decision-making.

Economic assessment methods regarding the management of 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 separately in the following sections. Where relevant, the potential role of ecosystem service maps as input into economic assessments is highlighted.

The economic assessment methods are illustrated with example applications. During the course of the ESMERALDA project, further example applications, particularly with respect to the MAES process, will be collected and included in the final version of this report.

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[1] Assessment is defined in the ESMERALDA glossary 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."

[2] The term "mapping" is understood broadly as the description and representation of spatially variable phenomena. 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 mapping of ecosystem services examines spatial variation in the economic value of ecosystem services.

Summary

- This report provides an overview of economic methods for the assessment of ecosystem services: 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 assessments, ecosystem service accounting and corporate ecosystem service reviews.
- The choice of which assessment method to use is largely determined by the type of decision problem and the availability of information. To understand the differences between economic assessment methods, we describe the procedural steps of each approach, provide brief example applications and discuss the strengths and weaknesses of each approach.
- **Cost-effectiveness analysis** (CEA) can be used for decisions that involve selecting between alternative options to achieve a single specific policy goal and where all costs can be expressed in monetary terms. This approach compares alternative policy options only in terms of their costs.
- **Cost-benefit analysis** (CBA) can be used to select between alternative policy options for which all impacts can be quantified in monetary terms. This 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).
- **Multi-criteria analysis** (MCA) is an applicable assessment method in the situation that the relevant criteria (costs and benefits) to the decision cannot be expressed in monetised values, but can only be expressed in other units or in qualitative terms (i.e. impacts can be ranked in order of importance).
- **Ecosystem service assessment** is an appraisal of the status and trends in the provision of ecosystem services in a specified geographic area. The general aim of an ecosystem service assessment is to highlight and quantify the importance of ecosystem services to society. This is a general framework that may focus on biophysical indicators and does not necessarily include economic information. Nevertheless, we include it in this overview of assessment methods because it is distinct from the other reviewed methods and important to ESMERALDA.
- **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 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 develop strategies to manage business risks and opportunities arising from their company's dependence and impact on ecosystems.
- 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.

1. Introduction to economic assessment methods

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. Note that this report focuses on economic assessment methods and not valuation methods, which are described separately in ESMERALDA report D3.2 and summarised in Annex 1 of this report.

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 assessment** is an appraisal of the status and trends in the provision of ecosystem services in a specified geographic area. The general aim of an ecosystem service assessment is to highlight and quantify the importance of ecosystem services to society. This is an overarching framework that can take many forms, possibly without including economic information. Nevertheless, we include it here as a framework that is distinct from the other methods reviewed and important to ESMERALDA.
- **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), 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.

2. Cost-Effectiveness Analysis

Cost-effectiveness analysis (CEA) involves identifying the lowest cost option to achieve a given objective.[3] 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.

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.

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.

[3] 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.

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.

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:

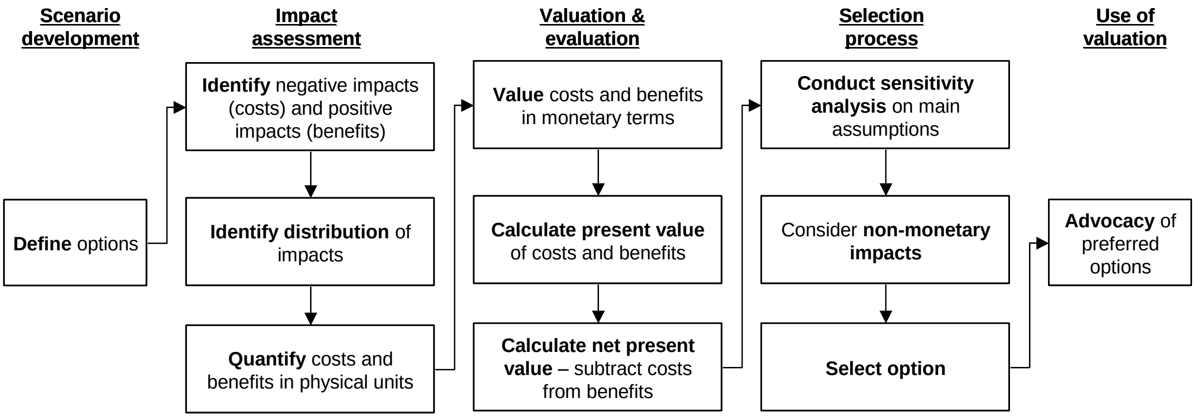

Figure 1: Methodological steps in cost-benefit analysis (source: Brander and van Beukering, 2015)

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 will develop 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.

An example application of a spatial CBA is provided by Brander et al. (2015), who estimate 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.

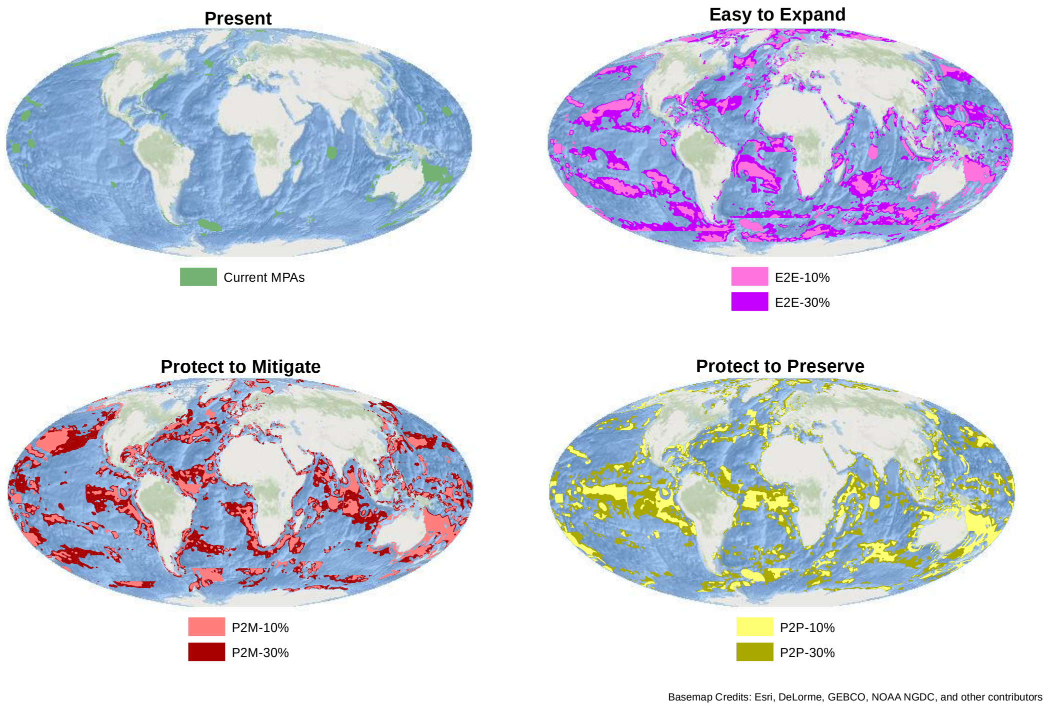

Figure 2. Current and future global distributions of marine protected areas

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 3. 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.

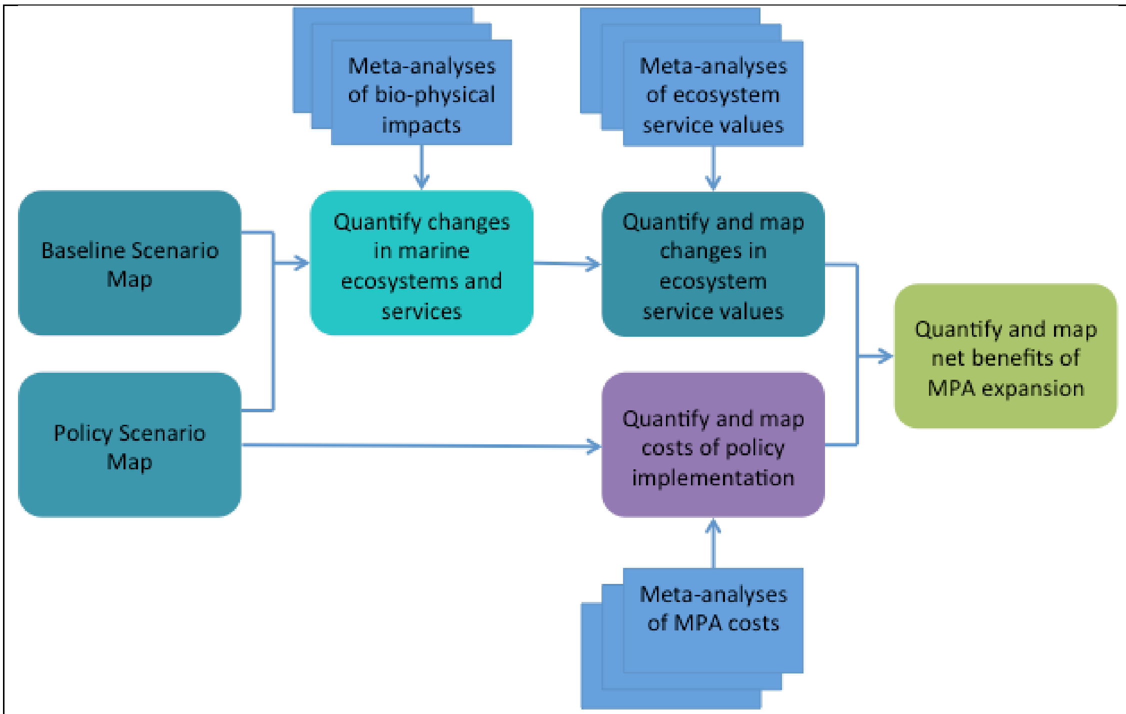

Figure 3. Methodological framework for assessing the net benefits of expanding marine protected areas. Adapted from Figure 2, Balmford et al. (2011); and Figure 2, Hussain et al. (2011).

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.

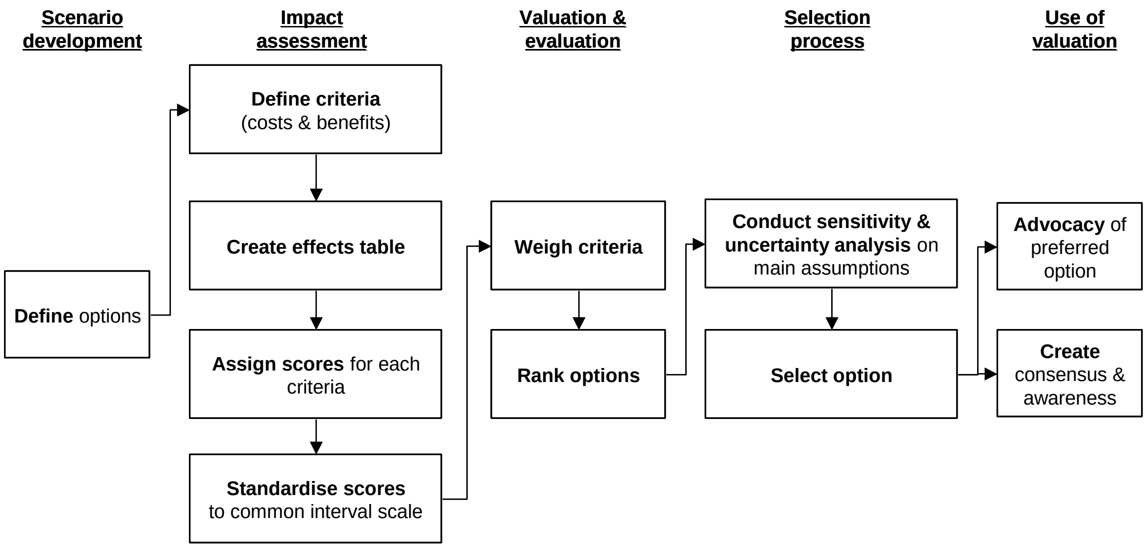

Figure 4: Methodological steps in multi-criteria analysis (source: Brander and van Beukering, 2015)

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.[4]

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

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

MCA is, however, heavily reliant on the judgement of the analytical team for defining alternatives and criteria, estimating the relative importance of criteria and, to some extent, in calculating and inputting

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

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.

An example of a spatial MCA of ecosystem restoration options is provided by Newton et al (2012), who examine the potential of 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. Ecosystem Service Assessments

An ecosystem service assessment is an appraisal of the status and trends in the provision of ecosystem services in a specified geographic area. Ecosystem service assessments can take various forms in terms of the scale of assessment (ecosystem, municipal, sub-national, national etc.), ecosystems included (key habitats, terrestrial, marine etc.) and the type of information produced (bio-physical quantification, economic values, analysis of future scenario). The general aim of an ecosystem service assessment is to highlight and quantify the importance of ecosystem services to society. An ecosystem service assessment may therefore incorporate a broad range of economic methods but differs from economic evaluations such CEA, CBA and MCA, which aim to appraise specific policies or investments; and also differs from ecosystem service accounting, which aims to provide a structured way of measuring the economic significance of nature that is consistent with existing macro-economic accounts. Ecosystem service assessments may involve the development and assessment of possible future land use scenarios. Examples of ecosystem service assessments include The Economics of Ecosystems and Biodiversity (TEEB) countries studies (for Bhutan, Liberia, Ecuador, Philippines and Tanzania)[5] and 'TEEB inspired' country studies (e.g. for Belgium, Germany, Netherlands, the Nordic countries, Poland, Portugal, Slovakia, UK).[6]

To date, arguably the most comprehensive ecosystem service assessment is the UK National Ecosystem Assessment (UK NEA), conducted in the period 2009-2011. The UK NEA was the first

[5] http://www.teebweb.org/areas-of-work/teeb-country-studies/
[6] http://www.teebweb.org/areas-of-work/teeb-country-studies-2/

analysis of the UK's natural environment in terms of the benefits it provides to society and continuing economic prosperity. A follow on phase for the UK NEA was also undertaken to further develop and promote the use of knowledge in decision and policy making. Here we provide a brief summary of the economic analysis conducted for the UK NEA.

The economic valuation component of the UK NEA was coordinated by the Centre for Social and Economic Research on the Global Environment (CSERGE) at the University of East Anglia and conducted by a team of 47 researchers from 17 different institutions. The analysis covers all UK terrestrial and marine habitats but with varying degrees of completeness in terms of the ES provided by each ecosystem. The variation in coverage is largely driven by the availability of data on the provision and value of ES for specific ecosystems. The specific ecosystems that are addressed include: marine habitats, coastal margins, woodlands, plantation forests, moorland, peatlands, agricultural land, crop land, inland wetlands, coastal wetlands, rivers, freshwater, National Parks, National Trust sites, nature recreation sites, local green spaces, urban green space. Table 1 provides an overview of which ecosystems are valued for specific ecosystems services. It is evident that, due to data availability, some ecosystems are more readily valued than others. For example, agricultural land and inland and coastal wetlands are valued for five separate ecosystem services.

The valuation work of the UK NEA addresses only a subset of all ecosystem services due to data limitations for many ecosystems and services. The specific ecosystem services that are valued at a national scale in the UK NEA include: marine fisheries, other marine-based biotic resources, pollination services, climate inputs to agriculture, timber, renewable energy, water supply, water quality, flood protection, carbon sequestration, climate as a source of amenity value, amenity value of green spaces, recreational use of natural areas, and non-use values for biodiversity (existence and bequest). Due to data limitations, these ecosystem services are not valued for all ecosystems that provide them.

The scale of analysis for the valuations of ES varies across from highly local (e.g., 1 km grid cells for the amenity value of green spaces; 2 km grid cells for climate as an input to agricultural production; 5 km grid cells for recreational use of nature areas; individual wetland sites) to national (e.g., timber production, non-use biodiversity values). The scale of reported aggregation of values also varies across ES from regional (e.g., climate as an input to agricultural production), country (e.g., outdoor recreational use of nature areas), to national (UK) aggregation (e.g., water supply provided by wetlands).

6. Ecosystem Service Accounting

6.1. 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).

6.2. System of Environmental-Economic Accounts (SEEA)

The System of Environmental-Economic Accounting (SEEA) provides detailed methodological guidance on how to prepare environmental-economic accounts.[7] 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,

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

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.[8] 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, TEEB Secretariat at UNEP and 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.

6.3. 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), 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 bio-physical 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 2 main phases, a feasibility and design phase (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

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

MAES initiative. Member States will be able to link into this system. The main project partners are Eurostat, the European Environment Agency, DG Environment, the Joint Research Centre and DG Research and Innovation.

The KIP will connect relevant existing projects (in particular ESMERALDA) and data collection exercises (such as LUCAS – land use/cover statistics[9]) 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.

The ecosystem accounting system will provide maps, tables and accounts and will be designed to support and inform policy development and implementation in the EU and will be established on the basis of MAES, the SEEA EEA and other relevant methodological guidance. The system will be designed so that its data layers and other information outcomes are fit for purpose for policy-makers, analysts and researchers as they prepare various policy evaluations and decisions. It will contribute to better planning and implementation, as well as monitoring of progress towards achieving objectives and meeting communication goals. Examples range from UK work on forest spatially disaggregated accounts, which helped support forest management decisions, to the publication of national natural wealth figures in Canada and Australia to complement economic performance figures. By focussing on ecosystems and their services, this KIP addresses an important gap in terms of knowledge, data and tools, for national accounting and related indicators.

6.4. 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.[10] The 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.

[9] http://ec.europa.eu/eurostat/web/lucas/overview

[10] The Roadmap and related documents on natural capital accounting can be found at: http://www.ons.gov.uk/ons/guide-method/user-guidance/natural-capital/index.html

6.5. 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).

6.6. European Environment Agency (EEA)

The European Environment Agency has developed a framework for Simplified Ecosystem Capital Accounts (SECA) (Weber, 2011). The basic statistical unit is the Socio-Ecological Landscape Unit (SELU), derived from the Corine land cover maps and additional geo-environmental information on a 1km grid. The main division of landscape units is between mountains, highlands, lowlands, coasts, and rivers. The terrestrial landscapes are subdivided in urban areas, broad pattern agriculture, agricultural associations and mosaics, pastures and natural grasslands, forest tree cover, other dominant natural land cover, and composite land cover.

Within these landscape units, SECA focuses on three groups of services: biomass/carbon production, freshwater production and functional services. The latter measure the capacity or potential of ecosystems to deliver ecosystem services in a sustainable way.[11] A final composite index is the Ecosystem Potential Unit Equivalent (EPUE).

The monetary valuation approach of SECA is related to the concept of Consumption of Fixed Capital (CFC). Translated to ecosystems this refers to the depreciation of ecosystem capital. The EEA gives a few examples of this depreciation: "the cost of keeping below the maximum of 2 degrees global warming target", "REDD (Reducing Emissions from Deforestation and forest Degradation) payments", and "the costs of remediation measures to restore or maintain 'good environmental quality of the river basins' under the Water Framework Directive". Unit costs per EPUE are to be derived by experts from the analysis of real expenditures or costs of restoration programs. "Estimates of unitary costs have to be carried out by ecosystem types/issues/regions" (Weber, 2011, p.23).

6.7. 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

[11] This is measured by indicators such as the Green Background Landscape (GBL) index, the Mesh Effective Size (MEFF) index, the Sated Social Nature Value (SSNV) index, the Landscape Ecosystem Potential (LEP) index, etc.

macro-economic and sector impacts of different WFD implementation scenarios (Brouwer et al., 2008; Dellink et al., 2012).

6.8. Spanish Agro-forestry Accounts System

In Spain, an 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 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.

7. 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).[12] 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 5. 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 5. 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

[12] 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.

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 1.

Table 1. 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.

8. Distributional considerations

8.1 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 report 4.3 provides an overview and guidance on social 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.

8.2 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.

8.3 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.[13]

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.[14] 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.[15]

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

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14 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.

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

9. Summary guidance on economic assessment methods

This section provides summary guidance on the use of economic assessment methods for ecosystem services. In order to provide guidance on methods, 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 2 provides a definition of each tier. The assignment of economic assessment methods to tiers 1-3, however, is not straightforward since each method can be applied with varying degrees of complexity and produce information with varying degrees of accuracy and detail. Nevertheless, we have attempted to make generalisations regarding the accuracy and complexity of each method, which is open to further discussion.

Table 2. Definition of tiers for economic assessment methods

	Accuracy	Detail	Technical Expertise	Data
Tier 1	Low accuracy and robustness of results (suitable for awareness raising)	Low 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	High accuracy and robustness of results (suitable for informing the selection of investments)	High level of detail and spatial specificity	Requires high levels of technical expertise across multiple disciplines	Requires collection of detailed new data from multiple sources e.g. region specific primary valuation studies

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 3 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 3. 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
Ecosystem Service Assessment	Used to highlight and quantify the importance of ecosystem services to society	Open and flexible process to allow assessment to focus on issues of highest public concern	Comprehensive high quality assessment can be highly resource (expertise and data) intensive	2
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

10. Next steps

This report is an initial draft that provides an overview of economic methods for the assessment of ecosystem services, illustrated with a few examples, together with tentative guidance on the strengths, weaknesses and 'tier' of each approach. During the course of the ESMERALDA project, this guidance will be further developed into a final report due February 2017.

The draft report will be revised to include additional example applications of each assessment method drawn from the joint WP3-WP4 survey of ecosystem service mapping and assessment case studies across Europe. In preparation for ESMERALDA workshop 3 (April 15-16 2016, Nottingham, UK), the draft report will be discussed with WP3 and WP4 project partners to develop a programme for presentation and discussion of all flexible mapping and assessment methods. At the workshop, the contents of this draft report will be presented and discussed with all project partners in order to agree the set of methods to be tested in WP5. Project partners involved in WP4.2 (BEF, CVGZ, IST, VITO, MCAST) will be asked to make specific contributions to the report, particularly in the form of example applications.

This report will undergo a further round of revision to reflect the lessons learned through the testing process in WP5. The methods outlined here will be tested in a selected set of case studies conducted by ESMERALDA project partners. Partners will be asked to report back their experience in applying economic assessment methods with respect to the suitability of the selected method to the policy question addressed; the effectiveness of the information provided for decision making; the technical expertise required; data availability; and data collection processes. To some extent, the feedback will require consultation with the case study stakeholders to obtain their opinions regarding the utility of the information produced.

The final report will include guidance on how to select which method to use for different decision making contexts.

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Annex 1: Economic valuation of ecosystem services

What is economic value?

Economic value of ecosystem services is a measure of the human welfare derived from the use or consumption 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 be used in combination with other forms of information (e.g. social 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'.

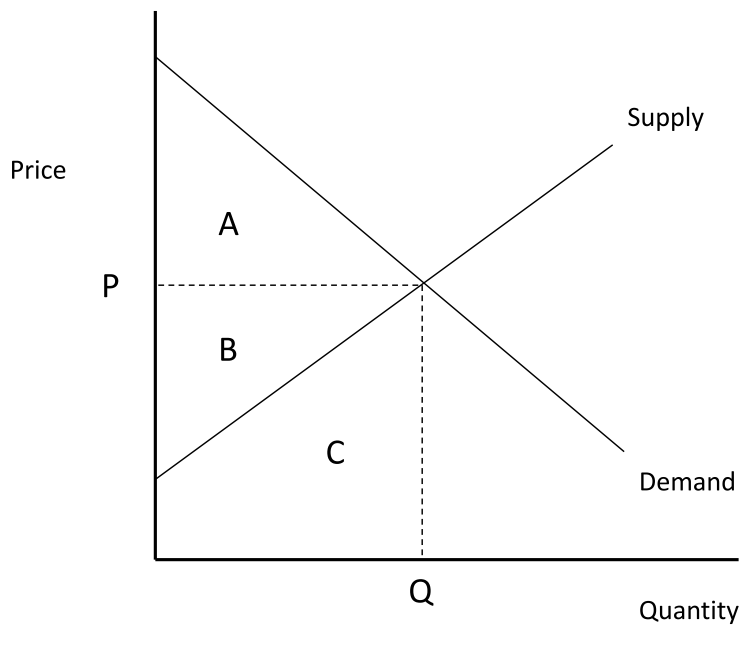

Figure A1: Demand and supply

In Figure A1, 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 A2 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 A2 as a vertical line. For the most part, bio-physical 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 (the more that we have of a service, the lower the additional welfare of consuming more). In this case, consumers don't pay a price for the quantity (Q) that is available to them and the entire area under the demand curve (D+E) represents their consumer surplus. It is useful to keep this 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 A2 also represents the conceptual value of ecosystem services that have a 'derived demand'.

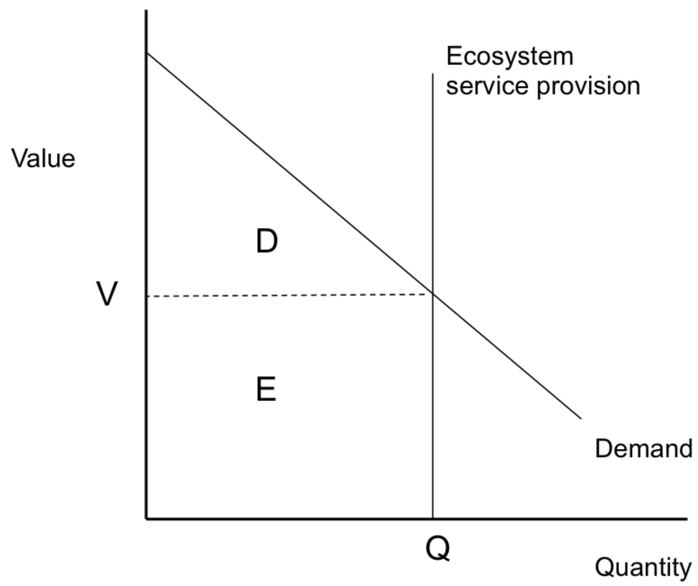

Figure A2: Demand and supply for a non-marketed ecosystem service

The **marginal value** of an ecosystem service is the contribution to wellbeing of one additional unit of the service (V in Figure A2). It is equivalent to the price of the service in a perfectly functioning market (P in Figure A1). 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).

Total Economic Value (TEV)

The concept of **Total Economic Value** (**TEV**) of an ecosystem is used to describe the comprehensive set of utilitarian values derived from that ecosystem. This concept is useful for identifying the different types of value that may be derived from an ecosystem. TEV comprises of **use values** and **non-use values**. Use values are the benefits that are derived from some physical use of the resource. **Direct use values** may derive from on-site extraction of resources (e.g. 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 A2. It is important to understand that the "total" in Total Economic Value refers to the aggregation of different sources of value rather than the sum of all value derived from a resource. TEV is a measure of total value as apposed 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.

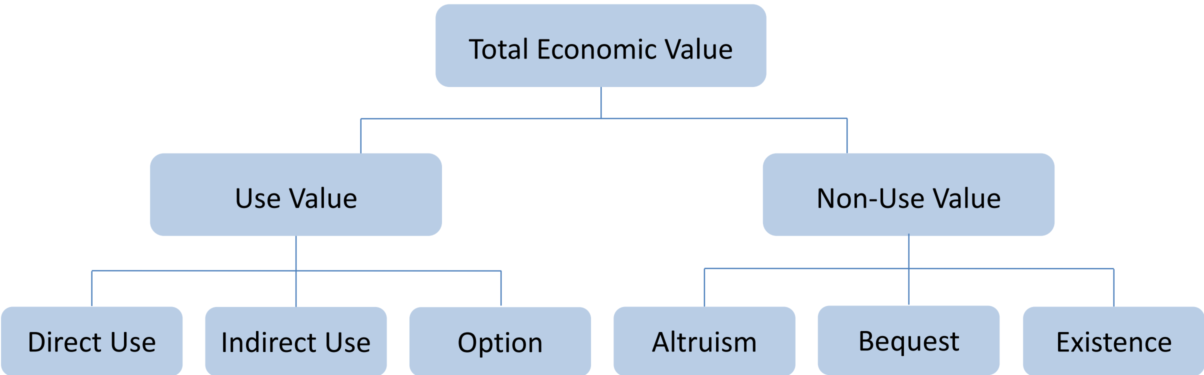

Figure 3: The components of Total Economic Value

The classification of different types of economic value within the concept of TEV is complementary to the classification of ecosystem services. Table A1 sets out the correspondence between categories of ecosystem service and components of TEV.

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

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

Exchange value

The **concept** of welfare value is used in most assessments of ecosystem service but it is not used in the **system of national accounts (SNA)** that is used to calculate gross domestic product (GDP) and other economic statistics. The SNA uses the concept **exchange value**, which is a measure of producer surplus plus the costs of production. In Figure A1 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.[16]

Mapping economic values for ecosystem services

The economic value of an ecosystem service is, as with any good or service, 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 assessment 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 values vary across space. It therefore includes not only studies that produce graphical value maps but also analyses that explicitly address spatial variability in values.

Value mapping thereby reveals additional information as compared to conventional site-specific ecosystem service 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. Thereby, spatially explicit ecosystem service value maps have specific advantages for several policy applications including ecosystem service accounting, land use policy evaluation, conservation planning and 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.

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

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

of substitutes and complements.[17] A value mapping application can address spatial variations in biophysical ecosystem service supply, socio-economic demand, or a combination of both aspects.

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) and economic methods are used to estimate spatially variable marginal values per unit of ecosystem service provided.

Methodologies for bio-physical mapping of ecosystem service supply are addressed in ESMERALDA report D3.3 and can be divided broadly into five main categories: (1) one-dimensional proxies for ecosystem services, such as land cover or land use; (2) non-validated models: ecological production functions (or models) based on likely causal combinations of explanatory variables, which are grounded on researcher or expert assumptions; (3) validated models: ecological production functions, which are calibrated based on primary or secondary data on ecosystem service supply; (4) representative samples of the study area: data on ecosystem service supply that is collected for the specific study area; and (5) implicit modelling of ecosystem service supply within a value function, i.e. the quantity of ecosystem service supply is modelled within the estimation of socio-economic benefits from the ecosystem service.

Economic methods for estimating ecosystem service demand, values and accounting for spatial variation are described in sections 2 and 3, for primary valuation methods and value transfer methods respectively.

Scaling up economic values

The currently available information on the value of ecosystem services is mostly for relatively small spatial scales (e.g. 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 scarcer, its marginal value will tend to increase. This means that multiplying a constant marginal value by the change in area of an ecosystem site, 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;

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

2012) propose a methodology to specifically address the challenge of scaling up ecosystem service values.

Methods for estimating economic values

A variety of methods have been developed for estimating the economic value of ecosystem services. These valuation methods are designed to span the range of valuation challenges raised by the application of economic analyses to the complexity of the natural environment. Table A2 provides an overview of primary valuation methods, typical applications, limitations and indicates which primary valuation methods can be used to value which ecosystem service. The term ‘primary valuation’ refers to methods and studies that produce a new or original value estimate for a specific ecosystem. The term is used to make a distinction from studies that use existing value information (i.e. value transfer, which is discussed in the next section).

An important distinction to be aware of between primary valuation methods is the difference between revealed preference methods (those that observe actual behaviour of the use of ecosystem services to elicit values) and stated preference methods (those that use public surveys to ask beneficiaries to state their preferences for, generally hypothetical, changes in the provision of ecosystem services). Revealed preference methods may be favoured since they reflect actual behaviour but are limited in their applicability to some ecosystem services. Stated preference methods on the other hand rely on responses recorded in surveys or experiments but are more flexible in their application.

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.

Table A2: 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
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 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 marketed goods that can substitute for ES	ES for which marketed substitutes are available	Bottled water as a substitute for clean drinking water	Only applicable where direct substitutes are available and observed to be purchased. 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 a man-made equivalent	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 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
Damage cost avoided	Estimate damage avoided due to ecosystem service	Ecosystems that provide storm or flood protection to houses or other assets	Coastal protection by dunes; river flow control by wetlands	Difficult to relate damage levels to ecosystem quality.	2
Net factor income	Revenue from sales of environment-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

Valuation method	Approach	Application to ecosystem services	Example ecosystem service	Limitations	Tier
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
Hedonic pricing	Estimate influence of environmental characteristics on price of marketed goods	Environmental characteristics that vary across goods (usually houses)	Urban open space; air quality	Technically difficult. High data requirements. Limited to ES that are spatially related to property locations.	3
Travel cost	Use data on travel costs and visit rates to estimate demand for recreation sites	Recreation sites	Outdoor open access recreation	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	Species loss; natural areas; air quality; water quality landscape aesthetics	Expensive and technically difficult to implement. Prone to biases in design and analysis	3
Choice modelling	Ask people to make trade-offs between ES and other goods to elicit willingness to pay	All ecosystem services	Species loss; natural areas; air quality; water quality; landscape aesthetics	Expensive and technically difficult to implement. Prone to 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	Species loss; natural areas; air quality; water quality; landscape aesthetics	Prone to biases due to group dynamics	3

Methods for transferring economic values

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 quick 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 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 is 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).

These three principal methods for transferring ecosystem service values are summarised in Table A3 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).

In order to provide guidance on methods, 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. The assignment of value transfer methods to tiers 1-3, however, is not straightforward since each method can be applied with varying degrees of accuracy, largely dependent on the similarity across characteristics of study and policy sites.

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

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