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Project General Coordinator: Jan Marcin Węśławski Project Scientific Manager: Julie Bremner Project Manager: Joanna Potrykus
Task Leader(s) Task 4.1: Anita Franco (IECS Ltd) Task 4.2: Carlo Fezzi (UNITN) Activity 4.2a: Carlo Fezzi (UNITN) Activity 4.2b: Carlo Fezzi (UNITN) Activity 4.2c: Johanna Forster (UEA) and Tasman Crowe (UCD) Task 4.3: Tiziana Luisetti (Cefas) and Kerry Turner (UEA) Activity 4.3a: Tiziana Luisetti (Cefas) Activity 4.3b: Silvia Ferrini (UEA) Activity 4.3c: Kerry Turner (UEA) Task 4.4: Michael Elliott and Daryl Burdon (IECS Ltd)
Author(s) Daryl Burdon (IECS Ltd.), Tiziana Luisetti (Cefas), Anita Franco (IECS Ltd.), Michael Elliott (IECS Ltd.), Eva Amorim (IECS Ltd.), Valentina Di Gennaro (UEA), Kerry Turner (UEA), Silvia Ferrini (UEA); Carlo Fezzi (UNITN), Sandra Notaro (UNITN), Nan Zhang (UNITN), Fernando Rodriguez (USAL), Ashley Cahillane (UCD), Marta Pascual (BC3), Benat Egidazu-de la Parte (BC3), Johanna Foster (UEA), Tasman Crowe (UCD), John Brannigan (UCD); Jonathan Atkins (IECS Ltd.); Steve Barnard (IECS Ltd.).
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Executive Summary

Marine biodiversity is in decline across the globe because of both anthropogenic activities and changing climatic conditions. Marine scientists and policymakers worldwide have emphasised the potential impact that threats to biodiversity have to the health of our planet and the welfare of society. Given that marine biodiversity underpins the delivery of ecosystem services, and subsequently the supply of benefits for society, there is a requirement for countries be able to value their coastal and marine biodiversity as a basis for cost-effective environmental management.

To assess the multifaceted value of marine biodiversity, and how these values change over time, a range of ecological, economic and socio-cultural valuation and accounting methods are required. Valuation of the societal benefits delivered by coastal and marine ecosystems provides the basis for assessing cost-effective environmental monitoring and management options. Some valuation methods are well established, while others are in their infancy. This handbook presents a range of valuation methods which have been developed, applied, refined and often integrated to value marine biodiversity across a range of European case study sites (MARBEFES Broad Belt Transects, BBTs). Methods and applications span between coastal and marine case study sites (see also MARBEFES Deliverable D2.1). Whilst the valuation methods presented have been developed, tested and validated within European Seas (including the Arctic, Baltic, Atlantic and Mediterranean; see also outputs, milestones and deliverables from MARBEFES WP2 for the testing of the methods in the BBTs), their application is of relevance to coastal and marine sites globally.

This handbook describes the methods for undertaking ecological (Guide 1), economic (Guide 2) and socio-cultural (Guide 3) valuation of marine biodiversity and presents case study boxes throughout which demonstrate applications of the approaches in the BBTs. The handbook presents further development and application of a Complementary Accounting Network which can be used to ensure that ecological, economic and socio-cultural values can be read side-by-side and therefore the multifaceted value of marine biodiversity be fully incorporated into management decisions. The research presented in this handbook is a first attempt to apply ecological, economic and socio-cultural valuation methods, either separately or integrated, consistently across multiple international case study sites.

Feedback is included on lessons learned from case study applications across European seas, concluding with the following set of recommendations for future research and uses of the outputs within marine management:

- Applying different valuation methods ideally requires a site-specific multidisciplinary team in which experts should be directly involved in the implementation of the methodologies rather than just providing guidance and support for implementing the method.
- The choice of valuation tools to apply in a case study should be related to the specific questions and challenges that are faced and the availability of site-specific data.
- Given that some of the tools rely on the same underlying data sets, valuation tools could be combined, for example Ecological Value Assessment with Physical Natural Capital Accounts (as in Egidazu-de la Parte et al., 2026), but also potentially with Monetary Natural Capital Accounts and Socio-Cultural Valuation.
- It is good practice to be cautious when taking decisions based on some exercise of economic valuation of ecosystem services – such studies provide valuable information for policy and management but underlying assumptions and caveats need to be understood.

- It is recommended that economic valuation studies should follow the established guidelines for revealed preferences and benefit transfer studies (e.g., Bateman & Kling, 2000; Johnston & Rosenberger, 2010).
- For the socio-cultural valuation surveys, there is no single engagement method that works well to capture data for all groups and therefore several approaches are needed to capture the breadth of responses that are required (e.g., online survey, face-to-face and focus groups).
- There is a need to bridge knowledge gaps across a multidisciplinary team, especially among natural scientists unfamiliar with economic accounting concepts, highlighting the need for early-stage capacity building.
- Future applications of both the Physical and Monetary Natural Capital Accounts would benefit from dedicated interdisciplinary training on the concepts and stronger emphasis on primary data collection for accounting purpose. These improvements would enhance both the robustness and usability of the approach, particularly at local scales.
- There is a need to improve inclusivity of social science and humanities expertise in ecosystem valuation research across geographies to ensure the widespread site-specific multidisciplinary expertise needed for holistic approaches to environmental valuation.
- Rather than seeking a full integration of environmental data into the Standard National Accounting (SNA) framework, we argue pragmatically, for the establishment of protocols as a Complementary Accounting Network (CAN) - a pluralistic, multi-dimensional framework designed to capture the full range of values that society derives from nature expressed through monetary (exchange and welfare) and non-monetary terms - to sit alongside the economic accounts and on the same timescale.

1. Introduction to the MARBEFES Handbook

This handbook has been developed as part of the Horizon Europe project MARBEFES, within *Work-Package 4 Ecological, Economic and Socio-Cultural Valuation*. In this handbook, we propose a nine-stage decision support procedure for the Broad Belt Transects (BBTs); although it is recognised that the methods presented can be applied to other case studies around the globe. Whilst the ambition is for all nine stages to be completed within each BBT, the proposed framework provides BBTs with several ‘optional’ stages depending on the expertise and resources available.

The aim of this report is to guide the reader through several ecological, economic and socio-cultural valuation methods that can be interlinked and/or are complementary: Standard National Income Accounting (SNA), biophysical data accounts and monetary value accounts (Ecosystem / Natural Capital Accounts), economic valuation techniques, marine Natural Capital Accounting. A more comprehensive environmental change indicators framework (Complementary Accounts Network) is also introduced in this context.

The main topics that will be explained in the following sections are:

1. Scoping study with DAPSI(W)R(M) framework – Introduction and Guide 2
2. Ecological Value Assessment for MARBEFES BBTs – Guide 1
3. Natural capital, ecosystem services (stock and flows) and societal benefits – Guide 2
4. Physical Natural Capital Accounts for MARBEFES BBTs - Guide 2
5. Monetary Natural Capital Accounts for MARBEFES BBTs (optional) - Guide 2
6. Apply research for primary data collection with one of the economic valuation methods/techniques available in the literature or Benefit Transfer (optional) - Guide 2
7. Apply research for primary data collection for socio-cultural valuation (optional) - Guide 3
8. Comprehensive indicators dashboard (Complementary Accounting Network, CAN), a step-by-step guide on the application of all guides
9. Repeat the process as a means of monitoring the BBT area.

The overall aim of MARBEFES is to place concepts, methods and techniques into a logical sequence that can provide decision support for policymakers across European seas. Decision support can be seen as a process and a set of tools which establishes an evidence base to aid choice making and trade off decisions.

In every BBT, the scope of the research to be conducted and the main issues and problems must first be identified and analyzed. In further detail, the question formulation process should lead to a characterization of the issues that need to be analyzed, the identification of system limits and restrictions, and the acknowledgment of policy concerns, objectives and evaluation methods.

A baseline description of the specific marine and coastal system under study, including geographical and spatial boundaries, biophysical information and socio-economic activity, is the ideal place to start for any decision support procedure.

The marine environment is a dynamic and complex system, which is influenced by both anthropogenic and natural changes. The DPSIR (Drivers-Pressures-State changes-Impacts-Responses) cause-consequence-response framework can be used as a problem structuring method (Gregory et al., 2013) which captures the key relationships between society and the environment (Atkins et al., 2011) through key scoping questions. Developed from an OECD approach which aimed to link anthropogenic Pressures with State changes and Impacts (OECD, 1994), the DPSIR framework encompasses the Drivers, which are the key demands by society and creates Pressures (the causes of the problems) and recognises that State changes (the change in background environmental status) and Impacts (the

changes in human welfare) then require management Responses from society. The DPSIR framework has been adopted by the European Environment Agency and others (e.g. EEA, 1999; Borja et al., 2006; Gray & Elliott, 2009) and it was successfully applied to several marine sectors (e.g., marine aggregates – Cooper et al., 2013). Subsequently it has undergone many iterations to make it harmonised and clarified (Patricio et al 2016).

Cooper (2013) then proposed that DPSIR should be modified as DPSWR (Drivers – Pressures - State changes – Welfare - Responses), to avoid potential confusion between the impacts on the environment i.e., State changes within the basic form of DPSIR and the impacts on human Welfare i.e. Impacts in its basic form. Such a distinction was then taken forward by the UK National Ecosystem Assessment Follow-On (UKNEAFO) project which refers to the application of a DPSWR model for the coastal and marine environment (Turner et al., 2015). In a further modification, Smyth et al., (2015) propose that DPSWR should become DAPSI(W)R (Drivers – Activities – Pressures - Impacts on Welfare - Responses) thus recognising that it is Activities that cause Pressures not the Drivers themselves and that Impacts are on Welfare. This modified DAPSI(W)R framework was applied in the context of the UK offshore wind energy development sector, with the focus on decommissioning as a management Response.

Within MARBEFES, the DAPSI(W)R(M) (Drivers – Activities – Pressures - State change - Impact (on Welfare) - Response (as management Measures)) approach was applied to assess the causes, consequences and responses to change in a holistic way (Figure 1.1 - Elliott et al., 2017). DAPSI(W)R(M) is of relevance to MARBEFES as it can be applied to categorise indicators of environmental change (e.g., GPC-IECS Ltd., 2021) and can be applied to scenarios analysis (e.g., Smyth et al., 2015; Turner et al., 2015) and therefore can be used to support management decisions. The DAPSI(W)R(M) approach links the natural and societal domain and can be informed by the ecosystem extent and condition (as depicted within the SEEA-EA framework) being captured within the State change (of the natural environment) and their respective effect on the delivery of societal benefits reflected within the Impacts (on the environment and ultimately on human Well-being).

By extending the DPSIR scoping framework to DAPSI(W)R(M), it is possible to identify more holistic management strategies that address the linkages between different environmental problems and their Drivers and Pressures (Elliott et al., 2017). Feedback loops between the management Responses and the Drivers, Activities and Pressures are of importance, as are the effects of natural change on the system. Figure 1.1 distinguishes between the Exogenic Unmanaged Pressures (ExUP), those pressures emanating from outside the sea area being managed and whose causes cannot be managed in that particular area, but for which we have to respond to the consequences of the Pressure, such as climate change, and the Endogenic Managed Pressures (EnMP) those occurring within the management area boundary and whose causes and consequences need managing and which can be managed, for example the activity and impacts of a particular sector such as commercial fisheries.

Feedback loops through the management Responses (as measures) allow to plan regular monitoring of the impact Drivers, Pressures and therefore Activities, have on the State of the marine environment and human Welfare; taking these natural and societal elements together creates the Social-Ecological System (Smith et al., 2025). A monitoring programme based on a DAPSI(W)R(M) may be supported by tools such as Marine Natural Capital Accounting, for example (see Section 3), as it will capture time-series data over time which allows managers to monitor any effects of the implementation of management measures on the marine environment and its natural resource uses in relations to the Drivers and Pressures. Wider feedback to policy assessment, which will include the environmental and socio-cultural valuation outputs that are not included in the natural capital accounting standard (SEEA), may be provided using a more holistic marine natural capital accounting framework such as the Complementary Accounting Network (CAN) (see Section 5.1).

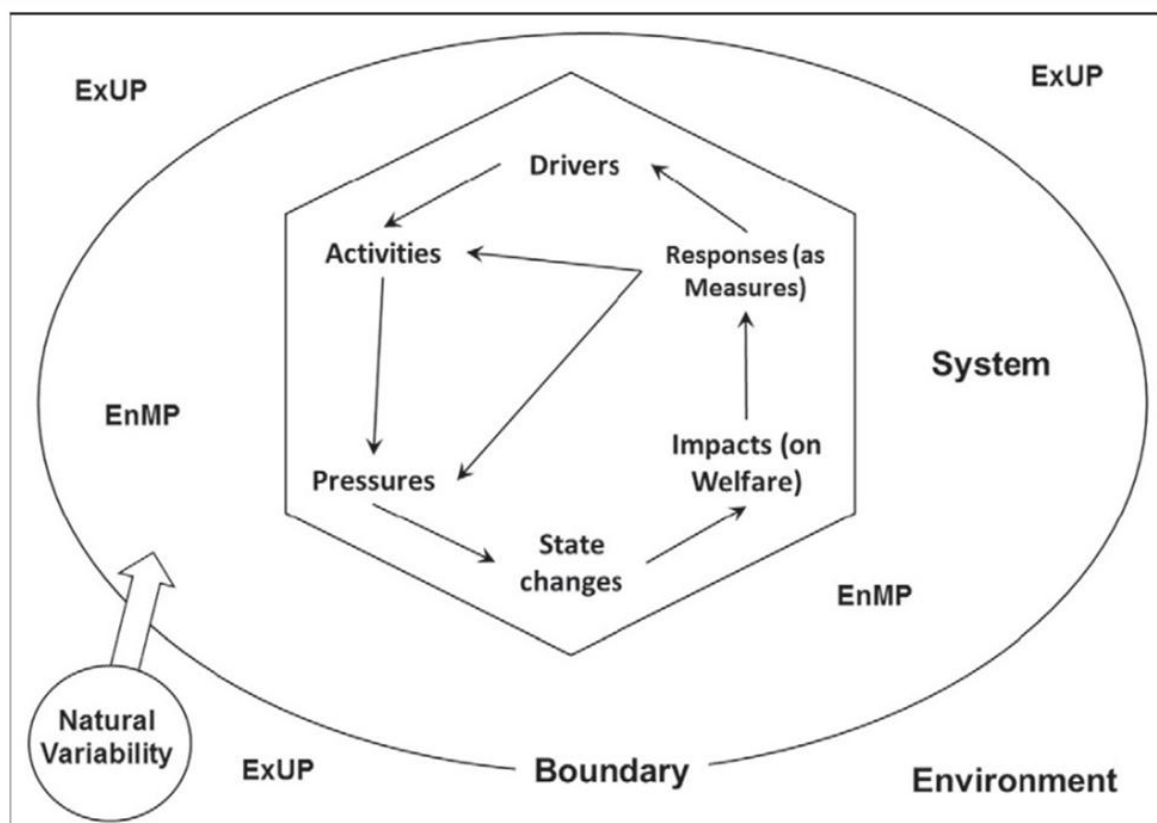


Figure 1.1: The DAPSI(W)R(M) framework as a problem structuring framework (Elliott et al., 2017). Key: ExUP = Exogenic Unmanaged Pressures; EnMP = Endogenic Managed Pressures (see text for further explanation).

Data collection, data analysis and valuation requirements (Guide 2) will all be influenced by the responses to the fundamental scoping questions. The scoping phase should lead to identifying spatial boundaries of the study area, biophysical data relevant, for example, to the condition and extent of habitats and socio-economic data. Biophysical information is used to assess the ecological health of the marine environment (Guide 1) (Ecological Value Assessment). The socio-economic information is the basic input into the Standard National Accounting (SNA) framework from which we obtain the Gross Domestic Product (GDP), an indicator to monitor how the national economy is growing in terms of economic activity and value. In this SNA framework, something is missing: the full contribution of the environment and its component ecosystems to the economy/society. In the MARBEFES project we orientate the decision support to the contribution of the marine environment (and its inherent natural capital) to the economy.

The SNA framework does not provide any information about the ecosystems and ecosystem services that underpin the economic activity of a specific country; how, for example, marine ecosystems (e.g. seagrass habitat) support wild fish stock provision. To understand the contribution of nature, the natural capital, to the economic activities taking place within a country, the SNA has been extended to include natural capital and ecosystem services accounting, in what is called the System of Environmental Economic Accounting-Ecosystem Accounting framework – SEEA EA (Guide 2).

The ‘ecosystem service’ term has been used since the 1970s (Westman, 1977) to highlight the importance of the ecosystems that underpin our society and the link between ecological functioning and economic activity. Environmental economists use the ecosystem service concept to formalise the links between nature and the societal benefits contributing to human wellbeing. Splitting ecosystem services from societal benefits is an important step for economic valuation as it is used to guard against

possible double counting of the benefits to society (Guide 2). Physical accounts require the compilation of biophysical data to link with environmental quality and ecosystem services flow (Guide 2), which can be informed by the EVA (Guide 1) and/or other data such as those collected for the Marine Strategy Framework Directive (MSFD) or Regional Sea specific assessments (such as OSPAR or HELCOM).

To produce marine natural capital accounts, nature's contribution to people (natural capital stocks and ecosystem services flows) needs to be translated into monetary terms (IPBES, 2022). Within an anthropocentric perspective, ecosystem services are the route to creating societal benefits which can be quantified in monetary units, using economic valuation methods. It is important to recognise that this is a familiar communal language, especially in finance ministries and policymaking circles (Guide 2).

Several valuation techniques aim to translate the importance of ecosystem services into monetary terms and each of them results in an economic value. For this guide, it is essential to clarify which can be used for natural capital accounting and which provides value estimates that do not fit into the conventional accounts. From this distinction, we can have exchange value and welfare value estimates. Only the exchange value can be used for building marine natural capital accounting following the SEEA-EA. The reason is that the marine natural capital accounting is an extension of SNA which has formal internationally agreed protocols and that all the indices must be in the form of market prices (known as exchange values). A section on the methods and data for building marine natural capital accounting is illustrated with examples (Guide 2).

While some societal benefits have a market and a price, others can have an impact on our well-being without the necessity to buy them (e.g., walking on a beach). For this, we can measure their impact in another way and thereby obtain more information about their relevance for us. From the economic valuation methods, we can get primary value estimates of societal benefits. When an actual onsite valuation is not possible, due to lack of time or other resources (e.g. lack of funding to obtain primary value estimates), the second-best option can be applied involving 'benefit transfer'. Benefit transfer is a set of techniques used to estimate the economic value of non-market ecosystem services by transferring the welfare measures for environmental resources, such as Willingness-to-Pay estimates, from original valuation studies ('primary studies') previously performed at other sites (the 'study sites') to the site in question (Guide 2).

When we have data from which to derive the value for each ecosystem service, we can build the accounts. Data referring to economic activity can be found in the national SNA. To move to marine natural capital accounting, we need both biophysical data and monetary valuation in exchange prices. If a BBT wants to focus on more comprehensive well-being valuation, including those ecosystem services that do not have market prices, then economic valuation in welfare value terms can be used with the appropriate valuation methods employed (Guide 2).

The non-monetary social and cultural values that relate to how people derive meaning, pleasure, education and community from the marine environment can also be estimated (Guide 3). This is important when the initial scope question is understanding the societal benefits that cannot be monetised or that are public good such as local identity, aesthetic stimulation and inspiration, cultural heritage, emotional, spiritual and wellbeing connections to nature and environmental and indigenous knowledge.

If a BBT has environmental change datasets covering ecological data (Guide 1), physical and monetary accounts estimated with exchange value data (Guide 2) and socio-cultural data (Guide 3) as well as having the value of each ecosystem service for several years (i.e. in a time series), then it is possible to use all this information in a Complementary Accounting Network (CAN) framework (Section 5).

The CAN framework is a dashboard of relevant indicators (e.g. population, GDP, quantity of pollutants in water, quantitative information about increases or decreases in an ecosystem service in exchange

and welfare value). Such indicators are required to assess the behaviour, state and trajectory of change due to natural or anthropogenic events (Atkins et al., 2015).

The 9-stage decision support procedure illustrated in this handout is important because it leads to the monitoring phase (at the end of the year) whilst providing evidence for policymakers to implement prevention, mitigation and compensation initiatives in a 'looking forward' analysis.

Each section and guide, of this handout has been produced as a stand-alone document but provided in the sequence in which it should be applied (see the Conclusions Section 5 for details).

We accept that collecting data is difficult, expensive and time-consuming! The minimum requirement that we suggest is to look for at least one year to start this journey in accounting and valuation.

2. Guide 1: Ecological Value Assessment

2.1. Introduction

This document provides a step-by-step guidance to apply the Ecological Value Assessment (EVA), a method to value nature for its own, intrinsic sake without reference to any benefits for society. The EVA guidance has undergone a series of revisions since its first design in March 2023, following the testing of the method by the BBTs and the consequent feedback received throughout the MARBEFES project (see Section 2.3 and Section 6.3). This document updates the original guidance.

The EVA method and guidance have been designed and formulated for general applicability to any marine area (including the BBT case study areas in MARBEFES). As the method allows some flexibility in its implementation (e.g., depending on data availability), additional specific guidelines have been defined for its use within MARBEFES to standardise the approach implemented by the different BBTs as much as possible. Such additional guidelines are distinguished from the generic guidance provided in each of the methodological steps/sections.

2.1.1. Aims and structure

The aim of this document is to provide guidance on the methodological protocol to undertake Ecological Value assessment (hereafter EVA) in marine areas (the term ‘Ecological Valuation’, as used in the MARBEFES WP4 description, has been replaced by Ecological Value (EV) assessment to avoid misunderstandings about the (non-monetary) nature of this approach).

A brief outline of what the EVA entails is given as background in Section 2.1.2, as reviewed from the available literature on the approach. Additional details and definitions are provided in Appendix 2.1.

Section 2.2 provides the step-by-step guidance to undertake the EVA process. For each step, the rationale behind the proposed methodology is provided, together with the methodological guidelines (often given in tabular format) and additional specific guidelines for use by the BBTs in MARBEFES.

Section 2.3 includes a preliminary overview of the method testing during the implementation by the BBTs, along with a list of Questions & Answers (Q&As) is also included (Section 2.3.2) as developed throughout the testing process. The Q&As recorded the feedback and discussions about the method and provide justification to the subsequent method revisions that have been implemented iteratively.

2.1.2 What is Ecological Value Assessment (EVA)

The concept and methodology for the assessment of the Ecological Value of an area was first developed by Derous et al., (2007a, 2007b) and has since been adapted and applied in various case studies, including the Dutch and Belgian parts of the North Sea (Derous, 2007), the Isles of Scilly archipelago (Vanden Eede, 2007), the Polish EEZ in the Baltic Sea (Węśławski et al., 2009), the Basque continental shelf (Pascual et al., 2011), the Belgian coastline (Vanden Eede et al., 2014) and the Portuguese continental shelf (Gomes et al., 2018). This guidance document builds on this literature in deriving and applying the Ecological Value assessment.

It is of note that the literature above refers to this method as (Marine) Biological Valuation with the term Biological Value (BV) used to indicate the ecosystem property that is being assessed. To ensure clarity, the terms Ecological Value (EV) and Ecological Value assessment (EVA) are used in this guidance instead (see Box 2.1); although they are conceptually equivalent to the terms in the literature, they are preferred as they account for the wider applicability of the approach to ecosystem components, such as habitats, that are not solely biological and also represent the underlying geophysical structure and hydrodynamic processes. Although habitats were less represented in the previous uses of the method, which mostly used species as features for the assessment, it was considered important to include them in this guidance.

Box 2.1 – EVA key definitions

Ecological Value (EV) is the inherent value of the qualities of an ecosystem. It is a property of the ecosystem and its biodiversity that is independent from any human association (e.g., uses, pressures or social interests) and is measured in non-monetary terms (modified from Smith and Theberge 1986, Derous et al., 2007a, Gomes et al., 2018).

Ecological Value Assessment (EVA) is the process of assigning ecological value to an area through the integration of the inherent value of the ecosystem qualities within (modified from Derous et al., 2007a, 2007b).

Rarity (R) is this is the degree to which an area is characterized by unique, rare or distinct ecosystem components (landscapes, habitats, communities, species, ecological functions, geomorphological and/or hydrological characteristics) for which no alternatives exist (modified from Derous et al., 2007a, 2007b).

Aggregation-Fitness consequences (AF):

o **Aggregation** represents any of (i) the degree to which an area is a site where most individuals of a species are aggregated for some part of the year; (ii) a site which most individuals use for some important function in their life history; or (iii) a site where some structural property or ecological process occurs at an exceptionally high level (DFO 2004, Derous et al., 2007a, b).

o **Fitness consequences** represent the degree to which an area is a site where the activity(ies)/functions undertaken by the component species there have fitness consequences, i.e., contribute significantly to the survival or reproduction of a species or population (DFO 2004).

Key definitions for the EVA approach are given in Box 2.1. Essential aspects are:

- Ecosystem qualities that do not reflect the inherent value of biodiversity (e.g., utilitarian/instrumental value related to the uses humans make of the environment) do not contribute to EV.
- EV is assessed separately for the multiple Ecosystem Components (ECs) characterising the ecosystem within the study area (e.g., species groups, habitats and habitat mosaics, ecosystem processes/functions) which are then integrated across ECs.
- EVA is a spatial assessment, i.e., the EV is assigned to spatial units (subzones) within a study area (the BBT in MARBEFES) and the EV of a subzone reflects the ecological value of that subzone relative to the other subzones in the study area.

An ecosystem has many qualities, but not all of them are required to be included in determining its inherent ecological value and therefore are not required as suitable assessment criteria for Ecological Value (see Appendix 1 for further details). The assessment criteria that have been used for Ecological Value are Rarity (R) and Aggregation-Fitness, as defined in Box 2.1.

As a measure of functional quality, the criterion ‘Fitness consequences’ partially overlaps with functional habitats identified under ‘Aggregation’. Therefore, the merging of these two criteria into one (Aggregation-Fitness consequences) has been suggested, as undertaken in Derous et al. (2007b and derived applications).

Biodiversity is also a key inherent property of an ecosystem, as advocated in the literature on Biological Valuation. However, this is not included in the assessment here as a standalone criterion,

but rather attributes of biodiversity are used in the measurement of the metrics characterising the above-mentioned criteria (see Section 2.3 for details). This usually refers to taxonomic structural biodiversity but in current literature, it also includes genetic diversity, functional diversity, physiological diversity, etc. (e.g., Thrush et al., 2021).

The operationalisation guidance of the EVA process is given in Section 2.2.

EVA is a method that is likely to be used by the scientists or science advisor to a policy implementer or policy maker rather than the latter themselves (see Borja et al., 2025).

2.2. EVA Protocol – a step-by-step guide

2.2.1 The EVA process

The EVA process can be separated into three main phases, with several steps in each phase:

Phase 1: Define the study system and its boundaries

This phase includes defining the spatial system (study area and its subdivision in spatial assessment units or subzones) and identifying the relevant ecosystem components, the data available for the assessment and the time over which they were determined.

Phase 2: Assess the EV

EVA then involves answering a set of questions which characterise key qualities of the ecosystem relevant to EV. The questions are answered by calculating a set of metrics (using specific algorithms) separately for each ecosystem component (EC) in each subzone and then integrating the results across metrics (questions) and ECs. The resulting EV is mapped across subzones in the BBT.

Phase 3: Assess Confidence

Different sources of uncertainty are considered and combined to determine the confidence in the EV assigned to a subzone. The resulting confidence is mapped across subzones in the study area.

Figure 2.1 gives the graphical conceptual model of the EVA process.

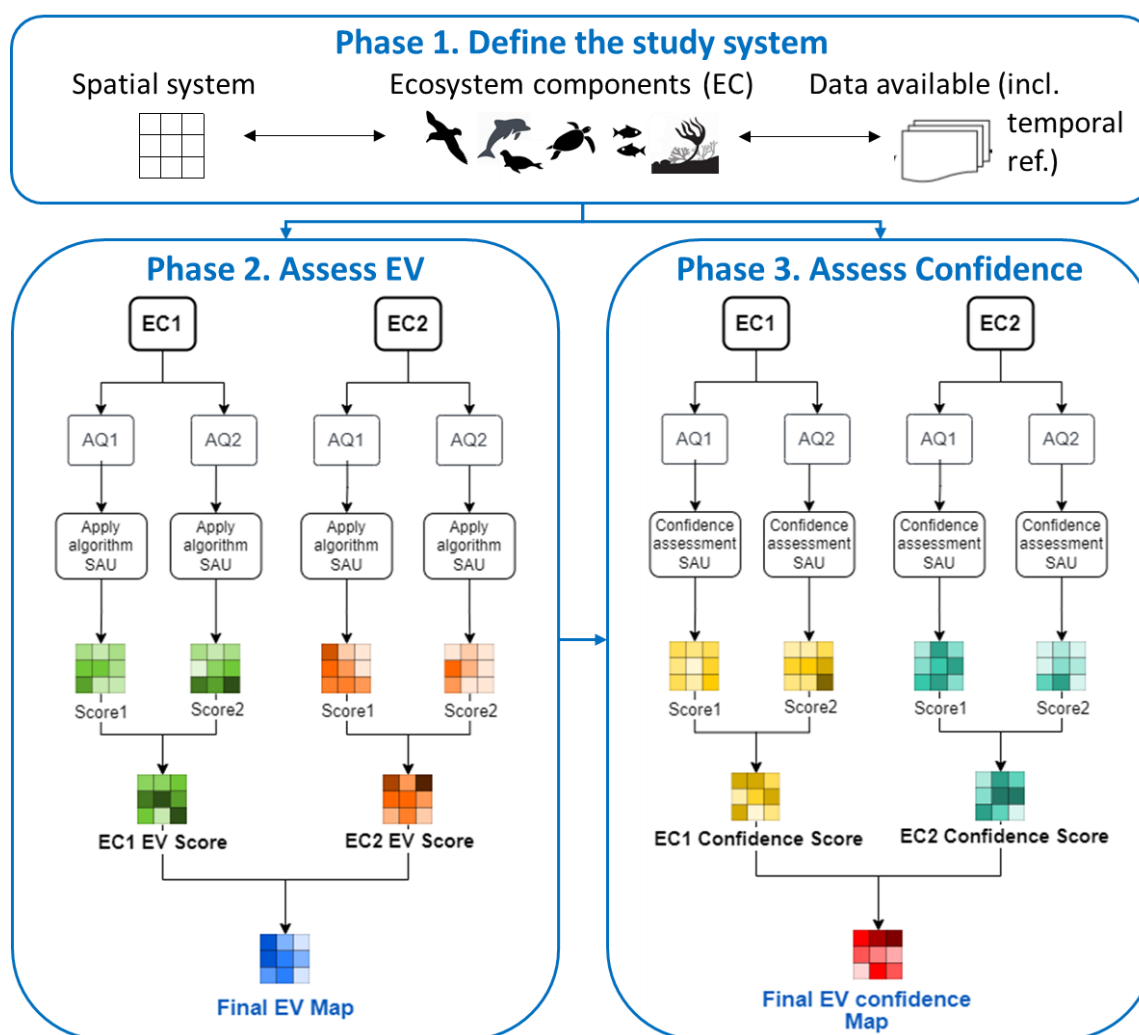


Figure 2.1: The EVA process (AQ = assessment question; SAU = spatial assessment unit or subzone; EV = ecological value).

Detailed guidance on each step of the process is provided in the sections below.

2.2.2 Define the study system (Phase 1)

2.2.2.1 Spatial system and Ecosystem Components

General principles

The EVA should be applied to as many components of the ecosystem as possible (both structural and functional/processes) (Deros et al., 2007a, 2007b) and all levels of biological organisation (from individuals and species populations, through communities to ecosystem level) should be assessed (if data are available).

The study area should be clearly delineated and divided into subzones that constitute the spatial assessment units. Given the nature of the EV, subzones should be identified based on their ecological and physical relevance to the ECs being considered in the assessment (hence the combination of spatial system and ECs in this section of the guidance). Practicality aspects should also be considered (e.g., data availability, spatial scale of sample data, size of the study area) to allow the comparison of the EV between defined subzones.

Deros et al. (2007a, 2007b) recommend the use of habitat classification schemes (e.g., EUNIS) or marine landscapes for zoning, which should incorporate bottom and pelagic habitat features.

However, this might lead to high variability in the size between subzones, within and between different BBTs, depending on the variability in habitat distribution and in the habitat classifications used. Such variability may create issues in the standardisation of the assessment across zones and study areas.

Alternatively, a zoning approach using a regular raster grid to divide the study area into subzones (grid cells) may be used (Derous et al., 2007a, 2007b). The spatial resolution of the grid cells (i.e., the size of the subzones) may be variable depending on the EC under consideration, responding to ecologically relevant reasoning (Pascual et al., 2011) as detailed below.

The selection of the resolution should consider:

- The EC being assessed (smaller grid sizes should be chosen for benthic ecosystem components (habitats or species) which have no or low mobility, while larger grid sizes are more appropriate for highly mobile species such as fishes, birds and sea mammals).
- The compatibility of the selected grids, which, when combined, should align between ECs. This can be achieved by nesting smaller grids (e.g. defined for benthic ECs) within larger grids (e.g. defined for highly mobile ECs) and it is necessary to integrate the EV across ECs spatially, later in the process.
- Data availability (data poor areas/components might result in lower resolutions, to allow good coverage of the study area - however, too-large grid cells could result in the loss of site-specific information).

Other types of spatial zonation, e.g., based on spatial management units (MPAs, ICES rectangles, etc.), or impact areas in the marine environment, should be avoided for the spatial assessment of EV as they introduce a human element in the assessment which is outside the scope of the EVA. However, these additional zones can be applied *a posteriori*, to compare the spatial distribution of EV (based on inherent value of biodiversity) with other spatially meaningful representations of the marine system (e.g., associated with other values that may be attributed to biodiversity). This comparison may be undertaken by aggregating or disaggregating the assessed subzones into these zones. The reason for this comparison is outside the scope of the EVA and will depend on specific interests (e.g., to validate the designation of conservation areas, identify areas of higher risk or concern where anthropogenic impacts may occur).

Additional spatial units may be defined and the spatial results of the assessment (in SAUs) may be spatially aggregated/disaggregated into these additional polygons through GIS. These additional units may be defined at the scale of management area (ecoregion, governance scales, ICES zones, protection areas, etc.), depending on the specific interest within the study area (e.g., to compare the EV of protected/unprotected areas, or differently impacted areas within a marine area).

Method guidance

The use of a regular grid is the most applied zonation approach in the site-specific applications of EVA (as mentioned in Section 2.1.2). The scales applied in those studies have been considered in providing the guidance below (Tables 2.1 and 2.2).

Grid cells may be of square or hexagonal. The use of hexagonal grids is recommended as, compared to square grids, they allow more flexibility and better spatial representation of marine biological components and information. They are also more suitable for the evaluation of different connectivity measures. Their projection and visualisation are also more precise and aesthetically pleasing, which is a criterion also valued in this application. Hexagonal meshes can be produced both in GIS software (ArcGis or QGis) and in R or Matlab. However, it is acknowledged that the use of square grids may be easier (e.g., to aggregate/re-scale data from grids of variable spatial resolution) and therefore the choice is left to the user, based on their GIS skills and level of expertise.

The list of the Ecosystem Components (ECs) that can be for the assessment is given in Table 2.2. It is of note that previous versions of the biological valuation approach only use species groups (defined through a combination of taxonomic and functional criteria, e.g., birds, marine mammals, demersal fish) as ECs, with the assessment questions mostly focusing on different aspects of the species diversity within these groups (this is also used as a proxy for functions; see Section 2.2.3). This emphasis may be because most current monitoring programmes undertaken at the broader scale (hence with more chances to provide better coverage of a broad study area) focus on structural aspects of species biodiversity.

This EVA guidance also includes habitats as ECs to account for diversity at the higher level (communities, assemblages, biotopes, ecosystems). Where there is possible overlap (e.g., between macrobenthos and benthic habitats assessments), this has been considered in the formulation of the algorithms to answer the assessment questions (see Section 2.2.3).

Whether or not an EC is used for the assessment of a study area will depend on the relevance of an EC in the area (e.g., marine reptiles might not be a relevant EC if they do not occur in the area or the region) and on the availability and distribution of the evidence-base for the assessment. It is of note, while EVA assigns value based on evidence of the ECs distribution within the study area, for some ECs (e.g., highly mobile species), this may be the result of events affecting populations outside the assessment area.

Additional guidelines for use in MARBEFES: In MARBEFES, the study area is synonymous to a specific BBT area. For standardisation of the EVA implementation across BBTs, the minimum requirement is that the EVA is undertaken using the benthic ECs - macrobenthos and benthic habitats, with EUNIS Level 3 classification used to identify habitat types. The use of a hexagonal grid is also required (see video tutorial on how to create hexagonal grids using QGIS at <https://nc-marbefes.iopan.pl/nextcloud/s/retGAGYs97B6fRt>).

Table 2.1: EVA guidance: Spatial system.

Spatial System	
1.	Identify the study area by delineating its boundaries in GIS.
2.	Divide the area into subzones by applying a regular raster hexagonal (or square) grid. Grids of different resolution are defined for different ECs (see Table 2.2) and grids of finer resolution should be nested within those of coarser resolution. Each grid cell will constitute a subzone, i.e. the spatial unit for which the EV is calculated (for the relevant EC).
3.	Any data-based evidence for the assessment is to be assigned to each cell within the grid. Monitoring data points/tracks are the most common type of data-based evidence, but spatial interpolation/extrapolation of these data (e.g., based on spatial or habitat suitability models) may also be used where appropriate to improve the spatial coverage of the grid cells in the study area. Expert judgment may also be used for this, where deemed appropriate. If expert judgment is used when/where data are not available, this must also apply to individual subzones distinguished by the grid cells. Where there is no evidence (of any nature) supporting the assessment in a subzone, no value (e.g., for a metric for an EC) is assigned to that subzone (hence resulting in a spatial gap in the mapped results).
4.	Additional zonation can be applied by identifying natural zones (e.g., regions, depth strata) or management zones (e.g., MPAs, impact areas). These zones will only be used <i>a posteriori</i> , i.e., EV is calculated for the grid as specified in point 2 above and the subzones defined that grid can be aggregated into larger natural or management zones for other assessments.

Table 2.2: EVA guidance: Ecosystem Components.

Ecosystem Components		
EC type	Ecosystem component (EC)	Subzone size (see Figure 2.2)
Species group	Seabirds	3 km
	Marine mammals and/or Marine reptiles They can be assessed together or as separate ECs, depending on relevance and data availability)	3 km
	Fish Likely differentiated into ECs for Demersal fish and Pelagic/nektonic fish given different nature of the evidence base (e.g., different monitoring methods/programmes). Pelagic molluscs (e.g., squids) may be included in this EC as they are sampled in the same programmes.	3 km
	Zoobenthos Likely differentiated into ECs for Macrobenthos, Epibenthos, Hyperbenthos, Meiobenthos and possibly further between hard and soft substrata (where appropriate), given different nature of the evidence base (e.g., different monitoring methods/programmes).	250 m
	Phytobenthos Likely differentiated into ECs for Algae and Plants (seagrasses, saltmarsh) and further between hard and soft substrata (where appropriate), given different nature of the evidence base (e.g., different monitoring methods/programmes).	250 m

Ecosystem Components		
EC type	Ecosystem component (EC)	Subzone size (see Figure 2.2)
	Plankton Likely differentiated into ECs for Zooplankton and Phytoplankton	3 km
Habitats	Marine benthic habitats	250 m or 3 km*
	Marine pelagic habitats Possible differentiation based on vertical/depth zonation	3 km
* The grid cell size may vary depending on the scale at which the habitat types are defined and used in the assessment (e.g., for benthic habitats broadly defined, for example using EUNIS Level 3 habitat type classification, the use of the larger grid cell size is suggested to allow the habitat diversity within a grid cell to be adequately represented).		

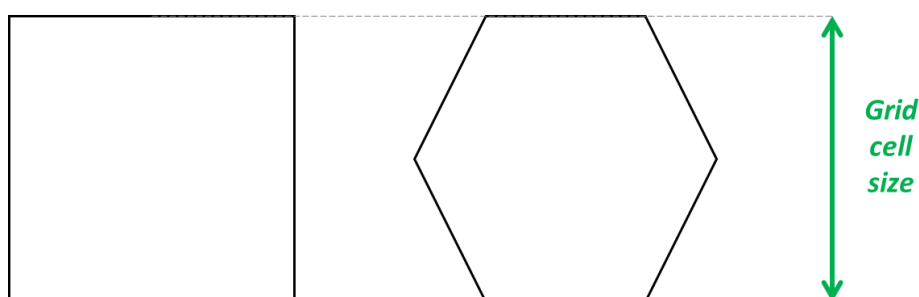


Figure 2.2: Subzone (grid cell) size measured for a square or a hexagonal grid cell.

2.2.2.2 Evidence base

Type of evidence

The application of the EVA to case studies in the literature (see Section 2.1.2) mostly relies on the use of monitoring data available for the study area and the EVA methodology used (assessment questions and associated algorithms) reflects this.

To allow wider applicability of the method to case study areas (or ECs) for which data may be sparse or lacking, the methodology has been modified to include other data-derived evidence, for example interpolated or modelled maps that can be derived from the monitoring data or that can be obtained from publicly available data layers (e.g., EUNIS habitat maps, Copernicus oceanographic layers modelled from optical characteristics of the sea, EMODnet). Evidence based on expert judgment may also be used where data are lacking, as necessary and if appropriate. The type of evidence base used for the assessment will influence the confidence in the resulting EV (see Section 2.2.4).

Examples of typical data-based evidence (directly from monitoring or derived through modelling) are given in Table 2.3 for each of the ECs. As data requirements for the assessment may also vary depending on the assessment questions (e.g., some require species presence/absence, other estimates of density), additional details on this are given in Section 2.2.3. Additional guidelines for use in MARBEFES: The use of monitoring data should be prioritised over model-based evidence or expert judgement for the assessment in the BBT areas.

Table 2.3: EVA guidance: Examples of data-based evidence types for the different ECs.

Ecosystem components (EC)	Typical monitoring data sources	Model-based data
Seabirds	Boat/airplane-based systematic visual surveys (transect-base) Breeding colonies visual surveys (coastal)	Inter/extrapolation across transect area with Spatial Analysis
Marine mammals Marine reptiles	Boat/airplane-based systematic visual surveys (transect-base) Opportunistic sightings	Inter/extrapolation across transect area with Spatial Analysis
Demersal fish	Bottom/beam trawl surveys (tow line) from broad scale survey programmes (e.g., ICES DATRAS or eggs and larvae surveys) or localised / site specific surveys (which may also include visual or acoustic surveys)	Habitat suitability/species distribution models (some may be valid for species as a whole e.g., sandeel; but note that most are developed for Essential Fish Habitats, hence they refer to a particular life stage/function rather than to the species as a whole)
Pelagic fish	Pelagic trawl surveys (tow line) from broad scale survey programmes (e.g., ICES DATRAS or eggs and larvae surveys) or localised / site specific surveys (which may also include acoustic surveys)	As for demersal fish
Epibenthos	Bottom/beam trawl surveys (tow line)	As for demersal fish (e.g., crustaceans)
Macro(zoo)benthos	Grab surveys, Coring (point data)	Habitat suitability models for individual species or communities (note that EUNIS habitat classification maps are often the result of community model predictions and can be used as proxy for macrobenthos, depending on which metric is to be measured)
Macro(phyto)benthos (incl. macroalgae and/or plants)	Quadrat surveys, visual census (based on transects/point or aerial photography).	
Meiobenthos	Grab surveys, Coring (point data)	
Zooplankton	Plankton nets (tow line), CPR	
Phytoplankton	Plankton nets (tow line), CPR, remote sensing (which may also include acoustic surveys) (where species ID and a measure of abundance can be obtained)	
Marine benthic habitats	Habitat maps, e.g. based on EUNIS broad scale habitat (Level 3) classification or biotopes (Level 4) (EUNIS, 2022)	
Marine pelagic habitats	Habitat maps, e.g. based on EUNIS broad scale habitat (Level 3) classification or biotopes (Level 4) (EUNIS, 2022)	

Spatial availability of the evidence

The evidence available in a study area (both from expert judgment and data-based) are likely to have variable coverage of the subzones. This could be shown, for example, as the percentage of subzones with evidence available for the assessment (for each EC) and, where monitoring data are used, the data availability may be measured as the number of data points (or observations) used in each

subzone to calculate a metric in response to an assessment question. The latter (data availability) is one of the sources of uncertainty that is considered in the confidence assessment (see Section 2.2.4).

Spatial gaps may occur in subzones and for ECs for which there is no evidence available to support the assessment. As a result, these subzones may remain unassessed (for EV overall, or limited to a specific EC or to a specific question/metric for the EC) if there are no other alternative sources of evidence for their assessment. However, spatial interpolation/extrapolation of the available data over the wider study area (e.g., based on spatial or habitat suitability models for certain types of data; see examples in Table 2.3) may be used to increase the spatial coverage of the assessment. In the absence of data, expert judgment may also be used, if deemed appropriate.

Whether an EVA is undertaken for a subzone and, if so, what evidence is used and its availability, are considerations that contribute to the confidence associated with the resulting EV (see Section 2.2.4).

Temporal reference of the evidence

The evidence available for an EC in a study area may also vary in terms of its temporal reference (time period covered, inter- or intra-annual frequency of the data collection, etc.). This should always be recorded for any type of evidence used in the assessment.

Datasets from multiple monitoring programmes and studies in multiple years can be combined to cover the full study area, as well as seasonal and interannual variabilities, assuring the availability of maximum information for each EC. However, when combining data from multiple sources, representativity and consistency (e.g., in the sampling regime, methods, seasonality) should be considered for the standardisation and harmonisation of the data into an integrated database and for the comparability across subzones in the study area. Therefore, the EV of each subzone is a relative value given the data available at the time. Data from different time periods may differ in methods, precision/accuracy and analysis and so the data may have to be harmonised to the lowest common denominator. As more (or better) evidence becomes available, the EVA for the study area may be revisited and refined.

The temporal coverage of the datasets should be limited to avoid using data with temporal trends and to avoid regime shifts caused by changing oceanographic conditions due to climate change (Pascual et al., 2011, DFO 2011). As a guideline for use in MARBEFES, a maximum of 10 years is suggested to define the time period for the data used in the EVA for the BBTs. As an example, in comparison, it is suggested that Ecologically and Biologically Significant Areas (EBSAs) are reassessed every 5 years (DFO, 2011).

2.2.3 Assess EV (Phase 2)

General principles and approach

The EVA is based on the assessment of the ecosystem in an area against criteria such as Rarity (R) and Aggregation-Fitness consequences (AF), as previously indicated in Section 2.1.2 (Box 2.1).

To assess each criterion at different ecological organizational levels, a set of assessment questions should be addressed by examining all available data and the presence of certain processes and the ecological structure in the area. Table 2.4 suggests the possible topics on which these questions may focus, as mapped against the EVA criteria and the different biodiversity levels (species, community, habitat, etc). It should be noted that this is not an exhaustive list of topics, nor is there the requirement to answer all these questions, as practicality aspects (e.g., data availability) come into play when operationalising the EVA process. Specific categories of features to be identified within an ecosystem component (e.g., rare species, habitat-forming species or ecologically significant species) should be selected based on expert judgement, supported by the grey and published literature.

Each assessment question needs to be translated into a metric to be measured following a specific mathematical algorithm, that will be applied to each ecosystem component at the subzone level

allowing subzones to be quantitatively assessed relatively to each other (Figure 2.1). The algorithms can be developed for different types of data, such as densities or presence/absence for species components and aim at assigning a standardised numeric value (ranging typically between 1 and 5) to each metric for each question answered for each EC.

The numeric results thus obtained are then integrated first across all metrics/questions within each ecosystem component, so that an EC-specific EV is assessed at the subzone level and can be mapped across subzones in the study area (Figure 2.1). Average (Pascual et al., 2011) or sum (Deraus 2007) has typically been used to aggregate the metric standardised values to obtain the EV map for each ecosystem component, with equal weight given to all assessment questions. After adding the values Derous (2007) created 5 classes from 1 to 5 to re-standardise the EV into the original 1-5 range of its component metrics.

The last step of the EVA process is the aggregation of the EC-specific EV values across ecosystem components and into an overall EV that can be mapped across subzones in the study area (Figure 2.1). Averaging is typically used for this step (Deraus, 2007, as shown in the method implementation in Section 1.2).

Table 2.4: Assessment topics related to the different structures and processes of biodiversity and coupled to the proposed EVA criteria (adapted from Derous, 2007). The asterisk identifies topics/questions that were indicated under the Rarity criterion in Derous (2007) and subsequent applications, but that were considered here more appropriate under the criterion 'Aggregation-fitness consequence'. This is because their main value derives from the function that they provide in supporting the ecosystem and its components (e.g., as a pivotal element of a food web, or forming a structured habitat that provide key support to other species) rather than for their rarity.

Organizational level of biodiversity		Criteria	
		Rarity	Aggregation-Fitness consequences
Structure	Species/ Population level	• High number of rare species • High abundance of rare species	• High percentage of a species population • High abundance of a certain species population • High counts of many species • High densities of species (otherwise restricted distribution in the study area) • High abundance of focal species • Presence of habitats formed by keystone species (habitat forming species and ecological significant species)* • Presence of indicator species or indicator conditions* • High abundance of umbrella species* • Presence of ecological significant (keystone) species with controlling influence on other species present*
	Community level	• Presence of distinctive/unique communities (with respect to species richness and abundance)	• High species richness • Presence of species living in symbiosis with each other

Organizational level of biodiversity		Criteria	
		Rarity	Aggregation-Fitness consequences
		• Presence of endemic species • Presence of unique biomes • High level of ecological heterogeneity	• High total biomass
	Ecosystem level	• Complex topography or seabed morphology • High substrate diversity • Subzone is outstanding example representing significant geological processes in the development of landforms • Presence of unique/distinctive ecosystems • Presence of unique/distinctive oceanographic features	• Presence of oceanographic features that cause species to aggregate (e.g., natural refugia?) • Subzone is critical for nutrient cycling*
Processes	Species/Population level	• Species retention (high/low)	• Existence of migration routes • Existence of refuge areas used in adverse conditions • Existence of wintering/resting/feeding sites • Existence of key sites for reproduction (spawning/breeding) • Existence of key sites for recruitment (nursery/rearing)
	Community level		• Presence of species living in mutualism with each other • High natural productivity
	Ecosystem level	• Presence of unique distinctive oceanographic processes • Presence of upwelling areas	• Presence of oceanographic processes causing species to aggregate (e.g., nutrient retention, upwelling)

Method guidance

The approach described above is intended to be flexible, as it allows for including a varying number of ecosystem components and Assessment Questions (depending on data availability for each component). However, to operationalise and standardise the approach, a selected set of Assessment Questions (AQs), translating into metrics and algorithms to measure them, have been suggested (Table 2.5 and Table 2.6). A decision system flow-chart has also been designed (Figure 2.3) to support the user in the selection of the AQs relevant to their assessment.

The AQs have been proposed and algorithms for the AQ calculation and their selection formulated (modified from those applied in the literature on EVA) considering the following:

- The AQs and associated algorithms are designed to be applied separately to each of the ECs selected for the assessment and defined based either on species groups (taxonomic and traits-based) and habitats (see also Section 2.2.2.1). Species and habitats are referred to as ‘features’ of an EC in this guidance. The original EVA protocol applied in the literature was operationalised only for species groups ECs (e.g. macrobenthos and birds in Vanden Eede et al., 2014), accounting for aspects of biodiversity at the species assemblage level. In this guidance, it has been extended to include habitat ECs as well, to improve the coverage at the

community and ecosystem levels in the assessment. Where there is possible overlap in the assessments (e.g., between macrobenthos and benthic habitats), this has been considered in the formulation of the algorithms to answer the assessment questions to avoid double-counting (see footnotes in Table 2.5).

- AQs are provided which require to be addressed using either qualitative (e.g., presence/absence) or quantitative data (e.g., species density, habitat extent). A qualitative and a quantitative version of the same question (e.g. AQ1 and AQ2) has been defined for most questions. Where only presence/absence data are available for the species or habitat types characterising the EC, the user would only be able to measure the qualitative AQs. If quantitative data are available, the quantitative AQs should be prioritised (see Figure 2.3). In the latter case, a higher confidence will be associated with the EVA (as Data Reliability will increase; see Section 2.2.4).
- Unlike the EVA protocols from the literature, the option is given here of using expert judgement to derive the metric values in the subzones of a study area where there are no data-based products (quantitative or qualitative) for an EC. This option is an alternative to not being able to undertake the EVA through a lack of data. Therefore, the user can decide whether the assessment (of a metric, for an EC, in a subzone) is possible and which evidence-base is most appropriate or available. Whichever option is taken, this should be recorded and the confidence assessment will take it into account (e.g., with higher confidence weight associated with metrics measured on data, compared to expert judgement; see Section 2.2.4).
- With regards to the questions on rare features (AQs #1-6), rarity has been defined at different spatial scales, from local to national and regional (the latter possibly including the international scale). Most of the algorithms applied for EVA in the literature accounted for rarity at the local scale (i.e., with reference to the occurrence of the species within the study area); this was probably as the EVA protocols were data-driven and data are more likely to be available for a specific study area as opposed to the wider regional or national waters covered by the study area. Consideration of rarity at the regional and national scale has been included in this guidance, with the option of using data (if available) or alternative evidence sources (see Table 2.6). It is emphasised that a species or habitat may be rare at one scale but not at another (e.g., a species may be rare in the study area, but not in the larger region where the study area occurs). The more scales that are considered in the assessment of rare features, then the higher the confidence associated with the EVA (due to higher Data Reliability; see Section 2.2.4).

Despite the protocol adaptations mentioned above, it is acknowledged that some aspects of biodiversity (e.g., genetic diversity, processes) are still not adequately represented in the AQs. However, considerations of practicality are also made, in that the proposed AQs have data requirements that most likely match the data that are collected and available from small and large scales monitoring programmes (e.g., for MSFD and other EU directive compliance), whereas other types of data (e.g., genetic biodiversity, process rates) may not yet be available consistently within an EC or at the spatial scale necessary for the EVA within the study area.

The final computed EV is essentially a multi-metric index and, as such, it has the flexibility of being easily adapted to the specific user's needs and capabilities. Where possible, it is aimed to standardise the set of possible questions assessed, but acknowledging that not all questions may be answered in all areas (this is captured by the confidence assessment; see Section 2.2.4).

Additional guidelines for use in MARBEFES: As a minimum, BBTs should answer the question(s) accounting for rarity at the local scale (AQs #1 and #2), but, where possible, regional and national rare features should also be assessed (AQs #3-6).

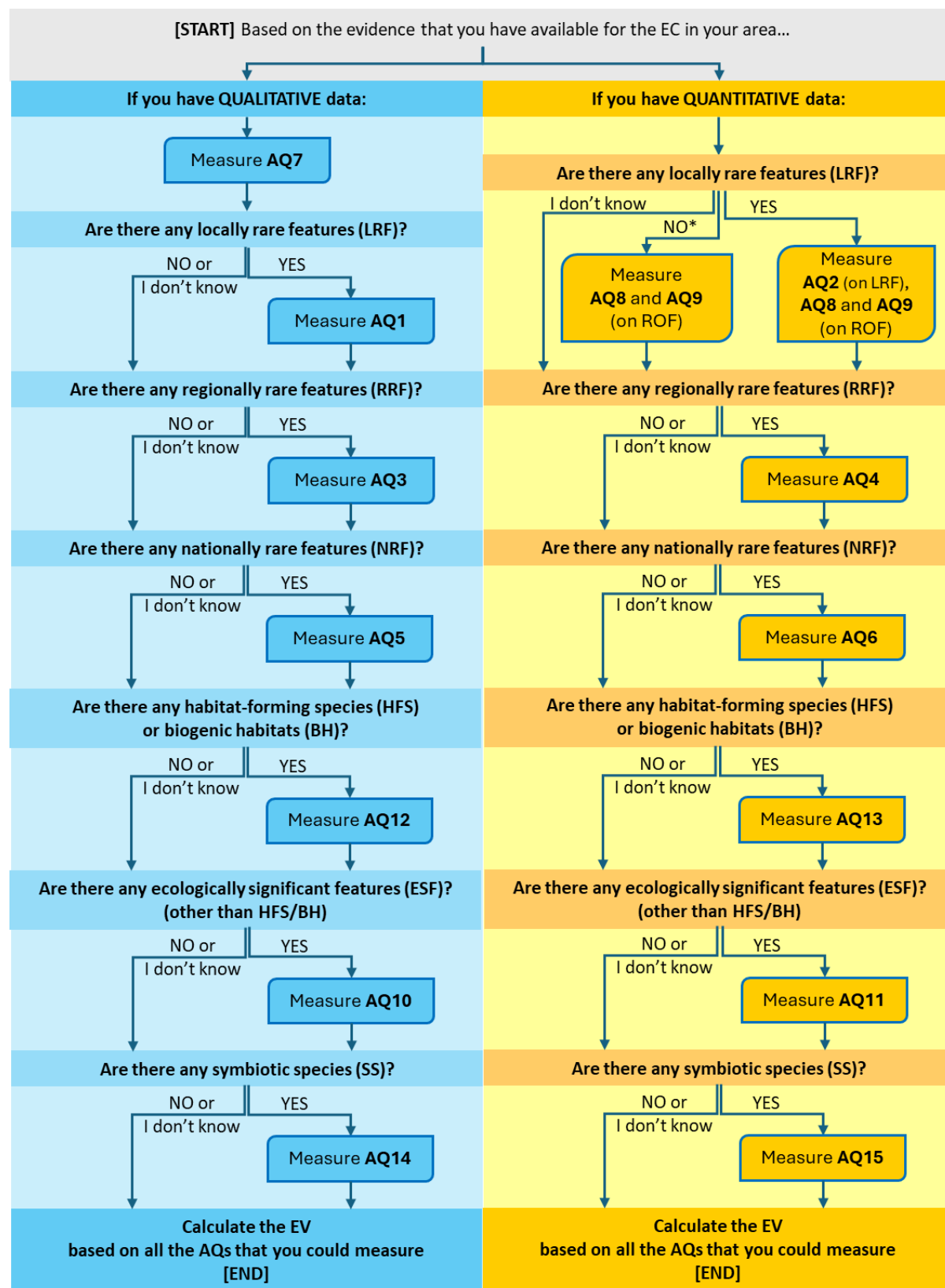


Figure 2.3: Decision system flow-chart supporting the user in selecting the AQs (metrics) to measure.

Table 2.5: EVA guidance: Proposed assessment questions (draft) mapped against the EV criteria and with respective metrics and algorithms to be applied to answer the question. Data requirements are also indicated. Where reference to feature selection criteria is given, see Table 2.6 for guidance and evidence sources. Notes are given at the bottom of the table.

AQ#	Assessment question (AQ)	EV criterion	EC	Metric name	Algorithm	Data required	Notes
1	Is the subzone characterized by the presence of many locally rare features?	R	All ECs	LRF (qual) Locally rare features (species or habitats), qualitative	1) Select LRF (see feature selection criteria) for the EC of interest. 2) Create 2 occurrence classes for the LRF in a subzone with value of 0 where it is absent and value of 5 where it is present. Leave blank where presence or absence cannot be ascertained (e.g., no evidence coverage for the LRF in a subzone). 3) Repeat steps 1 and 2 separately for each LRF in the BBT, then average the values across LRFs in every subzone. The resulting metric for this AQ will be a value between 0-5 (or blank where not assessed).	Spatial distribution (presence/absence) of species/habitats in the BBT area based on monitoring data (sample data/observations) or data-based model/spatial predictions allowing wider coverage of the BBT (e.g., spatial interpolation of data, maps from logistic regression model using relevant environmental predictors, EUNIS habitat maps). Where data-based evidence is not available, expert judgment may be used to assign class scores to subzones based on presence or absence of the species/habitat.	
2	Is the subzone characterized by high abundance of many locally rare features?	R	All ECs	LRF (quant) Locally rare features (species or habitats), quantitative	1) Select LRF (see feature selection criteria) for the EC of interest. 2) If appropriate and needed, interpolate abundance data for the feature to the chosen subzones. 3) Extract the abundance value (or proxy for it) for the LRF in each subzone (<i>LRF-abundance</i>). 4) Assign an <i>LRF-value</i> to each subzone by rescaling <i>LRF-abundance</i> to values between 0 and 5 using the following formula: $LRF-value = 5 * LRF-abundance / maximum\ LRF-abundance,$ where <i>maximum LRF-abundance</i> is measured from the range of abundance values for the specific LRF (species or habitat) obtained across subzones in the	Spatial distribution of the abundance of the feature (species/habitat) in the BBT area. Abundance of a feature in a subzone may be measured as density of the species, spatial extent/relative coverage of the habitat (provided the habitat base map is at higher spatial resolution than the grid identifying the subzones; if not, then the metric LRF (qual) is only used for habitats) or other suitable proxy (depending on evidence available). Feature abundance may be obtained from monitoring data (sample data/observations) or data-based model/spatial predictions allowing wider coverage of the BBT (e.g., spatial interpolation of data, maps from regression model using relevant environmental predictors, EUNIS habitat maps). Where data-	(1)

AQ#	Assessment question (AQ)	EV criterion	EC	Metric name	Algorithm	Data required	Notes
					BBT. Leave blank (no <i>LRF-value</i>) where the <i>LRF-value</i> cannot be ascertained (e.g., no <i>LRF-abundance</i> available in a subzone). 5) Repeat steps 1 to 4 separately for each LRF in the BBT, then average the values across LRFs in every subzone. The resulting metric for this AQ will be a value between 0-5 (or blank where not assessed).	based evidence is not available, expert judgment may be used to assign an LRF-value (between 0-5) to subzones representing species density or habitat extent (spatial coverage) within.	
3	Is the subzone characterized by the presence of many regionally rare features?	R	All ECs	RRF (qual) Regionally rare features (species or habitats), qualitative	The algorithm for AQ3 [RRF (qual)] is the same as for AQ1 [LRF (qual)] but applied to RRF instead of LRF.	Data requirements as for LRF (qual), but for RRF	
4	Is the subzone characterized by high abundance of many regionally rare features?	R	All ECs	RRF (quant) Regionally rare features (species or habitats), quantitative	The algorithm for AQ4 [RRF (quant)] is the same as for AQ2 [LRF (quant)] but applied to RRF instead of LRF.	Data requirements as for LRF (quant), but for RRF	(1)
5	Is the subzone characterized by the presence of many nationally rare features?	R	All ECs	NRF (qual) Nationally rare features (species or habitats), qualitative	The algorithm for AQ5 [NRF (qual)] is the same as for AQ1 [LRF (qual)] but applied to NRF instead of LRF.	Data requirements as for LRF (qual), but for NRF	
6	Is the subzone characterized by high abundance of many nationally rare features?	R	All ECs	NRF (quant) Nationally rare features (species or habitats), quantitative	The algorithm for AQ6 [NRF (quant)] is the same as for AQ2 [LRF (quant)] but applied to NRF instead of LRF.	Data requirements as for LRF (quant), but for NRF	(1)

AQ#	Assessment question (AQ)	EV criterion	EC	Metric name	Algorithm	Data required	Notes
7	Is the number of features in the subzone high?	AF	All ECs	FN (qual) Feature (species or habitat) number, qualitative	1) Consider all species or habitats for the selected EC in the BBT. 2) Create 2 occurrence classes for each species/habitat in a subzone with value 0 where it is absent and value 5 where it is present. Leave blank where presence or absence cannot be ascertained (e.g., no evidence coverage in a subzone). 3) Average the values across the species/habitats within an EC for the subzone. The resulting metric for this AQ will be a value between 0-5 (or blank where not assessed).	Data requirements for LRF (qual), but for all features relevant to the EC.	(2)
8	Is the subzone characterized by high counts of many species?	AF	All ECs	ROF1 (quant) Regularly occurring features (species or habitats), quantitative	The algorithm for AQ8 [ROF1 (quant)] is the same as for AQ2 [LRF (quant)] but applied to ROF instead of LRF.	Data requirements as for LRF (quant), but for ROF.	(1)
9	Is the abundance of certain species very high in the subzone?	AF	All ECs	ROF2 (quant) Regularly occurring features (species or habitats), quantitative	1) Select ROF (see feature selection criteria) for the EC of interest. 2) Determine the mean abundance (or proxy for it) of the individual ROF for the whole study area (=X) 3) Calculate the mean abundance (or proxy for it) of the individual ROF in every subzone (=Xi). Leave blank where abundance cannot be ascertained. 4) Calculate the ratio Xi/X for the individual ROF in each subzone. 5) Determine the 5% subzones with the highest ratio (across all subzones where data for the individual ROF exist, including those where the data say that the ROF is absent;	Data requirements as for LRF (quant), but for ROF.	(1)

AQ#	Assessment question (AQ)	EV criterion	EC	Metric name	Algorithm	Data required	Notes
					these are also the 5% subzones with the highest abundance X_i). Calculate the percentage of the abundance (X_i) of the ROF that occurs in these 5% most important subzones ($=Y$) 6) Determine in how many subzones the individual ROF occurs ($=Z$, i.e. the number of subzones where $X_i > 0$). 7) Calculate the ratio Y/Z which is the aggregation coefficient for the ROF. 8) Multiply the ratio Y/Z with the ratio X_i/X for the ROF (<i>ROF-unscaled</i>). Leave blank where the metric could not be calculated. 8) Assign a <i>ROF-value</i> to each subzone by rescaling <i>ROF-unscaled</i> to values between 0 and 5 using the following formula: $ROF\text{-}value = 5 * ROF\text{-}unscaled / \text{maximum } ROF\text{-}unscaled,$ Where <i>maximum ROF-unscaled</i> is measured from the range of <i>ROF-unscaled</i> values obtained across subzones in the BBT. Leave blank (no <i>ROF-value</i>) where the <i>ROF-value</i> cannot be ascertained. 10) Repeat steps 1 to 9 separately for each ROF in the BBT, then average the values across ROFs in every subzone. The resulting metric for this AQ will be a value between 0-5 (or blank where not assessed).		

AQ#	Assessment question (AQ)	EV criterion	EC	Metric name	Algorithm	Data required	Notes
10	Is the presence of ecologically significant features high in the subzone?	AF	All ECs	ESF (qual) Ecologically significant features (species or habitats), qualitative	The algorithm for AQ10 [ESF (qual)] is the same as for AQ1 [LRF (qual)] but applied to ESF instead of LRF.	Data requirements as for LRF (qual), but for ESFs; incl. maps of essential habitats (e.g., EFH based on occurrence) in the BBT.	(3)
11	Is the abundance of ecologically significant features high in the subzone?	AF	All ECs	ESF (quant) Ecologically significant features (species or habitats), quantitative	The algorithm for AQ11 [ESF (quant)] is the same as for AQ2 [LRF (quant)] but applied to ESF instead of LRF.	Data requirements as for LRF (quant), but for ESFs; incl. maps of essential habitats (e.g., EFH based on abundance) in the BBT	(1)
12	Is the presence of habitat-forming species (or species-formed habitats, i.e., biogenic habitats) high in the subzone?	AF	Benthic ECs only (e.g., Macrobenthos, Epibenthos, Algae, Plants, marine benthic habitats)	HFS/BH (qual) Habitat-forming species/biogenic habitats, qualitative	The algorithm for AQ12 [HFS/BH (qual)] is the same as for AQ1 [LRF (qual)], but applied to HFS/BH instead of LRF.	Data requirements as for LRF (qual), but for HFS/BH.	(4)
13	Is the abundance of habitat-forming species (or species-formed habitats, i.e., biogenic habitats) high in the subzone?	AF	Benthic ECs only (e.g., Macrobenthos, Epibenthos, Algae, Plants, marine benthic habitats)	HFS/BH (quant) Habitat-forming species/biogenic habitats, quantitative	The algorithm for AQ13 [HFS/BH (quant)] is the same as for AQ2 [LRF (quant)] but applied to HFS/BH instead of LRF.	Data requirements as for LRF (quant), but for HFS/BH.	(1)

AQ#	Assessment question (AQ)	EV criterion	EC	Metric name	Algorithm	Data required	Notes
14	Is the presence of mutualist or symbiotic species high in the subzone?	AF	All ECs (except for habitats)	SS (qual) Symbiotic species, qualitative	The algorithm for AQ14 [SS (qual)] is the same as for AQ1 [LRF (qual)] but applied to SS instead of LRF.	Data requirements as for LRF (qual), but for SS.	(4)
15	Is the abundance of symbiotic species high in the subzone?	AF	All ECs (except for habitats)	SS (quant) Symbiotic species, quantitative	The algorithm for AQ15 [SS (quant)] is the same as for AQ1 [LRF (qual)] but applied to SS instead of LRF.	Data requirements as for LRF (quant), but for SS.	
Notes: (1) Relative (or %) spatial coverage used as habitat-equivalent of the density for species. (2) This metric corresponds to the species richness question for species in EV assessment protocols applied in the literature. (3) To avoid double counting between ESF, HFS/BH and SS, if a species or habitat is selected in one of these metrics, then it should not contribute to the others. (4) The same feature may be identified in different ECs, as the habitat-forming species (e.g., <i>Sabellaria</i> , seagrasses) or the respective species-formed (biogenic) habitat type (e.g., <i>Sabellaria</i> reefs, seagrass beds). The choice of which EC to use in the assessment depends on data availability and duplication across ECs should be avoided.							

Table 2.6: EVA guidance: protocols for the selection of specific features relevant to AQs in Table 2.5.

Feature selection criteria				
Type	Reference AQ#	Selection criterion	Evidence base	Note
Locally rare feature (LRF)	AQ1, AQ2	Species or habitat occurring in <5% of the subzones in the BBT	Spatial distribution (presence/absence) of species/habitats in the BBT area.	1. Area of occupancy (by a species or habitat) is used for identification of rare features 2. Percentage threshold for identifying rarity of a species/habitat (based on data) may be changed (increased) if justified by limited data coverage of the subzones. Expert judgement is to be applied for this choice. 3. A feature (species/habitat) may be rare at one scale but not at another. 4. The type of evidence base used for the selection affects the confidence associated with the assessment.
Regionally rare feature (RRF)	AQ3, AQ4	Species occurring in <2% of 50 x 50 km squares of each of three bathymetric zones (littoral, sublittoral, bathyal/abyssal) in the region / Habitat occurring in <2% of 50 x 50km squares in the region, OR Species or habitat identified as regionally rare from accredited national/international lists, OR based on expert judgement	Spatial distribution (presence/absence) of species/habitats in the regional waters; OR Accredited list of rare species/habitats in the region (e.g. from OSPAR, HELCOM); OR Expert judgement	
Nationally rare feature (NRF)	AQ5, AQ6	Species occurring in <0.5% of 10 x 10 km squares within national waters / habitat type restricted to a limited number of locations in territorial waters, OR Species or habitat identified as nationally rare from accredited national/international lists, OR based on expert judgement	Spatial distribution (presence/absence) of species/habitats in the national waters; OR Accredited list of nationally rare species/habitats; OR Expert judgement	
Regularly occurring feature (in the study area) (ROF)	AQ8, AQ9	Species/habitats occurring in >5% of the subzones for the study area (i.e. all species/habitat excluding locally rare ones)	Spatial distribution (presence/absence) of species/habitats in the BBT area	5. Complementary to locally rare species (AQ #1 and #2)
Ecologically significant feature (ESF)	AQ10, AQ11	Species or habitats which have a significant ecological role in supporting the ecosystem structure and functioning. E.g. keystone species; species which constitute important food sources of certain seabirds (e.g., <i>Mesopodopsis slabberi</i> , sandeels) or have a	DEVOTES keystone catalogue; Species/habitat evidence for EBSAs in the region; Expert judgement, supported by the literature on the role of species and habitats in the region/ecosystem of interest.	6. There may be some overlap between ESF and HFS. ESF corresponding to HFS or biogenic habitats should be included in HFS metric instead, to avoid double counting. 7. Examples of ESF (from the literature) are given in Appendix 2. Work undertaken

Feature selection criteria				
Type	Reference AQ#	Selection criterion	Evidence base	Note
		key/pivotal role in the food web (e.g., keystone, top predator); species which are important for recruitment of fish stocks (e.g., fish larvae); habitats which play a key function in supporting viable populations of certain species (e.g., areas or habitats identified as essential habitats for fish/shellfish nursery, spawning, feeding or other functions).		under MARBEFES WP3 may also contribute to identify ESF.
Habitat-forming species or biogenic habitat (HFS/BH)	AQ12, AQ13	These can be identified as the species with habitat forming role or as the respective biogenic habitat. Their distribution is indicative of habitats often supporting diverse biota and key functions (e.g., refuge from predators).	DEVOTES keystone catalogue; Species/habitat evidence for EBSAs in the region; Expert judgement, supported by the literature on the role of species and habitats in the region/ecosystem of interest.	8. The same feature may be identified in different ECs, as the habitat-forming species (e.g., <i>Sabellaria</i> , seagrasses) or the respective species-formed (biogenic) habitat type (e.g., <i>Sabellaria</i> reefs, seagrass beds). The choice of which EC to use in the assessment depends on data availability and duplication across ECs should be avoided.
Symbiotic species (SS)	AQ14, AQ15	Species living in a symbiotic relationship with another species	Expert judgement, supported by the literature on the symbiotic interactions between species as relevant to the region/ecosystem of interest.	9. Where there is overlap between species or habitats identified as ESF, HFS/BH, SS, only include the species/habitat once to avoid double counting in the assessment.

The AQs are answered by applying the metrics and algorithms in Table 2.5 separately to the different ECs (and their relevant subsets of features) in each of the subzones in the study area. As a result, values ranging 1-5 are assigned to each AQ in each subzone for each EC. Where the assessment is not possible (e.g., due to insufficient data or knowledge for an EC in the area or in specific subzones of it), no AQ value will be attributed to the subzone.

The EC-specific values attributed to the different AQs in each subzone are to be aggregated by considering the maximum value across the calculated AQs to obtain the subzone-specific EC-specific EV. The Total EV is then obtained for each subzone by also considering the maximum value across the EC-specific EVs and the results are graphically represented by mapping on the grid for the study area. The overall aggregation procedure is illustrated in Figure 2.4.

Unlike the use of the average as the aggregation method (as with Pascual et al., 2011), which may dampen the variability of the resulting EVs across grid cells, the use of the maximum value as aggregation method allows higher value components of the biodiversity to be prioritised and represented. It is acknowledged that this may lead to different ECs and/or related AQs being responsible for the resulting Total EV in different grid cells. The importance of mapping the intermediate assessment results (maps of the EC-specific EVs and related AQs) is also emphasised, as these will ascertain the reason behind the value of total EV assigned to a grid cell (EC and AQ determining it).

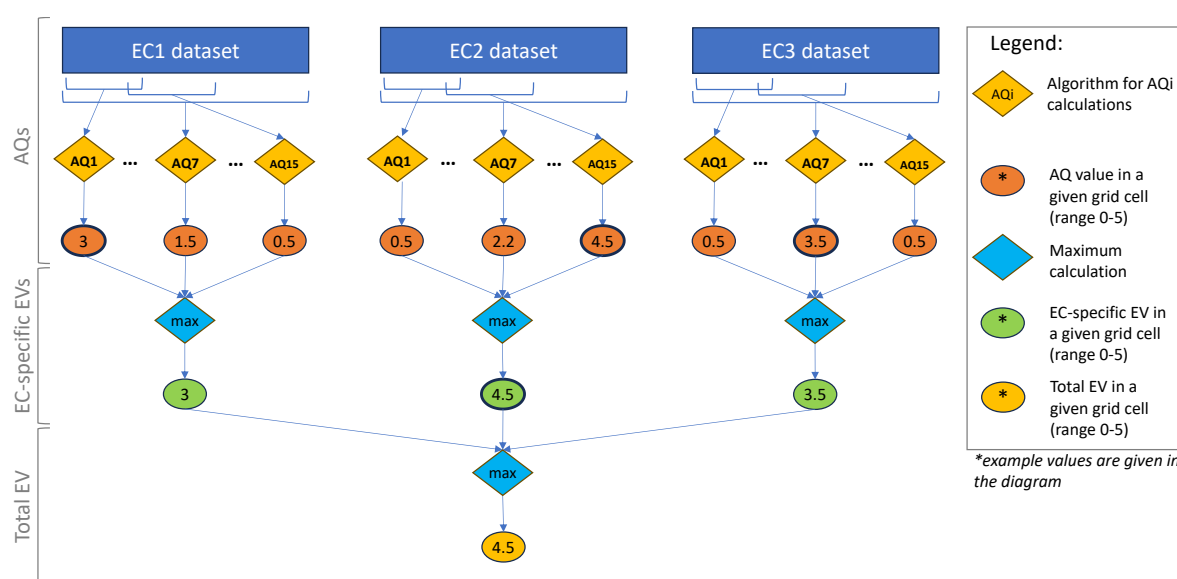


Figure 2.4: Aggregation procedure from individual AQ values (measured for a given EC) to the total EV (across all ECs assessed).

When calculating the Total EV, it is acknowledged that the different EC-specific EVs might have differing spatial reference, given that different ECs may be represented using subzones (grid cells) of different spatial resolution (see Section 2.2.2.1). If this is the case, the EC-specific EVs measured at the higher spatial resolution (i.e., using smaller grid cell size) should be aggregated at the lower resolution (i.e., at the scale of the larger grid cell size) to harmonise spatial scales and combine EVs across ECs. The overlap between smaller and larger grid cells may vary depending on the chosen cell shape (Figure 2.5). In any case, the aggregation of the EV values from the higher resolution grid (smaller cells) into the lower resolution grid (larger cells) should be undertaken using the maximum (i.e., the maximum EV across all small cells overlapping, partially or completely, with the larger cell should be used to represent the aggregated EV for the larger cell), in line with the aggregation method applied for the EV calculations.

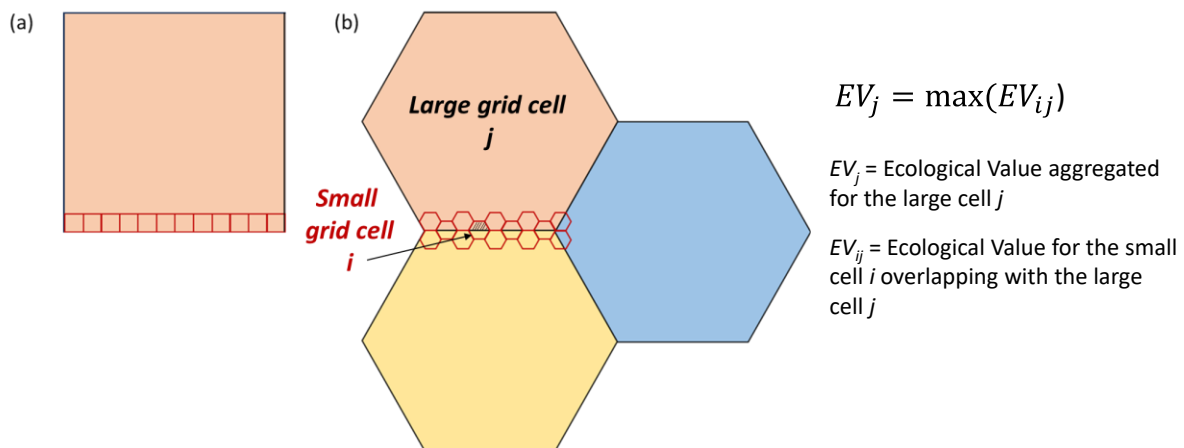


Figure 2.5: EVA guidance: Example of variable overlap between small and large hexagonal grid cells and EV aggregation formula (maximum).

The metrics calculated for each AQ assessed for each EC, as well as the derived EC-specific EV and Total EV are expressed as values ranging between 0-5. On mapping of these values, a 5-class scale may be used which distinguishes the following discrete categories: Very Low (VL, values 0-1), Low (L, values 1-2), Medium (M, values 2-3), High (H, values 3-4), Very High (VH, values 4-5).

A rescaling of the measure (presence/absence or abundance) relevant to each AQ is implied within the AQ algorithms (Table 2.5) to obtain standardised values within the 0-5 range and allow their aggregation into EC-specific EV and Total EV. The rescaling is based on the range of variability of such measure across subzones within an area, and therefore the resulting values (and the EV derived from them) express the relative ecological value of a subzone compared to the other subzones of the study area. This must be taken into account if comparing the results from different EVA implementations (study areas, e.g., BBTs in MARBEFES), as e.g., a value of 5 assigned to an AQ for an EC in an area may have a different meaning, in absolute terms (e.g. as the abundance of the species being scored by the metric), for the same AQ and EC in another area (due to the different range of variability of the measure behind the score across the subzones in the area). In turn, the comparison of the EVA process between study areas (as opposed to the results) is not affected.

To make comparisons of EV values between study areas (e.g., BBTs in MARBEFES), the rescaling procedure in the algorithms would need to be modified to account for the wider variability of a metric across the multiple study areas rather than within the individual area. However, some of the factors affecting the assessment results at this wider scale might not reflect actual differences in the inherent ecological value between the ecosystems considered. For example, these could be due to variability in monitoring programmes (different sampling methods, regimes etc) affecting how the same measure for an AQ is obtained in different study areas, thereby affecting the comparability of the data. Biogeographic factors might also influence the EV assessment at the wider scale (between areas), e.g., latitudinal variability in the distribution of the species and habitats considered, natural variability in biodiversity between the regions where the areas occur, etc. The possible influence of these factors should be taken into consideration for the comparison of EV results across areas to avoid misinterpretations.

Tools aiding the implementation of EVA have been created and validated, namely an Excel tool implementing Phase 2 of EVA (calculation of AQs and EV from EC gridded data) is available in Zenodo (Franco, 2026) and a Shiny app also dealing with the spatial component of the method (Razinkovas-Baziukas and Franco, 2026). These tools are being integrated into the MARBEFES toolbox (WP5; <https://marbefes.lifewatch.eu>).

2.2.4 Assess Confidence (Phase 3)

Following the above calculations, it is important to indicate confidence in the results. This can be derived as a measure of uncertainty which is associated to the EV in the relevant literature through the assessment of data availability and reliability:

- Data Availability (DAV) is determined by the number of (non-replicate) samples (or observations) of each ecosystem component taken (or made) in each subzone. This includes the inherent sampling and analytical variability which compound the field variability.
- Data Reliability (REL) accounts for the number of AQs that could be answered for each subzone in relation to the maximum number of AQs that may be applied (expressed as a ratio, or proportion).

Each of these uncertainty elements may be classified as Low (value 1), Medium (value 2) or High (value 3), based on the value range across the subzones in the study area and measured at the EC level (for DAV, the scores assigned to the different AQs are averaged to obtain a DAV for the EC). DAV and REL may be mapped separately or together (using a combination of colours and patterns to identify the combinations of the two variables). For example, Pascual et al. (2011) derived a map for 'reliability and sampling effort' to display the quality and amount of the data, respectively, to assess the information used for the EVA.

Given the modifications made to the original EVA protocol in this guidance (as in the above sections), the sources of uncertainty that may be associated with the estimated EV have been considered and updated (compared to the original method). These include the data availability and reliability components as per the original protocol applied in the literature (e.g., Pascual et al., 2011), with additional indicators of data availability being also added to account for distribution of mapped data and prediction. Additional sources of uncertainty are also considered, as specified in Table 2.7. The way in which these uncertainty sources influence the confidence in the resulting EVA is also indicated in Table 2.7.

Table 2.7: EVA guidance: Sources of uncertainty contributing to the confidence assessment.

Source of uncertainty	What is assessed and its relationship with the confidence in the EVA	Where the uncertainty source is accounted for in the proposed confidence assessment
(a) Assessment Reliability	How many of the AQs have been answered in the assessment of an EC in a subzone? Expressed as a proportion of the maximum number of AQs one may apply based on the AQ selection procedure in Figure 2.3. The higher is the proportion of AQs answered, the higher is the confidence associated with the EVA.	Equation 1 below
(b) Evidence Base for the assessment	Is the AQ (for an EC in a subzone) answered based on data or expert judgement? A higher confidence is associated with the use of data compared to expert judgement.	Weight scoring as per Table 2.8
(c) Data Use for the assessment	Where the assessment is based on data, is the metric to answer an AQ for an EC in a subzone derived directly from monitoring data or indirectly from model predictions? A higher confidence is associated with the use of monitoring data compared to model predictions. The latter may include for example habitat suitability models to extrapolate species occurrence or density based on environmental predictors, spatial interpolation models etc, which may be used to ensure full or better spatial coverage of the study	Weight scoring as per Table 2.8

Source of uncertainty	What is assessed and its relationship with the confidence in the EVA	Where the uncertainty source is accounted for in the proposed confidence assessment
	area. This type of product (data-based modelled maps) may be generated by the assessor based on the monitoring data available in the BBT, or may be derived from pre-existing maps for the area (e.g., EUNIS habitat maps, Essential Fish Habitat maps, modelled oceanographic variable maps from Copernicus Marine Data sources).	
(d) Data Availability	Where the assessment is based on monitoring data, how many data (samples/observations) are available and have been used for the assessment of the AQ for an EC in a subzone? Where the assessment is based on modelled spatial distribution of data (e.g. EUNIS habitat maps, or indeed any other spatial predictions or interpolation of distribution of species for different ecosystem components), the data availability may be measured as % mapped area in a subzone. The higher the data availability supporting the assessment, the higher is the associated confidence.	Weight scoring as per Table 2.8
(e) Data Quality	Are the data used for the assessment qualitative (presence/absence) or quantitative (abundance, extent etc)? A higher confidence is associated with the use of quantitative data compared to qualitative ones.	Table 2.5 + Equation 1

The confidence assessment designed in this guidance considers the sources of uncertainty specified in Table 2.7, as well as the practicality of assessing them as part of the EVA process. It is acknowledged that the sources of uncertainty could be further detailed and assumptions are made on how confidence is affected. For example, where expert judgment is applied (source (b) in Table 2.7), the confidence may vary depending for example on the range and type of expertise used to derive the judgment and how this is obtained, which may affect bias and other pitfalls of this approach. Where the use of data-derived models and products is considered (source (c) in Table 2.7), a lower confidence is assumed on average to express the preferred direct use of monitoring data. However, it is acknowledged that the relative confidence of monitoring data and data-derived products is affected by a number of factors, such as for example whether the sampling method used is the most appropriate to estimate the metric under consideration, the model predictive ability, etc. (see for example a discussion of the different lines of evidence and the confidence assessment for Essential Fish Habitat maps in Franco et al., 2022).

We propose that the confidence associated with the EVA of each EC on each subzone is expressed as a single index, based on the equation below:

$$Confidence = \frac{1}{N} \sum_{i=1}^N AQ_i w_i \quad (\text{Equation 1})$$

where:

AQ_i identifies the i^{th} Assessment Question, which in the equation is assigned a value of 0 if the AQ has not been used in the assessment or, if used, a value of 0.7 to qualitative AQs (AQ1, AQ3, AQ5, AQ7, AQ10, AQ12 or AQ14) and 1 to quantitative AQs (AQ2, AQ4, AQ6, AQ8, AQ9, AQ11, AQ13 or AQ15);

N is the maximum number of AQs that one may apply, as illustrated in Table 2.8. This varies depending on the EC assessed (as not all ECs can be assessed using all AQs) and whether qualitative or quantitative data are used (as this affects the AQ selection procedure);

w_i is the weight assigned to each AQ _{i} , where answered, which takes into consideration which evidence has been used to answer the question, how the data have been used and their availability (sources b, c and d in Table 2.7). This is a value between 1 and 5, assigned to the AQ as indicated in Table 2.9 and illustrated in Figure 2.6.

Table 2.8. Possible maximum number of AQs (N in Equation 1) that may be applied depending on the type of EC and the quality of data used for confidence assessment in EVA.

Type of EC	Qualitative data	Quantitative data
Species-based EC, benthic	7	8
Habitat-based EC, benthic	6	7
Species-based EC, non-benthic	6	7
Habitat-based EC, non-benthic	5	6

As indicated in Table 2.7, the assessment reliability component of the confidence assessment (source (a) in Table 2.7), is included in the formulation of the confidence equation itself. By assigning 0 to non-answered AQs and dividing by the maximum number of available AQs that may be assessed in theory for an EC, the confidence score considers the proportion of questions answered. While the data quality (source (e) in Table 2.7) is not explicitly included in the confidence weighting, it is incorporated in the assessment reliability component, given the guidance related to answering the qualitative and quantitative AQs as per Section 2.3.

The Confidence Index resulting from Equation 1 expresses the confidence associated with the assessment process that has led to the EC-specific EV resulting for a given subzone and ranges between 0 and 5. To express the confidence for the total EV integrating multiple ECs in a subzone, the confidence associated with the EC selected to represent the total EV in the subzone (i.e., the EC having the highest EV compared to the other ECs in that subzone) is to be used.

For mapping purposes, the confidence score may be classified on a 3-level scale as follows: Low confidence (score ≤ 1.67), Medium confidence (score > 1.67 and ≤ 3.33) and High confidence (score > 3.33 and ≤ 5). The confidence map (produced for each EC-specific assessment or for the EVA overall) provides a spatial product to be read in parallel with the respective EV maps.

Table 2.9: EVA guidance: weight (w_i) to be assigned to answered AQs for each EC in each subzone in the confidence assessment (Equation 1).

Base for the assessment (b) & Data use (c)		Data availability (d)*					
		Low	Medium-low	Medium	Medium-High	High	Not known
Expert judgment		1					
Data based	Indirect use of the data (model products)	1	1	2	3	3	3
	Direct use of monitoring data	1	2	3	4	5	-
*Note: The data availability in each subzone (for a given EC) is measured as the number of samples/observations available in the subzone where monitoring data are directly used for the assessment, or as the % mapped area in the subzone where interpolated or otherwise modelled spatial distribution maps are used as source data. For the data availability categorisation (as low, medium-low etc), the range of the data availability measure is calculated across all the subzones in the BBT (Range=Min – Max) and it is divided in 5 classes. Where data availability is measured as % mapped area within a grid cell (e.g., from habitat model maps), the Min-Max range is fixed (0-100%) and the data availability is classified as follows: Low with % mapped area values $\leq 20\%$, Medium-low with values $>20\%$ and $\leq 40\%$, Medium with values $>40\%$ and $\leq 60\%$, Medium-high with values $>60\%$ and $\leq 80\%$, High with values $>80\%$. Where data availability is measured as number of samples/observations (e.g., from species monitoring data), the Min-Max range is variable and the data distribution within this range might be skewed. Therefore, for these type of data availability measure, the 5 classes should be established using 20% inter-quantile ranges: Low with data availability values falling within the 1st/lowest 20% quantile (including $\leq 20\%$ of data), Medium-low with values within the 2nd 20% quantile ($>20\%$ and $\leq 40\%$ of data), Medium with values within the 3rd 20% quantile ($>40\%$ and $\leq 60\%$ of data), Medium-high with values within the 4th 20% quantile ($>60\%$ and $\leq 80\%$ of data), High with values within the 5th/uppermost 20% quantile ($>80\%$ of data).							

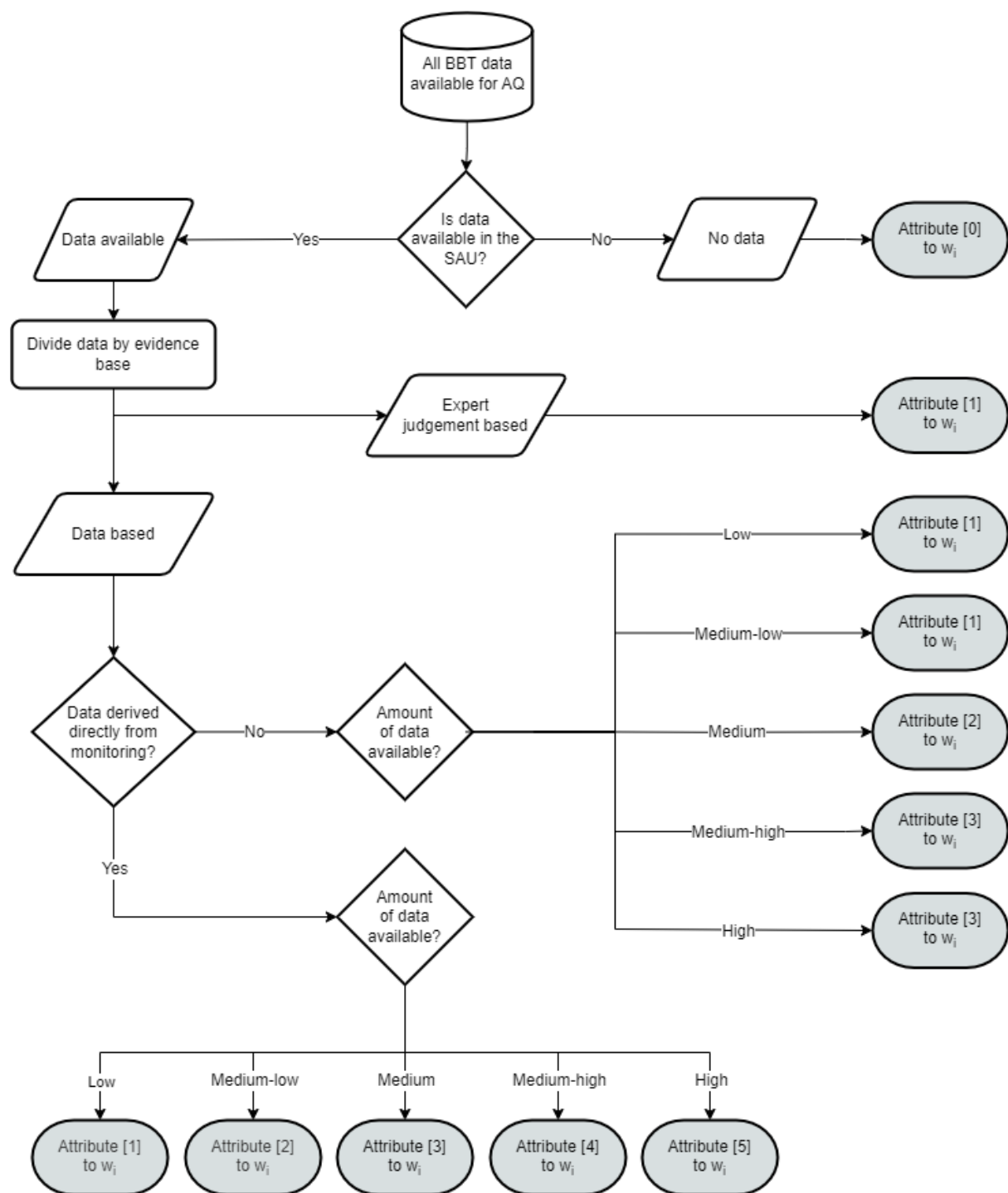


Figure 2.6: EVA guidance: Confidence weight (w_i) assessment for each AQ and EC in a subzone (SAU) based on criteria in Table 2.8.

2.3. EVA testing in MARBEFES BBTs and protocol revisions

This section summarises the testing process undertaken in MARBEFES at the 12 BBT sites, which has led to revisions of the EVA protocol throughout the development process (see Table 2.10 for a summary of the changes made since the original protocol was designed in March 2023). The feedback and main discussions about the method implementation have been summarised as a set of questions and answers (Q&A) which have been used to provide clarifications to the BBTs on the method use and to revise the protocol, if needed to improve its applicability, amend errors and reduce obstacles for its implementation.

Table 2.10. Summary of the versions of the EVA guidance produced following iterations in the method application and revision process, with indication of the main changes made.

Version date	Comment/Change made	Modified by
28.03.2023	First draft	A. Franco, E. Amorim
18.04.2023	Document revised after comments by M. Elliott and D. Burdon	A. Franco, E. Amorim
09.06.2023	Guidance revised after comments received up to EVA workshop on 9 June 2023	A. Franco
24.07.2023	Guidance revised following discussions (incl. use hexagonal grid and revised algorithms for question scoring in Table 2.5, plus other minor amendments)	A. Franco
22.08.2023	Final revisions	A. Franco
04.12.2023	Amendment/clarification to hexagon grid size	A. Franco
12.01.2024	Amendment to confidence assessment to account for additional metrics of data availability (section 2.2.4, Tables 2.7 and 2.8 and Figure 2.4)	A. Franco
16.01.2024	Clarification added to algorithm for AQ9 (step 5) in Table 2.5	A. Franco
29.01.2024	'Intrinsic' value replaced by 'inherent' value	A. Franco
14.02.2024	Details added on how to combine EV from ECs with different grid size into total EC (section 2.2.3)	A. Franco
21.10.2024	Amended confidence score categorisation (section 2.2.4, Table 2.8)	A. Franco
27.11.2024	Changes to aggregation method for EV calculation (maximum instead of average; section 2.2.3, also incl. spatial grid cells aggregation) and classification of data availability for w_i estimate (Table 2.8 in section 2.2.4)	A. Franco
05.02.2025	Change to confidence aggregation between ECs following changes in version dated 27.11.2024 (section 2.2.4)	A. Franco
06.06.2025	Change to AQ selection procedure (with prioritisation of quantitative AQs over qualitative ones, if quantitative data are available) (section 2.2.3, with new decision system flow-chart (Figure 2.3) for AQ selection)	A. Franco
10.06.2025	Confidence assessment updated following changes in AQ selection procedure (section 2.2.4, formula 1). Changes to spatial system, to include possible use of square grid cells and larger scale for habitat ECs (section 2.2.2.1, Table 2.2, Figure 2.2)	A. Franco

2.3.1 Considerations on the EVA protocol testing and use

The EVA was implemented at the 12 BBT areas in MARBEFES to test the method functionalities and identify possible areas for improvement. The implementation was undertaken between July 2023 and June 2025, although for most BBTs the bulk of the EVA implementation was completed by the summer-autumn 2024. The considerations below regard the implementation process and the exploration and discussion of the initial results obtained by the BBTs in July 2024.

There was a variable degree of adherence by the BBTs to the full EVA protocol. This was not unexpected, given the flexibility of the EVA method, which allows the user to adjust the assessment based on their own interests and data availability (e.g., selection of ECs and AQs to include in the assessment). The 'gold standard' for the application of EVA to an area would be to include the full range of ECs relevant to that area and for which data are available to undertake the assessment. Ideally, all the species or habitat assemblages in the area would be included and all the quantitative AQs would be measured for the assessment using a wide variety of data sources (ideally combining monitoring data to increase accuracy and confidence with modelling data to increase spatial coverage and possibly future predictions). However, it is acknowledged that real-life constraints may apply to the assessment, e.g., due to limitations in the data availability for an area or the user level of skill to perform the assessment. For example, suitable/sufficient data may only be available to assess some ECs and, in some cases, these may only qualitative data (presence/absence), thus limiting the assessment to the calculations of qualitative AQs only hence with inherent lower confidence associated. Operational constraints may also apply, for example due to time and resource limitations (as within MARBEFES) which may restrict the ability to source relevant data or to analyse them in all aspects. The flexibility of the EVA method allows its implementation even under these constraints, albeit this might happen at the expense of the representativity and accuracy of the results. Hence the importance of always undertaking the confidence assessment as an integral part of the EVA implementation.

Considering the operational constraints within MARBEFES, a minimum standard was set for the implementation of EVA by the BBTs, also to ensure consistency in the testing. This required that benthic ECs (macrobenthos and benthic habitats) were assessed by the BBTs and that the assessment of benthic species groups prioritised the use of monitoring data over modelling ones (favouring higher confidence but likely at the expense of spatial coverage), while EUNIS habitat maps (based on Level 3 habitat classification) were used for the benthic habitats EC. Most BBTs conformed to this minimum requirement, although some went beyond it by also including additional ECs and data. For example: fish, zooplankton and phytoplankton data were also used in the Curonian lagoon and Lithuanian coast BBT5; fish (demersal and pelagic) cetaceans, seabirds and phytoplankton data were also used for the assessment in the Irish Sea BBT8; fish, waterbirds, phytoplankton, phytobenthos (seagrasses and saltmarshes) and macroalgae (intertidal and subtidal) data were also used in the southern Gulf of Biscay BBT9. In some cases, alternative or additional data sources were used as deemed more suitable for the assessment in a case study area (e.g., more detailed habitat types than EUNIS level 3m as obtained from field surveys and local studies in the Gulf of Oristano BBT11 or in the Menorca channel BBT10; species abundance or biomass from spatial extrapolation or species distribution models for macrozoobenthos in the marine area of the Lithuanian BBT5 or in the Gulf of Biscay BBT9).

When considering the spatial system for the EVA implementation, most of the BBTs applied the 250 m hexagonal spatial grid as requested by the initial EVA protocol for the benthic ECs. However, the small spatial grid proved to be computationally demanding, especially in areas with a high habitat diversity and heterogeneity (e.g., Archipelago Sea BBT4) and in some areas a larger grid size was used (e.g., 3km in BBT5, 500 m in BBT10). In some cases, there was also a notable variability between the environments included in a BBT (also reflected by different data collection methods and availability) and the EVA was applied separately to them (e.g., lagoon or estuarine areas and the adjacent coastal marine area included in BBT5, BBT9 and BBT11). Consideration of this and of the results obtained for the assessment based on benthic habitat EC led to a revision of the recommendations on grid size

(allowing higher flexibility in the choice, depending on the data used), as explained in the Q&A section below (Q (06/06/2025) and resulting in the protocol provided in Section 2.2.

There was also a marked variability in the number of AQs being included in the assessments in the different BBT areas (e.g., the benthic habitat EC assessment including from only AQ7 in BBT5 to all AQs relevant to this EC in BBT9). Often, this was due to limitations in the availability of relevant data (e.g., qualitative v. quantitative) or other evidence (e.g., to identify rare species/habitats at different scales or ecologically significant species/habitats) time available to source/analyse the data), or in the ability to source them within the time allowed within the project task. However, in some cases, it was due to a misunderstanding of the protocol requirements. These were clarified with the BBTs and, where needed, the EVA protocol was amended to avoid them.

The considerations presented here only refer to the methodological aspects of the EVA implementation by the BBTs and do not address the interpretation of the specific results obtained by the BBTs. The Ecological Value (EV) calculated and mapped by the different BBTs cannot be directly compared across the BBTs. This is because, in addition to the variability in the protocol implementation between BBTs, the EV value calculated for an area is expressed in relative terms and this prevents comparability of results across different areas. Although the EV resulting from the BBT implementations are expressed on a common standard scale (0-5), its calculation relies on intermediate steps requiring the averaging of values between species or habitats specific for the BBTs and rescaling the AQ intermediate values (to a 0-5 range) based on the maximum observed across grid cells in the case study area (BBT) being assessed. As a result, the maximum value of 5 for the standardised EV corresponds to the condition of maximum biodiversity observed within an area and its meaning (in terms of actual degree of biodiversity it represents) may change between BBTs. As mentioned in Section 2.2.3, the only way to compare the EV results between BBTs would be to disaggregate the EV value tracking the calculations back to the initial steps of the implementation of the AQ algorithms (separately for the individual ECs) and to define a different rescaling procedure which takes account of the variability of an AQ across multiple BBTs (rather than just within it). This was outside the remit of WP4 (and Task 4.1 in particular), which focused on testing the EVA methodology rather than analyse and interpret its results (the latter is left to the individual BBTs).

Exchanges and checks with the BBTs were maintained throughout the testing of EVA, to ensure that issues, possible errors (both in the protocol and in its use by the BBTs) and areas of improvement were identified in a timely manner, allowing the BBTs to correct their assessment accordingly, when possible. This flexibility in the implementation of the EVA protocol changes was more likely at the earlier stages of the implementation by the BBTs. As the process advanced, and more time was spent by the BBTs to obtain suitable results, revising the EVA implementation following protocol revisions became more difficult. While the BBTs have put great effort in amending their assessment following protocol changes up to the end of 2024, it is acknowledged that later protocol revisions (see Table 2.9) may not have been fully implemented by the BBT, nor they were required to do so. Nevertheless, the feedback obtained from the BBTs and the assessment of their experience in applying the EVA method has been invaluable to identify and correct possible weaknesses in the protocol and to improve the resulting protocol presented in Section 2.2.

2.3.2 Q&A

This section collates comments and questions and the associated answers and clarifications about the EVA process developed in MARBEFES (WP4, Task 4.1). The Q&A are organised according to the phase of the EVA protocol they are relevant to (or in section 'Other' where of general relevance). The date of when the item was added (or last modified) is also indicated to provide temporal reference throughout the process of testing and revision of the EVA method.

2.3.2.1 Phase 1 – Define the study system

Spatial system

Q (11/07/2023): What is the **spatial scale of evaluation** - BBT limits only? Region? all EU area ? Difference is very important - rare, protected and important species in one place (e.g. *Fucus* in Poland) is a trivial and common species in Finland and not taken in consideration in Lithuania as there is no habitat for this. The valuations separated by administrative/political borders are very difficult to compare - highest value in Baltic and highest value in Ireland will mean something completely different in terms of biodiversity

A: There are two spatial scales to be considered in the EVA: one is the scale of the study area being assessed (defined by the BBT boundaries), the other the spatial scale at which the features (species and habitats) in the study area are characterised, e.g. in relation to their rarity. The questionnaire in EVA includes multiple questions for rarity which reflect the different scales at which rarity may be characterised (at local, within the study area; regional; or national). If in one BBT there is enough information to characterise the rarity of the species/habitats within on multiple scales (e.g. locally rare, nationally rare), there multiple questions can be answered and they will contribute to the final assessment. If there is no information available, e.g. on rarity at national or regional scale, then the BBT can assess local rarity based on the species/habitat distribution within the BBT (as per guidance). Only that question (on local rarity) will contribute to the final EV, although a lower confidence would likely be associated with that result as some the questions on regional/national rarity were not assessed.

As for the boundaries of the spatial scales for rarity, the local (BBT) and national scale are probably the easiest ones. For the regional scale, there will need to be clarity on which region the rarity assessment refers to.

Q (11/07/2023): **Grid for plankton** has resolution of 250 x 250 m. This is not consistent with wider grid (3 x 3 km resolution) for pelagic habitats. Patterns and dynamics for water column/plankton are more likely to occur at the wider spatial scale.

A: It is agreed that the grid scale for plankton should be changed from 250x250m to 3x3km. ***The EVA guidance document has been updated accordingly.***

Q (11/07/2023): **Can the assessment be applied to marine and coastal areas, also including estuaries?**

A: The ecological relevance of the zonation (gridding) is ensured by identifying different grid sizes as appropriate to the different ECs: larger grid cells (3x3km) for more mobile species or pelagic habitats, smaller grid cells (250x250m) for benthic species and habitats. This means that assessment can be applied to marine and coastal areas, including estuaries.

Q (11/07/2023): EVA requires for a study area (BBT) to be divided into **subzones that are meaningful from an ecological perspective**, as relevant to the ECs considered in the assessment. How are these identified?

A: Subzones may be identified for example through the biogeographic/habitat zonation of an area, or as often applied in the literature, through the application of a regular spatial grid. The EVA protocol developed in MARBEFES adopts the latter approach for standardisation within and across BBTs. It is therefore emphasised that the terms 'subzone', 'spatial assessment unit (SAU)' and 'grid cell' are used interchangeably as synonyms in the proposed EVA protocol. The 'ecological meaningfulness' of the resulting zonation (gridding) is ensured by identifying different grid sizes as appropriate to the different ECs: larger grid cells (3x3km) for more mobile species or pelagic habitats, smaller grid cells (250x250m) for benthic species and habitats. The use of a regular grid defined independently from the habitat distribution also allows the use of habitats as an EC to be assessed within each grid cell.

Q (11/07/2023): Is there a potential issue with **grid cells crossing between different biological zones/habitats** (including for example coastal/estuarine boundaries) and how could this affect the assessment? (as the area within a grid cell is assessed as one spatial unit).

A: Whether this is an issue would depend on the EC considered, as, e.g. mobile species may distribute across multiple habitats/biozones and therefore the habitat boundary may be less relevant than for other ECs. The different scaling of the grid according to ECs takes this into account and therefore should allow mitigation of this potential issue.

Q (11/07/2023): We will be working with a **1km x 1km grid** as we consider the 250m grid too detailed for our BBT, particularly as most of the area is >50m (data comes from remote monitoring and spatial extrapolation)

A: It does not appear to be an issue to adapt the grid to your specific case study if the proposed pre-defined grid scale is not suitable. For comparability with the other BBTs, your 1x1km grid cell could be divided into 16 cells of 250x250m size (all with the same value as the bigger grid cell they fall within), but this can be done *a posteriori*, if comparison between BBTs is needed.

Q (11/07/2023): About the grid, could we work with **hexagonal grid** (better for marine areas) and with [lat./long.] degrees as unit of measure?

As for using degrees, it will not be an issue as any measurement using degrees as units can always be transformed back into one using m/km.

As for using a hexagonal grid instead of a square one, it has been highlighted that hexagonal grids allow more flexibility and better spatial representation of marine biological components and information. They are also more suitable for the evaluation of different connectivity measures. But the most remarkable thing is that their projection and visualisation are more precise and aesthetically pleasing, which should be a criterion to be valued in this type of projects.

Some of the data that are used for EVA will probably feed into the ARIES platform and then into the Natural Capital Physical Accounting and it has been confirmed that ARIES can work with any grid type proposed. It is essential, however, that **all BBTs use the same type of grid**.

Following the comments above, it has been decided to ask **BBTs to use hexagonal grids** (of different size, depending on the ECs) instead of a square one (11/07/2023).

For those not familiar with hexagonal grids here is some advice (from Andrés Ospina-Alvarez at CSIC): *Both in GIS software (ArcGis or QGis) and in R or Matlab, hexagonal meshes can be produced in a simple way. In QGis, for example, it only involves two menus and three or four clicks, so there should be no problem in changing from square to hexagonal grids. [see video-tutorial on how to create hexagonal grids using QGis – file ‘Makinga Hexbin Grid for a BBT’ in NextCloud at <https://nc-marbefes.iopan.pl/nextcloud/s/retGAGYs97B6fRt>]*

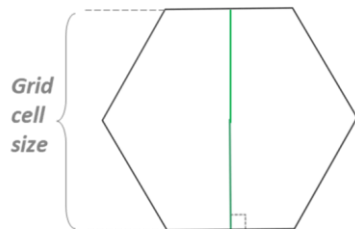
The EVA guidance document has been updated accordingly.

Q (04/12/2023): Regarding **grid size**, by 'Maximum width' do you mean twice the apothem of the hexagon or twice the radius of the circumscribed circle? The QGIS tool requests indicate the vertical distance from the centres, equal to twice the apothem?

A: When the grid was changed from square to hexagonal, the grid cell size was changed from e.g. 3x3km square to 3km maximum width (meaning the diameter of the circumscribed circle) to keep the area of a grid cell closer to the one of the original square.

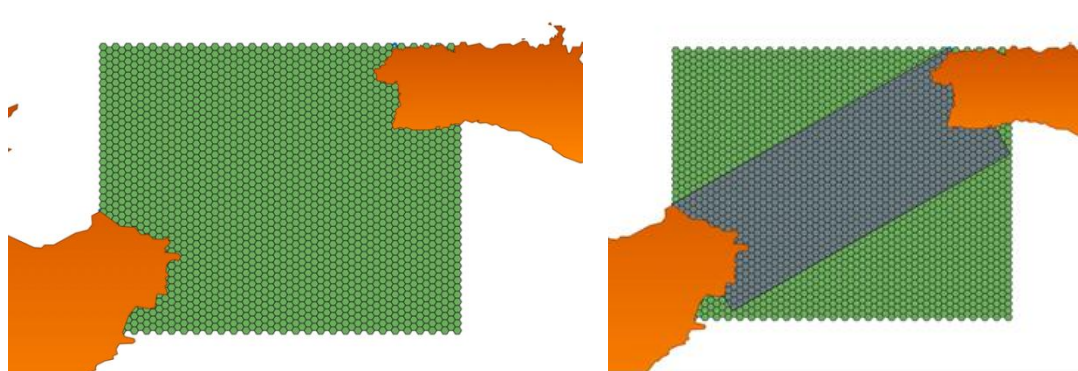
Using the indicated size as the length of the apothem (as in the figure below) would lead to a hexagon area which is 3.5 times bigger than the area of the original square (e.g., 31.2km² instead of 9km² using the 3km size). However, it was realised that using the maximum width for the hexagon size, would reduce the area to 65% of the original square (e.g., 5.85km² instead of 9km²).

The best option would be to use the size indicated in the guidance as the measure of **twice the apothem (i.e. the vertical distance between two sides of the hexagon, through its centre – see figure below)**. This would lead to an area which is 86% of the original square (e.g., using the size of 3km, hence apothem = 1.5km, the hexagon area would be 7.8km², compared to 9km² of the square 3x3km cell). *The EVA guidance document has been updated accordingly.*



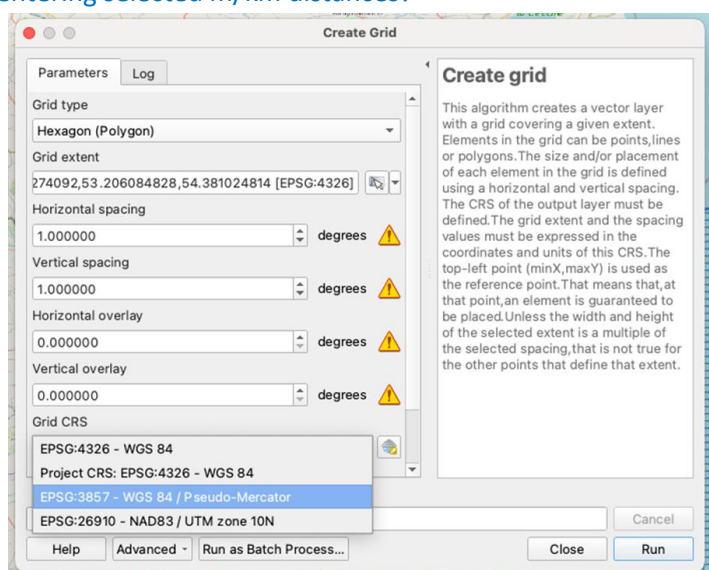
Q (04/12/2023): Using the **BBT boundary** as "Grid extent," the grid created does not completely fill the BBT extension, but there are holes on the sides of the layer (Figure 1). This also happens in the demo video (Figure 2). Can we expand the "grid extent" and then intersect?

A: Yes, you should create an expanded grid extent and then intersect as shown in the video (<https://nc-marbefes.iopan.pl/nextcloud/s/retGAGYs97B6fRt>) This way you will not have holes...



Q (04/12/2023; updated 12/01/2024): Using the **WGS 84 (4326) projection** to define hexagons, it is necessary, as shown in the demo video, to use values in degrees to **specify grid size** (as horizontal and vertical spacings – see screenshot below). In this way, hexagonal polygons with non-uniform metric dimensions will be generated. In this case we can't really insert the length of 2x apothem (e.g., 3 km) into horizontal and vertical spacing, only degrees allowed, UNLESS we swap to pseudo-Mercator projection, which allows m/km input. My question is in this case should we stick to the original

projection (and recalculate the degrees to relevant area etc.) or do we have some other possibility of entering selected m/km distances?



A: Two conversion methods can be used, as per advice provided by Willem Boone at VLIZ (below). We recommend to **use the second option** where possible. Note that the text below refers to the example of a 3km grid size; this value should be adjusted to the actual grid size required for the specific ecosystem component.

- 1) *The most straightforward method would indeed be to calculate how the defined distance in meters at a certain BBT can be converted to lat/long degrees [e.g., using online calculators as <http://www.csgnetwork.com/degreenllavcalc.html>]. In principle, latitude axis, 3km will always correspond to 0.027° Lat (considering a spherical geodetic datum i.e., the world is a perfect sphere). For longitude there is a larger difference as longitudes come closer together towards the North Pole, so their distance in meters becomes smaller. See example.*

Specific Latitude	Length Of A Degree Of Latitude In Meters	Length Of A Degree Of Longitude In Meters	Lat degrees corresponding to 3km	Lon degrees responding to 3km
30	110852	96486	0.0271	0.0311
40	111034	85393	0.0270	0.0351
50	111229	71695	0.0270	0.0418
60	111412	55799	0.0269	0.0538
70	111561	38186	0.0269	0.0786
80	111659	19393	0.0269	0.1547

Concluding on this, using this conversion method will result in errors especially in longitude axis. If the BBT has a wide north-south range or is located higher up north, it is not recommended since the error will increase.

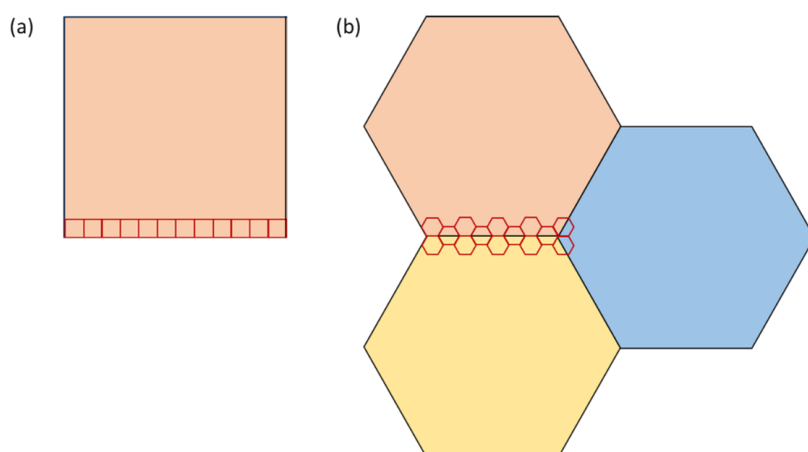
- 2) *An alternative method could be:*

- a. *Chose a local CRS with metric system (e.g. zone in <https://epsg.io/32600> -> codes 32601-32660, for Belgium that would be EPSG32631N). *Using WGS 84 / Pseudo-Mercator is not a good option as it is a global metric CRS and local errors will be larger.*
- b. *Create the BBT grid in QGIS using this metric CRS, use 3000m as spacing.*
- c. *[Re-project](#) the resulting shapefile/gpkg to WGS84 - EPSG4326.*

Q (14/02/2024): The assessment of an overall EV will require results to be combined across ECs. Different ECs may have different grid size (250 m v. 3 km), so how should the EVs from different ECs assessed at different spatial resolution be combined?

A: EC-specific EV values obtained at the higher resolution (250m grid size) should be aggregated at the lower resolution grid scale (3km grid size) before they can be combined with the EV values for the ECs assessed at the lower resolution. This rescaling would consist in averaging the EV values across all the small grid cells contained in a larger grid cell to obtain the resulting EV value for the larger grid cell.

While 144 (12 x 12) smaller (250 m) square cells would have fitted exactly within a larger (3 km) square cells (see figure (a) below), this is not the case when using hexagonal grids (figure (b) below). The area of the larger cell is still 144 times the area of the smaller cell, but some cells on the border may be partially included in different larger cells.



There are different methods that could be used to aggregate EV values from smaller cells into an EV value at the larger cell scale. Advice was sought from GIS experts (*) and it was agreed that the best option for the MARBEFES EVA approach is to use a weighted average, where the weights are determined by the proportion of the area of each small cell that overlaps with the large cell. Compared to other methods, this approach provides a more accurate representation of the EV values across different scales. It is acknowledged that it requires additional computations (including the determination of the overlapping areas and their respective proportions) compared to a simple overlap method, but these should be easily undertaken in GIS. **The EVA guidance document has been updated accordingly.**

(*) Thank you to Andrés Ospina-Alvarez (CSIC), Giorgio Massaro (CNR) and Marta Pascual (BC3) for the advice provided.

Q (06/06/2025): **Final considerations on the spatial grid system** after seeing experience and results of implementation of EVA in the BBTS areas

A: These considerations regard both the shape and size of the grid cells.

Shape of grid cells: The guidance to the BBTs was changed from using square to hexagonal grid cells following considerations as per Q (11/07/2023) above. After seeing the proposed approach implemented by the BBTS and receiving the feedback on use and challenges, it is clear that both grid shapes have their pros and cons. Square grid cells may be considered not as good as hexagonal cells to represent marine components and their connectivity, but are easier to use and to re-scale (e.g., when EV results obtained at different grid spatial resolution are to be rescaled for aggregation into a common grid). As such, they are more likely to be implemented even by users with basic GIS skills/experience. As a result, **the guidance has been modified (for general use)** to recommend the use of hexagonal grid cells, but also allow the use of square grid cells instead so as to remove a possible obstacle in the EVA implementation.

Size of grid cells: The size of grid cells determines the spatial resolution of the assessment within a study area and it may change depending on the EC considered and the type of evidence used to inform its assessment (e.g., to account for the spatial resolution and representativity of the sample data). For the assessment of benthic ECs, a smaller grid size (250 m) was suggested compared to that one used for most of the other ECs (3 km) considering the spatial resolution and representativity of typical benthic samples; e.g., in the worst-case scenario where the assessment of a grid cell area is based solely on the benthic community from one benthic grab sample, it would be unlikely for those data to represent the benthic features over a 3km-wide area hence the choice of a smaller grid size). However, after seeing the method implemented by the BBTs, it became clear that the same might not apply when benthic habitats are the EC being assessed. In this case, the EVA considers the diversity of habitats (based on presence and/or extent) within a grid cell, and therefore it is implied that such habitats are defined/identified (hence mapped) at a higher spatial resolution than the one of the grid cell. This may not be the case if the habitat types are identified in broader ecological terms (e.g., EUNIS Level 3 habitat types, as used by the BBTs), which may also reflect on the inability to differentiate them at lower spatial scales hence to assess their diversity within a smaller grid cell area (resulting in the homogeneous coverage of the grid cell area by the individual broad scale habitat). This was evident from the BBT's EVA results based on EUNIS Level 3 habitats, often showing a homogeneous low EV due to the presence on one individual habitat type within most of the 250 m grid cell in the study area, which higher levels only detected where the grid cells overlapped with the borders between the different broad scale habitats. The same assessment undertaken using 3 km grid cells (e.g., as tested in the Irish Sea BBT8) showed that a larger grid cell size allowed to better capture the diversity of habitat types defined by the EUNIS Level 3 classification. Taking this into consideration, **the guidance has been modified (for general use)** to allow some flexibility in the grid cell size and to allow the user to make a judgment call about the most suited spatial resolution depending on the scale at which the habitat types are defined and used in the assessment.

Ecosystem components (ECs)

Q (11/07/2023): What is the **limit of habitats and domains** with which we are dealing - benthos? pelagial? both? Valuation shall be set separately for seabed and water column as both domains are independent to a high degree - ecological valuation of seabed will not match this for pelagial in the same area

A: Benthic and water column habitats can be considered as separate components (as are, for example planktonic and benthic species), to be assessed separately in the EVA, if there is sufficient data (e.g. habitat maps for the two domains within coverage of the BBT). This means that you will have an EV for benthic habitats and an EV for pelagic habitats (as well as EV for the other species groups components, i.e. fish, marine mammals, seabirds etc). As you say, the resulting EV maps for the different components might not match and show different distribution of the EC-specific EV within the BBT. The overall EV will combine these EC-specific EVs.

Q (11/07/2023): What are the **organism categories we are going to value** - selected taxa (as I have proposed Crustacea, Polychaeta, Mollusca, Echinodermata, fish, macrophytes) again selection of taxa will set the value - e.g. all sea mammals and seabirds are rare and protected in many countries and their presence will outweigh all other organisms

A: In the guidance there is a list of the possible Ecosystem Components that can be used, but it doesn't mean that all are to be assessed in each BBT. The choice of the ecosystem components (organism categories for species) one BBT should consider for the EVA is up to the BBT, as it depends on the evidence that is available to undertake the assessment in that BBT. For example, some BBTs may have sufficient evidence (data or expert knowledge) to undertake an assessment (by answering as many questions as possible) based on birds, fish and benthic infauna, others may only be able to consider birds and might have data for benthic infauna only including annelids). As regards the granularity of how EC are defined, there is some suggestion in the guidance (e.g., demersal and pelagic fish fauna,

or benthic infauna and epifauna may be assessed separately if there is sufficient evidence to allow it). However, it is recommended not to resolve EC at a too high taxonomic separation level (differentiating polychaetes, molluscs etc). It is suggested using taxonomic/functional groupings that are commonly applied (e.g., in MSFD assessments), as suggested in the guidance.

As for the risk of one quality of one EC outweighing the others, this is unlikely in the EVA as multiple qualities (assessed with different questions/metrics) for multiple ECs are combined in the final EV. In your example, the metric associated with the rarity of seabirds is only one among many other qualities that can potentially contribute to the final EV. Also note that the questions about rarity assess the diversity of e.g., rare species (at a given spatial scale) within each subzone of the BBT, but the metric is expressed as a relative score which depends on the diversity measured across the whole BBT. For example, if the range of nationally rare species in the BBT (across its subzones) is between 0-6 for fish and 0-22 for birds, a subzone with 6 rare fish and one with 22 rare seabirds will both have score 5 for that metric and they will both contribute equally to the calculation of EV (first within the specific EC, then across all ECs).

Q (04/12/2023): In which EC or where do the data related to **microplastics** or anthropogenic impacts go?

A: NOT HERE! Ecosystem Components are just for BIOLOGICAL DATA. NOT HUMAN-PRESSURE RELATED DATA.

Q (04/12/2023): We have information about the **microplastic** distribution along the BBT beaches. Do they fall under the EVA?

A: NO, that information is not for EVA

Q (12/12/2023): There is an ecosystem component for seabirds, but a big part of the coastal ecosystem in a BBT may also be **waterbirds** (incl. wintering). Is this OK to create an EC for birds from coastal wetlands etc.?

A: Yes, you can create a separate layer for waterbirds. As these are generally recoded from coastal surveys (while roosting or feeding in intertidal areas), the range of this dataset will only cover the coastal cells of your BBT and the other cells will be blank (no data). You would then include EV-waterbirds and EV-seabirds separately with the other EC in the final EV (but EV-waterbirds will only contribute to the total EV in the coastal grid cells).

Q (15/12/2023): Have we agreed finally on the taxonomic level of evaluation - which taxa to consider? If one will focus on fishes and other on meiofauna we will travel to nowhere, of course habitats will be different in all sites (still in one EU classification). I have suggested to have common and the same for all BBT set of taxa for biological valuation, namely:

- 1) seabirds - species
- 2) sea mammals - species
- 3) fishes - species
- 4) crustaceans benthic and pelagic - species
- 5) Molluscs - benthic and pelagic - species
- 6) macrophytes (both algae and vascular) - species
- 7) Polychaeta - family level
- 8) Echinodermata- species

A: The full list of ecosystem components is in the guidance. BBTs should assess as many as possible, but not all of them will be assessed by all BBTs and it is not expected that the BBTs to include all the same set of components, as it depends on the data they have. Some may only have data on fish and

macrobenthos, others on macrobenthos and seabirds, so their EVA will be dependent on the EC they are able to include. As the EVA is based on the aggregation of ecological value at different levels (from individual metrics for each component, to EV across metrics for the whole component, to total EV across components), the final EV can be disaggregated to make ad-hoc comparisons if needed (e.g., for the same EC used by some BBTs), but with the caveat that the EV in different BBTs may have different meaning (as the standardisation of EV scores is based on the range of variability of the metrics in the individual BBT).

Even when the same EC is assessed by different BBTs (species groups or EUNIS habitats), we do not expect the identity of the species (or the habitats) included will be always the same across BBTs, due to biogeographical differences in the distribution of species and variable morpho-ecological characteristics of the different BBT areas. E.g., some BBTs might only have sedimentary/soft bottom habitats, while others might also have hard substrata. It will be all right, as long as the habitat classification system used by the different BBTs is the same (EUNIS 2022, broad scale habitat (Level 2) as minimum, biotopes (Level 4) where possible).

As regards the taxonomic level used to distinguish the EC, the EC are identified at a high taxonomic level groups (as indicated in the guidance), which may be combined with some functional grouping also considering that the data behind the different groups may be collected with different monitoring methods and therefore might not be comparable (e.g. demersal and pelagic fish could be assessed as separate EC). It is not suggested splitting the EC taxonomically to finer levels; e.g., polychaetes, molluscs, echinoderms and crustaceans would be assessed at the higher taxonomic level as part of the macrobenthic community; this could be distinguished between intertidal and subtidal communities, if the data exist in the BBT, as the sampling method may differ. Where epibenthic invertebrates (including larger crustaceans) are also sampled (with methods other than grabs/cores, e.g., epibenthic dredges/trawls) these could be included as a separate EC (as the data from epibenthic dredge would not be comparable with those from grabs or cores).

Q (15/12/2023): How can one use "macrobenthos" as a measure of biodiversity, without defining the species identity?

A: When the guidance indicates that macrobenthos is the EC assessed, this means that the metrics for EVA will be assessed on this community, but, of course, they require data for the individual species to measure the biodiversity metrics for the community (depending on the metric, all the macrobenthic species in the community may be considered in the measure of biodiversity, or only subsets of them, e.g. according to their rarity at different geographical scales, or their functional/ecological importance). The EVA requires both data on presence/absence and abundance/density, although, in the absence of the latter, a 'simplified' version of the EVA can be applied (only based on the metrics requiring p/a), although this reduces the confidence in the results.

Q (15/12/2023): Of course you may use the trawl, grab, core, traps etc for information of biodiversity, as long as what we need is presence/absence - species list, the qualitative, not quantitative information.

A: Even if only presence/absence (not just presence) data are used, caution is needed in combining data from different survey methods, as different survey methods may have for example different selectivity and this might affect the data comparability (especially for records of absence). For example, if trawl and grab data are combined to qualitatively characterise the macrobenthic community (no abundance), a mollusc species might be present in subzone 'A' where grab samples have been taken (hence a value of '1' in the dataset), but it might not be present in the samples collected with trawling in subzone 'B'. This does not necessarily mean that that mollusc is absent from subzone 'B', as trawling would not have effectively sampled that part of the macrobenthic community and therefore its absence from the trawl sample in subzone 'B' cannot be taken as indicative of absence from the subzone (i.e., with a value of '0' in the combined dataset). The two datasets should be treated separately (e.g., trawling data for epibenthic community, grab data for macrobenthic

community) and the data for the species effectively sampled by each of these methods should be considered as representative of the two communities.

Evidence base

Q (04/09/2023): What to do if there are **spatial gaps in the evidence base** for an EC? (i.e., it only covers part of the BBT)

A: The evidence (data, data products, expert judgments) available for an EC may only cover part of the BBT for different reasons. The EC may potentially occur only in parts of the BBT that are suitable (e.g., saltmarsh in intertidal areas only) and therefore the data for that EC only cover those suitable areas (these can be identified e.g., using depth zones). In other cases, the partial spatial coverage of the BBT may be associated with limited evidence availability, e.g., where the monitoring of an EC was only undertaken in a part of the BBT (e.g., an area of interest) but the EC may potentially be distributed in other areas of the BBT which were not monitored. In this latter case, where possible, modelling could be undertaken to extrapolate the monitoring results on the EC to the wider BBT area (e.g., using seabed characteristics and benthic data from monitoring to predict the distribution of benthic species or communities/habitats over the wider BBT area; or other species distribution models to predict habitat suitability of fish, marine mammals, seabird species). However, this may not be possible and the monitoring data (with spatial gaps) are directly used for the EVA.

In all these cases, the EVA assessment for the EC (and the rarity calculation in the specific; see also Q&A under 'Assessment criteria and questions') can only be applied to the areas of the BBT for which data are available, thus excluding any gap areas (n/a) from the calculations. The EC-specific results can then be combined in different ways, e.g., by calculating and mapping EV across the whole BBT using the ECs with widespread evidence and then 'zooming-in' to a specific area of the BBT (e.g., the intertidal zone, estuaries or the area where localised monitoring was undertaken), where additional relevant ECs can be added to the total EV assessment.

Q (04/12/2023): In case we are certain that a species cannot live in a specific area (e.g., *Patella ferruginea* cannot be found in the infralittoral zone), should we assign the **value "blank" or zero** to the subzone?

A: See also question above. The value should be BLANK (n/a) as zero is only used to characterise absence of a species from areas where it may occur (hence intertidal species should be marked as blanks in the subtidal parts of the BBT area and, similarly, species associated with hard substrata should be marked as blank on sedimentary substrata and vice versa).

Q (12/12/2023): Large mobile species such as whales or turtles - should we include **strandings**? What's often reported by volunteers is a dead animal.

A: It is preferable not to include strandings as they are not indicative of the actual spatial distribution of the species in the marine environment and their record in a (coastal) grid cell may be misleading.

Q (12/12/2023): We know that migrating species could be anywhere in the BBT and we just "happen" to catch them during a survey in a particular subzone/hexagon. In that case, that hexagon will have a higher EV than the neighbouring hexagon, where the species technically could have been just before, but hasn't been recorded. Consequently, that "empty" hexagon will have low EV despite being within the species wider occurrence area. In that case, can **species distribution models** "fill in" the gaps or should they not be combined in one EC with actual observations?

A: There are two aspects to this question:

1. There is a difference between 'zero data' (i.e. where a survey took place, but the species was not recorded) and 'no data' (i.e. where there was no survey, so it is not known whether the species occurs or not there). The former would be recorded as a zero in the dataset and therefore it would probably contribute to lower the EV for that grid cell (subzone/hexagon), compared to others where the survey produced a positive record of the species. The latter ('no data', i.e. blanks in the dataset)

would not contribute in any way to the final EV in the grid cell, which will be the mean of the EV of the ecological components for which data exist in that subzone/hexagon cell (e.g. in one grid cell the final EV may be the results of benthos + mammals + fish, while the EV in the next grid cell may only result from the assessment of benthos + fish, if there are no data on mammals for it; maps of EC-specific EV would allow you to see where the gaps are and therefore where each EC contributes to the final EV).

2. Any survey is limited in the spatial coverage it can achieve, so there will likely be 'empty' subzones that are not covered by a survey, but it does not mean that the species or habitat doesn't occur there. Species distribution models can certainly be used to extrapolate the survey results and make predictions where there was no survey, thus improving spatial coverage (and reducing gaps). Both sources can be combined for an EC (e.g., if there are predictive model maps for some species and only survey data for other species of the same EC), but make sure that zero data (observed or predicted absence) and no data (no observation or prediction, e.g. where the grid cell falls outside the survey area or model validity range) are reported correctly (as per point above). Also make sure it is recorded where model results have been used and there are survey data, as this will affect confidence calculations (phase 3).

Q (12/12/2023): The BBT may cover a pretty large area with **various sources of data**. For instance, ecosystem component "mammals" can comprise aerial surveys monitoring data for cetaceans, but for pinnipeds - only opportunistic sightings. In that case, is it OK to combine cetaceans and pinnipeds together as one EC?

A: As the two methods are very different (and also probably the time reference for the data, e.g. dates of the aerial survey vs. years over which opportunistic sightings have been recorded), the data are not really comparable. It is not advised to integrate the data into the same dataset on which the EV is calculated for marine mammals. However, if both datasets provide valuable information (especially if the sightings integrate long term evidence), they could be treated as two separate sub-EC. That is, EV could be calculated separately for cetaceans and for pinnipeds (but the latter would only be expected to have values >0 where sightings were recorded and blanks (not zero) elsewhere, as no sighting does not mean absence). Then the EV-cetaceans and EV-pinnipeds could be included (as sub-EC for mammals) in the final calculation of the total EV across all ECs.

Q (24/01/2024): Should we use preferably **monitoring data or data-derived evidence** (predictive models/spatial interpolations)?

A: At this stage, it is suggested that the assessment prioritises monitoring data over modelled ones (for species EC), if possible (i.e., use the best data available). The former have higher confidence, albeit with possible lower spatial coverage (hence more gaps in the grid) compared to model predictions. However, rather than thinking of these as alternative sources of evidence for the EVA, they could complement each other. The model predictions could be used to fill the gaps resulting from the EVA based on monitoring data alone, if needed (e.g., if monitoring data are too patchy, there are too many gaps with no assessment in the grid).

In the grid cells where both type of data are available, the direct use of monitoring data should be prioritised (higher confidence) over models for the EV assessment. However, the model data could inform the assessment in those grid cells for which no monitoring data are available. The confidence would be lower, but having an EV assessed, even at lower confidence, would be better than having no assessment at all.

This could mean double effort, as the assessment would have to be done twice on the same EC, on both monitoring data and model predictions (where these exist), but it would provide a more complete assessment of a BBT. As a starting point, it is suggested to concentrate on the assessment with the monitoring data and, based on the EV results (for each EC and overall), then one can see whether there is the need to fill the gaps.

Q (24/01/2024): Can monitoring data on **bird breeding colonies** be used for the assessment of the EC-Birds?

A: Data on breeding birds are generally recorded for species of seabirds only where their breeding colonies occur along the coast and as number of breeding pairs or other variables representing the use/importance of that breeding colony for the species. These data are not representative of the distribution of the bird species as a whole, but only of a specific habitat they use for a key aspect of their life cycle (breeding). Where there are no data on breeding colonies it might be inferred that there are no breeding colonies for a species (=absence/0), but not that there are no birds of that species at all.

These data might not be suitable to calculate some metrics/AQs, e.g., those based on the frequency of occurrence of a species (as rare or regularly occurring), as these data could only inform rarity of breeding colonies not of the bird species as a whole. However, one of the aspects accounted by some metrics in EVA (AQ10 and AQ11) is the presence of aggregations identifying areas playing important functions (e.g., breeding grounds). In these terms, the distribution of breeding colonies would provide evidence on the distribution of an ecologically significant habitat for birds, more than on the distribution of bird species themselves and this certainly is relevant to the ecological value of an area (increased where these functional habitats occur or are found with higher abundance of breeding pairs). The EVA undertaken based on the data on seabird breeding colonies could be based only these specific metrics/AQs only. Other data (e.g., at sea transect-based survey data) could be used to provide a more complete assessment of the bird species (other AQs also calculated).

Therefore, the EC-bird assessment could have multiple aspects (sub-components) assessed separately (where the different types of data are available): one based on the distribution of seabird species (e.g., from transect surveys at sea); another one based on the distribution of seabird breeding area (based on breeding colonies data and considering coastal grid cells only); and one on waterbirds (e.g., based on intertidal bird surveys and considering coastal grid cells only). EV could be calculated for each of these components (likely with gaps in the grid) and then combined to obtain the EV for the Birds EC.

Q (24/01/2024): Can **sightings** (e.g., of marine mammals or turtles) be used for the assessment of the relevant ECs?

A: Sighting data (e.g. for mammals or turtles) are obtained from opportunistic observations. Data are recorded only where a sighting of a species was made. Where there are no records of the species, its absence cannot be inferred, as the distribution of effort (observations) is not known, then those grid cells are blanks (no data). The resulting dataset would only include cells with values >0 and blanks. Therefore, as sighting data have no absences/zeros recorded, they cannot provide a complete picture of the species distribution in the marine area.

In addition, even the frequency of sightings in an area over the years may not be a reliable proxy for abundance of the species. As there is no information on the distribution of the effort, we cannot tell whether a higher frequency of records is the result of an actual higher occurrence of the species in an area (hence likely higher ecological value) or it is due to a higher number of people/observers visiting that area over the years, which would increase the chances of mammals/turtles being spotted compared to areas that are visited less frequently. If the latter, the higher EV calculated from sighting data might not necessarily correspond to an actual higher ecological value, just a higher effort.

Based on the above, this type of data may be considered unsuitable for the EVA assessment of the related ECs. *More advice is needed from people who normally work and have better knowledge of this type of data.*

Q (03/07/2024): For assessing EVA on macrobenthos we are going to interpolate abundance of all species

A: It is suggested that, in the first instance, original sampling data are used to do the macrobenthic assessment. This will likely lead to many gaps (blanks in the dataset and in the resulting maps) for those subzones not covered by the monitoring, but, where assessed, the EV would have a higher confidence as it is based on monitoring data rather than on modelled interpolations. This would probably be quicker, so the mapped results could be used to proceed with the physical accounts in AIRES. If there is more time, then the interpolations for all species in the assemblage and calculate the AQs and EVs based on the interpolated data and use the results to fill the gaps. This would have the advantage of a wider spatial coverage of subzones in your BBT, but at a cost for confidence (as assessment on modelled data has lower confidence than that one based on the source monitoring data). This can be considered when calculating and mapping confidence during Phase 3 of EVA.

Other

Q (22/08/2023): My question is if you already have or if **you will provide a data sheet for the EVA related data** with the organism groups and algorithms etc included? And would you like to receive the data as an excel sheet or in some other format? We have many data sources where the output varies, so I will likely not be able to calculate all values in ArcGIS, but will need to visually inspect some sources and fill in data by hand.

A: The choice of the specific organism groups that each BBT will consider (amongst those indicated in the EVA guidance on Table 3) is down to the BBT, as it will have to match data availability in the case study. Spreadsheets are not provided for the data at this stage, as the source data may be of different type (occurrence, species abundance, habitat extent, or other indicators for them) and may be derived directly from monitoring (point data, transects etc) or from data-derived products (e.g., existing maps of species or habitat distribution from models), or even from expert judgment. Each BBT is free to choose the best format for the source dataset, as long as it allows its spatialisation in the assessment (hexagonal) grid(s), which each BBT has to create for their case study area and for the ecological components they use. The tabular form of the spatialised/processed dataset is expected to be a matrix where each row is one of the hexagonal grid cells that have been identified for an ecosystem component in the BBT area and each column is a variable (indicator of presence/absence and/or abundance, extent, as derived from the original evidence) for each feature of an EC (species in each species group, or habitat types from the EUNIS habitat map). If data have to be combined from multiple outputs (e.g. from monitoring of different areas of the BBT or in different years), or expert judgment has been used to manually allocate data to the grid cells, this is acceptable. All the choices made should be noted in data selection and treatment as this may affect the confidence (Phase 3 of EVA).

As for the algorithms to calculate the metrics for EVA (Phase 2), these are specified in Table 5 of the EVA guidance, with Table 6 providing additional information on the criteria for the selection of the subset of species or habitats for an EC that are relevant to the calculation of the different metrics (are features, ecologically significant ones, etc).

WP4.1 is providing the guidance to the BBTs, but it is up to each BBT (in WP2) to collect, prepare and analyse the data and undertake EVA. Year 2 of the project will be dedicated to this, although this application is phased so that all BBTs are working on the same phase of the EVA and, if there are any issues, they can be collated to the benefit of all BBTs before the next phase is implemented. It is suggested that the BBTs should implement the first phase of data collection, preparation and mapping (using the hexagonal grids relevant to EVA) between now and the end of the year (December 2023). During this period (and for the rest of the EVA implementation by BBTs), the BBTs will be supported if they have any questions or doubts or difficulties by providing advice or specific help and then make the other BBTs aware of the solutions and revise the EVA guidance accordingly, if needed.

2.3.2.2 Phase 2 – Assess EV

Assessment criteria and questions

Q (11/07/2023): Can there be an issue with **double-counting** where the same species may be considered in different questions to represent similar functional characteristics (e.g. habitat forming species and ecological significant species)?

It is important to carefully define the features (species or habitats) to be considered in different AQs to avoid double-counting. Where specific features are selected to answer a question, it is necessary to ensure they are not selected for another question for the same reason (i.e. based on the same quality). For example, if a species is selected to answer the AQs about habitat forming species, that species should not be selected in the AQs on ecological significant species if the reason for attributing ecological significance to the species is associated with its habitat forming capability (but the species could be selected if there is another quality that determines its ecological significance). Similarly, double-counting might occur across ECs when the AQs on habitat forming species or biogenic habitats are answered (AQ12, AQ13). If those AQs are answered for the EC-benthic habitats considering a specific biogenic habitat (e.g. mussel beds), then the species forming that habitat (e.g. mussels) should not be considered to answer the same AQ for the EC-benthos.

Q (04/09/2023): We have undertaken the EVA on salt marsh and seagrass ECs so far. There is an issue regarding how local rarity is calculated. These **ECs only occur in estuaries and coastal parts of our BBT**, while there is large part of the BBT (sea area) where these ECs do not (and cannot) occur. Calculating local rarity based on the 5% occurrence threshold across the whole BBT (as per current guidance) would always lead to salt marsh and seagrass species being locally rare, even if they were distributed over all the estuarine and coastline (i.e., the areas where they can occur). Therefore, we have divided our BBT into two sub-areas (estuaries & coast and the rest of the sea area) where we undertake EVA separately considering the ECs relevant to the specific sub-area.

A: It is agreed that rarity should be calculated comparing the actual distribution of the EC with its potential area of distribution (within a BBT for local rarity, at wider scales for the other types of rarity; see Table 6 of the EVA guidance). Therefore, salt marsh and seagrass ECs (and their rarity) can only be assessed considering intertidal or/and shallow subtidal estuarine and coastal marine sub-areas of the BBT. As it has been suggested, the BBT area may be divided into sub-areas (e.g. based on depth zones) in which EVA is undertaken separately considering only the ECs relevant to each sub-area. The results will be spatially complementary and may be combined into a single final map of EV (albeit with the caveat that the EV values in different parts of the BBT may result from different combinations of ECs).

A similar approach applies to any cases when the evidence for an EC only covers part of the BBT, for the reasons described above or where monitoring of an EC was only undertaken in a part of the BBT (even if the EC may potentially be distributed in other areas of the BBT which were not monitored). Where possible, modelling could be undertaken to extrapolate the monitoring results on the EC to the wider BBT area (e.g., using seabed characteristics and benthic data from monitoring to predict the distribution of benthic species or communities/habitats over the wider BBT area; or other species distribution models to predict habitat suitability of fish, marine mammals, seabird species, etc). Where this is not possible and the localised monitoring data are directly used for the EVA, the EVA assessment in general for this EC, and the rarity calculation in the specific, can only be applied to the areas of the BBT for which data are available, this excluding any gap areas (n/a) from the calculations. The EC-specific results can then be combined in different ways, e.g., by calculating and mapping EV across the whole BBT using the ECs with widespread evidence and then ‘zooming-in’ to a specific area of the BBT, where additional ECs (from the localised monitoring) can be added to the total EV assessment.

Q (06/02/2025): *[following changes in AQ/EV aggregation method] In lieu of phytoplankton community data from any monitoring in the BBT, we used remote sensing data to give chlorophyll-a concentration as a proxy for phytoplankton biomass. Our assumption was that we could only answer*

AQs 7, 8 and 9 using these data because we have no species resolution. However, that means that AQ7 would be essentially the presence/absence of phytoplankton biomass and a subzone with any concentration of Chl-a would get a 5. This would be ok per se but now in the new way of calculating the EC-specific EV using the maximum, it means that the presence/absence of chl-a will completely override the more quantitative measures of where ecological value may vary with respect to increased phytoplankton biomass

A: The AQs assessed using qualitative data are likely to be scored higher than their respective quantitative AQ, as they only need to reflect the presence of any individual of the species (max. score 5), not the variability in their abundance (with max. score 5 only achieved with maximum abundance for the species in the data). There is the risk that, even if quantitative AQs are measured, they would be frequently ‘overpowered’ by the qualitative ones in generating the final EV (even if it is just one qualitative AQ v. multiple quantitative AQs), following the change in the method for aggregating EC-specific EV results into a total EV (see ‘Q (30/10/2024): Would averaging EV scores not homogenise the results across the map?’), now using maximum value for aggregation. Taking this into consideration, the protocol for selecting AQs for the assessment has been revised to allow prioritising the quantitative AQ over the correspondent qualitative AQ, when quantitative data area available to measure it. A decision system flow-chart has been designed and **added in the EVA guidance (June 2025)** to support the user in the selection of AQs for the assessment. As an assessment based on quantitative data (hence AQs) is considered to have higher confidence than a qualitative one, the confidence assessment in the protocol has been modified accordingly.

Metrics calculations (algorithms) & aggregation

Q (11/07/2023): The **scoring method applied to the AQ metrics** was discussed at the EVA workshop on 9 June 2023, specifically how to assign classes ranging from 1 to 5 based on quantitative data.

A: The earlier iterations of the protocol included measure of continuous variables (abundance etc) and their classification into 5 discrete classes to which an integer value of 1 to 5 is assigned. Further calculations (averaging etc) were then undertaken on these class values, effectively treating the discrete classes as numbers. The results were then re-classified again into 5 classes. After discussions on how to improve this scoring method, the following changes have been made to the protocol. The continuous variables (abundance etc) measured to answer an AQ are rescaled into a 0-5 range, without classification (i.e. keeping them as continuous variables but on a common 0-5 scale). Scale starting with 0 gives better representation of absence and zero-abundance (classed as 1 before). The rescaled variables can be equally combined without the potential issue of treating discrete class values as continuous variables and having to re-classify them again afterwards. Classification of the values into 5 discrete classes is taken out of the algorithm and is only added at the end, on mapping the results: the EV results will also be values ranging 0-5 and one can map them as continuous variable or group them into 5 classes from very low (0-1 values) to very high (4-5 values). **The EVA guidance document has been updated accordingly.**

Q (04/12/2023): Will you provide a data sheet/**template** for the EVA related data with the organism groups and with formulas for the calculation of the algorithms etc included?

A: The choice of the specific organism groups that each BBT will consider (amongst those indicated in the EVA guidance on Table 3) is down to the BBT, as it will have to match data availability in the case study. We are not going to provide any spreadsheet/template for the data, as the source data may be of different type (presence, abundance, or other indicator for it) and may be derived directly from monitoring (point data, transects etc) or from data-derived products (e.g., existing maps of species distribution from models), or even from expert judgment. Each BBT is free to choose the best format for the source dataset, as long as it allows its spatialisation in the assessment grid(s) (which each BBT has to create for their case study area and for the ecological components they use).

The tabular form of the spatialised/processed dataset would essentially be a data matrix (e.g. an Excel spreadsheet) where each row is one of the hexagonal grid cells that have been identified for an

ecosystem component in the BBT area and each column is a variable (indicator of presence/absence and/or abundance, as derived from the original evidence) for each feature of an EC (species in each species group, or habitat types from the EUNIS habitat map). If there is the need to combine data from multiple outputs or use expert judgment to manually allocate data to the grid cells, this is acceptable. However, choices made should be recorded in the data selection and treatment as this may affect the confidence (Phase 3 of EVA). Separate data matrices may be created for each EC and add columns with the calculations of the algorithms for the different questions.

No template has been provided with the formulae for the metrics calculation as these can be derived from the algorithm steps in Table 5 of the EVA guidance (Phase 2), with Table 6 also providing additional information on the criteria for the selection of the subset of species or habitat for an EC as relevant to the different questions. The formula a BBT will apply will depend on which species/habitats are selected from the data to answer a specific question (and their location as a column in the data matrix) and the BBT may choose to create separate formulae for each step of an algorithm (hence the calculation of a metric is undertaken over multiple columns) or to combine the steps in a single formula showing the scored result. It is up to the BBT to find the way that suits them most.

Q (04/12/2023): In AQ#9, Algorithm Column, point 5: What does it mean “Calculate the percentage of the abundance of the ROF that occurs in these 5% most important subzones (=Y)”

A: Once the top 5% subzones have been identified with the highest abundance ratio for a ROF species (e.g. if there are 200 subzones with data and the abundance ratio X_i/X has been calculated, these would be the 10 subzones with the highest value), then the abundance of the ROF recorded in each of these top 5% grid cells can be summed ($\sum X_i$ across top 5% cells) and divided by the total abundance of the ROF in the BBT ($\sum X_i$ across all grid cells). The resulting number is then multiplied by 100 to give the required percentage Y (but this may also be calculated as a proportion, i.e. without multiplying it by 100, as long as this is done for all cells – the result would not change once the final ROF-value is rescaled). ***The EVA guidance document has been updated (with clarification on this aspect) accordingly.***

Q (29/01/2024): Regional and National features: Answering AQ1 and AQ2 concerning the local features is not an issue, but we are not too sure on how to apply the regional and national features. Specifically, how do we apply the 50 x 50km and 10 x 10km search on the hexagonal grids? It's probably something easy to define in the code but we are not sure on how to approach this.

A: The 50x50km and 10x10km grids that can be used to identify which features are regionally or nationally rare are independent from the hexagonal grids. The species/habitat distribution would be mapped at the regional or national scale using the 10 or 50 km grids to see whether the species can be classified as rare at that scale (based on the criteria in Table 2.6 of the EVA guidance). The species/habitats distribution within the BBT then are to be mapped using the relevant hexagonal grid and those that have been identified as regionally or nationally rare will be used for the calculations of AQ3 – AQ6.

If there are no data that allow to map the wider distribution of the features over the region or national marine area to assess their rarity at that scale, then other sources providing that classification can be used (e.g., if there are lists of nationally/regionally rare species or habitats in the literature) or, alternatively, expert judgment. If no information or expert judgment on either or both national and regional rarity is available, then the question based on those specific features can be skipped.

Q (29/01/2024): Number of features and count of species: For the AQ7 and AQ8, could you clarify the difference between these two questions.

A: AQ7 is calculated taking into account the presence/absence distribution of ALL the features (species or habitats) for the EC (including both rare and frequently occurring ones) for which data are available in the BBT. AQ8 is calculated taking into account only the regularly occurring features (ROFs) in the

BBT (hence excluding the locally rare species or habitats) and it needs quantitative data for them (abundance or proxy for it).

Q (29/01/2024): Mutualistic or symbiotic species: Is there a centralized list for specific species that we should test or is this BBT/region/dataset specific? For bird species or some benthic species this is more likely to be well documented, but for phytoplankton and zooplankton this is not so easy to identify. We were just wondering what the agreement was around this.

A: There is no centralized list of symbiotic species, it is up to each BBT to identify them from the species pool they have in their area. However, having a centralized list shared between all BBTs is a good idea. *As a starting point, a shared Excel worksheet ('EVA_Features_classification.xlsx') has been added in Next Cloud for BBTs to list the symbiotic species they have identified, as well as the ecologically significant features and the habitat forming species (or biogenic habitats) to be identified for the purpose of the EVA. This way, the information can be exchanged between BBTs.*

Q (05/07/2024): AQs that one should be able to answer as a minimum?

A: The AQs that one will be able to answer for a given EC (an assemblage of species or of habitats) depend on the EC and the data available:

1. AQ7 is the AQ that should be able to be measured in any case for the EC being assessed as: (i) it is based on qualitative information (presence/absence), so it should be able to be measured whether or not there are qualitative or quantitative data; (ii) it considers all the species or habitats present in the assemblage being assessed and does not require any additional knowledge or ability to identify specific features of interest within that assemblage (LRF, RRF, ESF etc).
2. Whether the other AQs can be assessed depends on the ability to identify features of interest (LRF, RRF, NRF, ROF, ESF, HFS/BH, SS) in the assemblage being assessed. This will be influenced by the knowledge and data available (e.g., there may be no knowledge of species being identified as rare in the wider region of the study area, nor data to assess this spatially (as per the RRF criterion in Table 6 of the EVA guidance), so it is not possible to assess either AQ3 or AQ4; there may be a dataset that only includes main/common species and based on this there might not be LRF in it, so it is not possible to assess AQ1 or AQ2 as the feature of interest is not present in the dataset).
3. If there are only qualitative data (presence/absence), then the other AQs that may be assessed (depending on the factors described above) include AQ1, AQ3, AQ5, AQ10, AQ11 and AQ13 for any EC, as well as AQ15 for species EC only.
4. If there are quantitative data, a general rule is that, if the quantitative version of a specific AQ can be measured in one subzone (e.g. AQ2, AQ4, AQ11), then it should be possible to also measure the qualitative version of the same AQ in that subzone (e.g. AQ1, AQ3, AQ10) as well as AQ7.
5. If there are quantitative data and AQ2 (LRF) can be measured, then it should be possible to also measure AQ8 and AQ9 as these AQs focus on ROF, which are all the other species or habitats in the assemblage (other than LRF) and require quantitative data. If it is not possible to establish which features in the assemblage are LRF, then it would also not be possible to identify which features are ROF and therefore either AQ1/2 or AQ8/9 could not be measured. However, if it is possible to assess the features in the assemblage for local rarity and it is possible to establish that none of those in the data is LRF (see example given in point 2), then it would be possible to say that all features in the dataset are ROF and therefore it is possible to apply AQs 8 and 9 (provided that there are quantitative data).

Q (05/07/2024): **Number of subzones assessed per AQ:** Can the number of subzones assessed for a given EC be different between AQs?

A: The possible cases are these:

1. There are abundance data for X subzones (sbz) (excluding blanks): in this case it should be possible to answer multiple AQs (AQ7 as a minimum, plus others depending on whether it is possible to identify LRF, RRF, NRF, ESF etc. in the data), including both quantitative and qualitative AQs (as if there are abundance data, then it is also possible to know presence/absence). Irrespective of which AQs are answered, the number of subzones assessed for them should always be X (with variable number of zeros).
2. There are presence data for Y sbz (excluding blanks): in this case it is possible to answer multiple AQs (AQ7 as a minimum, plus others depending on whether it is possible to identify LRF, RRF, NRF, ESF etc. in the data), but only amongst those requiring qualitative data (hence no AQ2, AQ4, AQ6, AQ8-9, AQ11, AQ13, AQ15). No matter which qualitative AQs are answered, the number of subzones assessed for them should always be Y (with variable number of zeros).
3. There are abundance data for X sbz, but also presence/absence data that cover additional Y sbz: in this case it should be possible to answer multiple AQs (AQ7 as a minimum, plus others depending on whether LRF, RRF, NRF, ESF etc. can be identified in your data). It should be possible to answer the qualitative AQs in all the sbz where there are data (of any type), while it is possible to answer the quantitative AQs only in the sbz where there are abundance data (subzones with only presence/absence data will have blanks for the quantitative AQs). In this case, it is reasonable to expect a difference in the number of subzones between AQs, in that, all the quantitative AQs assessed will only have measures (incl. 0) in X sbz, while all the qualitative AQs being assessed will have measures (incl. 0) in X+Y sbz.

Q (06/07/2025): **Clarification about AQ standardisation/scaling** – should an AQ value be expected to cover the full 0-5 range across the grid cells assessed in an area?

A: Although the AQ values may theoretically vary on between 0 and 5, the full range is represented by the values across the grid cells, as the theoretical maximum diversity might not be realised in any of these areas. To clarify, see example below:

For example, if AQ7 is being answered for benthic habitats and the BBT has 9 habitats. Each habitat is assigned 0 or 5 in a subzone depending on its absence or presence. There will only be AQ7 observed max=5 if all 9 habitats occur in one subzone/grid cell, but this might not happen and it might be that the highest habitat diversity in any subzone in the BBT is 6 habitats, which would lead to AQ7 observed max = 3.33 $[(5+5+5+5+5+5+0+0+0)/9]$. Therefore, the observed max for the AQ may be lower than the theoretical max (5).

Q (30/10/2024): **Wouldn't averaging EV scores homogenise the results across the map?**

A: The aggregation of values (AQs into EC-specific EV and EC-specific EV into Total EV) in the EVA method is currently done by averaging the values (first across AQs and then across ECs). It has been commented that averaging might 'homogenise' the results across the maps, and this seems a valid point also by looking at the EV maps resulting from the work done in the BBTs. Based on this, the aggregation method has changed, whereby the maximum value is used (across AQs to derive the EC-specific EV and across ECs to derive the total EV) instead of the average. ***The EVA protocol has been changed accordingly.*** It is highlighted that this change in method has consequences for the way AQs are selected for the assessment and also for how the confidence is represented. These parts of the protocol have been changed accordingly, as also highlighted in the relevant questions reported here (see sections on 'Assessment criteria and questions' and 'Phase 3 - Confidence').

Other

Q (11/07/2023): Each EC is currently given equal weight in the calculation of the final EV. Can alternative **weighting options for the ECs** be explored, e.g., to also account for different conservation or other type of human value assigned to different ECs?

A: Each EC is given equal weight in the calculation of the final EV because the assessment measures the intrinsic ecological value and all ECs are considered equally important to support the ecosystem and its functioning. Giving different weights to the ECs, e.g. based on the different conservation or other type of human value assigned to different ECs, would lead to a different type of assessment, considering that the key principle of EVA is to assess the intrinsic ecological value of an ecosystem, independently from human interactions/uses, hence solely from an eco-centric perspective. Identifying which ECs are more important (hence bring more weight) from a purely ecological point of view would require philosophical discussions and expert input. The work in Tasks 4.2 and 4.3 may also help in defining different weights from a more anthropocentric perspective (e.g., based on goods and benefits to society and economic value). At the EVA workshop on 9 June 2023, it was agreed that the EVA giving equal weighting to the ECs should be prioritised for the baseline application in the BBTs, but discussions about alternative weightings may be undertaken throughout the project and the results applied as additional implementations, where appropriate. The Analytical Hierarchical Process (AHP) methodology was suggested to derive different weights from an expert judgment perspective; this involves collecting information from local experts and performing repeated pairwise judgments to produce consistent relative weightings.

Q (11/07/2023): Can **EVs be compared between different BBTs across Europe**?

A: No. The metrics (and resulting EVs) are expressed as a relative score which depends on the diversity measured across the subzones (grid cells) in the individual BBT. For example, if the range of nationally rare seabird species in one BBT (across its subzones) is between 0-6 and 0-22 in another BBT, a subzone with 6 rare seabirds in the first BBT and one with 22 rare seabirds in the second BBT will both score 5 for that metric and they will both contribute equally to the calculation of EV.

In order for the EV results to be comparable between BBTs, a different re-scaling procedure for the AQs would be needed. This would involve applying the algorithms to obtain the EV scores using the variability range (min-max) of each metric (AQ) ACROSS all the BBTs (instead of WITHIN a BBT, as done in current protocol), ensuring that the final EV is comparable between them. However, there are factors of variability at the wider geographical scale that may influence the variability of the metrics and therefore of the resulting EV across BBTs but that might not necessarily reflect a change in intrinsic value of the ecosystem (e.g., naturally low diversity areas/regions, effects of biogeography). These should be considered.

2.3.2.3 Phase 3 – Confidence

Q (12/01/2024): Available mapped data (e.g., EUNIS habitat maps or indeed any other spatial predictions of distribution of species for different ecosystem components) may have incomplete coverage of an area and therefore **% of mapped area** may be <100% in some subzones/grid cells. This **may also account for data availability** and therefore affect the confidence assessment. However, currently, data availability is only measured as number of samples/observations available in the subzone (Table 8 of the EVA guidance) and only for data directly derived from monitoring. How can the aspect of data availability represented by % area mapped/covered by the data be included in the confidence assessment.

A: The % of area covered by the data (monitored or predicted by a spatial model) is certainly an important aspect of data availability for certain types of data (e.g. EUNIS maps for the habitat EC). ***The EVA guidance document has been updated accordingly (with changes to the protocol for confidence assessment in the text, Tables 7 and 8 and Figure 3)***

Q (30/10/2024): Amendments to the **classification of data availability for monitoring data** (to define w_i)

A: Where monitoring data are used to inform the EVA, data availability is measured as number of samples or observations used for the AQ calculations in a grid cell and it is classified by dividing its range (min-max across all grid cells) into 5 equally-sized classes. The EVA implementation undertaken MARBEFES (e.g., in the North Sea BBT7) has shown that the range of number of samples used to inform the EV calculations may be very wide (e.g., up to around 1000 samples in some cells for BBT7) and a skewed distribution is not uncommon (e.g., only very few cells would have data availability close to the maximum, and most of cells would be closer to the minimum). As a result, the current method for classifying data availability across their range (using fixed/equal-sized classes dividing the range) would lead to most of the grid cells in the area having low data availability, leading to low-confidence grid cells dominating the map (as a reflection of the skewed distribution of the data availability across grid cells). Taking this into consideration, the approach for classifying data availability within their range has been changed, whereby the data availability range is to be divided into **20% quantiles** (hence accounting for the distribution of data within the range) to derive the five classes. **The EVA guidance document has been updated accordingly.**

An illustration of the problem and the protocol change applied is given in the figures below:



Proposed changes to EVA protocol



2. Phase 3 - Confidence:

Step: classification of data availability (to define w_i)

Problem: data availability may vary widely across grid cells and distribution may be right skewed (example: # of samples in the grid cells ranges 3 to 500, but only one grid cell has 500 samples, and most grid cells have <5 samples). Classifying the data availability (from low to high, 5 classes) by dividing the min-max range into 5 equally-sized classes may lead to confidence map with most cells with low confidence.

Solution: use 20% quantiles to divide the min-max range



EVA Phase 3: Assess Confidence (current protocol, Oct-24)



$$Confidence = \frac{1}{N} \sum_{i=1}^N AQ_i w_i$$

w_i = weight based on evidence used to calculate the AQs (essentially, the confidence in the evidence-base used for the AQ calculations)

Table 8. EVA guidance: weight (w_i) to be assigned to answered AQ for each EC in each subzone in the confidence assessment (Equation 1).

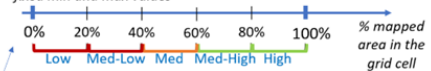
Base for the assessment (b) & Data use (c)	Data availability (d)*				
	Low	Medium-low	Medium	Medium-High	High
Expert judgment					
Data based					
Indirect use of the data (model products)	1	1	2	3	3
Direct use of monitoring data	1	2	3	4	5

*Note: The data availability in each subzone (for a given EC) is measured as the number of samples/observations available in the subzone where monitoring data are directly used for the assessment, or as the % mapped area in the subzone where interpolated or otherwise modelled spatial distribution maps are used as source data.

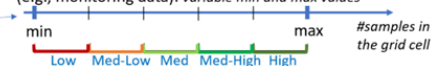
For the data availability categorisation (as low, medium-low etc), the range of the data availability measure is calculated across all the subzones in the BBT (Range=Min – Max) and it is divided in 5 (X=Range/5). The data availability in a subzone is classed as Low with a value between Min and Min+X, Medium-low between Min+X and Min+2X, Medium between Min+2X and Min+3X, Medium-high between Min+3X and Min+4X, and High between Min+4X and Max. Note that, where data availability is measured as % mapped area, Min = 0% and Max = 100% (i.e., data availability is low with % mapped area values ≤20%, medium-low with values >20% and ≤40%, etc).

Data availability classification:

•Where measured as % mapped area (e.g., habitat models): fixed min and max values



•Where measured as number of samples/observations (e.g., monitoring data): variable min and max values





EVA Phase 3: Assess Confidence (current protocol, Oct-24)



w_i = weight based on evidence used to calculate the AQ_i s (essentially, the confidence in the evidence base used for the AQ calculations)

$$Confidence = \frac{1}{N} \sum_{i=1}^N AQ_i w_i$$

Table 8. EVA guidance: weight (w_i) to be assigned to answered AQ for each EC in each subzone in the confidence assessment (Equation 1).

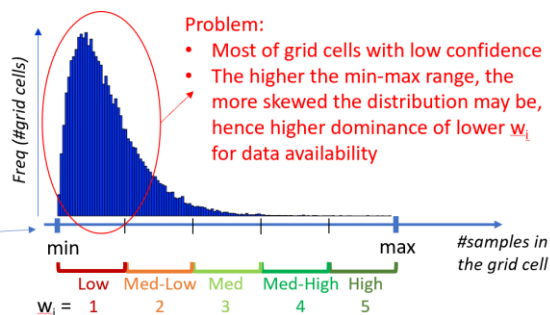
Base for the assessment (b) & Data use (c)		Data availability (d)*					Not known
		Low	Medium-low	Medium	Medium-High	High	
Expert judgment		1					
Data based	Indirect use of the data (model products)	1	1	2	3	3	3
	Direct use of monitoring data	1	2	3	4	5	-

*Note: The data availability in each subzone (for a given EC) is measured as the number of samples/observations available in the subzone where monitoring data are directly used for the assessment, or as the % mapped area in the subzone where interpolated or otherwise modelled spatial distribution maps are used as source data.

For the data availability categorisation (as low, medium-low etc), the range of the data availability measure is calculated across all the subzones in the BBT (Range=Min – Max) and it is divided in 5 (X=Range/5). The data availability in a subzone is classed as Low with a value between Min and Min+X, Medium-low between Min+X and Min+2X, Medium between Min+2X and Min+3X, Medium-high between Min+3X and Min+4X, and High between Min+4X and Max. Note that, where data availability is measured as % mapped area, Min = 0% and Max = 100% (i.e., data availability is low with % mapped area values ≤20%, medium-low with values >20% and ≤40%, etc).

Data availability classification:

- Where measured as number of samples/observations (e.g., monitoring data): variable min and max values



7



EVA Phase 3: Assess Confidence (proposed change, Oct-24)



w_i = weight based on evidence used to calculate the AQ_i s (essentially, the confidence in the evidence base used for the AQ calculations)

$$Confidence = \frac{1}{N} \sum_{i=1}^N AQ_i w_i$$

Table 8. EVA guidance: weight (w_i) to be assigned to answered AQ for each EC in each subzone in the confidence assessment (Equation 1).

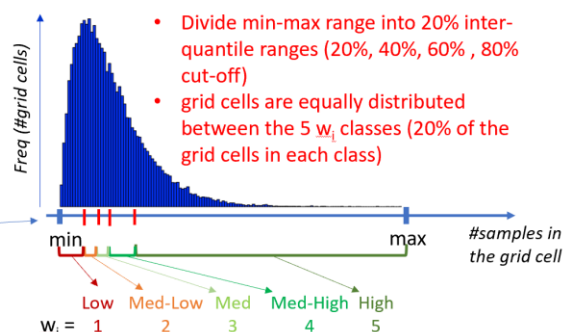
Base for the assessment (b) & Data use (c)		Data availability (d)*					Not known
		Low	Medium-low	Medium	Medium-High	High	
Expert judgment		1					
Data based	Indirect use of the data (model products)	1	1	2	3	3	3
	Direct use of monitoring data	1	2	3	4	5	-

*Note: The data availability in each subzone (for a given EC) is measured as the number of samples/observations available in the subzone where monitoring data are directly used for the assessment, or as the % mapped area in the subzone where interpolated or otherwise modelled spatial distribution maps are used as source data.

For the data availability categorisation (as low, medium-low etc), the range of the data availability measure is calculated across all the subzones in the BBT (Range=Min – Max) and it is divided in 5 (X=Range/5). The data availability in a subzone is classed as Low with a value between Min and Min+X, Medium-low between Min+X and Min+2X, Medium between Min+2X and Min+3X, Medium-high between Min+3X and Min+4X, and High between Min+4X and Max. Note that, where data availability is measured as % mapped area, Min = 0% and Max = 100% (i.e., data availability is low with % mapped area values ≤20%, medium-low with values >20% and ≤40%, etc).

Data availability:

- Where measured as number of samples/observations (e.g., monitoring data): variable min and max values



(no changes where data availability is measured based on % area mapped)

8

Q (30/10/2024): [following changes in AQ/EV aggregation method] I have one question though about calculating final confidence scores - not exactly sure how to approach it (is it using confidence scores from the EC that had the highest EV in the subzone, or is the highest confidence score from whichever EC had the highest confidence in the subzone?) I'm assuming it's the first one, but please let me know if that's correct and I'll amend accordingly.

A: Following the change in the method for aggregating EC -specific EV results into a total EV (see 'Q (30/10/2024): Would averaging EV scores not homogenise the results across the map?'), the way confidence values are transferred between result representation was adjusted as follows. To express the confidence for the total EV integrating multiple EC s in a subzone, the confidence associated with the EC selected to represent the total EV in the subzone (i.e., the EC having the highest EV compared to the other EC s in that subzone) is to be used. Therefore, the answer to the question above is that the first option is more appropriate to give the confidence associated with the actual EV in the cell/subzone. **The EVA guidance document has been updated accordingly.**

Other

Q (11/07/2023): **Additional work on biological valuation** in the Baltic was done by Węslawski et al. (2009)

A: **This has been added to the EVA guidance.**

Q (11/07/2023): **Interaction of EVA with other WPs** in MARBEFES

A: Once the EVA tool has been tested and implemented in the BBTs, it might be possible to incorporate information resulting from WP3 into the EVA to enhance the tool (e.g., incorporating outputs from the ecological models and data from innovative monitoring methods to inform the assessment). This would also be an opportunity to explore incorporating more functional aspects into the tool. EVA tool features may also be considered in WP6, when addressing upscaling, relevance to other areas, and potentially other challenges.

Q (11/07/2023): **The proposed EVA approach is only one of many approaches** that can be used to assess the ecological value of an ecosystem. Can BBTs apply other approaches?

A: It is acceptable that there are other approaches, but, for standardisation, the same EVA protocol should be applied across BBTs, as a minimum. Should the BBTs want to implement alternative approaches, they are free to do it in addition to the standardised approach. If so, circulating the information on the pros and cons of different approaches would be of benefit.

Q (27/11/2023): **A phased implementation of the EVA protocol by the BBTs** has been agreed, with meetings and discussions at the end of each assessment phase.

A: The timeline for the phased implementation of the EVA protocol by the BBTs within Year 2 of the project is as follows:

- **Phase 1:** the BBT defines the study system by mapping the BBT area and the spatial grids for the relevant ECs; identifies, gathers and maps (on the selected grids) the relevant evidence for the assessment; prepares the data (e.g., where they need to be aggregated or standardised from multiple surveys if needed); identifies which subset of species or habitats are relevant to the different questions of the assessment (based on Table 6 of the guidance); extracts the values for each grid cell (e.g., as average of the data points within a cell) into the data matrix that will be used to calculate the metrics during Phase 2. The data matrix will have rows = grid cells in the BBT, columns = variable (indicator of presence/absence and/or abundance, as derived from the original evidence) for each feature of an EC (species in each species group, or habitat types from the EUNIS habitat map). Timeline: Month 13-16/17 (September 2023 to December 2023/ January 2024)
- **Phase 2:** the BBT applies the algorithm for the calculation of the different metrics (see Table 6 of the EVA guidance) to the data matrix to obtain the resulting score values for each question; combines the scores across questions for each EC into the EC-specific EV; combines the scores across EC into the total EV; maps the resulting scores back into the BBT grid. Timeline: Month 17-20/21 (January to April/May 2024).
- **Phase 3:** the BBT calculates the confidence criteria and overall score and maps the results into the BBT grid. Month 21-24 (May to August 2024).

Please note that BBTs are not asked to provide any output for each phase. The timeline above is there just to help the BBTs to organise their work for the implementation of the EVA protocol and so that all BBTs are working on the same phase. This way, if a BBT has an issue during one phase, the solution can be shared as it could also help other BBTs who are at the same stage of the EVA implementation. It is planned to organise a meeting with the BBTs at the end of each phase to discuss the challenges faced by different groups and find solutions to overcome similar issues, thus progressing to the next phase. Support can be given to individual BBTs during each phase, e.g., to

check or review intermediate outputs and spreadsheets and help overcome possible issues, where needed.

Q (21/10/2024): I have created a few slides (below) summarising the guidance on the EVA confidence assessment and also including a couple of mock-up examples of how the confidence score is calculated.

A: .



EVA Phase 3: Assess Confidence



A **CONFIDENCE SCORE** is associated to each EV result, considering multiple sources of uncertainty:

$$\text{Confidence} = \frac{1}{N} \sum_{i=1}^N AQ_i w_i$$

$AQ_i = 1$ if the AQ_i has been answered, 0 if not

$N = \text{max. number of } AQ\text{'s defined for the EC (13 for habitat-based ECs, 15 for species-based ECs)}$

$w_i = \text{weight based on evidence used; see Table 8}$

As a result, confidence score varies between 0 and 5

MARBEFES WP4.1 EVA: Confidence Assessment (Oct-24)

Table 8. EVA guidance: weight (w_i) to be assigned to answered AQ for each EC in each subzone in the confidence assessment (Equation 1).

Base for the assessment (b) & Data use (c)		Data availability (d)*					
		Low	Medium-low	Medium	Medium-High	High	Not known
Expert judgment		1					
Data based	Indirect use of the data (model products)	1	1	2	3	3	3
	Direct use of monitoring data	1	2	3	4	5	-

*Note: The **data availability** in each subzone (for a given EC) is measured as the **number of samples/observations available** in the subzone where monitoring data are directly used for the assessment, or as the **% mapped area** in the subzone where interpolated or otherwise modelled spatial distribution maps are used as source data.

For the data availability categorisation (as low, medium-low etc), the range of the data availability measure is calculated across all the subzones in the BBT (Range=Min – Max) and it is divided in 5 (X=Range/5). The data availability in a subzone is classed as Low with a value between Min and Min+X, Medium-low between Min+X and Min+2X, Medium between Min+2X and Min+3X, Medium-high between Min+3X and Min+4X, and High between Min+4X and Max. Note that, where data availability is measured as % mapped area, Min = 0% and Max = 100% (i.e., data availability is low with % mapped area values ≤20%, medium-low with values >20% and ≤40%, etc).

4



EVA Phase 3: Assess Confidence



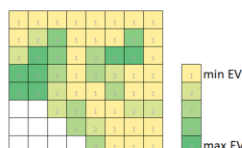
Example (mock-up): Benthic habitats EC

$$\text{Confidence} = \frac{1}{N} \sum_{i=1}^N AQ_i w_i$$

(value ranging 0-5)

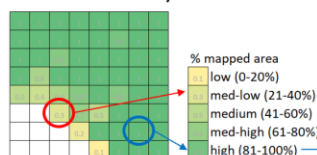
EV map:

EV calculated based on $AQ1, AQ2, AQ7, AQ8, AQ9, AQ12, AQ13$



- Data source: EUNIS habitat map

- Data availability:



Base for the assessment (b) & Data use (c)		Data availability (d)*					
		Low	Medium-low	Medium	Medium-High	High	Not known
Expert judgment		1					
Data based	Indirect use of the data (model products)	1	1	2	3	3	3
	Direct use of monitoring data	1	2	3	4	5	-

$$\text{Confidence} = \frac{1}{13} (1 + 1 + 0 + 0 + 0 + 0 + 1 + 1 + 1 + 0 + 0 + 1 + 1) * 1 = 0.54$$

$$\text{Confidence} = \frac{1}{13} (AQ1 \cdot AQ2 \cdot AQ3 \cdot AQ4 \cdot AQ5 \cdot AQ6 \cdot AQ7 \cdot AQ8 \cdot AQ9 \cdot AQ10 \cdot AQ11 \cdot AQ12 \cdot AQ13) * 3 = 1.62$$

MARBEFES WP4.1 EVA: Confidence Assessment (Oct-24)

13 AQs for benthic habitats EC

$w=3$ for all AQs as the same data have been used

5



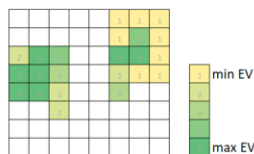
EVA Phase 3: Assess Confidence



Example (mock-up): Benthic species EC

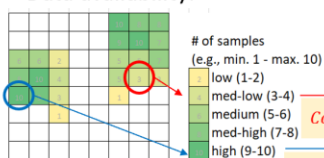
EV map:

EV calculated based on
AQ1, AQ2, AQ7, AQ8,
AQ9, AQ10, AQ11,
AQ12, AQ13, AQ14,
AQ15



- Data source: monitoring data (grab survey)

- Data availability:



Base for the assessment (b) & Data use (c)		Data availability (d)*					
		Low	Medium-low	Medium	Medium-High	High	Not known
Expert judgment		1					
Data based	Indirect use of the data (model products)	1	1	2	3	3	3
	Direct use of monitoring data	1	2	3	4	5	-

$$Confidence = \frac{1}{15} (1 + 1 + 0 + 0 + 0 + 0 + 1 + 1 + 1 + 1 + 1 + 1 + 1 + 1 + 1) * 2 = 1.47$$

$$Confidence = \frac{1}{15} (1 + 1 + 0 + 0 + 0 + 0 + 1 + 1 + 1 + 1 + 1 + 1 + 1 + 1 + 1) * 5 = 3.67$$

MARBEFES WP4.1 EVA: Confidence Assessment (Oct-24)

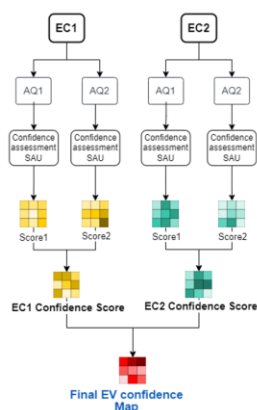
15 AQs for benthic habitats EC

w=3 for all AQs as the same data have been used

6



EVA Phase 3: Assess Confidence



- Confidence calculated for each EC/dataset in each grid cell
e.g., *Confidence scores for EC1 & Confidence scores for EC2*
- Total confidence averaged across ECs in each subzone (i)
e.g., $(\text{Confidence score EC1} + \text{Confidence score EC2})/2$
- For mapping of confidence (for individual EC or for total EV), confidence score classified on a 3-level scale:

Confidence class (score range)

Low (0-1.67]
Medium (1.67-3.33]
High (3.33-5]

MARBEFES WP4.1 EVA: Confidence Assessment (Oct-24)

7

3. Guide 2: Environmental Valuation & Natural Capital Accounting

3.1 Background to Natural Capital and Natural Capital Accounting

As illustrated in Figure 3.1, the marine environment can be structured into Natural Capital, which includes stocks (Ecosystem Assets) and flows (Marine Processes and Ecosystem Services), which with an input of Complementary Capital (e.g., built, human and social capital) can lead to Societal Benefits. In this context, Marine Processes “relate to inter-relationships between the physico-chemical and biological attributes” and they underpin the ecosystem services flows that are “products from nature that can be turned into benefits with varying degrees of human input”. An ‘eco-centric’ approach is undertaken when it is recognised that the marine processes exist (and some of them are not even known by humans) and have their own ecological value, while the term ecosystem services is used mainly within an ‘anthropo-centric’ perspective to identify the services that nature provides to society. In environmental economics, it is important to distinguish between ecosystem services and benefits that nature provides to humans. Ecological processes and functions underpin ecosystem services, whereas benefits are the goods, or the welfare benefits, that humans use or benefit from nature. It is solely on the benefits that a monetary value can be assigned. It is important to conciliate these complementary views in natural capital accounting as they interact with each other during all stages of the approach. Natural Capital Accounts, in fact, aim to reveal the contribution of Nature to the economy and the impact of the economy on the use of the natural resources.

All these terms are defined by the United Kingdom Natural Capital Committee (NCC, 2019) and are embedded within Figure 3.1. The figure recognises that ecosystem services are different to societal benefits, as ecosystem services are a naturally occurring process in the natural environment domain, whereas societal benefits are secured from ecosystem services often through the input of complementary capital (built, human and social) in the human domain. However, not all the societal benefits, especially the non-SNA ones (see Sections 1 and 3) require an input of other forms of capital for society to enjoy them (e.g., healthy climate). The natural capital assets (that include the marine habitats and species) are linked to the benefits obtained by society (valued through their support to human well-being and the economy) through the flow of ecosystem service provided by the assets (Figure 3.1).

Figure 3.1 has been adapted from the coastal ecosystem services framework developed by Turner et al. (2015) and reflects the definitions of terms developed by the NCC (2019). The framework in Figure 3.1 is in line with the Common International Classification of Ecosystem Services (CICES) V5.1 Framework (Haines-Young & Potschin, 2018) with respect to ecosystem services, however Figure 3.1 separates ecosystem services and societal benefits which is important for developing the economic valuations (Fisher & Turner, 2008) within MARBEFES.

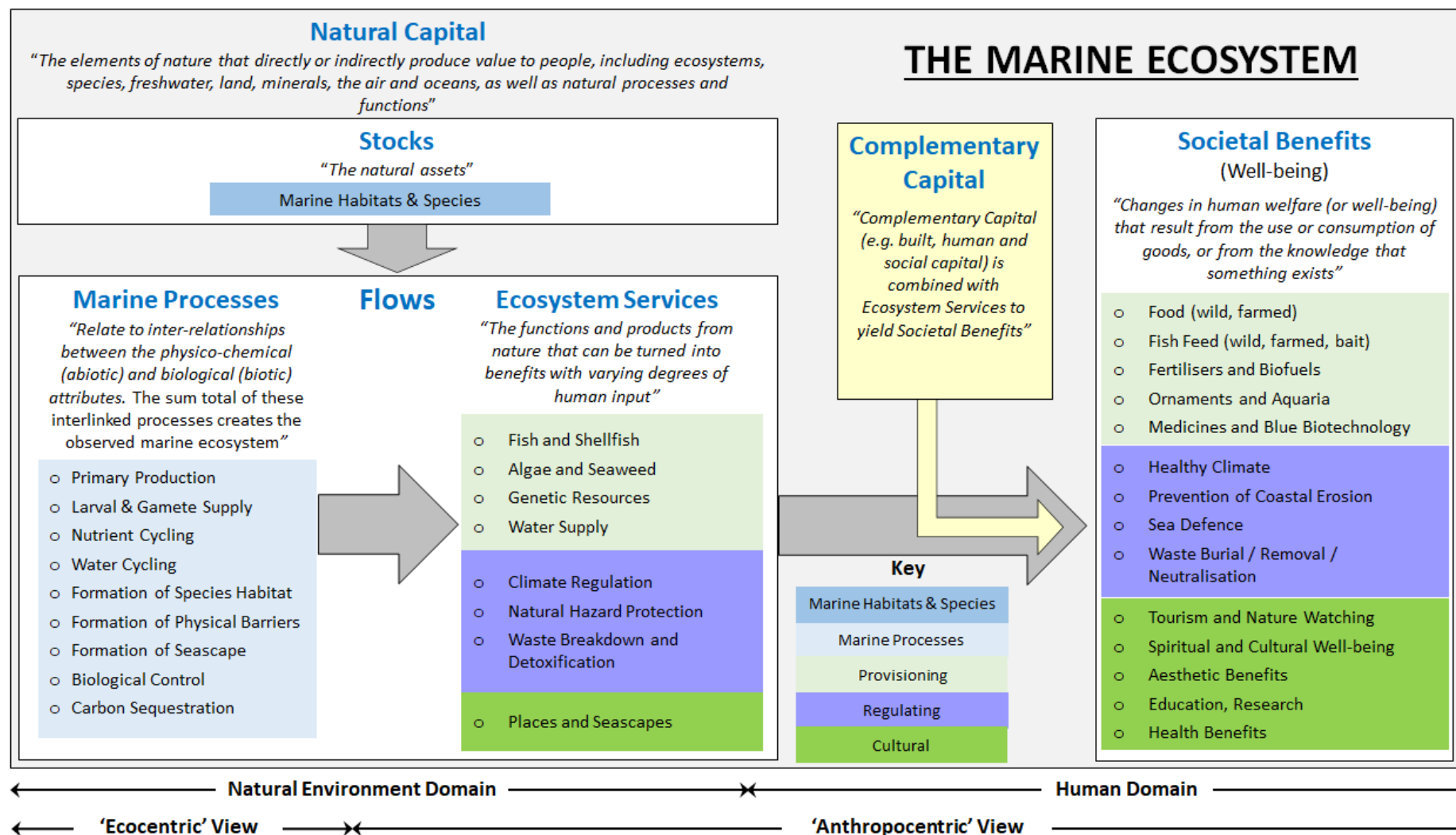


Figure 3.1: Conceptual framework linking natural capital, ecosystem services and societal benefits (adapted from Turner et al., 2015; Burdon et al., 2022; Burdon et al., 2024).

This vision of nature as an asset (or stock) gives stewardship to humans to sustainably manage the natural capital, which provides society with fundamental natural resources of value to people and their lives. In its Review of The Economics of Biodiversity, Dasgupta (2021) uses the term Natural Capital interchangeably with nature, natural environment, natural world and biosphere – it all depends on the lenses of the analysis that are utilised. The economic reason behind the use of ‘Natural Capital’ relates to the asset management of natural resources, as if it was a portfolio to manage, supported by biodiversity which helps reduce portfolio uncertainty (Dasgupta, 2021). In MARBEFES, such as in the Dasgupta Review, the focus is on the living and renewable (self-regenerative) part of the Natural Capital assets only. The natural capital approach to policy and decision-making considers the value of the natural environment for people and the economy, providing a helpful tool to support the protection and management of the natural environment and the engagement of stakeholders within management decisions (Barnard & Atkins, 2022; Burdon et al., 2022). The natural capital approach, as illustrated in Figure 3.1, takes an anthropocentric perspective, if there were not humans benefitting from the ecosystem services that nature provides, there will not be any societal benefits to value. For example, tourism was not in the list of benefits until the last 200 hundred years when society started going on holidays. Similarly, if people were all vegetarians, fish to eat would not be a societal benefit. Following Mace et al. (2015), natural capital assets can be defined in terms of their biophysical features, the types of benefits provided and the management context. In the case of the marine environment, three types of natural capital asset can be identified: habitat assets (e.g., EUNIS level 3 habitats), species assets (e.g., commercial fish and shellfish species) and the water column (Rees et al., 2022). The focus of this document is on habitat assets given our knowledge of these assets from survey work and 2D modelling across European seas and natural capital accounting, thus far, has focussed on these assets alone. The incorporation of water column assets into natural capital is yet to be developed and would require improvements in our capability of 3D modelling our seas and the delivery of ecosystem services and societal benefits from these assets.

The System of Environmental-Economic Accounting Ecosystem Accounting (SEEA-EA) is the globally adopted statistical standard on ecosystem accounts that the *EU-wide methodology* (Vallecillo et al., 2022) used to map, assess and achieve good condition of ecosystems. The reference framework is useful to assess the condition of EU ecosystems, so they continue delivering societal benefits, as required by the EU Biodiversity Strategy for 2030. *SEEA-EA is a spatially based, integrated statistical framework for organizing biophysical information about ecosystems, measuring ecosystem services, tracking changes in ecosystem extent and condition, valuing ecosystem services and assets and linking this information to measures of economic and human activity* (United Nations et al., 2021). In the EU, Eurostat is bringing forward the necessary amendments in the Regulation (EU) No 691/2011 on European environmental economic accounts for inclusion of a new module on ecosystem accounts based on the SEEA-EA (Vallecillo et al., 2022). In the UK, a white paper by the Department for Environment, Food and Rural Affairs (Defra) entitled ‘The Natural Choice: securing the value of nature’ (HM Government, 2011), enshrined the commitment of the UK to put natural capital at the heart of government accounting. Since 2012, the Office for National Statistics (ONS) and Defra have collaborated on the development of natural capital accounts which are in line with the SEEA EA framework (ONS, 2022).

The following sub-sections aim to guide the BBTs towards the preliminary steps needed to fill in natural capital accounts in a simple but complete way that is easily understood and applied by non-experts in situations of limited data availability. Therefore, the aim is not to alter the SEEA-EA in any way, or modify the accounts, but rather to provide different step-by-step strategies to collect data and analyse what happens to the natural capital, either at the national or spatially disaggregated level, in each BBT; which is typical for the *thematic accounts*, such as the Ocean or Marine Natural Capital Accounts (United Nations, 2021), which are still in an experimental phase. This guidance should provide the BBTs with a transparent framework to enable them to gather the relevant data and

complete natural capital accounts for their BBT. Where required, additional guidance and support was provided by the authors throughout the MARBEFES project.

3.2 SNA and SEEA EA Accounting approaches

To understand the reason for using the Natural Capital Accounting approach following the SEEA-EA guidelines (United Nations, 2021), we need to take a step back and explain where the System of National Accounting (SNA) comes from and why need to extend it to include the environment and its natural capital assets.

After the 1929 economic crisis and World War II, the United States government aimed to foster economic recovery and looked for a useful tool to prevent, monitor and react to similar events. By the end of 1930s, the SNA was developed to provide an overview of the main dimensions of the economic system such as production, consumption and savings. The Gross Domestic Product (GDP) was - and still is - the core indicator of this methodology. It is used to indicate and to monitor how the economy of a country is changing over time; it quantifies the economic activities (production, income/expenditure) of countries over the period of a given past year including the sum of national factors (expenditures, exports, imports, investments). In other words, the GDP is used as a measure of economic performance.

The SNA relies on input-output tables (Figure 3.2) that summarize the economic relationships in an economic system meaning i.e., the way firms trade with each other and produce items for consumption and investment. Looking at the so-called 'Blue Economy' in the coastal/marine environment, we can calculate, for example, how much the aquaculture sector buys from the fishery sector, how many people are employed in each of these activities and the fish consumption. From this data we can then estimate the monetary value of this activity. Figure 3.2 shows a simple example of an input-output table consistent with SNA rules referring to the marine environment and some of the related economic sectors. In this example table a very simplified version of the national economy is shown. Firms are aggregated into only two types of industry: fishing (sector A) and aquaculture (sector B). What are the inputs that each sector requires? Fishing activity relies mainly on the input from the wild fish stock (natural capital) and primary input such as labour (e.g., numbers of fishermen) and capital (e.g., vessels); the aquaculture activity needs intermediate inputs from the fishing industry (e.g., feed to cultivate fish) as well as labour and capital such as employees and infrastructure. All the information collated in the SNA is in monetary values.

		OUTPUT STRUCTURE							
INPUT STRUCTURE		Intermediate use/output		Final use/output	Export			DEMAND	
		Sector A (Fisheries)	Sector B (Acquaculture)	Household consumption	Household(e)	Sector A(e)	Sector B(e)		
	Intermediate input	Sector A (Fisheries)	0	6	10	0	0	0	16
		Sector B (Acquaculture)	0	0	20	0	0	0	20
		Sector A(i)	0	0	0				0
Primary input		Sector B(i)	0	0	0				0
		Labour	10	10					20
		Capital	6	4					10
OUTPUT		16	20	30	0	0	0		

Figure 3.2: A simplified example of input-output table consistent with SNA.

Analysing more in detail the input-output Table in Figure 3.2, in the third column, which refers to the intermediate use by the industry or output, the orange area shows the relationship between the two sectors and the red circle highlights the contribution of the fisheries sector (6 million euros in the example) to aquaculture. The blue circle highlights the household fish consumptions by sectors and in the green area we see the primary inputs (labour and capital) used by each industry. In this example, international trade imports and exports are not considered for simplicity. For a more technical explanation see Annex 3.

An SNA input-output table, such as in the example in Figure 3.2, does not provide any information about the natural capital in the form of ecosystems and ecosystem services that underpin the economic activity of a specific country; for example, how marine ecosystems (e.g. seagrass habitat) support wild fish stock provision and the benefits that the economy gets for, in this case the fish food production. Following this reasoning, the fish stock can be considered as an input to the fisheries sector that uses also vessels, human labour and many other inputs. To estimate the value of natural capital for the fishery sector, we must separate the contribution of each component and thus isolate the value generated by the natural environment.

To understand the contribution of nature, the natural capital, to the economic activities taking place within a country, the SNA has been extended to include natural capital and ecosystem services (ecosystem) accounting. The international standard used for ecosystem accounting can be found in the System of Environmental-Economic Accounting (SEEA) Central Framework (CF) and Ecosystem Accounting (EA) (United Nations et al., 2021). These two documents, SEEA-CF and SEEA-EA provide guidance on environmental-economic accounting which is made of a set of tables that make information comparable across space and time. The SEEA-EA, is the framework proposed for helping the development of Natural Capital Accounts; it provides a guide on how collect physical and monetary data about ecosystems, their health and their contribution to human well-being and economy in a standardised way. Figure 3.3 graphically summarises the link between the SNA and SEEA-EA approach.

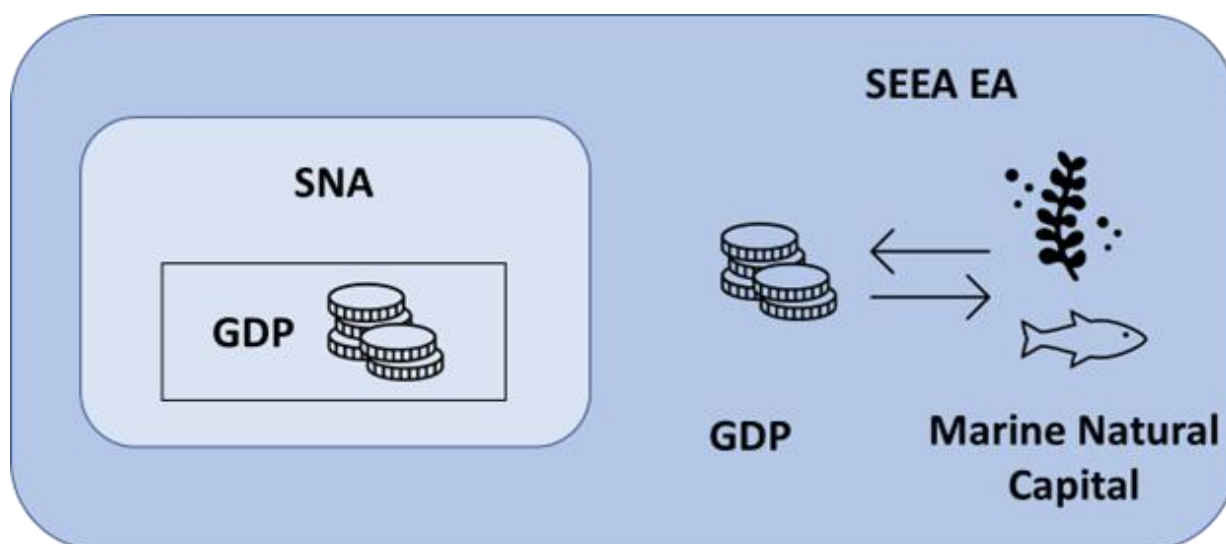


Figure 3.3: SNA and SEEA-EA approach.

To support the development of the SEEA-EA for the marine environment (see Section 3.3 for details), the first challenge is to define the spatial location and prevailing legislative power over the habitats

(e.g., the seagrass beds need to be within a country's territorial boundaries and therefore within its governance regime) and Physical Accounts (see Section 3.4). The second challenge is to value in monetary terms the contribution of relevant habitats and ecosystem services to the economy (see Section 3.6).

Both SNA and SEEA-EA mainly focus on the aggregate level (e.g., national) and require access to official statistics data and potential surveys; however, it can also be useful to work on a smaller scale as the local and regional level, if information is available, to make comparison between them and understand and explore potential benefits and limits of each scale.

3.3 Marine Natural Capital Accounts

All EU Member States and the UK have now produced Natural Capital Accounts (NCAs). Some countries in Europe, such as the UK and the Netherlands, have also produced marine NCAs. This section explores the use of marine NCAs beyond the national scale (see Section 3.2 for a summary on national NCA). The focus here will be on the spatially disaggregated scale, which can be local or even smaller (downscaling), such as a Marine Protected Area, or wider, such as the regional sea scale (upscaling). MARBEFES guidelines will provide support to implement marine NCAs, following the SEEA-EA framework, at the downscale within a context that might see the mandatory provision of NCAs by European member states, if the new proposal for a specific new EU Regulation will go ahead. As reported by the SEEA FAQ website (SEEA, 2023), Natural Capital Accounting is the umbrella term for the use of accounting to value (measure and report on) the natural capital, whereas ecosystem accounting (EA) is the coherent framework of accounting to be used (SEEA EA, 2021a). Therefore, ecosystem, or natural capital, accounting *brings together economic and environmental information to measure the contribution of the environment to the economy, and the impact of economic activities on the environment* (Luisetti & Schratzberger, 2022). The System of Environmental-Economic Accounting—Ecosystem Accounting (SEEA-EA) *is a spatially-based, integrated statistical framework for organizing biophysical information about ecosystems, measuring ecosystem services, tracking changes in ecosystem extent and condition, valuing ecosystem services and assets and linking this information to measures of economic and human activity* (UN, 2021).

MARBEFES has focussed on potential applications for the further development of the Ocean Accounts (OA) framework presented in the SEEA-EA (UN, 2021) and in Loureiro et al. (2023) (see Section 3.4). GOAP defines Ocean Accounts as *integrated records of regularly compiled and comparable data concerning ocean environment assets (e.g., extent/condition of mangroves), economic activity (e.g., sale of fish) and social conditions (e.g., coastal employment)* (GOAP, 2022). These are somewhat wider in their remit than the standard Ecosystem Accounts, as they include social conditions. As highlighted in Cummins et al. (2023), NCA frameworks have been progressing but their application to the Ocean is still evolving. For example, there is no statistical standard for OA yet, although these should still follow the SEEA statistical framework (central framework and ecosystem accounting).

3.3.1 Policy and Decision Making with Marine Natural Capital Accounting

The importance of marine Natural Capital Accounting (NCA) to support policy and decision making, especially in support to national monitoring, is clearly stated in the SEEA-EA (UN, 2021a). Examples of international policy documents where the use of NCA is welcome include the *Post-2020 Global Biodiversity Framework* and the *Sustainable Development Goals* (SDGs), the Paris Agreement on climate change (and its successive CoP meetings) and the Convention to Combat Desertification (UNCCD). NCA can also inform Multilateral Environmental Agreements. The four thematic accounts in the SEEA-EA cover four policy themes: biodiversity, climate change, oceans and urban areas. NCA can be used to mainstream the topics of the thematic accounts, such as biodiversity informing the *Post-2020 Global Biodiversity Framework*. NCA could also be used for policy scenario analysis to support policymaking (see *Policy Scenario Analysis using SEEA Ecosystem Accounting* (UN, 2021b). Ecosystem Accounting provides the statistical frameworks necessary to inform complex decision making

regarding the use of the natural resources with new data and indicators consistently and coherently organised. NCA indicators can be communicated using the DAPSI(W)R(M) framework, providing information on the S and the I of the framework, but other indicators could also possibly inform the D and P of the DAPSI(W)R(M) (UN, 2021). In the SEEA research agenda, the oceans are one of the ecosystem types for which specific methods of accounting needs further development – this is the aim of these MARBEFES guidelines. Further development of appropriate accounting data and indicators will also be able to inform evaluation of policy and projects. The PEAf framework explored the possibility of using natural capital accounting to monitor and evaluate projects at the intersection between environment and poverty.

To be of use for policy analysis, the accounting data need to provide long-run trends to verify changes. The OECD report Biodiversity, Natural Capital and the Economy: A Policy Guide for Finance, Economic and Environment Ministers (OECD, 2021) recommends the use of NCA following the SEEA standard to measure and mainstream biodiversity and natural capital more widely, so to increase the supply and demand of NCA at all scales. Increase in demand will support the necessary sampling to collect data compatible with NCA.

3.3.2 National, Regional and Spatially Disaggregated Marine Natural Capital Accounts

The issue of spatial scaling when dealing with ecosystem services is particularly relevant as ecosystem services can be provided from one area but benefit a larger area than they are located in and this can happen in different ways. Fisher et al. (2009) describe three options:

1. The ecosystem service provision and the benefit happen at the same location (e.g., provision of raw materials);
2. The ecosystem service is provided in all direction benefitting the surrounding area (e.g., pollination) and beyond (e.g., carbon sequestration and storage; migratory fish species);
3. The ecosystem service is provided in a specific direction only (e.g., from uphill to downstream such as in water regulation or in the case of a coastal wetlands that provides coastal protection from flooding).

Knowing where the ecosystem service is provided and which areas it is providing a benefit to is important for management decisions on welfare based also on their users and beneficiaries. Therefore, it is crucial that information about the extent and condition of the ecosystems and the ecosystem services an area provides are collected and coherently organised in the physical natural capital accounts tables (see Section 3.4). Natural capital accounting might be needed at different spatial scales, for example at a Regional Sea scale to a local MPA scale. We describe this as going from the aggregated to the disaggregated scale. Spatial disaggregation is also known as downscaling and it is ‘the process by which information at a coarse spatial scale is translated to finer scales while maintaining consistency with the original data set’ (Vincent et al., 2018). In NCA this is not always possible. So, we refer to spatially disaggregated (sub-national) accounts also when the scale considered is smaller, but the data (information on extent and condition) are specifically collected for that smaller scale area. This is the case in the UK where marine natural capital accounts exist, but the information at the country scale (England, Scotland and Wales) is based on different data than a downscale of the national (UK) accounts. As the Office of National Statistics (ONS) report in their *Principles of UK Natural Capital Accounting* (ONS, 2023) ideally Principle D3, by which there is a consistent methodology between national scale accounts and spatially disaggregated accounts (i.e., the same methods are used), is followed. In practice, it may not be possible to scale-down or scale-up (from sub-national to national accounts) because, as in the case of the UK, for example, the agricultural structure in Wales is very different than in England. In such cases, the best alternative is using the best available data and methods at the scale of interest (ONS, 2023).

Statistical data are collected at the administrative unit level, whereas for ecosystem accounting grid-based data are needed, which highlights the need for spatially disaggregated data. This is one of the

issues identified by the SEEA-EA revision team in 2020 ‘Outstanding issues in accounting for ecosystem services’ and that MARBEFES will address.

The SEEA-EA revision team also suggested that it is up to the accounts compiler to understand the need for either spatial disaggregation or biophysical modelling. For example, in the *Initial Natural Capital Accounts for the UK Marine and Coastal Environment*, it was highlighted that for blue carbon some limitations exist in sub dividing the area using the EUNIS habitat classification system because EUNIS boundaries do not match the carbon processing (Grilli et al., 2021). Thornton et al. (2019) report as an example the separation between mud and sand where the transition between permeability regimes and oxygenation of the sediments as a result is key for carbon processing.

The MARBEFES guidelines and applications, support answering two fundamental questions put forward by the SEEA-EA revision team (2020) which are still outstanding despite the publication of the SEEA-EA (UN, 2021):

1. What are some examples (i.e., countries and ecosystem services) of where disaggregation has been useful for ecosystem accounting?
2. How can we better guide compilers on how best to use spatial disaggregation, based on past successful examples?

Cummins et al. (2023) report that, to date, within the marine NCA / Ocean accounts written in the English language most of the testing has been done to a sub-national scale rather than at the national scale, although there are some exceptions (e.g., UK, Netherlands, Norway and Mauritius). Some pilots have been implemented in Canada, China, Malaysia, Samoa, Thailand and Vietnam, supported by the UN Economic and Social Commission for Asia Pacific (ESCAP) and statistics on the ocean economy are held in the United States, Portugal and China (Grilli et al., 2021). The limitations reported by Cummins et al. (2023) include: not reporting any condition scores, which makes the Oceans Accounting reporting incomplete, with the exception of the UK and two cases in Australia (Geopraphe Marine Park and the Great Barrier Reef); most of the studies not being publicly available preventing transparency and traceability of the methods used limiting their applicability to marine policy and ocean governance; the scale of the area considered for OA varied from the national scale to the local scale (e.g., a single bay); ecosystems easy to map and ecosystem services easy to value have been mostly considered (e.g., those for which data already exist); the valuation methodology for the ecosystem services varies, preventing any standardization and comparison; anthropogenic marine biomes (e.g., related to offshore infrastructure) were not reported in any of the case studies; measures of error or uncertainty are often missing, whereas confidence intervals around the estimates of extent and condition are important to assess the precision of the economic valuation assessments.

At the BBT level we are talking about spatially disaggregated needs, which can be further drilled down to a local or specific geographical scale (e.g., MPA). It is important to be aware that disaggregation may also happen for ecosystems, when the disaggregation is by ecosystem type, for ecosystem services, as it happens for example in the SEEA-CF on emissions flow and for stocks, such as the carbon stock that can be subdivided as the one in biomass and that in the ocean (UN, 2021).

Similarly, it is possible to compile marine natural capital accounts scaling up the area, as the Netherlands, with the support of OSPAR, did for the North-East Atlantic Area (Alarcon Blazquez, 2021). In this case, as for downscaling, it was not an aggregation of all the marine natural capital accounts for each EEZ in the North-East Atlantic Area but a specific assessment of the ecosystems and ecosystem services provided by the area.

3.3.3 Regional Sea Marine Natural Capital Accounts for migratory fish species and connectivity

National ecosystem accounting standards (the SEEA) are very recent and ecosystem Ocean Accounts are in development, but currently there is no regional sea ecosystem accounts standard. The only regional sea ecosystem accounts example we have is the one of the Northeast Atlantic (Alarcon

Blazquez et al., 2023). Inter country input-output tables (see Section 3.2) are relatively new too and have been prompted by different projects worldwide such as the World Input-Output Database project (WIOD) and the EXIOBASE project financed by the European Commission within the 7th framework programme, as they are emerging in different fields of economics (Eurostat, 2019), which may soon include ecosystem accounts.

One of the greatest challenges for any potential regional sea ecosystem accounting standard is understanding how to account for migratory species, such as fish, as they represent a shared stock in a shared space. In the OSPAR report FAO landings data have been used (Alarcon Blazquez et al., 2023), but inter-country input-output tables that account for the contribution of each country based on their quota and the international agreements in place would probably be more appropriate. The development of any national marine natural capital / ocean ecosystem account will have to take account of this necessity at the regional sea level. Thornton et al. (2019) highlight that a harmonized method within countries to account for fish in natural capital accounts is necessary to avoid double counting at the macro-level, which stresses the importance of inter-country input-output tables. In Thornton et al. (2019) the quantity landed by UK fleets both in the UK and in foreign ports was reported, as both contribute to the national economy, with evident chances of double counting at the macro-level. The national approach suggested by Grilli et al. (2021) is modelling the biophysical processes involved in the production of fish stocks (i.e., the life cycle of fish) whilst considering the quotas assigned to each country within the governance of the regional sea the country belongs to, together with the conventional economic elements of a resource rent approach (see Section 3.5).

3.3.4 Marine Natural Capital Accounts challenges, limitations and opportunities to support policy decision making

The Ocean Accounting (OA) framework, which is used for marine natural capital accounting, is based on established (i.e., SEEA-EA) and novel accounting standards (United Nations et al., 2021; Loureiro et al., 2023). As with the other natural capital accounts, OA can support monitoring an evaluation of policy programs and related targets. However, OA also integrate the support to adaptive governance. A graphical representation of OA can be found in Figure 3.4.

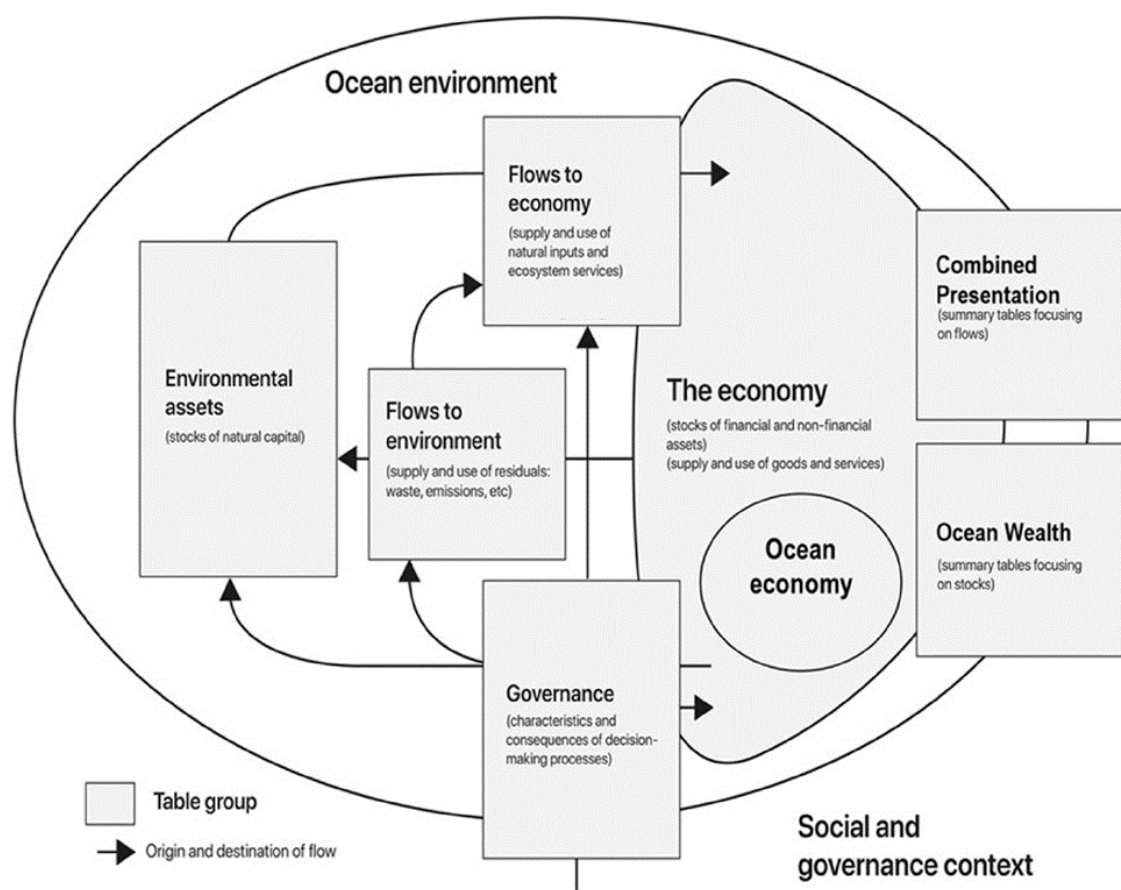


Figure 3.4: The general structure of the Ocean Accounts Framework. Source: GOAP Technical Guidance on Ocean Accounting, Version 1.0 (GOAP, 2021).

As illustrated in Loureiro et al. (2023) OA are informed by standard statistical accounts (SNA and SEEA CF and EA), but they are going to be complemented by novel ones which include: social accounts, risk accounts, governance accounts and summary accounts (see Figure 3.5). The model presented in Figure 3.5 is called 'The Stocks and Flows Model' (SaFM) and its novel accounts are being tested worldwide, but especially in the Ocean-Pacific Region, representing the still experimental components of the OA framework. SaFM follow a similar conceptual idea as the Complementary Accounting Network (CAN) that MARBEFES will be testing (see Section 5.1). The difference between the two lies in the fact that SaFM uses 'satellite accounts' where CAN uses complementary accounts. The Ocean Economy Satellite Accounts (OESA) are a set of accounts based on SNA information which are usually not as detailed in the SNA accounts (e.g. traditional fishing and fish processing) implemented by some countries, such as Portugal, the USA, Canada, Fiji and Indonesia (Loureiro et al., 2023). CAN is more interested in providing the wider set of information as possible to the policy decision maker in a succinct way (i.e. through accounts), which are called a dashboard of indicators in the CAN framework (see Section 5.1).

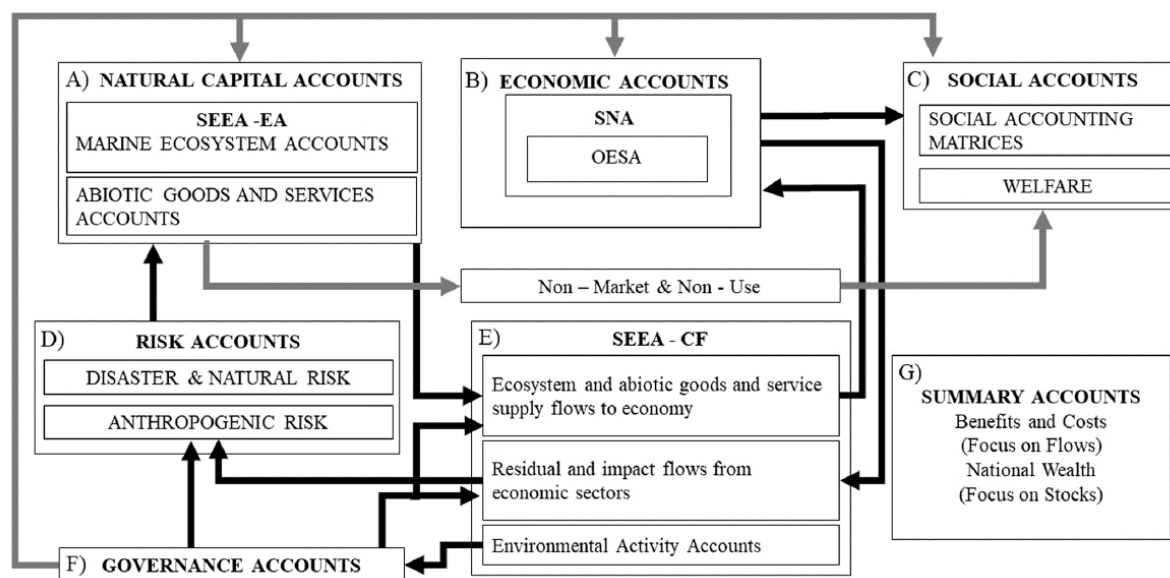


Figure 3.5: The Stock and Flows Model (SaFM) as reported in Loureiro et al. (2023).

The SaFM Model and the CAN framework both stem from the need to go *Beyond GDP* (Stiglitz et al, 2018), meaning that SNA and SEEA indicators may not be enough to explain the wider value of Nature. The CAN framework goes beyond the use of statistical indicators only, which is on what the SaFM model is still based on, to include shadow pricing (the actual economic value of nature as expressed, for example, by a Willingness-To-Pay measure – see Section 3.6) and the value of the less tangible cultural benefits (see Section 4, Guide 3). Also, as with the SEEA-EA, CAN includes ecological indicators, but it goes beyond that to also accommodate information on the inherent value of Nature based on an ecological value assessment (see Section 2, Guide 1). Figure 3.6 shows how all these components are integrated in the CAN framework.

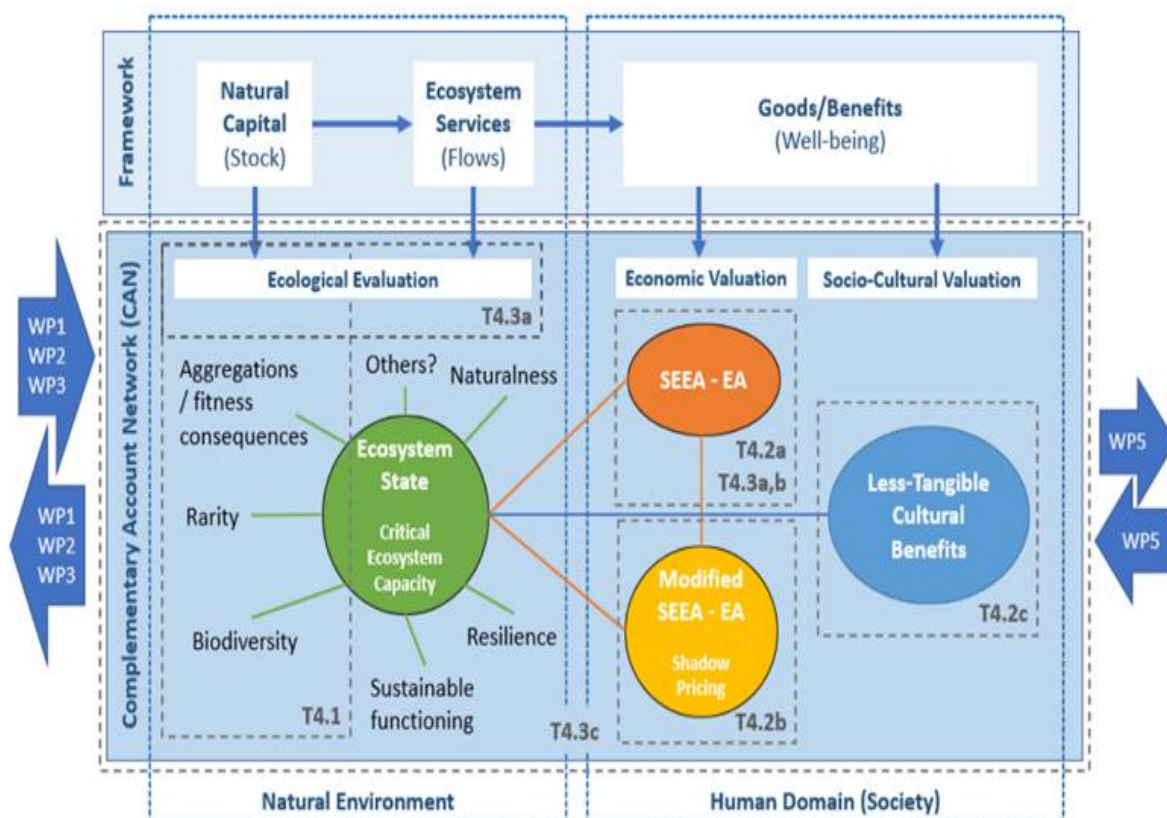


Figure 3.6: MARBEFES representation of the Complementary Accounting Network (CAN) (the side arrows represent information flows to and from the other Work Packages).

The SaFM model and the CAN framework are bringing us closer to a complete set of wealth accounts as envisaged by the Commission report on ‘The Measurement of Performance and Social Progress’ (Stiglitz et al., 2008); which highlights the opportunities this new way of thinking in natural capital accounts could bring to society.

According to Grilli et al. (2021), based on the UK experience of developing marine natural capital accounts, a series of more technical challenges currently faced by marine NCA include:

- The realisation that the mapping of the extent of the marine environment is challenging and far from complete; this limits the potential protection of the marine environment as well as its sustainable development and use of marine space. The need of further granular spatial disaggregation at the local and/or specific ecosystem scale;
- The development of ecosystem condition accounts is still limited mostly due to data limitations and the need to select the key indicators for ecosystem condition together with the reference baseline (see also Vallecillo et al., 2022);
- The economic valuation of provisioning, regulating and cultural services should be improved often with ad hoc data collected for natural capital accounting purposes, such as specific costs involved in the fishing industry, as well as accounting for discards and recreational fishing;
- Limitations related to the accounting and valuation of regulating services are linked to the limited development of ecosystem condition tables and to the different boundaries of the EUNIS habitat mapping compared to the boundaries involved in the processing of some ecosystem services (such as carbon sequestration);

- The accounting and valuation of cultural services is likely to be underestimated due to the restrictive rules around their valuation in accounting and the lack of specific data.

The aim of these MARBEFES guidelines and the use of a CAN framework is to overcome the listed limitations and make the most of the opportunities the information in the accounts could bring, from monitoring to evaluation (pre- and post-implementation) of sustainable development policies.

3.4 Guidance for Marine Natural Capital Accounting in the BBTs

In the MARBEFES project, a seven-step process was developed to support the production of natural capital accounts within each of the BBTs. The seven-step process is illustrated in Figure 3.7 and is further described below. It is highlighted that of the three types of natural capital stock mentioned above, the process developed here only considers habitat assets (e.g., saltmarshes) and species (e.g., fish stocks). The spatial unit considered in the SEEA-EA it is called Ecosystem Accounting Area (EAA). Since an EAA limits can be as wide as a country, or as small as a Marine Protected Area (MPA), an EAA can include several habitat types although reported in a disaggregated manner (UN, 2021) – MARBEFES refers to this as the habitat mosaic.

There are two main sets of tables needed for developing Marine Natural Capital Accounts:

- **physical accounts** (a set of tables on the environmental characteristics of the ecosystems considered); and
- **monetary accounts** (a set of tables highlighting the role of natural resources on the economy of the EAA through economic activities).

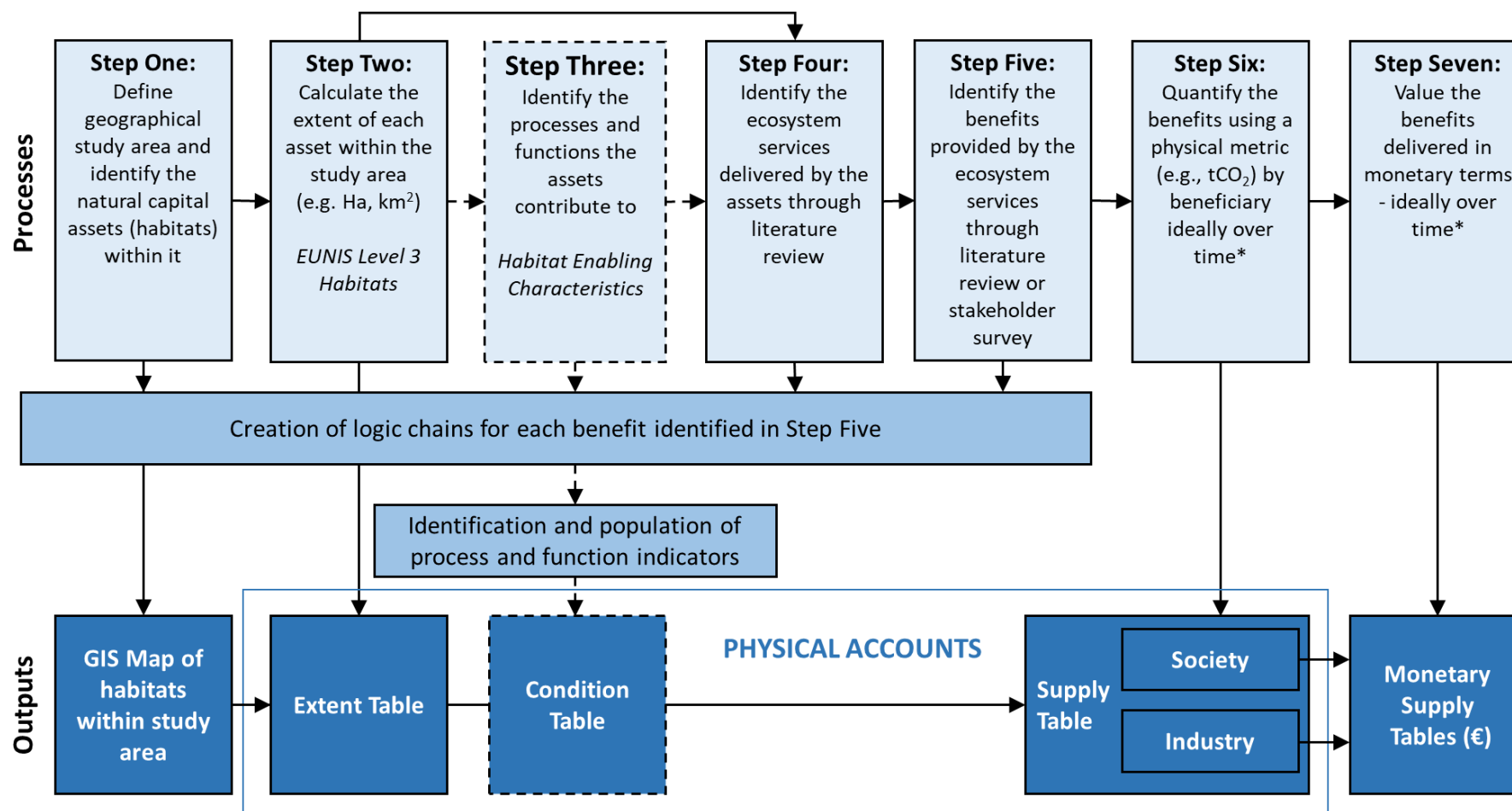
In this section, we focus on how to develop physical accounts (Steps 1-6 in Figure 3.7). Ecosystem assets, or ecosystem asset areas (EAAs), are defined in the SEEA-EA as *spatially contiguous areas that can be considered as internally homogeneous with respect to ecosystem type [defined as a distinct set of abiotic and biotic components and their interactions], condition and service flows* (Vallecillo et al., 2022). Section 3.4 in this document focuses on the development of the monetary accounts and its related (monetary) supply and use tables (Step 7 in Figure 3.7).

Physical accounts tables provide information, often presented separately, on:

- **Ecosystem Extent:** The spatial extent (e.g., km²; Ha) of the asset considered (Step 2 output in Figure 3.7).
- **Ecosystem Condition:** The condition of the asset considered (i.e., the ecosystem health) assessed through a set of appropriate biophysical indicators (Step 3 output in Figure 3.7). This step can be linked with outputs from the EVA approach (see Section 5.2.1).
- **Supply and Use (Physical):** The estimated flow of benefits that the asset(s) provide(s) in physical terms (e.g. tonnes of fish catch; tonnes of CO₂ stored etc) for specific beneficiaries (e.g. society, industry) (Step 6 output in Figure 3.7). This information can be collected in one supply and use table or in separate supply and use tables – see Annex 3.

The extent table, and preferably all tables in the physical accounts, report the habitat classification system used to assess the spatial extent and condition of the asset considered. For the purposes of the MARBEFES project, it has been agreed that the EUNIS classification system (minimum Level 3) will be applied across all work packages. Each BBT is required to define the geographical boundaries of their study location – this may be the boundary of the BBT, a smaller portion of the BBT, or the national boundary(s) within which the BBT is located. If the BBT includes more than one national area then the adjoining country participants will need to agree how to merge their outputs. Once the boundary is defined, GIS maps can be produced (output from MARBEFES WP2) which identify which EUNIS Level 3 habitats are present within the boundary (Step 1 in Figure 3.7). This step identifies the area of interest and, by using a standardised habitat classification system, allows for comparisons between different natural capital accounts over space (e.g., between habitats within a BBT, between BBTs or

between countries) and time (e.g. over different time periods). A detailed flow chart on how to achieve data collection to compile the physical accounts table is given in Figure 3.7.



* From accounting period to accounting period

Figure 3.7: Seven-step process for creating marine natural capital accounts within the BBTs. Light blue boxes (black text) represent the process; dark blue boxes (white text) reflect the outputs. Assessing condition is challenging – hence it is represented by dashed lines within the figure.

3.4.1 Logic Chain Development

To develop the physical attributes accounts, some preliminary work to identify the types of data which are needed is required to structure the links between natural capital assets, flows of ecosystem services and the delivery of benefits. Logic chains are a basic tool to help gather and present the fundamental biophysical and socio-economic information necessary to complete the marine natural capital accounts physical tables (extent, condition and supply) (Figure 3.8). Given the complexity of the system to be examined, logic chains are the outcome of a combined effort between natural scientists, economists and social scientists and their output is a visually simplified version of that complex system. Logic chains have been applied in many research studies (e.g., Burdon et al., 2022; Barnard et al., 2022) and specifically with respect to natural capital accounting as they can link the condition of the ecosystem and ecosystem services considered to their provision of societal benefits to humans and hence measure its economic value (e.g., Lusardi et al., 2018; Thornton et al., 2019).

Logic chains are generally read from left to right, however the focus of the logic chains developed in MARBEFES (Figure 3.8) is on the Societal Benefits in each BBT and therefore, following the approach adapted by Thornton et al. (2019), it is recommended that they are developed working from right (Societal Benefits) to left (BBT and EUNIS Habitats). An example of how the MARBEFES logic chain template can be populated can be found in Annex 4.

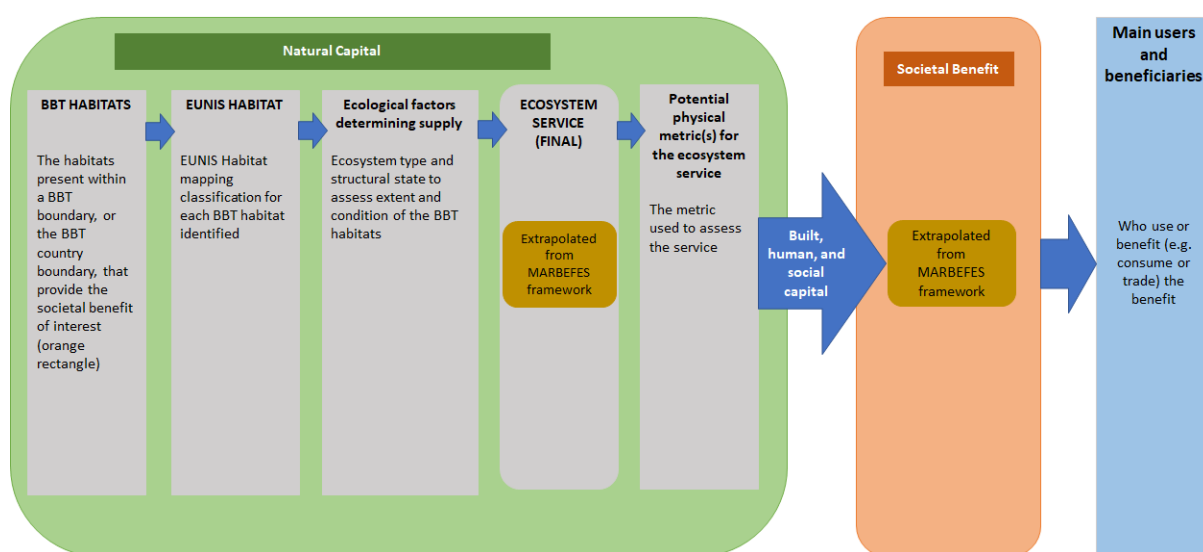


Figure 3.8: MARBEFES logic chain template. Modified from Thornton et al. (2019) and United Nations (2021).

Five of the logic chains developed within the *Initial natural capital accounts for the UK marine and coastal environment report* (Thornton et al., 2019) were discussed at a MARBEFES logic chain workshop¹. As it was custom for early natural capital accounts and it is now a standard in the SEEA-EA (United Nation, 2021), logic chains were developed to clarify data availability and data needs to populate the natural capital accounting tables. An early example in Europe is the Natural Capital Accounting in the Netherlands report (Statistics Netherlands and WUR, 2021). The selected five logic

¹ MARBEFES Logic Chain workshop, University of East Anglia (UK), 2-3 February 2023.

chains were based on five benefits that were selected from the 14 societal benefits identified in the ecosystem services framework in Figure 3.1 by deciding the relevance of each benefit for the UK and the availability of data, including the feasibility to collect new data:

1. wild food (commercial fish species) (ES: (all) wild fish).
2. healthy climate (ES: carbon sequestration and storage).
3. coastal and marine recreation and nature watching² (ES: places and seascapes).
4. erosion and coastal flood prevention (ES: natural hazard protection).
5. clean water (ES: waste (nutrient) remediation).

However, other societal benefits listed in Figure 3.1, could also be considered depending on the specific interests of each BBT, including less-tangible benefits, such as aesthetic benefits and spiritual and cultural well-being (ES: Seascape), or the harvest of seaweed for the production of cosmetics (ES: algae and seaweed).

The initial information gathered through the logic chains will be used to draft the extent and the supply tables, which are mandatory for natural capital accounting. Accounting for ecosystem condition is crucial for ecosystem accounting as, for example, it can identify ecosystems that need protection or restoration. However, assessing condition is challenging – hence it is represented by dashed lines within Figure 3.7. The condition table is required by the decision/policy-maker when deciding on management interventions, but it is very demanding in terms of data requirements, therefore it may not always be possible to develop one for all ecosystem services (see examples of condition tables in Annex 3). New EU guidance (Vallecillo et al., 2022) presents an *EU-wide methodology to map, assess and achieve good condition of ecosystems*, which is fully aligned with the SEEA-EA framework³. Ecosystem condition is defined in the SEEA-EA as *the quality of an ecosystem measured in terms of its abiotic and biotic characteristics* (United Nations, 2021) and is usually assessed by measuring similarity (or distance) of a current ecosystem to a reference state (Vallecillo et al., 2022). Assessment for coastal and marine areas should be relatively straightforward as the data gathered for the ecosystem status for the Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD) are considered by the EU-wide methodology equivalent to the concept of ecosystem condition in the SEEA-EA. However, comparison between ecosystems can often only happen within a Member State as each Member State reports data that often are collected using different approaches, limiting comparison between Member States and integration of these data under EU legislation for ecosystem condition and also for policy applications. Other data coming from monitoring programmes (e.g., LUCAS and Copernicus) as well as geospatial data coming from research outputs could also be used to inform the natural capital accounts tables. Another challenge comes from defining reference condition (a condition against which past, present and future ecosystem condition is compared) and reference levels and thresholds of good ecological (WFD) or environmental (MSFD) condition. The EU-wide

² This could be broken down into specific categories: recreational fishing (ES: places and seascapes and fish); diving (ES: places and seascapes and fish); kayaking (ES: places and seascapes)

³ The adoption and transposition of EU laws, principles and practices into UK law has been carried out through the European (Withdrawal) Act 2018, followed by the creation of new domestic legislation such as the new landmark Environment Act 2021. The Policy paper "Biodiversity 2020: A strategy for England's wildlife and ecosystem services" (Defra, 2011) continues to provide a comprehensive picture of how the UK is implementing both international and EU commitments and drives the requirement for Natural Capital accounts.

methodology suggests different methodological options (i.e., 'optimal condition', sustainable thresholds' and 'contemporary condition') (see Vallecillo et al., 2022).

The figures reported in the physical accounts tables indicate the supply of the environmental resource at a specific time (e.g., year 2023). Therefore, for each year presented, they are a static picture of the state of the environment. However, the accounting period may be either every few years, or every year and natural capital accounts usually report the supply provided at each of several available years (for example, five years: 2019-2023). This helps visualise any change occurring between accounting periods and/or years. If the accounting period is 2019-2023, for example, but BBTs only have data for 2019 for some benefits, they will have to make assumptions for the ecosystem services flow in other years. Changes within a year can be captured comparing the opening stock (e.g., the extent of an asset at the beginning of the reporting period) and the closing stock (e.g., the extent of an asset at the end of the reporting period). Hence, to determine changes incurred over time, time series data are required. In general, natural capital accounts are retrospective in nature, so they can only assess what the trend has been. Projections of future trends will require environmental, ecological and economic modelling to provide insights into the sustainable use of the resource; this is not yet part of current natural capital accounts although some experiments exist (La Notte et al., 2021).

Annex 3 provides an example of physical accounts tables (extent, condition, supply and use). These show that SEEA-EA tables can be quite complex, although they indicate the way in which data can be reported and therefore can also be simplified depending on data availability (see data requirements in Figure 3.11 below). For example, ideally, extent account tables will have a record of the opening extent (e.g., the spatial extent of seagrasses at the beginning of the accounting period, which could be the start of the calendar year) and a record of the closing extent which takes into account any spatial extent modification that took place during the accounting period. However, data on the opening extent and any subsequent modification of the spatial extent is not always available. In that case, the account table will show only the data that are available highlighting the gaps that prevent the completion of full natural capital accounts. However, at a smaller scale, for example, these data are very valuable to monitor the progress of a restoration project. Linking physical accounts to measures of economic activity and beneficiaries is examined in Section 5.

3.4.2 Completion of Natural Capital Physical Accounts Tables

The flow diagram in Figure 3.11 shows the steps needed to gather the data to complete the physical accounts tables (extent, condition and supply) and the monetary accounts in Section 3.4 (examples of natural capital physical accounts tables can be found in Annex 3) and where this information can be found. In MARBEFES, accounting tables were developed and completed differently depending on data availability in the different BBTs.⁴ Some text explaining the process and methodologies followed, as well as the data used and why, usually accompanies the physical and monetary accounting tables. It is of note that the EU-wide methodology (Vallecillo et al., 2022) suggests the use of the same six classes of condition indicators (i.e., physical, chemical, structural compositional, functional and landscape/seascape) used in the SEEA-EA. In MARBEFES, information on the indicators to assess

⁴ There will be a minimum requirement for all BBTs but some may go further depending on data availability – skill/data rich areas vs. skills/data poor areas. The aim is to get all the BBTs doing at least some of the same things.

condition based on the WFD and the MSFD were found in the WP3 activities. However, physical condition variables (e.g., acidification, sea temperature rise) are currently not present in the MSFD GES indicators. Vallecillo et al. (2022) provides a list of variables for the condition assessment of marine ecosystems. Details on the economic valuation data needed to fill in the monetary tables and how to link physical ecosystem accounts to monetary accounts, is given in Figure 3.13 below.

Before turning to Figure 3.11, BBTs should determine whether their country is already undertaking natural capital accounts (see Figure 3.9 and Figure 3.10 below). If they are, it is highly likely that most of the biophysical and monetary data have been collected by their relevant National Statistical Office (GDP tables) and Ministry of the Environment (e.g., MSFD and WFD indicators and related data). Further details on how to link physical and monetary accounts can be found in Section 5.

According to the 2024 Global Assessment of Environmental-Economic Accounting and Supporting Statistics (United Nations, 2025), 94 countries around the world have been compiling SEEA accounts within the last five years. 67 countries are at stage III (regular compilation and dissemination)*, with all of them compiling the SEEA Central Framework (see Section 1) and 53 countries compiling the SEEA-EA (the focus of Section 2) (United Nations, 2025).

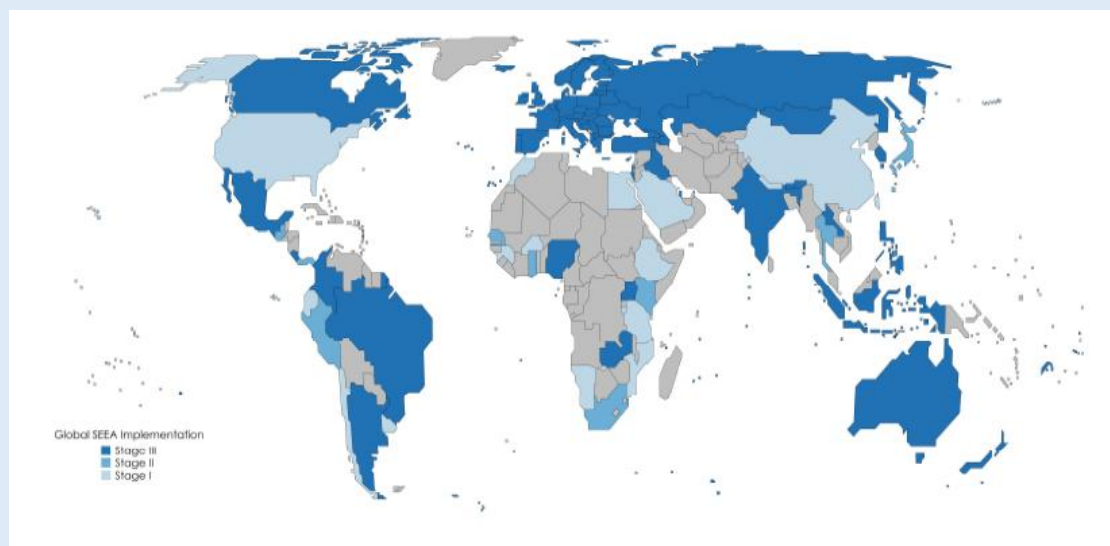


Figure 3.9: Stages of SEEA implementation worldwide (2024 Global Assessment of Environmental-Economic Accounting and Supporting Statistics (2025).

Only few countries worldwide are compiling the SEEA thematic accounts, such as the Ocean Accounts (i.e. marine natural capital accounts), which are currently compiled by 13 countries (14%) (United Nations, 2025).

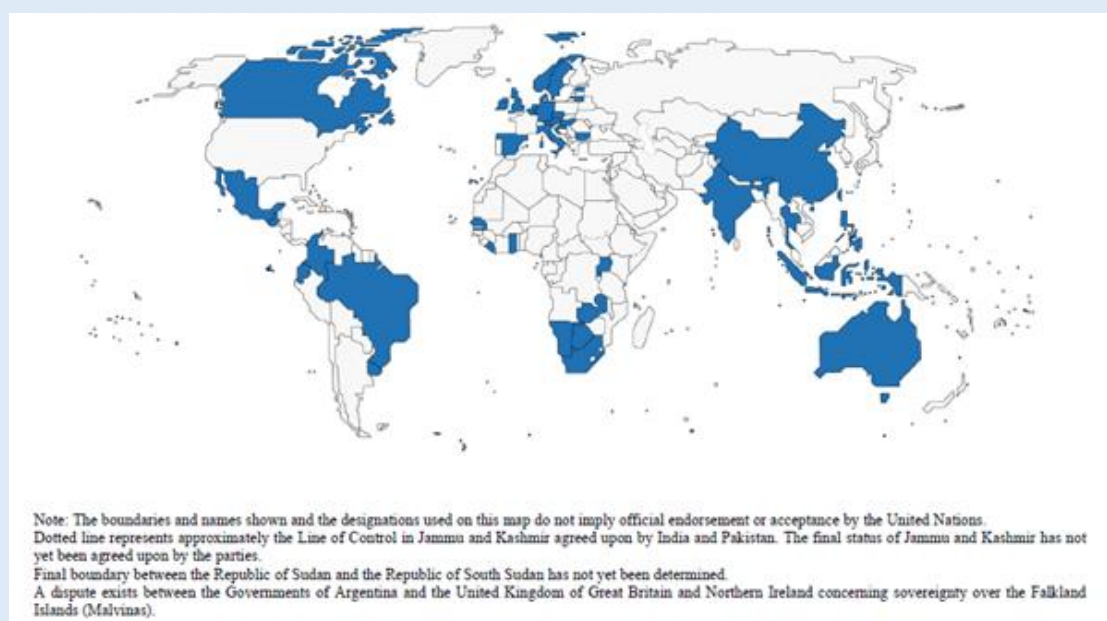


Figure 3.10: Countries implementing SEEA-EA accounts including thematic accounts (United Nations, 2023).

***A country falls into the regular compilation and dissemination stage if, at the time of the Assessment, it regularly publishes at least one account.**

The flow diagram (Figure 3.7) and the Logic Chains (Figure 3.8, Annex 4) are required by the BBTs as they indicate the biophysical data collection needed, when not already covered by existing data (e.g., MSFD and WFD, or other initiatives, data) to compile the natural capital accounts at the scale needed for their BBT, hence informing a key practical part of the MARBEFES project, i.e., the BBT fieldwork campaign. Fieldwork campaigns are a key action as they will determine the information available for the economic valuation reported in the natural capital accounts but also allow the development of other types of environmental valuation based on the scenarios developed by BBTs within WP5 (see Section 3.6). However, it is important to note that Natural Capital Accounts provide a static picture of the natural capital for the time considered. If the time is one year and data about the opening stock and the closing stock are collected (i.e., how much extent and ecosystem services flow is present at the beginning of the year and at the end of the year), it will be possible to assess if any change to the natural environment has occurred within that one-year period. If opening and closing stocks for one (or multiple consecutive) years are not available, then if data for previous time periods (e.g., previous years) are available then any change could be observed throughout the years. Changes in the natural environment are necessary for an economic welfare measure to be implemented (MARBEFES WP4, WP2 and WP5). Changes could be derived as described above, or, when available, even within the same period (e.g. time series data with a given year), depending on the benefit under assessment including less-tangible social and cultural benefits.

If the country in which a BBT or part of a BBT is situated is already compiling natural capital accounts, it is better for BBT leads to contact the relevant national authorities (usually the national office of statistics and/or their ministry of the environment). This is needed to verify what support MARBEFES could offer them or for which geographical area within the BBT it is relevant to collect data for natural capital accounts (e.g., an MPA). Natural capital accounts can be collected for an entire country or for spatially disaggregated investigations (e.g., an MPA); that primarily depends on the spatial and temporal availability of biophysical data related to the EUNIS habitat data beyond level 3.

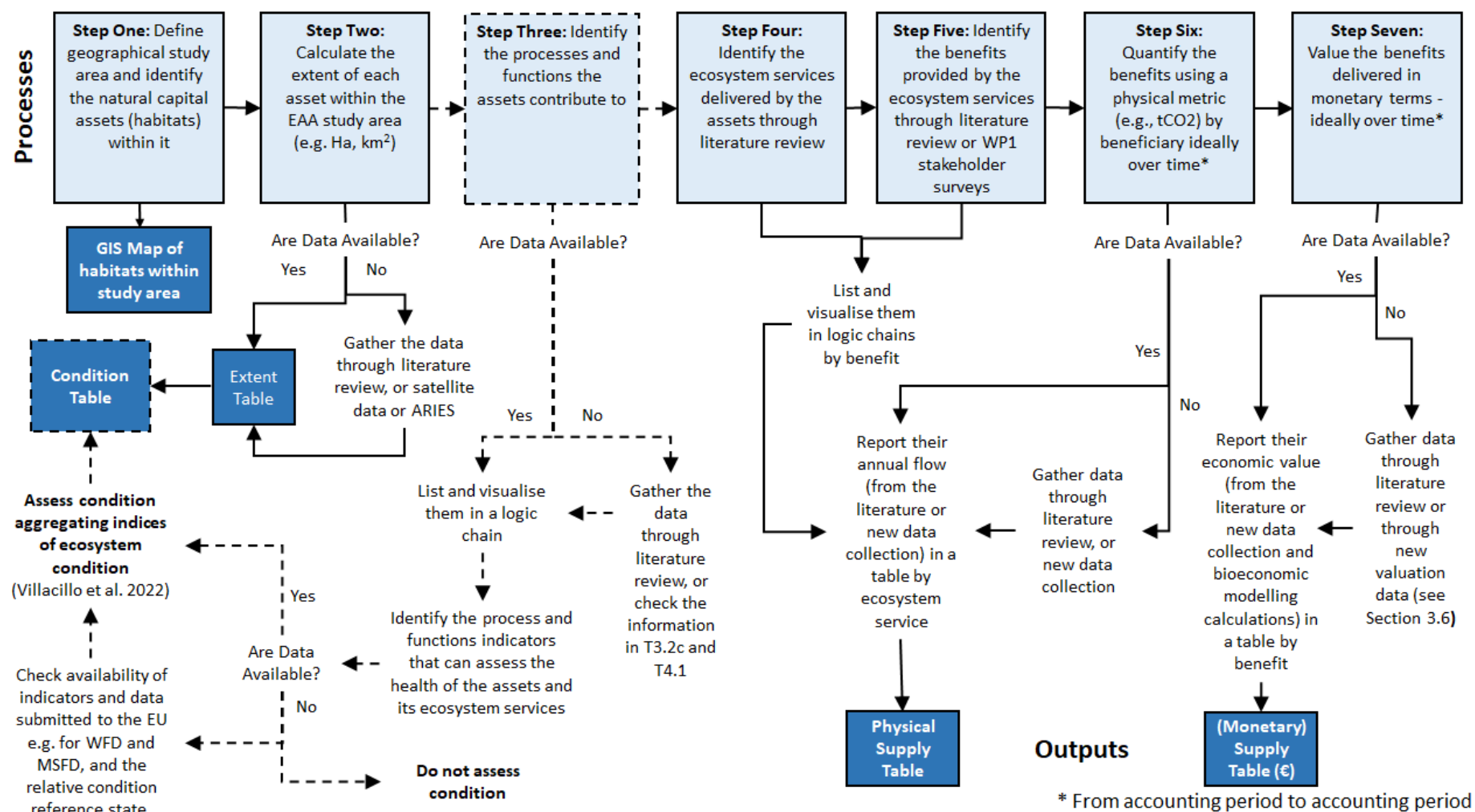


Figure 3.11: A MARBEFES step-by-step guide on how to fill in physical and monetary natural capital accounts tables in the BBTs.

3.5 Benefits of Completing Natural Capital Physical Accounts Tables

Natural capital accounts are developed to assess and monitor the contribution of natural resources to economic activity. So, they are most useful in a context of evaluation of a project or policy post-implementation (i.e. cost-effectiveness). Physical accounts tables provide basic information on the state of the environment (the stock and the flows of the natural capital, analogous to ecological structure and functioning) in a specific geographical area. When a condition table is also populated, this information can indicate at what level of the ecosystem an impact of economic activities is occurring. However, given the complexity of the marine environment and the fact that in the SEEA-EA only the societal benefits (SNA and non-SNA; see Section 1 and 3.4) for which a market value exists are reported, for policy and project decision making there might be the need to go beyond SEEA-EA accounts. In-depth impact of Drivers and Pressures can be better examined using the DAPSI(W)R(M) (Drivers - Activities – Pressures - State changes - Impact (on Welfare) – Responses (as Measures)) approach (see Section 1), which has been developed to assess the causes, consequences and responses to change in a holistic way (Elliott et al., 2017). The DAPSI(W)R(M) approach links the natural and societal domain and can be informed by the ecosystem extent and condition (as depicted within the SEEA-EA framework in Figure 3.12) being captured within the State change (of the natural environment) and their respective effect on the delivery of societal benefits reflected within the Impacts (on the environment and ultimately on human Well-being)⁵. If through the natural capital physical accounts table, a State change or Impact (on Welfare) is highlighted, a management or policy intervention, i.e., a Response (as management Measures) may be necessary. This equates to the Programmes-of-Measures in the WFD, MSFD, Natura 2000 Directives and the EU Nature Restoration Regulation.

Natural capital accounts tables and the DAPSI(W)R(M) framework can be further linked to the Bow-tie framework (WP5 guidance document on Bow-Tie analysis; Franco et al., 2026). Bow-Tie is an analytical approach for risk assessment and management, which can be adapted for opportunity assessment and management (further discussed and to be used by the BBTs in MARBEFES WP5; see also Cormier et al., 2019). The central knot of the Bow-Tie represents a particular risk (e.g., loss of biodiversity). The left-side of the Bow-Tie (Figure 3.12) lists pathways of potential causes of that event whereas the right-side lists consequences resulting from the event. Controls are positioned along the pathways of risks on the left (solutions to prevent the central event) and on the right (mitigation/compensation and recovery from the central event). Escalation factors which undermine or even enhance the effectiveness of a given control, can also be added with additional barriers. The Bow-Tie will therefore incorporate multiple causes and consequences of a given event, to analyse existing and possible controls that are used to prevent the causes of the event both individually and collectively and to mitigate and recover from consequences of the event (Figure 3.12, Cormier et al., 2019). In a standard Bow-tie framework, prevention controls would be implemented to reduce the likelihood of a State change outside tolerable levels (Cormier et al., 2018). Applying the Bow-tie structure with the DAPSI(W)R(M) framework can therefore be used to structure trade-offs and opportunity costs which may be investigated within scenarios and storylines, as it is planned in MARBEFES WP5.

An important limitation of natural capital accounts for policy decision making is that not all the societal benefits provided by coastal and marine ecosystem assets and costs of economic activities can be fully captured in natural capital accounts, given the need for economic valuation to be done within the exchange value methodologies (Section 3.1). For policy and (conservation or restoration) project decision making, instead it is important to have a wider understanding the socio-economic situation in a specific geographical context (Smith et al., 2025). Therefore, other economic valuation tools must be considered when the aim is to identify management and policy options. In MARBEFES, we have

⁵ In natural capital accounts, the only valuation method allowed is exchange values (i.e., market price), which means that in some cases this value will not coincide exactly with the welfare value for that benefit (See Section 4).

explored policy and management options using the Complementary Account Network (CAN) (Section 5), which can accommodate the assessment of welfare-values, using methods beyond exchange value (e.g., stated preference methods), for all the societal benefits of interest. The project will also consider indices for the less-tangible social and cultural benefits, allowing for monitoring overtime. This analysis of policy scenarios will therefore overlap with WP5 for scenario and storyline analysis and with WP6 for governance options.

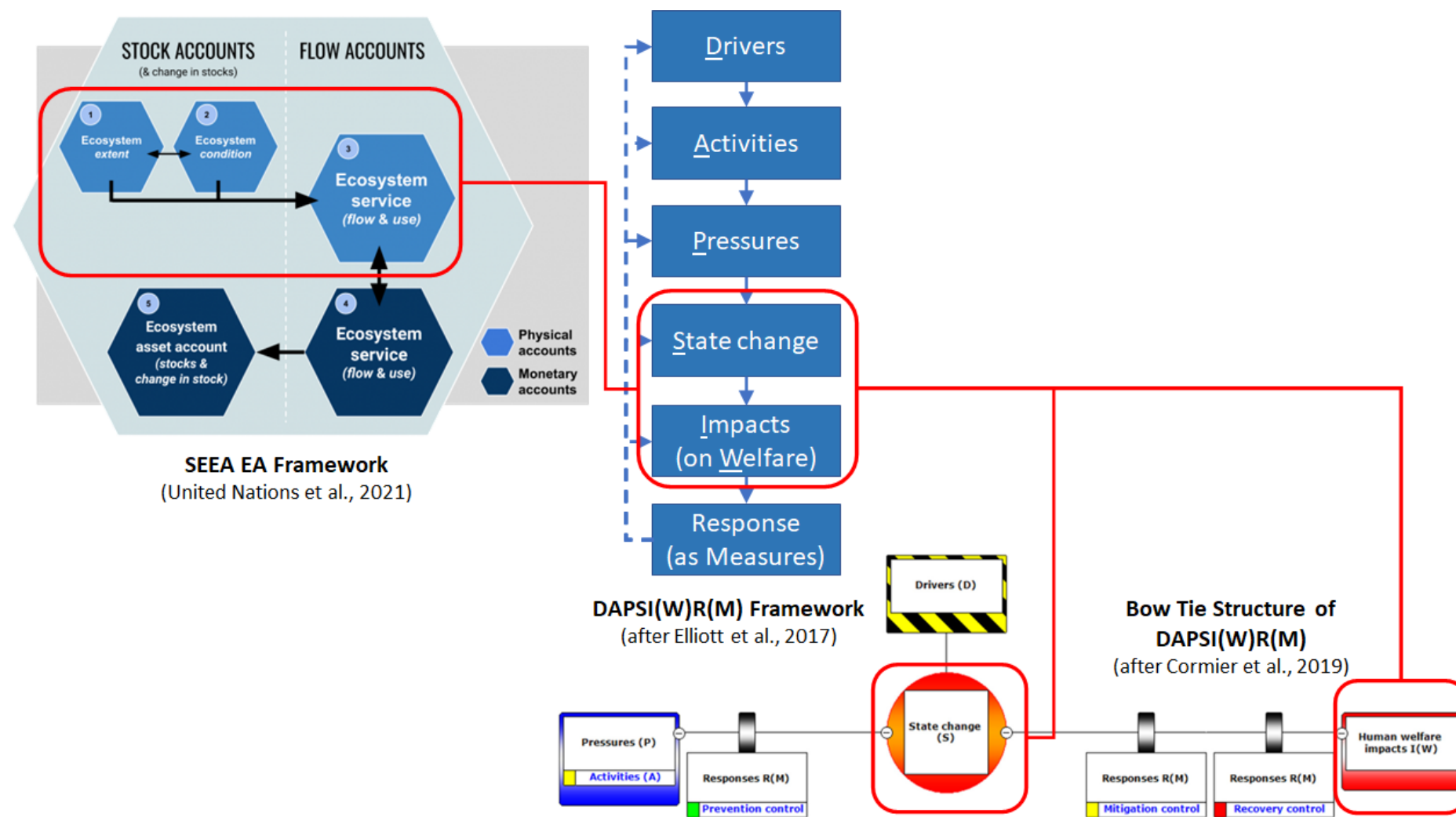


Figure 3.12: Linking the DAPSI(W)R(M), SEEA-EA and Bow-Tie approaches using the State change and Impacts (on Welfare) components.

3.6 Developing Monetary Natural Capital Accounts for MARBEFES BBTs

This section provides simplified guidelines on how to develop monetary natural capital accounts for areas such as the MARBEFES BBTs, based on the SEEA-EA (United Nation, 2021) guidelines. Using the information provided by natural capital physical accounts and logic chains (Sections 3.4 & 3.6), monetary accounts for the marine environment can be developed following the process in Figure 3.13.

The process in Figure 3.13 is only a simplified and practical way to understand the marine nature contribution to economic output and translate it into natural capital account supply and use tables (see Section 3.4) similar to those produced for the SNA. In general, this process aims to define the benefits, users and economic value of marine ecosystem services given economic methods and data available.

At the top of Figure 3.13, you can see the six-step process, with each step described below. The middle of the figure contains a few examples which are useful to build the supply and use tables of monetary accounts, while the bottom of the figure provides information required for the process obtained from physical accounts (e.g., supply and use tables, logic chains) and the main outcome of the monetary account (supply and use tables in monetary value).

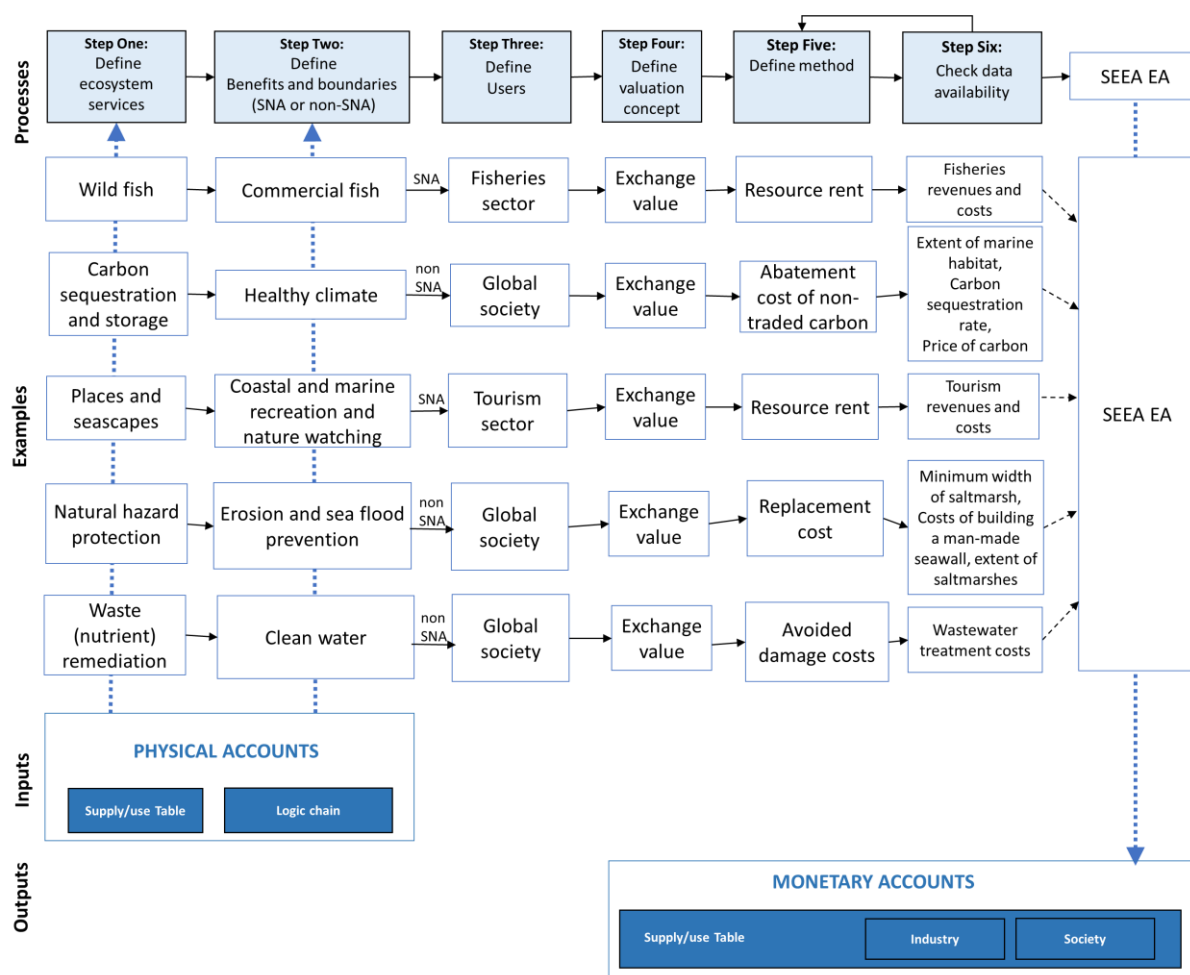


Figure 3.13: Six-step process for creating monetary natural capital accounts within the BBTs. Light blue boxes (black text) represent the process; white boxes (black text) are examples of marine ecosystem services; dark blue boxes (white text) reflect the information required in the process and the outputs.

STEP 1 Defining ecosystem services

A clear idea of which ecosystems are the object of the economic valuation is needed to truly reflect the value of nature. The aim of this step is the identification of the ecosystem services provided by the marine environment. In Section 3.2, ecosystem services may initially be identified within the framework in Figure 3.1, but more specifically through the logic chains that each BBT will come up with (see for example Annex 4). For the monetary accounts, it is useful to think of the ecosystem services (and their associated societal benefits) in relation to the users, also called economic units. Economic units refer to productive sectors, governments, households and global society. In the ecosystem services, the contribution of ecosystem to people is direct meaning that the relation goes from ecosystems to human well-being. In this relation the subjects are the ecosystem services that provide goods/services and the economic unit that obtain benefits from it (for example, recreation, food and carbon storage ecosystem services).

From an ecosystem accounting prospective and to fill the use table in monetary terms, the ecosystem services can be divided into:

- ecosystem services that do not need human input to provide societal benefits (e.g., carbon sequestration and storage) and for this reason can be classified as direct input to beneficiary well-being (usually the global society);
- ecosystem services that need human input to provide societal benefits (e.g., recreation, food) and for this they represent joint inputs to beneficiary well-being (usually sectors or household).

Figure 3.14 reports some examples of ecosystem services in relation to the classification in direct inputs and joint inputs in ecosystem accounting view. The carbon sequestration and storage ecosystem services are direct inputs to household well-being and in this case the users will be the global society. The place and seascape and wild fish ecosystem services are joint inputs to household final consumption as they need a human action, human capital or complementary assets (such as inputting energy, time, effort, skills or the ability to be sentient (Elliott, 2023)) to provide benefits (e.g., travel to a specific site or vessels to catch) and usually they contribute to the production sector such as tourism or fisheries sector.

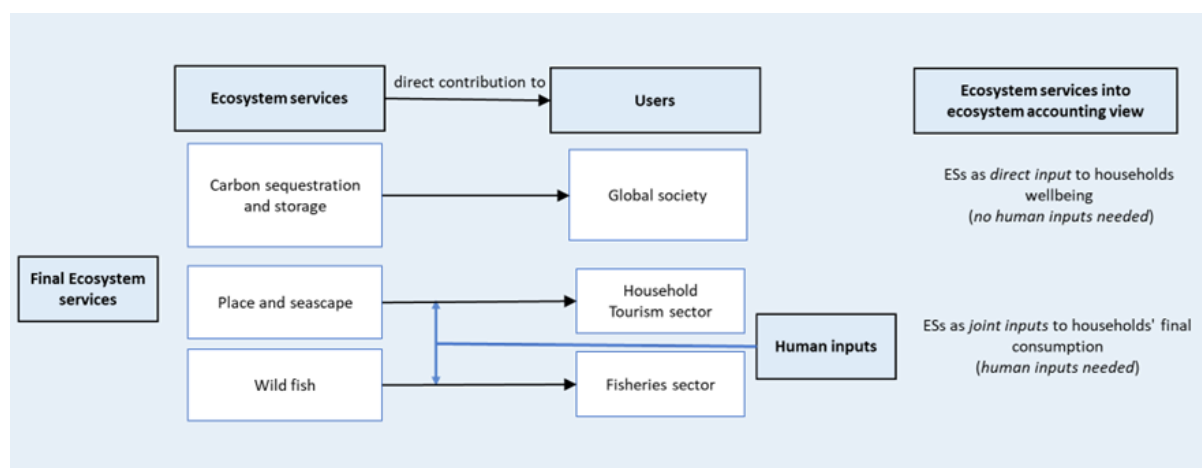


Figure 3.14: Ecosystem services classification in direct inputs and joint inputs.

STEP 2 Defining benefits and their boundaries (SNA or non-SNA)

In accounting and national income accounting (SNA), the production boundary refers to the conceptual line that separates economic activities included in the measurement of gross domestic

product (GDP) from those that are not. It defines which activities are considered productive and contribute to the economy.

The production boundary is important because it influences GDP calculations, economic statistics and policy decisions by determining what is counted as economic output. In the context of natural capital accounting, it is important to understand the link between the production boundary and benefits.

The term ‘benefits’ is used for all goods and services that are used and actively or passively enjoyed by people and society; termed societal benefits in Figure 3.1. Societal benefits are categorized in SNA-benefits and non-SNA benefits. The SNA-benefits refer to goods and services that are already exchanged in the market with a specific price and included in the SNA (within the production boundary that refers to all economic activities taken to contribute to economic performance, GDP). For example, commercial fish and recreation services are benefits provided respectively by the wild fish and places and seascapes ecosystem services. In the case of SNA benefits, ecosystem services are considered as joint inputs to the production system (e.g., fisheries or tourism sector) in addition to labour and infrastructure; the aim of natural accounting is to partially separate out the contribution of natural inputs to economic outputs. In other words, part of the revenues of fisheries and recreation services are due to the healthy functioning of marine ecosystems. This is why the SNA-benefit (within the production boundary) do not affect the overall value of economic performance, GDP.

Non-SNA benefits refer to goods and services that do not fit within the conventional SNA framework. Examples include a healthy climate, erosion and flood prevention and clean water. Ecosystem services related to these non-SNA benefits, such as carbon sequestration, are not accounted for in the SNA. Including non-SNA benefits in accounting expands the SNA production boundary, leading to changes in overall economic output (GDP), enriching economic statistics and providing additional insights into the economy’s dependency on and impact on the marine ecosystem. Moreover, this distinction is relevant to avoid overlaps with goods and services recorded in the SNA (SNA-benefits) and help to identify the users of ecosystem services as we explain in Step 3.

STEP 3 Defining users

The users of societal benefits obtained from ecosystem services are the economic units who benefit from the existence of them, which are industry sectors, households, governments and global society.

Knowing who is benefitting from a societal benefit is relevant to understand which economic valuation is more suitable for monetary accounting and where data can be found. For example, a seascape can be physically enjoyed by people through recreational opportunities and data may be found in national accounting from the statistics office and it will refer to the tourism sector as users.

STEPS 4 and 5 Defining valuation concepts and methods

Steps 4 and 5 are the core of the societal benefits economic valuation exercise. The societal benefit economic valuation in the SEEA-EA is based on the exchange value concept which is meant to be consistent with standard measures of products and assets recorded in the SNA. The exchange value is defined as “the value at which goods, services, labour or assets are in fact exchanged or else could be exchanged for cash”. In conventional national income accounts (SNA), exchange values refer to market prices that are defined as “the amounts of money that willing buyers pay to acquire something from willing sellers” (NCAVES & MAIA, 2022).

Although economic valuation based on exchange values is the only economic valuation allowed by the SEEA-EA guidelines, some societal benefits provided by ecosystem services (both SNA and non-SNA) may require, or could be better represented by, economic welfare measurements and non-monetary evaluations (see Section 4).

For example, the nature recreation value can be calculated with exchange prices (e.g., resource rent, Table 3.1) and this value refers mainly to the tourism sector or private sector users and data. In this case, this value helps to understand how certain habitats supports economic activities such as the

tourism sector. Moreover, this value also shows the indirect dependency of the economic sector on ecosystems; in other words, if the marine environment is suddenly degraded also market activities will be impacted (e.g., less tourism). This value is consistent with the SEEA-EA guidelines and it attributes to ecosystems the portion of the recreation economic value currently included in the SNA but solely attributed to the tourism sector (see Section 1).

The recreation value can also be enjoyed by people who live near the coast and go for a daily walk. This value refers to the direct human enjoyment of nature or the daily nature-based recreation activity that does not generate an economic activity and therefore is not included in the SNA approach, but still increases well-being. In fact, this value is relevant for people's well-being; several studies report that proximity to nature spaces increase physical activity, social interaction and health (Völker & Kistemann, 2011; Jennings & Bamkole, 2019; Garrett et al., 2023). This economic value can be measured in welfare values rather than exchange values (see Section 4), but if we follow the SEEA-EA guidelines, it cannot be integrated into the SNA tables. If there is a need to measure the economic benefits of nature recreation as welfare value as well as with other non-monetary indicators (e.g., wellbeing indices), then the CAN approach should be used (see Section 5).

The economic valuation methods that are recommended in the SEEA-EA are consistent with the SNA protocol and therefore based on the exchange value concept (see Step 4 and 5). An example of ecosystem accounting for recreational values provided by seagrass in Sardinia is provided in Box 3.1.

The economic valuation methods that are recommended in the context of SEEA-EA are classified as follows (United Nations, 2021):

1. Market-based or cost-based methods

- Unit resource rent
- Production function, cost function and profit function methods
- Replacement cost
- Damage costs or defensive expenditure
- Averting behaviour
- Payment for Ecosystem Services (PES) schemes
- Value transfer

2. Revealed Preference methods

- Hedonic pricing
- Marginal values from travel cost demand functions (simulated exchange)

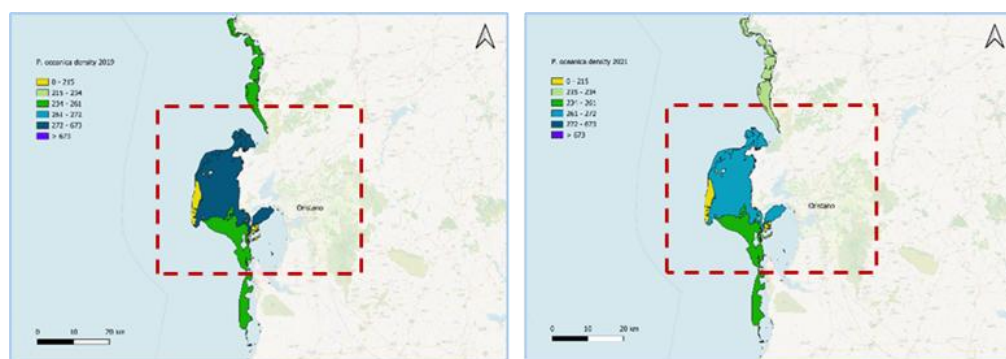
Box 3.1: Ecosystem Accounting for the recreation ecosystem service (Marine-Based Tourism (MBT)) provided by *Posidonia oceanica* in the Sardinia BBT


Fig 1 Extent and condition of *P. oceanica* meadows in the Sardinia BBT; a **decreasing trend** of the estimated *Posidonia oceanica* density is found in the area from 2019 (left hand side) to 2021 (right hand side); the red dotted line shows the boundaries of the Sardinia BBT.

Policy question: How does *Posidonia oceanica* contribute to the tourism industry?

Methodology: The travel expenditure (see Section 3.7) is applied to translate the number of arrivals due to the presence of *Posidonia oceanica* in the Sardinia BBT in monetary term. See the Bartolini et al. (2024) methodological approach which is downscaled to the case study in the Sardinia BBT.

Results and conclusion: We find that *P. oceanica* significantly contributes to the tourism sector in the Sardinia BBT, resulting in exchange values of MBT ecosystem services of € 126,341 in 2019 and € 88,645 in 2021.

Ecosystem service flow in physical terms: A positive and statistically significant relationship between the number of arrivals and the quality status of *Posidonia oceanica* results from the statistical model which indicates that a one-unit increase in *P. oceanica* density (e.g. from 169 to 170 shoots/m²) corresponds to an almost 1% (0.98)^[1] variation in the number of arrivals in the Sardinia BBT (proxy by Oristano municipality). This percentage is used to parcel out the arrivals driven by the presence of *P. oceanica* from the total tourists in Oristano in 2019 and 2021.

Ecosystem service flow in monetary terms: The travel expenditure method is used to convert the physical flow into monetary terms once the number of arrivals is quantified. The travel expenditure is computed using the travel distance and the expenditure of travel. The travel distance of the arrival flow from each Italian region to Oristano is based on the national tourist statistics^[2]. The travel expenditure is calculated by using a cost of 0.26 €/km according to Paletto et al., (2022).

Table 1: Marine Natural Capital Accounting: Supply and Use Table for Sardinia BBT

2019	Unit	Users			Ecosystem
Sardinia BBT		Tourism	Households	Government	<i>Posidonia oceanica</i>
SUPPLY					
Recreation(Tourism) ES	#arrivals				707
USE					
Recreation(Tourism) ES	#arrivals	707			

2019	Unit	Users			Ecosystem
Sardinia BBT		Tourism	Households	Government	<i>Posidonia oceanica</i>
SUPPLY					
Recreation(Tourism) ES	€				126,341
USE					
Recreation(Tourism) ES	€	126,341			

References

Bartolini, A., Di Gennaro, V., Reas, V., Mascolo, R. A., La Notte, A., Capriolo, A., & Ferrini, S. (2024). Ecosystem accounting for marine-based tourism provided by *Posidonia Oceanica* in Italy. *One Ecosystem*, 9, e129751;
 Paletto A, De Meo I, Grilli G, Notaro S (2022) Valuing nature-based recreation in forest areas in Italy: An application of Travel Cost Method (TCM). *Journal of Leisure Research* 54 (1): 26-45.

^[1] Pooled model estimates for Sardinia BBT (proxied by Oristano municipality)

^[2] https://www326.regione.toscana.it/prodext/Turismo_matrice/

The resource rent method computes the value of ecosystem services mainly using the difference of revenues and costs related to the benefits provided. For example, the value of fish caught by the fishing industry uses commercial fisheries revenues minus the operating costs and depreciation of infrastructure.

The production function method links the output of marketed good and services to a change in ecosystem services considered as an input of production holding other things equal. For instance, the reduction in fishery output because of reduction of wild fish ecosystem services.

The replacement cost method attributes the value by quantifying the cost of replacing an ecosystem service with an artificial or man-made infrastructure, products or technology. For example, the value of natural hazard protection ecosystem services is computed using the costs of building a seawall.

The damage cost method estimates the value of the ecosystem services using the costs that would incur to property, infrastructure and production due to the loss or reduction of ecosystem services. For example, the damage to roads, bridges, farms and property resulting from increased flooding after the loss of vegetation.

As with the previous methods, the mitigating or averting behaviour estimates the value of ecosystem services using the cost of mitigating or averting the negative effects of ecosystem services loss; for example, the costs of additional water treatment required to maintain water-quality standards after the loss of natural wetlands.

Payment for Ecosystem Services (PES) links the price of a potential buyers to the benefits provided by the ecosystem services. For example, hotels and tourism operators could pay for reef conservation due to the benefits associated with decreased beach erosion and species conservation (e.g., for scuba divers).

The value transfer technique uses average values of ecosystem services taken from existing and similar studies and adapts these to specific cases controlling for key factors that characterize the two contexts. For example, the value of recreational ecosystem services for visiting one beach in UK could be applied to another similar beach in another country.

The hedonic price method computes the value of ecosystem services through the difference in price ascribable to the presence of these ecosystem services. For example, the difference in price between houses overlooking the sea that benefit from clean air compared to similar ones that do not can estimate the value of the carbon sequestration and storage ecosystem services.

Table 3.1 summarises the methods of economic valuation consistent with SEEA-EA for natural capital accounting giving a description, data required for each and some advantages and disadvantages from the literature.

Table 3.1: Economic valuation methods consistent with SEEA EA for natural capital accounting.

Value	Method	Name	Description	Data needed
Exchange value	Market-based or cost-based methods	Unit resource rent	Resource rent computes “a value for an ecosystem service by taking the gross value of the final marketed good to which the ecosystem service provides an input and then deducting the cost of all other inputs, including labour, produced assets and intermediate inputs” (United Nations et al., (2021). The formula is showed below. Output less intermediate consumption less compensation of employees less other taxes on production plus other subsidies on production Equals gross operating surplus less consumption of fixed capital (depreciation) less return on produced assets less labour of self-employed persons Equals resource rent = depletion + net return on environmental assets (All values have to be in currency)	- Indicate for provisioning ecosystem services (e.g. wild fish and other natural aquatic biomass provisioning services) - Data on revenues and costs of production, taxes, depreciation of fixed capital needed
		Production function, cost function and profit function methods	Production function considers the ecosystem service as an input in the marketed good.	- Indicate for ecosystem services as joint inputs to production - Data intensive
		Replacement cost	Replacement cost estimates the cost of using the substitute alternative to provide the same benefits as provided by a single quantity unit of the ecosystem service	The existence of substitute or alternative and its costs (e.g., natural hazard protection substituting for a seawall)
		Damage costs or defensive expenditure	Avoided damage cost estimates the value of ecosystem services based on costs of damages that would occur due to the loss or poor condition of these services.	Indicate for regulating ecosystem services such as flood control, soil erosion, air filtration, global climate regulation services

Value	Method	Name	Description	Data needed
				Attention on the unit used required (for example, the value of air filtration services may be related to avoided health costs to governments but the beneficiaries are the global society)
		Averting behaviour	Expenditures for preventing or mitigating negative effects and damages of poor condition or polluted environment are used to compute the value of ecosystem services (for example, costs for extra filtration for purifying polluted water or air conditioning for avoiding polluted air)	- The value represents a lower bound estimate - Easier than damage costs method to compute - Attention of expenditure included needed (they must only reflect the cost of avoiding environmental impacts rather than taste or consumption preferences)
		Payment for Ecosystem Services (PES) schemes	PES programmes are financial policy instruments for conservation and sustainable management and use of ecosystem. They are based on the beneficiary-pays approach meaning that those who would benefit from the enhanced provision of ecosystem services (i.e. above that of the status quo) can pay resource owners or managers to change their management practices so as to incentivise higher (or additional) ecosystem service provision. Potential buyers may include the fishing, tourism, recreation and marine renewable energy industries; municipalities and governments; and so forth. For example, local hotels and tourism operators could pay for reef conservation due to the benefits associated with decreased beach erosion and species conservation (e.g. for scuba divers).	- PES programmes in the marine environment are challenging as marine resources such as fish, are mobile and hard to monitor, and property rights are often poorly defined and insecure - PES programmes involve local communities - Identification of beneficiaries and the amount of payments needed (e.g. revenues from commercial fishing licences, marine ecotourism revenue sharing and oil and gas taxation that is used to pay coastal communities for conservation)
		Value transfer	Value transfer or benefit transfer includes unit or function transfer value (e.g. meta-analysis). Unit transfer uses (and adjusts) monetary value derived from the mean or median value of a set of primary studies focused on similar ecosystem services and it usually expresses in unit such as £/hectare, £/visits, £/tonne. Value function transfer aims to produce secondary estimates using a function transfer produced in primary studies or developed via a meta regression analysis where the function should be characterized by case specific available info such as population, extension of site.	- Primary studies available - Literature review needed - Similarity with the context of primary studies (e.g. socio-economic and demographic characteristics, institutional context, geospatial characteristics, etc.) - The quality of estimates depends on data and studies available

Value	Method	Name	Description	Data needed
	Revealed Preference methods	Hedonic pricing	The hedonic price identifies the value of ecosystem services statically modelling the property prices as a function of several characteristics including the natural environment.	- Context specific (spatially related) - Data on market property needed (number of rooms, size, heating, proximity to parks, quality of air, etc.)
		Marginal values from travel cost demand functions (simulated exchange)	The simulated exchange method proposes to simulate, in a partial equilibrium context, the price that would occur if a good or service outside of the market, such as free access recreation in forests, were internalized. The method considers the demand, estimated using non-market valuation techniques, the supply and the market structure.	Previous non-market valuation techniques required
Welfare value	Stated Preference methods	Contingent Valuation; Choice experiment	See section 5 (Complementary Account Network)	

STEP 6 Data.

This step aims to check the availability of data and potential needs for data collection. As in the SNA, the data must ideally have a start date (opening the account) and an end date (closing the account). If time-series data are not immediately available, then a time should be chosen for which data do exist (start point for the account) and then the prospects for future data collection should be assessed.

Some examples of ecosystem services accounting and information needed are reported in the next paragraphs and we refer to Section 3.4 for details on the physical accounts.

The societal benefit of 'wild fish' (from provisioning ecosystem services) are valued using the resource rent approach. Data needed to compute the resource rent such as operating costs, intermediate consumption, compensation of employees, subsidies and taxes on extraction, etc. should be ideally found from annual financial balance sheet of companies operating in the fisheries sector. When these data are not available or not disclosed, the resource rent is derived from aggregate data using the input-output tables of the national accounting.

The healthy climate societal benefit (from carbon sequestration and storage regulating ecosystem service) valuation with the abatement cost of non-traded carbon needs information about:

- The extent of marine and coastal habitat providing the ecosystem services (see Section 3.4);
- Carbon sequestration and storage rate or intervals (low and high value rate) for each habitat (per tonne of CO₂ equivalent/ha/year) (see Section 3.4);
- The monetary value of carbon (usually express in £/tCO₂equivalent)

Societal benefits from the 'places and seascapes' ecosystem service can be valued using several methods such as resource rent or travel cost. Information to collect change depending on the chosen method. For the resource rent, similar to the wild fish societal benefit, data required are tourism revenues and costs, while the simulated exchange method needs:

- Identification of coastal and marine sites that provide tourism and recreation societal benefits (e.g. parks, marine protect areas) (see Section 3.4);
- previous primary studies with data about the cost per trip including travel, parking and admission costs (e.g. entrance fee, cost of fuel) and the total number of visits and time spent at the sites.

For the valuation of natural hazard protection, the replacement costs method can be applied. It requires information about:

- the extent of habitat (e.g. saltmarsh) (see Section 3.4);
- the minimum width of habitat needed to generate the coastal protection societal benefit;
- the cost of building an artificial habitat or man-made coastal defence such as a seawall that could provide that same societal benefit.

For waste nutrient remediation such as Nitrogen (N) and Phosphorous (P), information needed for monetary valuation with the avoided damage costs are:

- the remediation capacity threshold that varies for each habitat and contextual factors such as the presence of nutrients;
- the quantity of nutrients discharged into catchments or marine waters usually estimated from bioecological models;
- the cost for treating a unit (e.g. £/kg) of the same nutrient that would occur due to the loss or poor condition of natural ecosystem services for example referring in previous case studies and literature

In this step, it is of note that biophysical and economic data should be compatible with their units. For example, if the carbon sequestration rate is given in tCO₂equivalent/ha/year, then the cost of CO₂ equivalent must be in £/t given the extent of habitat in hectares.

Marine accounting is still in its infancy and data limitation issues are challenging many countries that are still experimenting how accounts tables can be built. Information, geographical scale and time series of data sets availability should be previously researched and catalogued to produce a set of consistent information. Data needed for monetary accounts could be found into local, regional, national or EU data sources and later transformed and rescaled for BBTs accounting purpose. However, further data collection across time should be part of the accounting approach and it is necessary for marine accounting development.

In general, the country statistical offices and Departments in charge of managing national accounting data are used as the main source for the ecosystem services related to the SNA-benefits. However, the level of aggregation and the lack of some necessary data to link the physical availability of natural resources and their value can be a significant issue to consider. For the ecosystem services related to non-SNA benefits, data depends mainly on the method chosen.

More pragmatically, the pilot natural capital accounts for marine and coastal environment applied in UK (Thornton et al., 2019) and Australia (IDEA Group, 2020) are presented as a case study at the national scale. The monetary valuation of wild fish ecosystem services is computed using the resource rent method in both studies. Data about revenues and costs of the fishery industry are from fisheries and aquaculture sector data from Office of National Statistics and Department of Primary Industries and Regional Development. Similar information for each European Country is available in the Scientific, Technical and Economic Committee for Fisheries website (Fish Processing Industry - European Commission (europa.eu) and data management is required to allow to use these data for accounting aims.

The avoid damage costs method is used for the carbon sequestration and storage ecosystem services valuation; once quantified the tonnes of CO₂ or CO₂ equivalent and the area per habitat, the economic value of carbon is used as a proxy for the price of carbon suggested by the Environmental Agency for the UK case study.

In Australia, the value of recreational societal benefits from the whale watching sector is derived from the total expenditure, value added, wages and salaries, gross operating surplus information using the resource rent approach; data are from the local Tourism Research Centre.

Table 3.2 summarises some societal benefits which are the object of monetary valuation, method and data sources used in these two case studies.

Table 3.2: Example of data sources and information for marine natural capital accounts.

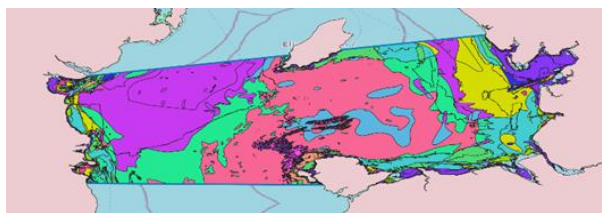
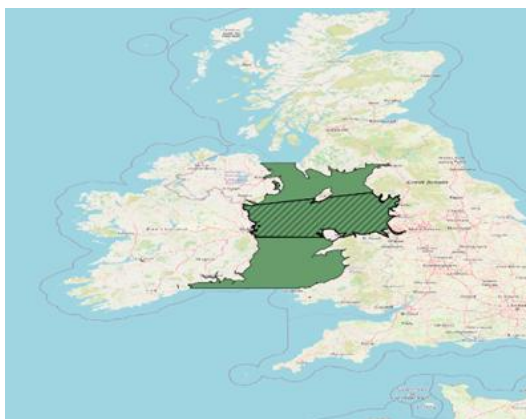
Societal Benefits	Method	Information	Geographical scale	Data sources
Wild fish	Resource rent	Revenues and operation Cost of fishing	UK	Fisheries and aquaculture sector data from national statistics office (national accounting ONS)
Wild fish	Resource rent	Revenues and operation Cost of fishing	Australia	Department of Primary Industries and Regional Development
Carbon sequestration and storage	Avoided damage costs	Price of carbon	UK	Department for Business, Energy and Industrial Strategy Valuation of energy use and greenhouse gas – supplementary guidance to the HM Treasury Green Book on Appraisal and Evaluation in Central Government. Available from: Valuation of energy use and greenhouse gas (GHG) emissions (publishing.service.gov.uk)
Recreational (from whale watching sector)	Resource rent	Whale watching tourism sector: Total expenditure, value added, wages and salaries, gross operating surplus	Australia	Tourism Research Australia
Natural hazard protection	Replacement cost	Annual capital cost of building a seawall	UK	Estimates by the Environmental Agency
Waste (nutrient) remediation	Avoided damage costs	Wastewater discharged from urban wastewater treatment plants, Cost avoided for the treatment of selected pollutants	UK	For estimate the quantity of UK urban waste discharge in coastal and marine waters: EU Urban Waste Water Treatment Directive (UWWTD) and the Irish Environmental Protection Agency (EPA); For the cost of wastewater treatment process: Hernandez-Sancho et al., (2010)

Other examples of experimental economic value of are provided by the environmental accounts, for example, by the UK national statistics office (ONS, 2021). Marine renewable energy such as offshore wind energy, marine fossil fuels, minerals such as sand and gravel (societal benefits provided by provisioning ecosystem services), flood protection, carbon sequestration, wastewater remediation (societal benefits provided by regulating ecosystem services) and coastal recreation, amenity value of sea views (societal benefits provided cultural ecosystem services) are all valued. Moreover, strengths, limitations and assumptions made of each monetary valuation are highlighted as well and it is a fundamental part of process that should be applied as standard to allow future natural capital accounting improvement and assessment

of reliability of marine ecosystem services value estimates (e.g., reporting intervals or variance of values, accessibility, transparency and reusability of data).

The monetary valuation of societal benefits allows us to fill the supply and use tables in monetary terms for the marine natural capital accounts for each year of the analysis. The supply table will provide information on which habitat provides a specific societal benefit in monetary terms, while the use table allows for recording the use of societal benefits by users (society and/or industry) – for an example see Annex 6.

Examples of applications of ecosystem accounting are presented for the Irish Sea BBT and the Archipelago Sea in Boxes 3.2 and 3.3, respectively.

Box 3.2: Ecosystem Accounting for the fish provisioning ecosystem service in the Irish Sea BBT

Left-side Fig: In green - Irish Sea fishing area (27.7.a), dashed lines - Irish Sea BBT; Right-side Fig: Habitats extent in the Irish Sea BBT and % area of BBT: MD32 Atlantic offshore circalittoral coarse sediment (29%), MD42 Atlantic offshore circalittoral coarse sediment (5%), MD52 Atlantic offshore circalittoral sand (18%), MD62 Atlantic offshore circalittoral mud (24%).

Policy question: How do marine habitats contribute to the fishery industry?

Study area: Irish Sea BBT (within Ireland, UK, Isle of Man)

Methodology: The species-habitat association - the quantity (kg) of fish caught per year in each of the habitats of the Irish Sea BBT – is identified by overlapping stocks condition sampling data with habitat maps using QGIS. Subsequently, the species-habitat association in percentage is used to allocate the landings data for the targeted species: *Gadus morhua* (COD), *Melanogrammus aeglefinus* (HAD), *Merlangius merlangus* (WHG), *Pleuronectes platessa* (PLE) and *Solea solea* (SOL) for 2021. The resource rent (see section 2.4) is applied to translate the kg of fish caught in the Irish Sea BBT habitats in monetary term.

Results: The ecological and economic data integration highlights the species-habitat association: for 2021, the Atlantic offshore circalittoral mud (MD62) contributes most to the provision of targeted fish species *Gadus morhua* (86%), *Melanogrammus aeglefinus* (78%), *Merlangius merlangus* (69%) and *Solea solea* (36%), while the Atlantic offshore circalittoral sand (MD52) for *Pleuronectes platessa* (47%). The unit resource rent for *Gadus morhua* (0.88 €/kg), *Melanogrammus aeglefinus* (2.15 €/kg), *Merlangius merlangus* (1.69 €/kg), *Pleuronectes platessa* (1.14 €/kg) and *Solea solea* (1.04 €/kg) are applied for the supply and use table in monetary terms.

In 2021, the marine ecosystem in the Irish Sea BBT provided 943,722 kg of fish (COD, HAD, WHG, PLE, SOL) to fisheries, worth €1.8 million.

Table 1: Marine Natural Capital Accounting: Supply and Use Table for Irish Sea BBT.

2021	Unit	Users			Ecosystem			
Irish Sea BBT		Fishery	Households	Government	MD32	MD42	MD52	MD62
SUPPLY								
<i>Gadus morhua</i> (COD)	€				431	-	5,139	46,958
<i>Melanogrammus aeglefinus</i> (HAD)	€				12,529	54,126	180,504	1,305,195
<i>Merlangius merlangus</i> (WHG)	€				6,606	245	17,424	83,227
<i>Pleuronectes platessa</i> (PLE)	€				4,731	357	21,461	22,896
<i>Solea solea</i> (SOL)	€				12,328	-	17,937	26,959
USE								
<i>Gadus morhua</i> (COD)	€	52,528						
<i>Melanogrammus aeglefinus</i> (HAD)	€	1,552,354						
<i>Merlangius merlangus</i> (WHG)	€	107,502						
<i>Pleuronectes platessa</i> (PLE)	€	49,445						
<i>Solea solea</i> (SOL)	€	57,224						

Box 3.3: Physical Natural Capital Account (Extent and Supply Tables) in BBT4 - Archipelago Sea (Baltic Sea)

The case study area is located in the Archipelago Sea, Northern Baltic Sea area, in Finland (Fig. 1). In this area, the ecosystem service “finfish” was assessed to reveal the economic value of the food provision service in the BBT4 area.

This service was estimated to link to five of the twelve EUNIS level 3 habitats¹ (Table 1; extent table, see M38).

The finfish data include catch data from 14 commercial fish species per year for years 2013-2022²: Baltic herring, sprat, whitefish, salmon, trout, smelt, bream, ide, roach, pike, perch, pikeperch, burbot and rainbow trout. These species form >99% of the total catch in the area. The data originate from four ICES statistical squares that overlap the case study area (Fig. 1). The catches were averaged across the squares each year and used as the estimate for catches. As the habitats form a high degree mosaic in the region, separating the catches per habitat was not feasible and a common quantity was used for all contributing habitats (Table 2; supply table, see M38). The catch areas also include geographic areas outside the case study area and the estimations may thus over- or underestimate the catches.

The economic value of the benefits is based on the data of fish prices per species per year. The pricing excludes smelt (no price data available), which formed 0.2-6.7% of the total annual catch. In addition, when price information for individual years was missing, the previous year prices were used. For Baltic herring, the average value of prices for human consumption and industrial use was used. The same price was assumed across different size categories. The catches and economic value of the caught finfish in the area decreased >30 % during the last two years of the data set (Table 2); a result worth further investigation.

Table 1. The ecosystem extent table

unit	Habitat				
	MC53: Baltic circalittoral sand	MC33: Baltic circalittoral coarse sediment	MC 13: Baltic circalittoral rock	MB53: Baltic infralittoral sand	MB13: Baltic infralittoral rock
area km2	0.00119	0.00285	0.04027	0.00272	0.04468

Table 2. The supply table indicating the quantity of benefits and the estimated value of the benefits in 2013-2022.

Ecosystem service	Unit of measure	Habitat					Supply									
		MC53: Baltic circalittoral sand	MC33: Baltic circalittoral coarse sediment	MC 13: Baltic circalittoral rock	MB53: Baltic infralittoral sand	MB13: Baltic infralittoral rock	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Wild fish	tons	common quantity for all habitats					4177.5	5585.3	5082.8	4470.0	5445.3	4755.3	5230.0	5168.3	3025.3	2133.0
Wild fish	€	common price for all habitats					1224.0	1441.0	1108.2	1071.5	1186.1	1125.6	1265.6	1287.1	848.4	746.8

¹EUNIS-habitat data are from EMODnet, ²finfish and price data from National Resources, ³price data from Statistics Finland

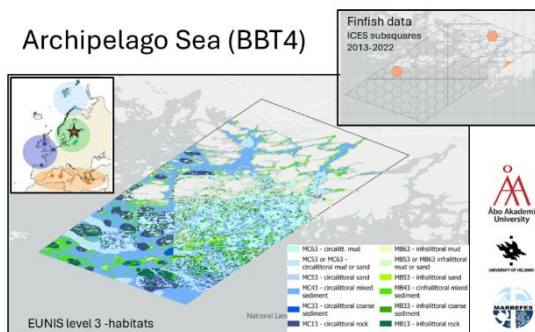


Fig. 1. The geographic study area in the Archipelago Sea (BBT4). The Map of Europe indicates the location of the study area (indicated by a star). The larger map indicates the 50 km wide geographic study area and the EUNIS level 3 habitats in it. The map in the top right corner indicates the cross-section of ICES statistical squares (i.e., the geographic resolution for the finfish data used in the assessment) with the study area.

3.7 Economic Valuation Methods

3.7.1 Economic Valuation

This section first introduces the definitions of different types of economic values, presents several examples and illustrates their potential role in informing academic and policy debates. Afterwards, it details the many different methodologies that can be used to estimate such values, including revealed preference methods, stated preference methods and benefit transfer. For each method we also present some detailed examples from the environmental economics literature.

Before exploring how economics can contribute to estimating the monetary value of marine natural resources, we need to clarify the meaning of the term “value” in economics. Firstly, economics is the science that studies how people make choices involving scarce resources. While many of such choices will involve money, the realm of economics is much broader and includes decisions that affect health, free-time, happiness and prosperity (the so-called Drivers as basic human needs in the DAPSI(W)R(M) model). This wide set of objectives is typically indicated with terms such as ‘wellbeing’ or ‘welfare’ (or ‘utility’) and represents the widest interpretation of economic value. Often, economists try to express such wellbeing in monetary units, even if no market transactions are involved, as money is familiar to everyone and comparable across different domains.

For example, let us consider a person living in a coastal town and going to one of the local beaches to swim in the sea. Let us assume that they go to the beach on foot and that the access to the beach is completely free. Therefore, this action does not involve any market transaction. However, this does not mean that the beach has no economic value for that person. They are going to the beach because that action increases their wellbeing (or ‘happiness’) and improves their physical and mental health. Therefore, the beach, despite being completely free, generates values. Economists have devised methods to quantify in monetary terms such values. We are going to return to such methods in Section 3.7.2.

As an economic compared to an ecological view, the anthropocentric nature of societal benefits means that economic valuation is the most appropriate framework for estimating their values. The approach that researchers follow is to understand the welfare generated by the societal benefits produced using those services. However, one cannot simply attribute all the value of such goods to ecosystem services.

For example, let us consider again the example of the wild fish ecosystem services to the societal benefit of fish food production. Let us consider commercial fisheries. Besides the underlying ecosystem services, the fisheries sector also uses vessels, human labour and many other inputs. To estimate the value of the societal benefit for fish food production, we must separate the contribution of the ecosystem services from those of all these other “human-made” inputs, thus isolating the value generated by the natural environment. Most of the approaches described in Section 3.6 are designed to take this into account for example the resource rent method.

Given the general nature of value in economics, researchers defined different types of economic values related to societal benefits. For example, Turner and Schaafsma (2015) summarise the total economic value of societal benefits as use value, option and quasi-option value and non-use values, as shown in Figure 3.15.

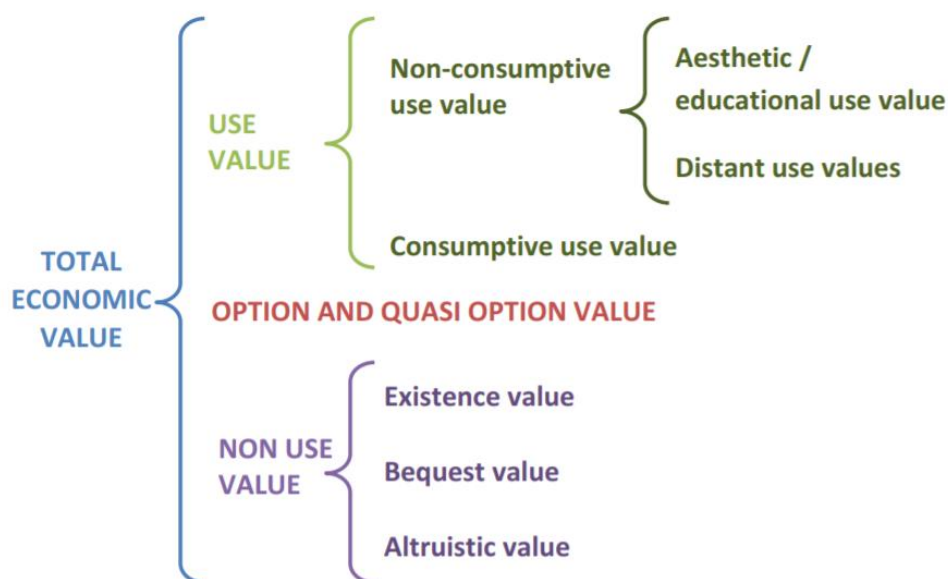


Figure 3.15: Total economic value (Turner & Schaafsma, 2015).

The first one is the *use value* i.e., the value that is generated by directly interacting with the natural environment, for example by eating fish caught offshore, or by swimming in a pristine coastal area. While the one in the first example is a market value (fish are traded in markets and generate monetary revenues), the one in the second example is a nonmarket value since access to the beach is normally free (however, note that if an entrance fee is introduced, a share of this non-market value can be converted into market transactions). In any case, both the value generated by fish consumption and beach visitation are examples of use values.

The second type of value is the *option value*. This is generated from knowing that environmental resources we currently do not use will still be there if we need them in the future. This is like an insurance policy. For example, biodiversity can hold option values because it allows ecosystems to become more resilient to future threats such as climate change. Quasi-option value is the value of information gained by delaying a decision to commit to some irreversible action (see Arrow & Fisher, 1974). For example, if a decision maker is deciding whether to preserve or to develop a piece of land, where the future value of preservation is unknown and will be discovered later, the quasi-option value is equal to the value of delaying the decision to develop until after the information pertaining to preservation is obtained.

Use values and option values can be estimated using Revealed Preferences (RP), using methods such as travel cost and hedonic pricing (Section 3.6.3.1).

There are then three types of values and they are all *non-use values*. The first one is the existence value: we can hold values for the sake of nature itself, independently from any use that humans may ever do of it. For example, a person may be happy to know that blue whales are still roaming in the ocean, even without ever planning to go whale watching. Another type of non-use value is the altruistic value: a person may never go whale watching, but they may be happy to know that other people can. Finally, the last type of non-use value is the bequest value: this is similar to the altruistic value, but it relates to future generations. Not surprisingly, non-use values are the hardest to measure empirically. The main approaches for such an estimation are Stated Preference methods, such as contingent valuations and choice experiments (Section 3.6.3.2).

The total economic value refers to the sum of use, option and non-use values. Total value changes in response to changes in the environment can be positive or negative and can also greatly vary among different individuals.

Economic values can contribute to better decision-making and environmental policies. However, we first need to acknowledge that economic values are just one consideration. Many values of nature exist that are outside the realm of economics. Scientific, social, moral, ethical and legal factors must also be considered (*à la* the ten-tenets for sustainable and successful environmental management (Elliott et al., 2025)).

Economic values can be used to improve our understanding of how nature generates benefits for mankind (termed societal benefits here), highlighting the non-market values that are often overshadowed by market ones. Since resources are finite, estimating non-market values can help policy makers to develop policies that deliver better outcomes for society. For example, economic values can highlight the cost of pollution and help develop taxes or other instruments designed to reduce it. They can help to understand the synergies and trade-offs between different groups of people and find efficient solutions. For example, valuation can help design Payments for Ecosystem Services. Examples are payments that water companies are providing to farmers to reduce their nitrate and pesticide applications, to improve river and estuarine water quality. It can highlight the impact of global environmental changes such as those fostered by climate change and their values for humanity. Examples are estimating the economic value of the impact of climate change on coral reefs (Fezzi et al., 2023) and measuring the value of seabed health, which is an important component of marine natural capital (Börger et al., 2020).

3.7.2 Economic Valuation Methods

As illustrated in the previous section, the valuation of societal benefits from natural resources constitutes a critical endeavour to inform policy decisions. Central to this pursuit are two primary methodologies: revealed preferences and stated preferences methods. The former leverages empirical data derived from observed behaviours and market transactions to discern valuations of societal benefits for individuals. By analysing how market prices or resource allocations change in response to environmental changes, economists can infer the implicit preferences individuals place on various environmental attributes, indicating the relative significance of factors such as clean air or scenic landscapes in individual decision-making processes.

Conversely, stated preferences methods engage individuals directly through structured surveys, hypothetical scenarios and choice experiments to extract their subjective evaluations and willingness-to-pay for diverse environmental aspects. This technique proves particularly valuable for assessing non-use environmental values such as preservation of biodiversity and the cultural heritage sites. By presenting respondents with scenarios mirroring real-world choices, researchers can quantify individual monetary commitments to environmental enhancements or avoidance of deterioration.

A third approach to environmental valuation is benefit transfer. It allows the estimation of economic values for societal benefits in settings where primary valuation data are lacking or impractical to obtain. This method involves transferring economic values from existing studies conducted in similar contexts to the target area or policy scenario. While a pragmatic approach, benefit transfer does require careful consideration of contextual differences and potential limitations to avoid large transfer errors.

3.6.3.1 Revealed Preferences Methods

The revealed preference methods use observed market behaviours (such as travelling choices to recreation sites) to infer the value consumers put on environmental attributes. The travel cost method and the hedonic pricing method are the most widely used reference methods.

The travel cost method aims at estimating the value people place on a particular environmental feature by analysing how much they are paying to visit it, in terms of travel and admission costs (if any). This method is often used for valuing recreational amenities such as parks and beaches. By analysing how many people visit the site at different price points, we can estimate the demand curve for the amenity and derive a value for it. The two most common types of travel costs approaches are: single-site and multi-site. We will illustrate both in detail in the next sections.

Hedonic pricing involves examining the relationship between the price of a good or service and its environmental characteristics. For example, researchers might look at how the price of a house varies based on its proximity to a river or to the coastline. By analysing such relationships, researchers can estimate the value people place on different environmental amenities.

The main advantages of revealed preference methods are that they are based on actual behaviour, rather than hypothetical scenarios. This means that they are less susceptible to all the different biases and inconsistencies that can arise with stated preferences. However, there are also some limitations. Firstly, they focus on use values only and therefore cannot capture the full range of benefits that people derive from the environment. Second, since they are based on historical data, they sometimes struggle to estimate the values created by large policy changes that generate environmental qualities that are outside the range of the observed data. To clarify this concept using an example in which there is the need to advise the policy-makers of a small island who are considering implementing a coral-reef restoration project adjacent to one of the beaches of the island. They wish to know the economic value that such a project would generate. This project aims to increase coral cover to 90% offshore of that specific beach. If the maximum current level of coral cover on the island is only 10%, any revealed preference method will need to rely on very strong assumptions to extrapolate so significantly the relationships estimated on such a limited range of data. Stated preferences do not share this weakness because respondents can be asked to value very diverse scenarios.

(1) The Travel Cost Method

The travel cost method is the most established revealed preference approach to estimate the recreational value of natural sites. It can also be used to value other public goods such as museums or cultural attractions. This approach estimates the “consumer surplus” associated with a visit, which represents the difference between the maximum amount a visitor is willing to pay for visiting the site and the actual cost incurred in traveling to the site. In less rigorous terms, it is the net increase in well-being that an individual experiences by visiting the site. This well-being can, as usual, be expressed in monetary terms. The travel cost method can be used to value the site access or some specific characteristics of sites, such as the amount of fish biomass or coral cover.

There are two main travel cost approaches: the single-site travel cost and the multi-site travel cost. The single-site travel cost method is simpler to implement and focuses on estimating the value of a specific site, for example, a beach, but can be also used to assess the value of a larger area (such as a region with several beaches). The multi-site travel cost method is more complex, allows estimating the value of multiple, heterogeneous sites within a region and the simulation of many different and spatially explicit policy scenarios with changes in both site accessibility and environmental quality (e.g., Fezzi et al, 2023).

Both methods rely on the following assumptions (Haab & McConnell, 2002):

- 1) Travel and time-cost are proxies for the price of a recreational trip. Travel time does not provide utility (well-being).
- 2) Trips are single-purpose journeys, taken to the site for the sole purpose of recreation. Multiple-purpose trips to different nearby sites are difficult to analyse, especially when the trips are for several days.
- 3) The decision unit is visits of roughly equal length to the site for each individual. For example, in a multi-site model, it would not be correct to mix a 15-minute walk in the local park with an entire day spent at the beach. Such substantial visit-length heterogeneity would call for separate models.

a) The single-site travel cost

In the single-site travel cost method, we are focusing on the value of a specific environmental site, such as a national park or beach. This model assumes that the cost of traveling to the site (including the access cost, if any) and the time required to get there are the primary factors influencing how often people visit the site. On average, people who live closer to the site and, therefore, experience lower travel costs, will visit the site more often than those who live further away and experience higher costs.

Based on this simple consideration, we can write the demand function for the site as follows:

$$x = f(TC, Z) \quad (1)$$

where x is the number of visits made to a site during a certain time interval (for example, a month, or the summer season), TC is the round-trip travel cost which includes fuel cost, access cost (parking, entrance ticket, etc.) and also the value of time and Z are respondents characteristics such as age, personal preferences, etc. This demand function is represented in Figure 3.16.

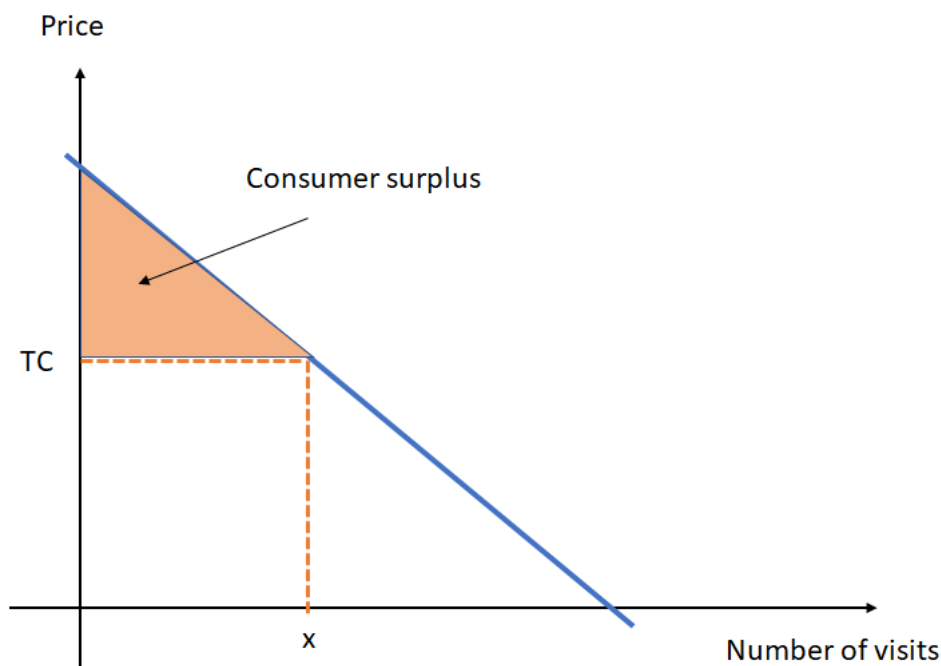


Figure 3.16: Single-site demand function and consumer surplus.

Figure 3.16 shows how the number of visits decreases with the travel cost. The triangle represents the 'consumer surplus', i.e., the willingness to pay for access, for an individual with travel cost equal to TC . It is the extra well-being that such an individual receives from all their visits to the site during the time-span considered in the analysis, i.e., the total well-being generated by the visits after we subtract their travel costs. People with different characteristics can have different demand curves. This can be represented by the variable Z , which shifts the demand curve to the left or the right. For more details about this framework see, for example, Haab and McConnell (2002). For a more technical explanation including the empirical modelling approach to estimate the demand curve, see Annex 7.

Box 3.4: Valuing coastal recreation in Florida, USA.

As an illustrative example of this method for valuing coastal recreation, we present the contribution by Shrestha et al. (2007). In their study, the authors analyse the demand for nature-based recreation in the Apalachicola River region of Florida. Considering the steps outlined before, their study can be detailed as follows:

1) Data Collection.

They adopt a two-step, on-site sampling method to collect the data for their single-site travel cost estimation. In the first step, they interviewed every second visitor whose age is over 18 at the recreation site, collecting information such as name, address, purpose of the trip, etc. They also gave each participant a survey questionnaire packet. In the second step, the participants complete the questionnaire and mail it back to the researcher within 2 weeks. The questionnaire collects information about participants' most satisfying recreation activities, travel costs, site preferences and socioeconomic characteristics.

2) Econometric modelling.

They first employed a Poisson regression model, since it is the most used analytical model in the single-site travel cost method (see the Annex 7 for an illustration of such model). However, they found overdispersion (a situation in which the variance of the distribution is higher than its mean) in their data, so they estimated a negative binomial model to address this issue.

3) Economic valuation.

After estimation, following the Poisson specification mentioned above, the economic value of nature-based recreation is estimated as:

$$WTP = -\frac{1}{\beta_c}$$

where β_c is the coefficient on the travel cost variable. The estimates show an average WTP per day of trip of \$74.

For methodological extensions and an application to the willingness to pay to access a coastal area in Florida see Ferrini and Fezzi (2010).

b) The Multi-site Travel Cost

The multi-site travel cost model is a widely used approach for valuing heterogeneous recreation sites in a region. The basic idea is to generalize the single-site approach to estimate the demand function and the welfare values associated with each of the many recreation sites in the area. These models allow the calculation of many spatially explicit and policy-relevant scenarios which include both access values and quality-changes values. For example, they can simulate the change in visits and values associated with a coral-reef restoration project implemented in different sites of a region. They can also be used to calculate in which sites this restoration project would generate the highest value for society (e.g. Fezzi et al., 2023). They can also simulate the economic impact of temporary site closures, such as those due to sediments from flash flood events.

When modelling recreation demands in multiple locations there exists a few different modelling strategies. Here we will consider the random utility model, which is the most established approach (e.g. Leggett et al., 2018; Dundas and von Haefen, 2020; Fezzi et al., 2023) and was first applied to the analysis of recreation choices by Hanemann (1978).

This method aims at understanding how people trade off travel cost, distance and the characteristics of the different sites to infer their values. Site characteristics include both natural features (e.g., a sandy or rocky beach, the % of coral cover and fish biomass) and anthropic ones (e.g., presence of lifeguards, showers, etc.).

For more technical explanation including the empirical econometric modelling required for this type of models, see Annex 7.

As an illustrative example of this method for valuing coastal recreation, see the recent publication by Fezzi et al. (2023). Their analysis includes data collection, model specification, estimation and scenarios related to coral reef restoration and climate change in almost 100 coastal sites on the Hawaiian island of Maui.

1) Data Collection.

To conduct their empirical analysis, they combine two distinct data sets. The first one contains information on the main characteristics of their case-study area, such as the location and types of recreation sites, the extent of the residential areas and the road network required to calculate travel costs. To identify the recreation sites, they merge information from travel guides, existing spatial datasets and online resources. Their study includes 176 recreation sites, subdivided into beaches and other coastal sites (94), city parks (38), hiking trails (31) and other site types (13). Given the focus of their analysis on coastal management, they also collect data on coral cover (provided by Asner et al., 2020) and fish biomass (provided by the Marine Biogeographic Assessment of the Main Hawaiian Islands).

The second data set concerns the outdoor recreation habits of the resident population and it is collected via an online survey. The key variables collected in this survey are the locations of the respondent homes and the outdoor sites visited in the four weeks prior to the survey. In total, they had 336 valid responses, who recorded a total of 2,703 recreation trips.

2) Econometric modelling.

They adopt the repeated discrete-choice travel-cost random utility model introduced by Morey et al. (1993). On each occasion, utility-maximizing individuals decide on whether to partake in outdoor recreation and on which site to visit, concerning the time constraints, travel cost and recreation site characteristics. They estimate three scenarios using data on 2,703 recreation trips taken to the 176 outdoor sites by 336 survey respondents, assuming 28 choice occasions per respondent (i.e., one choice per day).

3) Economic valuation.

With the estimation results of their best-fitting model, they simulate a series of spatially-explicit scenarios and measure the welfare losses/gained associated with them. For example, they find that the 2014-2015 mass coral-bleaching event has caused a total annual welfare loss of \$26.4 million for Maui's residents, due to loss of recreation opportunities. They also find that the benefits arising from different coral restoration strategies are highly heterogeneous and site dependent. They illustrate how wellbeing can vary up to a factor of more than 100 depending on the site in which coral is restored.

(2) The Hedonic Pricing Method

The exchange prices of goods and services reflect their characteristics, since the latter are related to production costs and are valued according to consumer preferences. The hedonic pricing method takes advantage of this relationship, usually based on the idea that real estate prices reflect environmental attributes such as views, air quality, or proximity to recreational areas among its characteristics. By deconstructing the market price of certain real estate properties into these characteristics, it is possible to identify which part corresponds to environmental attributes and, in this way, obtain information on the economic valuation of the corresponding ecosystem services. For a more technical explanation see Annex 7.

Environmental applications of this method came as early as that by Ridker and Henning (1967), who estimated the negative effects of air pollution on property values in the St Louis metropolitan area, using data from 167 dwelling units sold in 1960. Among the seminal papers for this method, Gardner and Pollakowski (1977) used market sales data for 179 units in Seattle sold during the period 1969-1974 to analyze the effect of proximity to water and water-related open space, related to amenity ecosystem services and Epp and Al-Ani (1979) analyzed real estate prices to determine the economic value of improvements in water quality in Pennsylvania rivers and streams.

Other recent papers include Sander and Polasky (2009) and Gibbons and Resende (2014), both suitable to put a value on ecosystem services linked to environmental amenities and Ma and Swinton (2011) for a wide set of ecosystem services. For ocean ecosystem services see Catma (2020), related to beach quality and Lee (2014) and Asche et al. (2021), related to fish characteristics.

A step-by-step application guide for the application of the hedonic pricing method to estimate the economic value of a certain environmental attribute, potentially related to real estate price, could be as follows:

1. Make sure that hedonic pricing is the appropriate method. First, the considered environmental attribute must be such that real estate prices are sensitive to its presence or its value. Second, a transaction database should be available, sufficiently large to allow the statistical separation of the effect of the environmental characteristic from the rest of the price constituents.
2. Standardize the data, including metrics of the studied environmental attribute (such as sea view, distance to a lake, level of noise...) and prices (purchase or rental, monetary unit, year...).
3. Use statistical or mathematical software to run a regression on a statistical model. Include structural and neighborhood characteristics as explanatory variables, avoiding non-relevant characteristics as well as those that are simultaneously determined by price.
4. Obtain the growth rate g in the real estate price that can be assigned to the environmental attribute. Then, g can be applied to the average price of the real estate properties to estimate the economic value of the environmental attribute, in this case as the present value of the future flows of the corresponding ecosystem service. This value can be adjusted by other variables not controlled in the estimated equation (household size, income, age...) to identify the willingness to pay per average person and extrapolate the value as needed.

Boto-García and Leoni (2023) provide an example to show how a hedonic pricing study is conducted step-by-step. They studied the implicit value of beach characteristics with data from the Spanish Balearic Island Airbnb rental market during the summer of 2016.

1. Airbnb listings are more scattered across the islands compared with hotels, which offers an advantage of large variations regarding both structural characteristics and beach characteristics. This paper used data for daily rates of 16,663 Airbnb listings, host attributes, listing structural characteristics and detailed information about the attributes of 263 beaches, to study the implicit economic value of several beach characteristics in Balearic Islands (Spain) in the summer of 2016. Data on Airbnb listings are from AirDNA and cover the entire Balearic archipelago.
2. They compute the average daily rate (ADR) as the mean price for booked days during August 2016, which is the dependent variable.
3. They ran a weight ordinary least square (WLS) regression, with the nature log of ADR as the dependent variable, beach attributes (including beach length, width and type, coastal frontage, tide, accessibility and degree of urbanization) as the core independent variables, controlling other structural characteristics, host attributes and the distance decay effect.
4. They calculated the average marginal effect for each beach amenity according to equation (14). They find that tourists are willing to pay around €6.24 more per day for a percentage increase in beach length, €15.4 for the presence of vegetation, €41.8 for beaches with difficult access, €30.2 and €26.5 if the beach has a dune-type of mountain-type frontage over an urban one, *ceteris paribus*. At the same time, they find no significant capitalization effects regarding the type of sand, calm tide, beach width, or the presence of protected natural spaces.

3.6.3.2 Stated Preference Methods

Stated Preference (SP) methods are survey-based approaches to non-market valuation. A sample of the reference population is asked to complete a questionnaire where a good or service is offered in a hypothetical market, for example, the improvements in water quality or biodiversity associated with an intervention policy. Willingness-to-pay (WTP) for the good is then estimated using statistical procedures.

The main advantage of SP methods over Revealed Preference (RP) methods is that they allow the estimation of non-use and option values (see Figure 3.15). Their main disadvantage is hypothetical bias since respondents state their preferences in a hypothetical market (Boyle, 2017). For example, respondents may provide an inaccurate WTP of the hypothetical goods/services due to a lack of understanding or familiarity.

SP methods are categorized into Contingent Valuation (CV) and Discrete Choice Experiments (DCE). In CV surveys, people are asked to state their willingness to pay (WTP) or willingness to accept (WTA) for a hypothetical change in the provision of a good (Mitchell & Carson, 1989). In DCE surveys, respondents are asked to make choices over goods or policies, defined in terms of attributes and levels in hypothetical alternative scenarios (Henser et al., 2005). That is, CV considers the good (e.g., the non-market value of a beach) as a whole and DCE as a “bundle of attributes” (Riera et al., 2012).

Designing a stated preference survey is a long and complex process. The validity and reliability of results depend on how carefully the creation process is carried out. Several basic steps should be followed and several issues should be carefully considered (Riera et al., 2012; Johnston et al., 2017). The basic steps to follow in conducting a stated preference survey are the following (Boyle, 2017):

1. Identify the change(s) in quantity or quality to be valued, which implies identifying the current provision of the good (the status quo) and the future one. This information is included in the valuation scenario.
2. Identify whose values are to be estimated, i.e., the affected or reference population.
3. Select a data collection mode: Data can be collected by self-administered or interviewer-administered surveys. Self-administered surveys are less expensive than interviewer-administered surveys. However, they can increase self-selection bias since respondents can examine the survey materials to see if they are interested in the topic and then decide to participate. They require respondents to be competent readers of the language. In interviewer-administered surveys, the interviewer can provide clarification if the survey is complex, but responses can be influenced. Today's most used data collection modes are face-to-face, e-mail, or internet-based surveys. E-surveys are gaining in popularity, mainly when relying on survey companies that provide a representative sample of the reference population at a relatively low cost and in a short time. On the other hand, face-to-face surveys allow to administer more complex and longer questionnaires.
4. Choose a sample size: A sample size is a matter of choosing an acceptable level of precision of the estimates within a given budget. As a general indication, a sample of 300-500 respondents can be considered acceptable, but the size depends on the dimension of the reference population.
5. Design the information component of the survey instrument:
 - a. Describe the item to be valued: A neutral and fair description of the item being valued, the current or baseline conditions and the new conditions must be included. This information can be presented in written form and accompanied by visual stimuli (e.g., graphs or pictures). The information included must be tested in focus groups and pre-tests since it affects stated preferences and influences the validity and reliability of estimates.
 - b. Select a provisioning mechanism: It is necessary to inform respondents how the change to be valued will be implemented (e.g., establishing protected areas to conserve the biodiversity of a beach).
 - c. Select a payment vehicle: Examples of payment vehicles are an entry ticket to a beach, parking fees, taxes, higher prices for water or electricity bills or accommodation facilities and voluntary donations. The choice of a payment vehicle requires balancing realism against payment vehicle rejection. For example, an entry ticket can be employed for marine outdoor recreation.
 - d. Select a decision rule: An example of a decision rule is that the good will be provided if aggregate benefits (aggregate WTP) exceed the cost of the project or the policy.

- e. Select a timeframe of payment: one-time payment (lump-sum), annual payments over the lifetime of the project (e.g., 20 years), payments every time the good is used (entry ticket), or perpetuity, can be considered, depending on the good to be valued.
 - f. Include substitutes and budget constraint reminders: Respondents should be reminded about substitutes and that they could spend their money otherwise.
6. Design the valuation question.
- a. Select a response format. Response formats are different in CV and DCE. They will be discussed in the following paragraphs when illustrating specific factors for the two methods.
 - b. Allow for values of 0: Respondents must be allowed to state they are unwilling to pay for the good.
 - c. Address protest and other misleading responses: zero value can be a real zero or protest response. People may truly hold a zero value since they do not want the provided good and are not willing to pay for it or give a response of €0 because they reject some component of the valuation scenario. A follow-up question is typically included after the valuation question to examine the reason for a zero answer. Protest responses are dropped from the data since they are not true zero, while real zeros are maintained.
- Strategic behavior occurs when respondents do not answer the valuation questions truthfully because they think they can influence the survey outcome by answering differently (Mariel et al., 2021). Questions can be included to detect strategic behaviors, or, statistical routines can be used to identify responses that have an undue influence on results.
7. Develop auxiliary questions for statistical analyses of valuation responses: Demographic questions and attitudinal questions are included in the questionnaire to collect data to be used in the analysis of responses to the valuation question (i.e., explaining the heterogeneity of preferences) and to provide information to support decision-making.

When the objective is to explain how individual variables influence willingness to pay (the heterogeneity of preferences), behavioural or psychological variables can also be used. The emotions that individuals feel during the interview can, for example, influence their choices (Notaro et al., 2019; Notaro & Grilli, 2022), such as cognitive aspects (Grilli & Notaro, 2019) or value orientations (Grilli et al., 2018). Including emotions as a factor influencing willingness to pay allows to integrate the economic and socio-cultural valuations (see Guide 3).

This integration is done differently depending on the Stated Preference Method (Contingent Valuation vs. Discrete Choice Experiments).

In Contingent Valuation, socio-economic, behavioural, or psychological variables directly enter the valuation function and explain willingness to pay. In Discrete Choice Experiments, interactions between these variables and the attributes of the good being evaluated are included in the models. Taking as an example Börger et al., 2020 (see Specific Factors for Discrete Choice Experiments), the influence of the emotional connection to ecosystems on preferences for protecting seabed can be investigated by asking individuals (with a Likert scale) about their sense of ownership and interacting this variable with the attribute “Seabed condition” (area of seabed restored to a natural condition). When the variables used to explain the heterogeneity of preferences are endogenous, i.e., they depend on other variables of the model itself, it is preferable to use hybrid models, which consider the problem of endogeneity (Grilli et al., 2018). This is the case of sense of ownership that can depend on how many years an individual lives in a place.

8. Pre-test and implement the survey: Focus groups, cognitive interviews (one-on-one interviews) and pre-tests are used to test the questionnaire and avoid pitfalls such as incorrect interpretation of the information, unwanted clues to respondents and rejection of information. It is crucial to translate expert language into understandable and valuable information for respondents (Mariel et al., 2021).

Survey pre-testing is a fundamental prerequisite for a proper survey design (Mitchell & Carson, 1989; Johnston et al., 2017).

9. Analyze data: The process of data analysis varies with the question format used (see Annex 8 Stated Preference Methods).
10. Report study: The report should include all the details on how each previous step was addressed.

One further critical point to address is hypothetical bias, the difference between hypothetical and actual values. This bias often leads to overestimating WTP (Bishop & Boyle, 2017). Several protocols have been proposed to remove overestimation. Considering actual values, certainty scale calibration approach allows respondents to express his/her confidence or certainty about their WTP (Champ et al., 2009). Otherwise, statistical procedures are used, such as the lower-bound nonparametric estimator of WTP (Haab & McConnell, 1997). Cheap talk, consequentiality, real talk and opt-out reminder are the most ex-ante-used approaches to mitigate hypothetical bias. Cheap talk (Cummings & Taylor, 1999) is a script that aims at making respondents aware of the hypothetical bias and its consequences on estimated WTP. The consequentiality script (Carson & Groves, 2007) informs respondents that their answers can influence the policy being investigated. A follow-up question to investigate the perception of consequentiality is commonly used. Real talk (Alfnes et al., 2010; Moser et al., 2014) and opt-out reminder (Ladenburg & Olsen, 2014) have been used in DCE. Real talk informs respondents that after the hypothetical choices, they will face a real choice situation where they have to buy the good. The opt-out reminder is a script that, just before each single choice card, reminds respondents to choose the opt-out alternative (the status quo) if they find the hypothetical alternatives are too expensive.

The following paragraphs illustrate some specific factors that characterize each method.

(1) Specific Factors for Contingent Valuation

Different elicitation questions are used in CV surveys. They differ in their incentive properties, the information collected from respondents, the sample size needed and the econometric approach (Vossler & Holladay, 2018).

The open-ended question (Hammack & Brown, 1974) directly asks respondents how much they are willing to pay for the specified change without offering any cue (For instance: “What is the maximum amount of money you are willing to pay for a coral restoration project at Lanikai Beach, Oahu?”). This format is not considered incentive compatible⁶ since it incentivizes strategic behaviors and protest responses, mainly because it is cognitively burdensome for respondents. Open-ended questions can be used when respondents are familiar with the good under evaluation (e.g., fishing). In the bidding game question (Davis, 1963), respondents are asked if they are willing to pay a specific amount (starting bid). If respondents answer “yes”, the bid is increased until they say “no”; if they say “no”, it is decreased until they say “yes”. The main disadvantage of this question format is the anchoring to the first proposed bid (starting point bias). With the payment card question (Mitchell & Carson, 1981), a list of values is given and respondents circle the highest amount they would be willing to pay. This question format avoids the starting point bias, by deciding on the bids offered in the card, particularly the maximum bid (Notaro & De Salvo, 2010), is a potential bias. The dichotomous-choice question (Bishop & Heberlein, 1979) is considered the most incentive-compatible question format because it mimics real market transactions and is easy to answer. It is asked to respond “yes” or “no” to a specific amount offered. Nevertheless, it suffers from the so-called “yea-saying”: people tend to say “yes” to the offered bid amount. Since this format only provides information on whether the WTP of each individual is above or below a given amount, the double-bounded question (Hanemann et al., 1991) has been introduced. With the double-bounded question, respondents who say “yes” to the offered amount are asked a higher amount and those who say “no” a lower amount. With the multiple-bounded question format (Welsh & Poe, 1998), different bid amounts are presented to respondents and the level of certainty the respondents would be willing to pay is asked for each amount (definitely yes, probably yes, not sure, probably

⁶ A mechanism is called incentive-compatible (IC) if every participant can achieve the best outcome to themselves just by acting according to their true preferences (Roughgarden, 2010).

no, definitely no). For example, Ramajo-Hernandez and Saz-Salazar (2012) used a dichotomous-choice question format preceded by a 0-value screen question:

Question 1: Considering that (1) the implementation of the proposed change in water quality costs money, (2) you already pay towards some water improvements as a part of your water bill and (3) any additional money you would pay to improve water quality will not be available to you for other purchases, would you be willing to contribute financially to such a project by means of an increase in your water bill? Yes, No, Don't know.

Question 2 (asked only to those respondents that answered "yes" to the previous question): Considering your willingness to pay, would you pay per month an extra amount of €A in your current water bill in order to enjoy the proposed improvement in water quality? Yes, No, Don't know

A bid design is required for all the illustrated question formats except for open-ended. Focus groups and pre-test can be used to determine the bid amounts.

(2) Specific Factors for Discrete Choice Experiments

DCE allows estimating preferences and willingness to pay for each characteristic (attribute) of the good to be valued. With CV, we can estimate, for example, the overall non-market value of a beach, while with DCE, the value of every single characteristic of the beach, such as the landscape, the biodiversity, the recreational value and in general, the values of each attribute that a beach has. At the same time, we can evaluate the beach's overall value by summing up the value of each attribute and the result of every possible beach management policy by combining the values of different attributes.

The DCE questionnaire does not differ from the CV questionnaire, except for the ways stated preferences are asked. In DCE, respondents choose between alternatives (policy scenarios) shown in a sequence of choice cards. Each alternative presents different combinations of characteristics (attributes) of the good being evaluated with different levels of achievement. In the environmental field, these alternatives typically represent the environmental changes due to intervention policies. Figure 3.17 shows the choice card example taken from (Börger et al., 2020). In this choice card, there are three alternatives (options A, B and C).

	Option A (business as usual)	Option B	Option C
Seabed condition Area of seabed restored to a natural condition 	35% of area	50% of area	35% of area
Coastal communities Jobs in traditional industries lost in 10 years 	10% of jobs	25% of jobs	10% of jobs
Seabird populations in the plan area with healthy population 	66% of species	66% of species	75% of species
Additional tax For your household per year 	£0	£20	£100

Figure 3.17: An example of a choice card (Börger et al., 2020).

Respondents choose the preferred alternative (scenario) among those proposed, i.e., the one that maximizes their utility function. One of the scenarios typically represents the current situation, i.e., the status quo. However, it can also represent the business-as-usual situation, showing what will happen if the project or the policy is not implemented. The status quo is, therefore, generally presented with a zero-price level. In the choice card shown in Figure 3.17, option A is the business-as-usual situation.

A key issue is identifying attributes and levels for the choice cards. Attributes must inform respondents about environmental change and provide the decision-maker with useful information for adopting the most appropriate policy from a social perspective. It is essential that attributes and levels are easily understood by respondents, are realistic and are relevant to them. To identify attributes and levels, interviews with experts and focus groups with potential users of the good are carried out and the questionnaire is pre-tested. Deciding the number of attributes, levels, alternatives and choice cards is also important since as the number of these elements increases, the cognitive effort of respondents increases too. The choice card with three alternatives is the most used in empirical studies, while presenting 9 - 12 choice cards is considered optimal (Caussade et al., 2005). Scarpa et al. (2011) showed that the efficiency of the response increases gradually in the first 9-11 choice cards because of a learning effect and then declines quite rapidly due to the fatigue effect.

Levels of attributes are combined in the choice cards through an experimental design. Experimental designs are mainly divided into full factorial and fractional designs. Full factorial designs are orthogonal designs - a mathematical condition that requires that the attributes must be statistically independent - and show all possible combinations of attributes and levels. Fractional designs are generated with a subset of combinations. Fractional designs can be based on the principle of orthogonality or efficiency. Efficient designs allow statistically significant estimates to be obtained with a smaller sample size, thus decreasing research costs (Rose & Bliemer, 2009). To reduce the number of choice cards presented to respondents, the design can be divided into blocks, so that each respondent is only presented with a subsample of choice cards.

In Börger et al. (2020), there are four attributes, each with three levels. They use the software Ngene to generate a D-Efficient design that consists of 18 choice cards with two blocks. The attributes and their levels are shown in Table 3.3.

Table 3.3: Attributes and levels in Börger et al. (2020).

Attributes	Levels
Seabed condition	35%/50%/70% area of seabed in the East Marine Plan restored to a natural condition
Coastal communities	10%/25%/50% of jobs in traditional industries lost in 10 years
Seabird population	66%/40%/75% of species in the East Marine Plan area with a healthy population
Additional tax (household per year)	£0; £10; £20; £50; £75; £100; £150

Note: Business-as-usual levels in *italics*.

Different elicitation formats can be used. "Pick one" asks respondents to choose the preferred alternative, "rating" to assign a score to each alternative (usually from 0 to 10), "rank order" to order the alternatives, or choose the most preferred and the least preferred alternatives. This question format is called Best-Worst (Louviere et al., 2013). In Börger et al. (2020), each respondent is asked to complete nine choice cards and choose the most preferred alternative in each of the choice sets.

For a more comprehensive illustration of the implementation phases of a stated preference study and biases, refer to widely accepted guidelines (Riera et al., 2012; Johnston et al., 2017).

We use two examples to show the basic steps of conducting an SP survey, see Table 3.4. Specifically, Ramajo-Hernandez and Saz-Salazar (2012) will be used to illustrate how surveys are designed in a CV study. Börger et al. (2020) will be used to show the survey design in DCE methods. Ramajo-Hernandez and Saz-Salazar (2012) adopt the contingent valuation method to assess the non-market benefits of water quality improvements in the Guadiana River Basin (GRB) related to the application of the European Water

Framework Directive (WFD). Börger et al. (2020) use a discrete choice experiment to value a set of ecosystem services in the UK section of the North Sea, with a focus on offshore seabed habitats, regarding both use value and non-use values.

Table 3.4: Illustrating the basic steps of conducting SP studies with examples.

Steps in designing a stated preference survey	Contingent Valuation (CV) Ramajo-Hernandez and Saz-Salazar (2012)	Discrete Choice Experiments (DCE) Börger et al. (2020)
1. Identify the change(s) in quantity or quality to be valued	Water quality changes from the current state to the target, defined by chemical variables such as biological oxygen demand (BOD5) and the level of phosphorus (P).	The changes in three attributes (Seabed condition, Coastal communities, Seabird population, see Table 3.3). The changes are based on the Good Environmental Status, a management target under the Marine Strategy Framework Directive.
2. Identify the affected or reference population	The target population is those living around Guadiana River Basin (GRB) area.	English population
3. Select a data collection mode	Face-to-face interview	Online survey conducted by a market research company.
4. Choose a sample size	Based on population distribution, a stratified random sampling procedure was adopted. A total of 505 individuals living in 39 municipalities of the GRB area were included in this study, corresponding to 4.5% of the reference population.	Respondents were recruited from a pre-existing survey panel owned by Snap Surveys using quota sampling with the characteristics of gender, age group and region. In total, 1,212 completed questionnaires were received.
5. Design the information component of the survey instrument	Two types of visual aids are employed to make the CVM scenario more understandable. The first uses images describing the current state of water quality and the target to reach once the WFD (Water Framework Directive) measures will be adopted. The second uses a water quality ladder (defined in terms of its suitability for specific recreational activities, ranging from “bad” to “excellent”) and asks respondents to value the changes in water quality from current status to the target quality status, “very good”. The payment vehicle is an increase in the water bill, as they believe it is plausible and familiar to the population surveyed, thus adding credibility to the hypothetical market. The time frame of the payment is every time the water bill is paid.	The selection of attributes to be valued was based on a review of marine valuation studies in the UK and further refined in an international expert and stakeholder workshop. The development of attributes and their description, as well as of the entire questionnaire was done by an interdisciplinary research team and based on recent recommendations for stated preference methods. Focus groups were held in four cities (Plymouth, Birmingham, Hull and Ipswich) across England to check attributes and levels and the questionnaire content. Nine participants (selected to ensure a balance in terms of age, income and gender) attended each focus group. The payment vehicle is a tax. The tax has to be paid by all households in England every year for the next 5 years. The plan will be realized if willingness-to-pay will cover the cost of implementing it.
6. Design the valuation question	A dichotomous-choice question was used. Ten different bids were used, ranging from a minimum value of 0.5 to a maximum of 9.0 euros, based on the responses from a pilot study. They allowed for values of zero asking before the dichotomous-choice question if the respondents were willing to contribute financially to the project by increasing their water bill. To the respondents who stated they were not willing to pay, a follow-up question was asked to differentiate protest responses from true zero responses.	A pick-one question format was used. The business-as-usual options offered at a zero price allowed the respondents to state a zero value.
7. Develop auxiliary questions for statistical analyses of valuation responses	They also collect information on attitudinal questions (about the environment in general and specifically about water resources and their pollution) and socio-demographic questions.	They collect information on whether respondents had visited the EMP area and the types of activities they engaged in the area and on UK coasts more generally, a set of attitudinal and socio-demographic questions are also included.
8. Pre-test and implement the survey	The pre-testing procedure consists of two focus groups and a pilot study of 50 interviews, aiming to detect sources of bias and identify unclear wording.	A pilot survey with 92 participants was conducted.
9. Analyze data	See Annex 8 Stated Preference Methods	See Annex 8 Stated Preference Methods

3.6.3.3 Benefit transfer

Benefit transfer or, more generally, value transfer, is a set of techniques used to estimate the economic value of non-market ecosystem services at a target site (the “policy site”, or site i) when an original valuation study is not feasible due to financial or time constraints. In such situations, benefit transfer techniques can be applied to transfer welfare measures, such as willingness to pay (WTP) estimates, from original valuation studies (“primary studies”) previously performed at other sites (study sites).

Benefit transfer is based on desk analysis of primary studies, allowing the avoidance of field work, so its main advantages come in the form of time and budget saving. The main disadvantage, as expected, is the higher likelihood of valuation errors, due to the inherent uncertainty in dealing with the differences between the study and the policy sites. In an ideal scenario, the policy site i and the study site j are well characterized and are known to be very similar in type, quality, extension and condition, as well as socio-economic context, making the transfer of values a simple task. However, real scenarios hardly ever are so simple.

The increasing availability and quality of primary studies will certainly help in enhancing the information set that forms the source of data for benefit transfer, though there are still many biomes and ecosystem services for which the existing information is not enough (or good enough) for reliable analysis. Naturally, the quality of the output of benefit transfer techniques cannot be better than the quality of the input data, no matter the fanciness of the applied transfer techniques. Note that even in the cases of poor data quality it might be a good idea to perform a simple benefit transfer analysis, at least as a screening test to assess the need of collecting primary data or carrying out an original valuation study.

The different approaches to benefit transfer and their related theoretical background, are reported in Annex 9.

Primary data may be obtained from studies previously performed by the analyst, but usually, this hypothetical set is not wide or rich enough to do benefit transfer. If this is the case, some databases can be accessed to retrieve information from primary studies and run the benefit transfer technique. Two sources of information clearly stand out due to the breadth of studies they contain:

- The Environmental Valuation Reference Inventory (EVRI), developed by the Government of Canada and accessible at <https://evri.ca/>. It is a searchable compilation of summaries of environmental valuation studies, covering their most relevant features, including economic value. Data are not parametrized; hence the analyst is expected to create her own set of variables from the text summaries.
- The Ecosystem Services Valuation Database (ESVD), sponsored by the Foundation for Sustainable Development and accessible at <https://www.esvd.net/>. Unlike the former, the data at ESVD are parametrized, so it returns a downloadable (CSV) list of studies with their most relevant features as variables, including the economic value. In contrast, it does not provide non-parametrized information as text summaries.

A step-by-step application guide for the application of benefit transfer techniques could be as follows:

1. Make sure that benefit transfer is the appropriate method. This usually happens if the goal is to estimate the economic value of a non-market ecosystem service, it is not feasible to perform an original valuation study due to budget or time constraints and there exist robust primary valuation studies performed at similar study sites.
2. Select the primary valuation studies, on the basis of the applied valuation method (suitable to reflect exchange value), robustness of the estimation and biome similarity.

3. Choose the benefit transfer method. If a study site has been selected because of its close similarities to the target site, it will be possible to apply a unit value transfer method, either plain or adjusted for differences in context variables such as income. If the similarities between the study and the target sites are not so close, it will be needed to estimate a value function or to perform a meta-analysis using the data from the study sites.
4. Adjust the value, if needed, to consider special features at the study or target sites that might push values upwards or downwards within the confidence interval of the estimated value of the target site, in accordance with the insights of the corresponding biome and ecosystem service literature.
5. Where appropriate, scale the estimated value for the target site to its area and population, following returns to scale criteria and best practices established for welfare analysis.
6. Report method, sources, assumptions, identified limitations and confidence intervals along with the estimated target site value.

Here we use Brander et al. (2007) as an example to show the steps of conducting benefit transfer, as mentioned earlier in this section. They collect 166 coral-reef valuation studies (resulting in 100 separate value observations) and conduct a meta-analysis to study the value of coral reefs and its variations.

1. This study is the first meta-analysis on coral reefs. There are more than 100 studies on coral reef valuation in the literature and the results vary significantly with the valuation methods and regions. The 'flood of numbers' necessitates the application of research synthesis techniques and in particular meta-analysis, to assess the results of this literature as a whole and identify the key explanatory factors that determine coral reef value. This study explicitly investigates the validity and robustness of value transfers based on this meta-analysis of coral reef recreational values.
2. This study focuses on recreational values to allow the definition of a standardised value in terms of visitor days and to produce a data set with a manageable degree of heterogeneity. In total, 166 coral reef valuation studies were collected in this study, however, only 49 of them contain sufficient information (including coral reef value, services being valued, location, year of valuation and valuation method used) to conduct the meta-analysis.
3. This study conducts a meta-analysis to study the recreational value of coral reefs.
4. The focus of this study is not to estimate the value of a target site but rather to study the factors affecting the recreational value of corals. They employ an n-1 (or jack-knife) data-splitting technique to produce 72 separate value transfer functions to examine the accuracy of their meta-regression results.
5. They find that the value transfer tends to overestimate low observed values and underestimate high values. Based on their study, they find that using estimated meta-regression functions for site-specific value transfer is not (yet) a sufficiently accurate approach and more high-quality valuation studies on coral reefs are needed.
6. This study reports high average and median transfer errors for the whole sample (186% and 79%) compared to other value transfer exercises because of the lack of enough high-quality studies on valuing coral reefs. Based on the results of this analysis, they suggest to not rely on value transfer to estimate site-specific recreational values for coral reefs.

4. Guide 3: Cultural and Social Values

4.1 Introduction to Cultural and Social Values

Non-monetary social and cultural values encompass important individual and collective knowledge, feelings, beliefs, customs and attitudes in relation to ecosystems. They are difficult to capture, but highly significant for understanding how people perceive and benefit from ecosystems. They relate to how people derive meaning, pleasure, education and community from their environments. Salient non-monetary Cultural Ecosystem Services (CES) can produce a range of societal benefits including values such as local identity, aesthetic stimulation and inspiration, cultural heritage, emotional, spiritual and wellbeing connections to nature and environmental and indigenous knowledge. These societal benefits have influence at broad scales of culture and history, as well as at the smaller scales of people's daily lives and practices. As Fish et al. (2016) argue, "cultural ecosystem services are about understanding modalities of living that people participate in, that constitute and reflect the values and histories people share, the material and symbolic practices they engage in and the places they inhabit". Understanding CES is becoming increasingly important to account for societal benefits that cannot be monetised, or that are shared among stakeholders and constitute a public good (Bullock, 2020). For the purposes of this handbook, the term CES is used to represent the societal benefits which cultural ecosystem services deliver however it is suggested that the distinction between CES and societal benefits is less important for some socio-cultural valuation methods.

Under the MARBEFES framework (Figure 3.1), CES associated with 'places and seascapes' can deliver the following societal benefits: tourism and nature watching, spiritual and cultural well-being, aesthetic benefits, education and research and health benefits. Yet, it is important to remember that the identities, feelings and meanings we might associate with particular environments vary widely, change over time, tend to be context-dependent and are not easily mapped, expressed, or measured. It is helpful, therefore, to cultivate tolerance for ambiguity, incoherence, contradiction, heterogeneity and mutability in the data during CES valuation. Sometimes just describing, rather than defining or ordering a datapoint is the best practice, as it may represent a particularly inchoate, complex, or contested CES.

The research on social and cultural values will be framed through a particular focus on emotional connection to ecosystems. The emerging field of affective ecocriticism argues that "place profoundly shapes our emotional lives" (Bladow & Ladino 2018) and that emotions often provide a significant foundation for environmental action or enjoyment, or else despair or stress at increased environmental damage. There is increased interest amongst scholars and policymakers to better understand people's emotional connections to ecosystems and how this can influence behaviour (McKinley et al., 2021; Bladow and Ladino 2018; Gladwin 2018). The sea has long elicited allure, attachment, fascination, care and concern for human communities. Water in particular has a "sensual charisma" (Chen et al., 2013) that draws human interest. Socio-cultural valuation can be used to forge and leverage emotional connections to nature within policy and decision-making to stimulate behavioural change for the environment. For example, participation in art competitions, questionnaires, or focus groups may cultivate a sense of ownership of and responsibility for different ecosystems. At the same time, it is also important to understand negative or frightening emotions surrounding ecosystems and ecosystem change, such as eco-anxiety or eco-grief (Cunsolo & Ellis, 2018; Halstead et al., 2021). A recent study of ocean literacy undertaken within the UK found that 'concern' was the most frequently cited emotion by the public when thinking about the marine environment (Defra, 2021). Such negative emotions may influence the perception of CES or highlight perceived threats to CES. Understanding these negative emotions can also help to address prevailing instances of "emotional disconnect" (McKinley et al., 2021) between European citizens and aquatic ecosystems, whereby emotional connection to ecosystems still does not lead to positive behaviour change.

Emotional connections to nature are also deeply tied to particular places (Gladwin 2018), therefore they can be explored through participatory mapping under the concept of 'sense of place' (e.g. Burdon

et al., 2022). As geographer Yi-Fu Tuan (1977) articulates: “What begins as undifferentiated space becomes place as we get to know it and better endow it with value”. A ‘sense of place’ is an emotional connection to place, often construed as a ‘home’ and which often engenders deep feelings of happiness, security, protectiveness and responsibility. There is much evidence to suggest that ‘sense of place’ has a significant influence on environmental perceptions and behaviours, such as environmental stewardship or conservation (Masterson et al., 2017; Gladwin, 2018). Due to its influence on individual and collective affective bonds between people and ecosystems, ‘sense of place’ can be understood as a key cultural societal benefit in its own right (Ryfield et al., 2019). Yet, it also overlaps with societal benefits such as ‘local identity’ and ‘cultural heritage.’ ‘Sense of place’ is practice-based, in that it arises from material and regular interactions between people and ecosystems (Ryfield et al., 2019). Analysing ‘sense of place’ therefore involves understanding the links between emotions, time spent engaging with a particular place and a sense of belonging or home. We will use mapping to understand how emotional connections to ecosystems are location-specific and develop over time. We will also use qualitative research, such as interviewing and surveying, to facilitate deeper exploration of the bonds formed between people and places. It is likely that stronger emotions can emerge from direct access to and experience with local places, yet there is a need to understand and cultivate emotional attachments to more distant and global spaces to meet current environmental challenges (Alaimo, 2019; McKinley et al., 2021). For example, how can we understand and cultivate emotions towards farther reaches of the sea to include it in a sense of planetary home? We will strive for a more expansive notion of ‘home’ to account for emotional connections to sea spaces that are less frequently seen or used.

The framework developed for ‘sense of place’ research in Ryfield et al. (2019), shown in Figure 4.1 and that was adopted in MARBEFES, uses four key research methods for gathering qualitative data about non-monetary CES. This is developed in steps: 1) **observational and experiential study of place**, to build an initial log of evidence of cultural uses and activities, cultural heritage, local communities and interactions with nature; 2) **conducting interviews**, focus groups and transect walks/journeys with experts and communities-of-interest (including drawing on additional social science frameworks such as the 3D Wellbeing Methodology developed by Coulthard et al., 2015); 3) **conducting online and field surveys** of participants from sectors such as: the public, government, business and industry, academia, arts and culture and voluntary groups. This is to collect data about social and cultural values - data will include narrative and visual responses to survey questions. It will be informed by the baseline of stakeholder perceptions (gathered in Work Packages 1 and 4) conducting map-based surveys of cultural representations of place which identify social and cultural values with specific locations and place-based experiences.

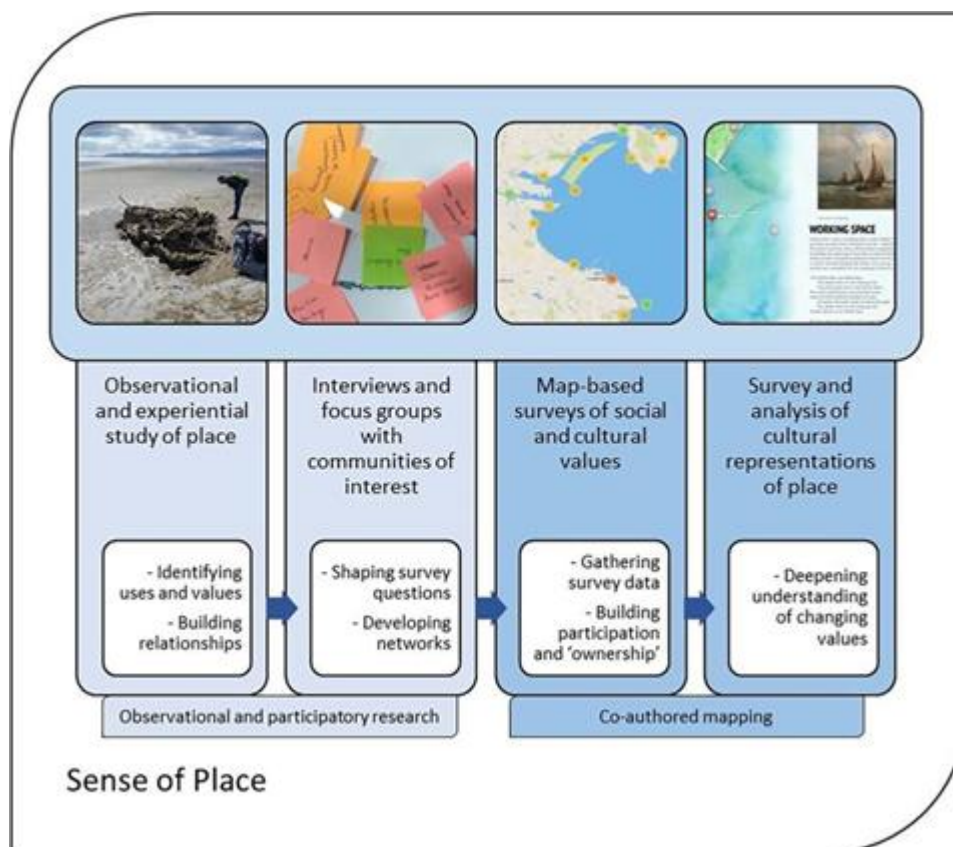


Figure 4.1: Methodology for investigating 'sense of place' in relation to CES (Ryfield et al., 2019).

As the MARBEFES project necessitates a common timescale for ecological, economic and socio-cultural valuation, our timescale is relatively short. Yet, this short timescale allows scope to focus more on density, diversity and richness of results across space within a BBT. Due to this, we will also add one more research method to our approach: 5) **arts-based creative workshops** to capture deep, intangible and complex emotional connections to ecosystems and to stimulate deliberative thinking around CES. As with the focus groups and surveys, these will be informed by the initial observational research and the stakeholder perceptions data from Work Package 1.

The following sections outline the five key research methods in more detail.

4.2 Socio-cultural valuation

4.2.1 Introduction

What is socio-cultural valuation?

Socio-cultural valuation is used to determine how and to what extent, environments are integral to the ways in which people derive benefits from social relationships and cultural activities and expressions. Social and cultural benefits are key determinants of the emotional and psychological well-being of individuals and groups. They include the processes that emerge from human thought, emotion and experience. Examples of how social and cultural benefits are derived from environments and ecosystems include sense of place, recreation, relaxation, local identity, artistic inspiration and sense of community. These benefits are difficult to capture with the quantitative (i.e., numerical) and empirical (i.e., verifiable) methods typically preferred in the Ecosystem Services field and Western policy contexts (Gould et al., 2019). Yet, they are highly significant for understanding how people perceive and benefit from ecosystems.

Under the Ecosystem Services framework, these types of socio-cultural benefits are called cultural ecosystem services and societal benefits. Fish et al. (2016) define societal benefits from cultural ecosystem service as *“the contributions ecosystems make to human well-being in terms of the identities they help frame, the experiences they help enable and the capabilities they help equip”*. They further explain that “cultural ecosystem services are about understanding modalities of living that people participate in, that constitute and reflect the values and histories people share, the material and symbolic practices they engage in and the places they inhabit”.

The MARBEFES framework (adapted from Turner et al., 2015; Burdon et al., 2022; Burdon et al., 2024) categorises cultural societal benefits into the following general types: tourism and nature watching, spiritual and cultural well-being, aesthetic benefits, education and research and health benefits (see the green boxes in Figure 3.1). These are overarching (and often overlapping) categories for the emotional, experiential and psychological benefits that people derive from environments. Such societal benefits relate to deep processes of meaning-making, community-building, sense of purpose and of pleasure. They have influence at broad scales of culture and history, as well as at the smaller scales of people’s daily lives and practices. Understanding societal benefits from cultural ecosystem services is becoming increasingly important to account for ecosystem benefits that cannot be monetised, or that are shared among stakeholders and constitute a public good (Bullock, 2020).

CES also encompass and illuminate two-way, reciprocal and interdependent relationships between people and nature (Buitendijk et al., 2024; McElwee et al., 2025). For example, sense of place is often enjoyed as an ongoing relationship, rather than a one-off benefit (Urquhart & Acott, 2013). People may value the quality of their relationship with a particular place—including the effort they put into the relationship, or the way a place can challenge them through weather, topography, or unpredictability. Benefits such as sense of place can also motivate people to care for and alter their environments over time, involving people themselves in the co-production of CES and associated societal benefits (Fish et al., 2016). Such a dynamic is difficult to fit into a linear model whereby ecosystem services and benefits flow in one direction from ecosystem structures, processes and functions. Instead, CES often indicate more cyclical, two-way human-nature relationships (Fish et al., 2016).

Socio-cultural valuation identifies and communicates cultural societal benefits to enhance policy and decision-making. Understanding cultural societal benefits can enhance social adoption of, and compliance with, environmental policies (Gould et al., 2019; McElwee et al., 2025). It can also make sure that environmental policies are inclusive of a diverse range of social groups and cultural needs. As the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al., 2018) argues, environmental assessment must be inclusive, comprehensive and incorporate a wide range of social interests and knowledge systems. Socio-cultural valuation furthers that goal.

4.2.2 The MARBEFES socio-cultural valuation methodology

The MARBEFES socio-cultural valuation methodology investigates three main categories in relation to human interactions with ecosystems: Sense of Place; Perceptions of Environmental Change and Environmental Responsibility.

Sense of place refers to the unique emotional and psychological connection that individuals or groups have to a specific environment or location. It elicits feelings of home and belonging, encompassing the memories, feelings and meanings attached to a location or environment. It arises from physical and cultural interactions with a place, often across generations. As such, the MARBEFES methodology emphasises understanding the links between emotions, time spent engaging with a particular place and a sense of belonging or home.

We are also interested in how participants perceive environmental change and how this perception informs their use and valuation of their environment. Perception of environmental change can affect how people feel attached to particular locations and also how they use and care for those locations.

It can also be a useful basis for planning management strategies and adaptation measures in local environments with stakeholders (Appadoo et al., 2023; Rogan et al., 2005; Wood et al., 2021). People may already be adapting their use and valuation of an environment if they perceive that environmental changes have resulted in greater risk to health or sustainability or if they perceive that the environment is under threat. There is some evidence to suggest that people are less likely to perceive environmental change in marine environments than terrestrial environments and that drivers of change in marine environments are less commonly understood (Ensor et al., 2018). Hence, the MARBEFES questionnaire invites participants to indicate how they understand the connection between environmental state and usage. The questionnaire might also invite participants to indicate their perception of possible future changes to understand whether anticipated changes affect usage and valuation.

Acts or expressions of care and responsibility for particular environments are good indicators of environmental awareness and socio-cultural value (Rayon-Viña et al., 2019). This type of interaction - such as litter-picking, avoiding vulnerable habitats, citizen science projects or planting pollinators - demonstrates understanding and care for the environment and contributes to environmental literacy (Kelly et al., 2020). It also indicates how participants regard the management of the environment and may inform the development of management strategies in relation to environmental care and responsibility.

To capture and analyse CES, MARBEFES used a mix of quantitative and qualitative methods to provide a holistic approach. We developed and applied **three overarching methods** which build on the foundational "sense of place" research developed by the *Cultural Value of Coastlines* project (Ryfield et al., 2019). These methods encompass:

- 1) observational study of place, to ground the research in particular places and capture information about human uses, groups, cultures, or conflicts that may not be readily available online;
- 2) participatory research, which includes methods such as interviews, focus groups, public, arts-based workshops and public questionnaires with experts and communities-of-interest. Participants come from various sectors such as the public, business, academia, government and voluntary groups. This also allows grounding the research in place and local knowledge. A key aim is to involve local stakeholders in the design and execution of the research and make sure it eventually benefits them and goes back into the local area;
- 3) gathering and analysing cultural representations of places. Cultural representations refer to creative media in any form, for example films, novels, plays, paintings, songs, or other forms. Cultural representations are rich archives of the social perceptions and feelings associated with particular places, especially over time and serve as evidence of the aesthetic inspiration and cultural heritage generated by ecosystems.

The participatory and cultural representation research require training in either social science or arts and humanities methods. It is important to note too that topics such as well-being, eco-anxiety and sense of place may be sensitive and should be treated as such. **Ethics approval** from your institution **will need to be gained** for this type of research. Also, should a research team lack the resources or expertise in this type of research, they could focus on just one or two of the methods or sub-methods which suit their capacities. The most desirable method to adopt would be the survey sub-method within the participatory research, which could stand alone or be combined with one or more of any of the other methods or sub-methods.

4.2.3 MARBEFES three key socio-cultural valuation research methods

4.2.3.1 Observational research

This phase of the research gathered information about how people interact with and experience their environment. Researchers started by walking around (or sailing or remote sensing) as much as possible

of the area to be assessed. This allows research teams to sense and observe the ways in which human communities make use of and benefit from local ecological conditions and services and to speak with the local people about the cultural activities they are engaged in. Researchers can go in teams and share perspectives and expertise while observing their surroundings.

Guidelines for observational research:

- Take notes or photographs to record experiences and observations. It can be helpful to describe observed interactions with landscapes or make a map of observed activities in particular locations.
- Focus on evidence to corroborate cultural benefits associated with particular places. For example, the aesthetic appreciation of the coast may be evident from curated beaches, wildlife presence, water bodies, caravan parks and shops selling ice-cream.
- Consider less direct uses of landscape, such as evidence that people may associate a beach with memories of learning to drive or family outings.
- Look out for evidence of how the community uses and cares for landscapes, such as clean-up or conservation groups and citizen science initiatives and it may be useful for identifying community contacts who may be useful to link in with later.
- Watch for evidence of hazards, abandonment, or a lack of human interaction to indicate negative associations with ecosystems.
- Look out for conflicting uses of landscape by different groups (e.g., birdwatchers may resent dogwalkers whose dogs when unleashed can disrupt key nesting sites). These may be helpful for understanding potential conflicts as well as how people value their environment.
- Look out for co-benefitting uses of landscape between groups (e.g., nature reserves creating more scenic and varied surroundings for swimmers).

These observational methods are not comprehensive and should be combined with other methods. They are also contingent on season, time of day, weather conditions and other factors. Yet, they are helpful for team building and building knowledge of local values, potential conflicts and possible networks of community groups to engage with.

4.2.3.2 Participatory Research

This phase of the research involves identifying local networks, experts or stakeholders to engage and invite into the research process. It should be commenced early to allow stakeholder participation in research design. It will use the preliminary information from stakeholder engagement and it can engage the same or different stakeholders (depending on research objectives, or concerns about stakeholder fatigue). Key stages may include:

- Interviews with local environmental managers, network leaders, heritage professionals, or council officials. Open interviews are more likely to address information gaps, but structured interviews with the use of maps, pictures, or situational questions may help to link particular societal benefits with particular places, as well as to explore particular CES in more depth.
- Advisory and focus groups. These may target individual sectors, or bring together different sectors (business, academic, governance, heritage and so forth). Use any knowledge of possible conflicts relating to ecosystem management or perceptions to make the meeting balanced (in terms of invited participants). Locate these meetings in a neutral setting. The Open Space method (Owen, 2008) may be useful for shaping the meetings. Open Space proposes one central question for in-depth discussion over the course of 2-3 hours and aims to produce a summary book that serves as a record of the discussion for participants, researchers and policymakers. It is useful because it can be deployed with small or large

groups, held in-person or online and it stimulates ownership and in-depth exploration of a research question amongst participants in a non-hierarchical fashion.

Suggested structure for advisory meetings in MARBEFES:

- Facilitator welcomes participants, introduces the structure for the day and guidelines for participation. They then propose one framing question, e.g. “How is this place important to you?”
- Participants propose subtopics for discussion and break into mini-discussion groups. Facilitators walk around as aids, but the discussions are set and directed by participants. Participants may also be responsible for note-taking or recording the sessions. The session ends after the mini-discussions come to a close (according to a set time, or when the participants collectively decide). Facilitators gather the notes, thank everyone and outline next steps. Facilitators draft a version of the discussion summary book as quickly as possible and distribute to the participants after the event for their input and to check whether they wish to add or amend anything.
- After seeking the participant inputs, facilitators work with a designer to publish a final version of the book. This book is then distributed (likely in electronic version) to the participants as a testament to their ideas and engagement. The book can be used for further research, community engagement, or policymaking purposes.
- **Creative and arts-based workshops.** These use techniques of drawing, writing, sculpting, craft-making or other artistic methods to capture and validate people’s emotions towards ecosystems. Following a practice-based method that utilises “writing as discovery” (Cook, 2013), we posit that the arts can allow for uninhibited, joyful, exploratory, emotional and personal articulations of CES. They can also help to forge familiarity with and emotional connections to ecosystems, engendering feelings of ownership and belonging (Bieling, 2014).

Guidelines for workshops (Cook, 2013):

- Begin with an introduction to the project and its key questions, aims and areas of assessment.
- Then, participants can be provided with guidelines and prompts to create their own artistic productions in response to research questions.
- The aim is not to create a highly accomplished or “good” piece of art, but rather to explore feelings and perceptions. Works may be speculative, exploring desired resource management strategies.
- Should participants agree, works may be displayed afterward in education centres, community halls or online to engender pride of place and community expression. Such outcomes should be planned and clearly communicated to participants from the beginning.
- **Public survey.** This should be the principal method for collecting information about CES. Surveys should be map-based, to enable participants to link the cultural societal benefits they identify to particular places. The survey should be open to any member of the public; while it is likely that only people with personal or local ties to the case study sites will uptake, it would also be welcome to gain responses from more geographically-removed participants as a comparison point.

Guidelines for survey creation (Brannigan et al., 2019a):

- Start with place of residence, then provide a questionnaire to tick activities most associated with area being assessed (informed by observational work and focus groups/interviews).

- Ask participants how they feel about particular places and why. Use open questions to elicit detail.
- Ask participants to consider questions of (i) proximity, (ii) access, (iii) frequency of visitation to their feelings about particular societal benefits.
- Asks participants whether they are aware who makes decisions about health and sustainability of their environment. Do they take part in decision-making, or take steps to care for their environment?
- End with some basic demographic information (age, gender etc.) to identify trends in survey uptake and results.

Conduct the survey on paper in target locations and online for a wider reach. Ask focus group participants to help with the distribution of the survey for wider uptake. One may use map-based platforms such as www.ushahidi.com. One can complete face-to-face surveys with demographic groups that you notice are not engaging with your survey.

You should prioritise getting a high volume of participation (e.g., at least 150 participants), for results to be representative and convincing. Also strive for results to be representative of a significant sample of users, visitors, or residents. Indicate where participants can see survey results so they can benefit from the experience and learn how other people value their environment. The survey can be designed to foster a sense of ownership of places, perhaps by inviting participants to share photographs, feelings, memories and stories.

4.2.3.3 Cultural Representations Analysis

This phase of the research involves assessment of cultural representations connected to the areas of interest, which include works of literature, theatre, painting, sculpture, music, photography and film. Environmental humanities frameworks can be deployed here to investigate how cultural representations provide a means to understand, critique, generate and map complex and wide-ranging cultural services, benefits, emotions, perceptions and motivations surrounding ecosystems (Gladwin, 2018; Brannigan, 2015). Gladwin (2018) suggests that cultural representations “appeal to how society uses narratives, images and metaphors as a way to frame values and emotions, which have the effect of reproducing social consciousness about environmental issues”. As this quote suggests, cultural representations can be a way to both understand and produce CES. A recent marine example is the Horizon Europe GES4SEAS Marine Art challenge which celebrates urban art inspired by the ocean ([MarinART Challenge: Urban Art for Europe’s Seas - GES4SEAS](https://www.ges4seas.eu/marinart-challenge)).

Cultural representations are typically highly crafted as works of aesthetic merit, often take a long time to produce and consume and thus may be difficult to connect directly to ecosystem services within short time frames (a novel may respond to ecosystems from ten years prior to its publication, for example). Thus, cultural representations will primarily be taken to provide evidence of what landscape features or cultural activities have been valued over **longer periods of time**.

Guidelines for cultural representations analysis in MARBEFES:

- Researchers start by **gathering, logging and analysing** a wide range and high quantity of cultural representations of particular places within the BBTs. Use databases such as: www.archive.org, Google Books, Google Images, Bing Images, IMDB, Amazon, EBSCO, LION, JSTOR, www.allmusic.com. Use place names, species names, or habitat features in searches. Consult local and national library catalogues and search engines.
- Focus on works that make **substantial reference to**, or depiction of, a particular place and **the extent to which** they associate that place with particular traits, values, feelings.
- Seek to **go beyond well-known works** and look for more obscure works to identify whether particular ecosystem features or cultural societal benefits have attracted more

attention than others. Consider less obvious genres, such as postcards or internet memes.

- If there is a paucity of cultural representations about your particular place, you could **invite people to submit stories, pictures, or films**. These can also be published on the project website.

Quantify and map the number of cultural representations of particular locations within the area being assessed. Also map the cultural values or activities associated with a place. These will reveal ‘hotspots’ which may have attracted more benefits for particular places (and may be of interest to policymakers). As well as quantitative information, cultural texts can provide qualitative information on the meanings, values and associations of particular places. Some places may be repeatedly associated with tranquility, for example, or may be iconic. These iconic places can be helpful to mobilise people for action. It is more effective to mobilise people through stories and images than scientific facts; iconic places, images, or narratives can be disseminated through project communications or community networks to inform, engage and inspire people.

4.2.3.4 Analyzing the socio-cultural data

Socio-economic valuation can display the elusive yet potent socio-cultural influences, phenomena and benefits which flow to and from particular ecosystems. Efforts should be made to make the results legible to other disciplines and non-specialist audiences and comparable to other ecological, monetary and non-monetary studies. Yet, it is important not to be too forceful when defining, categorising, or ordering CES, as this may misrepresent the data. Complex phenomena such as sense of place and identity may be difficult to divide into clear “benefits” separate from two-way relationships and material interactions with nature (McElwee et al., 2025). Moreover, we can be open to the possibility that not all CES can or should be fully captured, measured or quantified. Nonetheless, the data we collect can inform policy and decision making by highlighting important cultural benefits to protect and enhance, as well as important socio-cultural values to consider and mobilise for positive environmental action. The data can also have public benefit by fostering emotional connections to coastal and aquatic ecosystems and empowering people to understand, interpret and take responsibility for treasured ecosystems on local, national and global scales.

Observational findings

The findings from the observational research should be discussed and analysed amongst the researchers. The data should be treated **only as indicators** of CES and benefits/disbenefits that may prevail in a particular area. These findings can be used to direct lines of enquiry and approaches in other research methods, e.g., indicating influential groups or stakeholders to engage, informing survey questions, or informing search terms in the cultural representations research.

Participatory and cultural representations findings

The participatory and cultural representations research will yield both quantitative data—about the spatial distribution and locations of CES—and qualitative data—about the feelings and perceptions of people towards CES. The quantitative and qualitative data will need to be analysed separately first and later integrated (see Annex 11 for methods of analysing and integrating the data).

Interpreting the results

Results should be interpreted in an iterative fashion, with the researcher(s) revisiting and re-collecting data as frequently as possible. Researchers should seek to capture as wide a range of CES as feasible, allowing for the possibility that ambiguous, conflicting or anomalous CES may arise. The aim is to identify locations where CES are particularly evident and enjoyed (or suffered) and to generate insights about the depth, range, variety and quality of CES in those locations. As mentioned, researchers will be guided by research objectives concerning emotional connections to ecosystems and sense of place. As such, there will be an emphasis on understanding the variety and depth of emotions attached to

CES in particular places and the effect of variables such as direct use, access, familiarity, shared experiences and cultural assumptions in conditioning those emotions. Researchers can be open to a variety of such variables that may emerge from the data.

Culture is a broad category and it is likely that results will be wide-ranging and complex. For example, there may be confusion between whether culture represents a 'benefit' provided by an ecosystem (such as sense of place, wellbeing, or cultural heritage), or whether culture represents a set of individual or collective attitudes towards various ecosystems (such as a sense of responsibility, indifference, or pragmatism). The latter does not relate exclusively to cultural ecosystem services but can influence perceptions of and interactions with all types of ecosystem services (Scholte et al., 2015). Data about attitudes towards ecosystems more generally can be useful for decision makers and policymakers seeking to understand what groups and cultural values to accommodate and engage in a particular area (alongside trying to understand which cultural benefits to protect and enhance through policy).

Another thing to note is that such seemingly 'immaterial' phenomena such as emotions of fear, love, or disgust are also material phenomena in the way they can arise from material human interactions with environments (processes of commodification or extraction, for example) and influence ongoing material human interactions with environments (Brannigan et al., 2019b; Scholte et al., 2015; Tuan 1974). For example, people may be resistant to change in an ecosystem and mobilise against such phenomena as tourism development; such cultural movements would be indelibly tied to the material human uses of the ecosystem such as the tourism development, or even prior alterations to the ecosystem (Scholte et al., 2015, 69). It can be useful to link particular socio-cultural values to particular ecosystems and ecosystem characteristics where possible and to account for changes to these ecosystems and cultural values, to show the inseparability of the material and immaterial.

Disseminating the results

The results should be made as accessible as possible to the identified stakeholders and participants (e.g., members of the public, government officials, voluntary groups). The findings of your research will provide a good basis for engaging the community, policymakers and other researchers about key issues raised and the measures which might help to address them. It is important to set and manage participants' expectations about project outcomes from the beginning. As engagement with local people is a key element of CES research, the project team should already have good networks and contacts through which to disseminate the research findings and promote solutions and plans for management.

For each method of data collection deployed—observational study, local advisory groups, participatory survey, cultural representations and arts-based workshops—make a table listing the cultural societal benefits provided and the evidence (both quantitative and qualitative) of those benefits. From those tables, you can generate graphic representations of your results (such as bar charts, pie charts, treemaps, or scatter graphs), which are useful for communicating with others in the MARBEFES project (see Figure 4.2 using an example from the Cultural Value of Coastlines Project).

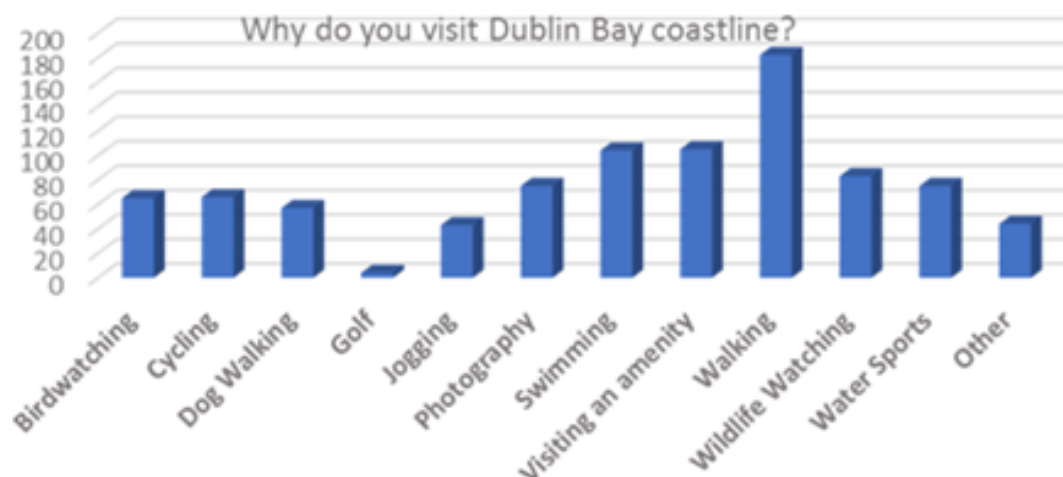


Figure 4.2: Bar chart demonstrating popularity of different CES benefits in Dublin Bay coastline from 2016-18 (Brannigan et al., 2019a).

The more detailed qualitative data collected from the interviews, focus groups, surveys, cultural representations and workshops can be used to ascertain connections between CES and emotions, or how people perceive the relationship between certain cultural societal benefits and ecosystem change. For example, they may show how the pollution of areas incite feelings of disgust, anger, or concern. Policymakers can use this data to identify, for example, which ecosystem changes have the most impact on people, or which species, places, or benefits are of particular interest and why. These qualitative data can also reveal perceptions on managing and sustaining cultural ecosystems, informing strategies that are in line with societal benefits and values.

You may also map results using GIS techniques to show different trends and metrics. In the map from the Cultural Value of Coastlines project in Figure 4.3, for example, a simple count of an approximate number of paintings and literary works that have been set in locations in Dublin Bay (circa 1750-2018) reveals which parts of the area have been valued most consistently by artists and writers. Overlaying the maps from different sources of data can reveal if there are strong and definite patterns of some places being valued more than others and for what reasons. Story maps, which are interactive maps that can show narrative text and images connected to particular places, can also be a useful way of communicating the societal benefits from CES and emotions attached to particular places (see Figure 4.4). Interesting images, quotes, or findings from the cultural representations analysis can be incorporated into the story maps and contribute to cultural heritage engagement amongst the public.

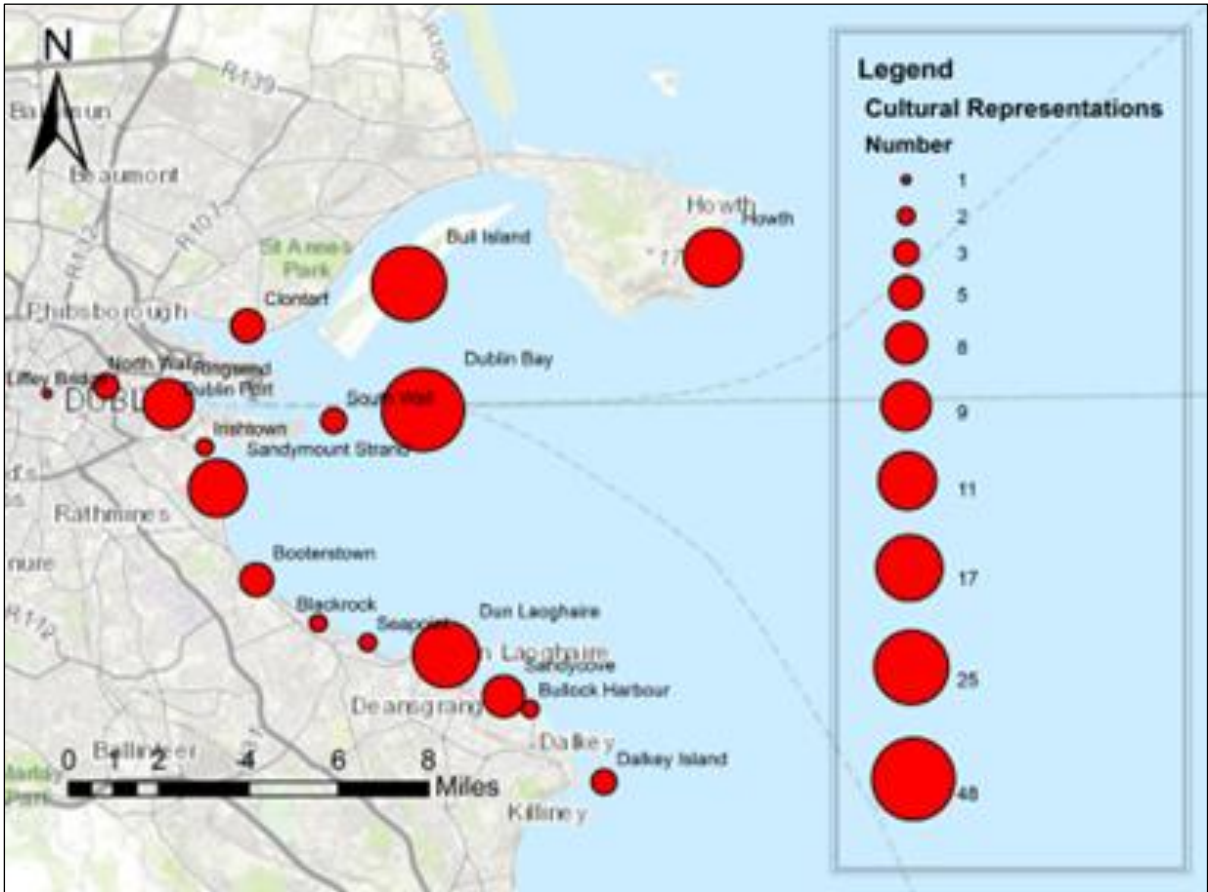


Figure 4.3: Heat map showing quantity of cultural representations of places in Dublin Bay circa 1750-2018 (Brannigan et al., 2019a).

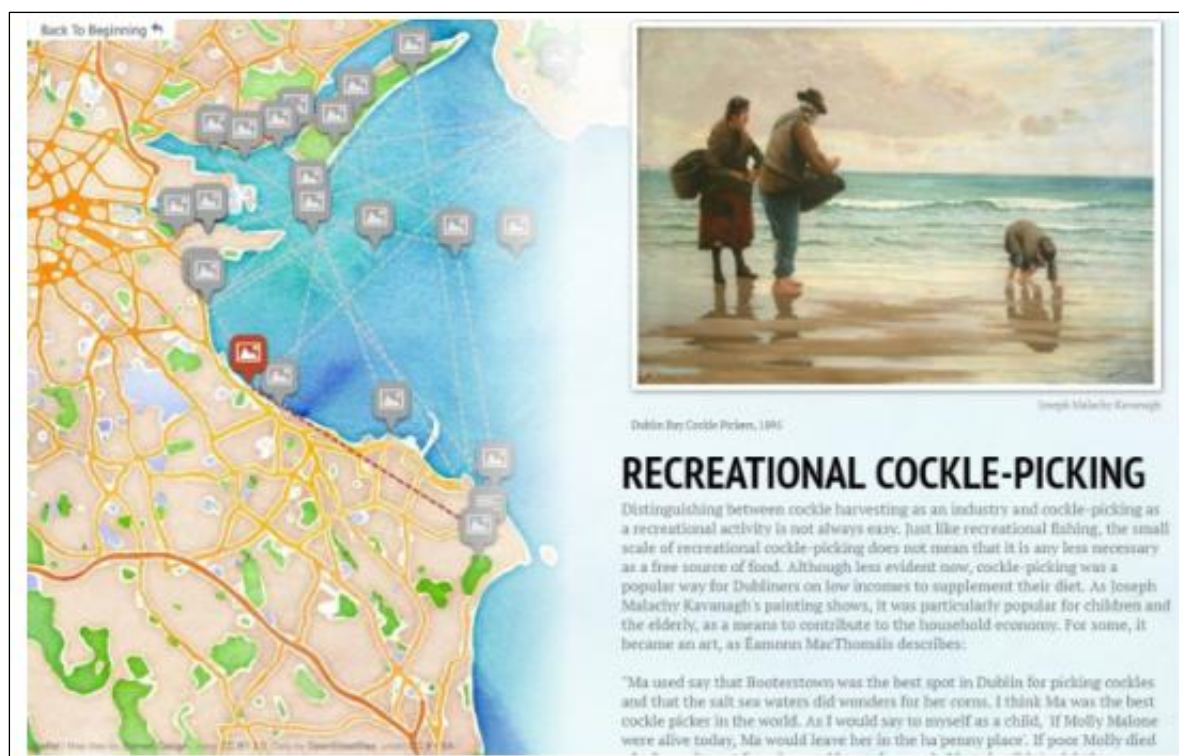


Figure 4.4: Image from story map of Dublin Bay (www.culturalvalueofcoastlines.com).

As socio-cultural valuation relies heavily on knowledge generated through expert and non-expert place-based interactions, research outputs should also be put back into the places that generated them. To achieve this, researchers may consider 'Town-hall' meetings, exhibitions, place-based readings or screenings of cultural representations, a project report launch, or a festival event as possible ways of engaging the community in a discussion and making the research accessible to them. These events could also invite community-led conversations on priorities for future action. How might communities mobilise strong emotions and place-attachments to mitigate risks to cultural ecosystem services? What policies or resource would communities like to see to support the protection and enhancement of cultural ecosystem services? While the MARBEFES researchers may not be able to answer these questions, the socio-cultural valuation research should provide foundations to empower and inform communities for future action. Strive to make the research accessible to all members of the community on a long-term basis so that they may use it to advance community goals.

5. Integration and Method Application

5.1 Changes in Ecosystem Services: Complementary Accounting Network (CAN) Approach

In their historical evolution, national accounts have always included a degree of pragmatism. Therefore, it can be argued that what is included or excluded from the accounts is open to debate (Lepenies, 2016). On the basis of this line of reasoning, we propose that the on-going efforts to expand the System of Environmental-Economic Accounting Ecosystem Accounting (SEEA-EA) accounting framework largely within the protocol of the Standard National Account (SNA) and which has produced some significant advances, should be continued over the medium to long term - to further refine methodologies, improve data availability and ensure broader adoption across different national contexts.

However, the constraints imposed by the need to present data in monetary exchange value terms will continue to exclude the true economic value of some ecosystem services and the holistic role played by biodiversity in the sustainable supply of services from the accounts. Therefore, we argue that there is a present and urgent requirement (given the state of the natural world) for a more comprehensive set of ecosystem/biodiversity change indicators. Rather than seeking a full integration of environmental data into the SNA framework, we argue pragmatically, for the establishment of protocols as a Complementary Accounting Network (CAN) - a pluralistic, multi-dimensional framework designed to capture the full range of values that society derives from nature expressed through monetary (exchange and welfare) and non-monetary terms - to sit alongside the economic accounts and on the same timescale (Turner et al., 2019). The complementary network needs to be populated by a range of data/indicators, building on the thematic accounts work supported by the SEEA-EA (2021). This path offers an immediate way forward, rather than trying to adjust the measures of production, consumption, income and the value of assets in the SNA to fully reflect biodiversity losses or gains. Some pilot work with the CAN framework has been undertaken using data for the East Anglian region in the UK. Some recreation accounts and a carbon account have been constructed and set alongside a GDP (gross domestic product) proxy indicator GVA (gross value added) (Badura et al., 2020). Figure 5.1 below illustrates the joint role of SEEA-EA and a CAN.

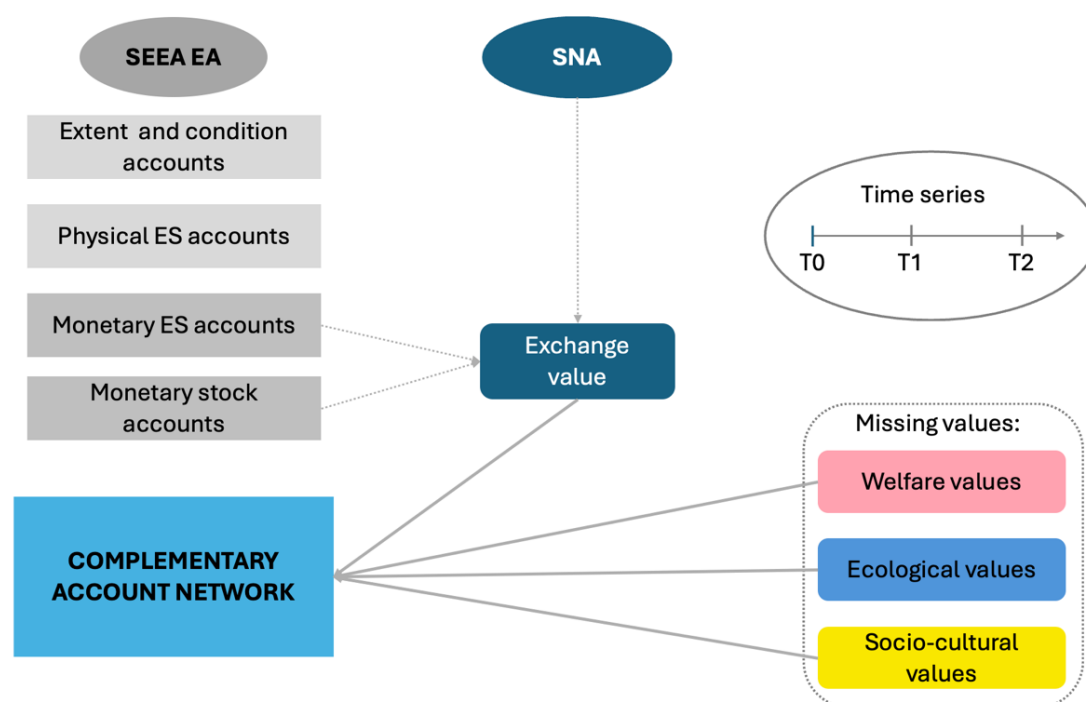


Figure 5.1: Complementary Account Network (CAN) setting.

A CAN should contain several relevant nodes, each of which encompasses a set of indicators and metrics, see Figure 5.2 below. Individual but networked nodes could contain the SNA/SEEA-EA account; an adjusted SEEA-EA account including shadow price values (i.e., stated preference values) for those ecosystem services outside of the exchange value calculus (see Section 4.2); further nodes containing, biodiversity indicators; ecosystem resilience indicators; naturalness indicators; and sustainable ecosystem functioning indicators; and cultural ecosystem services indicators can in principle be added, depending on data availability.

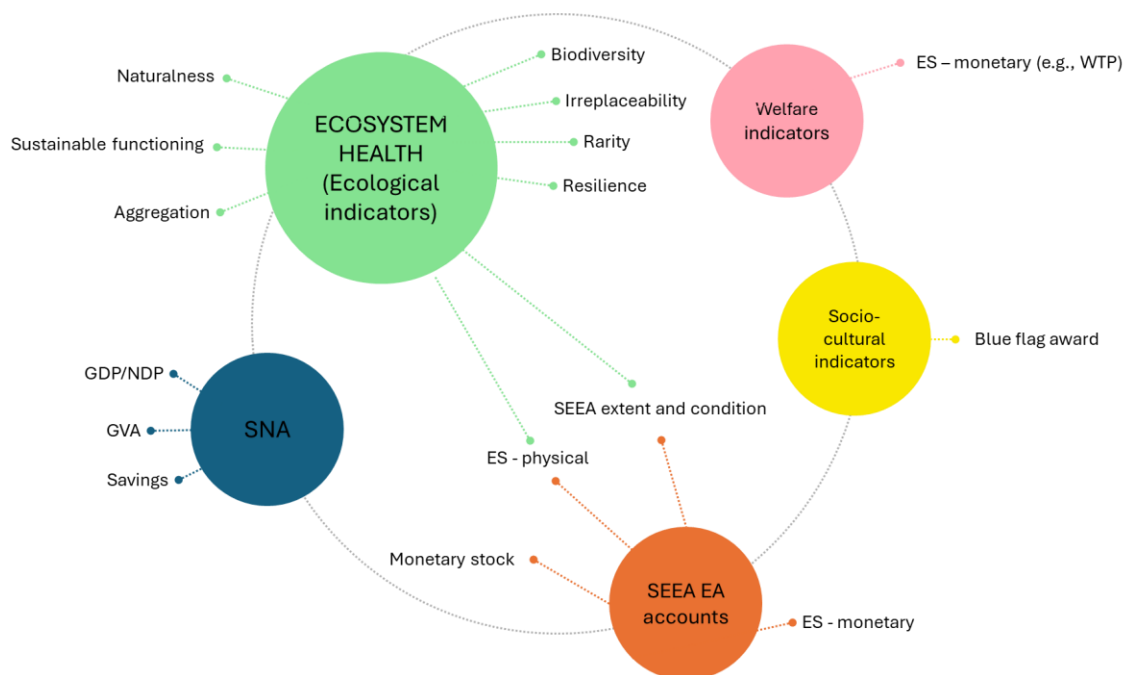


Figure 5.2: Complementary Accounting Network (CAN) and information nodes. The network structure shows that certain indicators provide inputs to multiple information nodes, thereby creating interconnections between them (for example, ecological indicators supplying information for the extent and condition components of SEEA-EA).

Within the nodes the indicator ‘accounts’ would need to be standardised in terms of timescale and all have an ‘opening point’ condition/extent and a ‘closing point’ condition/extent linked by time series data. The network concept serves to highlight the links between biodiversity and other ecosystem health characteristics, as well as to socio-cultural assets and services. Luisetti and Schratzberger (2022), for example, have proposed five sets of biodiversity related indicators for the marine environment covering physical condition (state index), composition (state index), structure (state index), function (state index) and seascape index.

Taking the compositional state index as an example, data requirements pose several challenges, including the need for representative data across ecosystems, space, time and species taxonomy. If direct observations on species are not available an alternative approach is needed. Ferrier (2011) has proposed that observations based on changes in the spatial extent and configuration of habitat by species can meet this need. Further research is needed to explore the links between thematic biodiversity accounts (for measuring and valuing biodiversity as an asset) and ecosystem services (King et al., 2021). But there is currently a tension between the Convention on Biological Diversity (CBD) and SEEA-EA over how biodiversity is reported. For the former, ecosystem diversity is a subset of biological diversity, while in the latter biodiversity accounting is a subset of ecosystem accounting. In the SEEA, the ecosystem asset accounts do not link directly to an ‘explicit’ ecosystem diversity account or fully to condition and services supply. Whether or not biodiversity is an asset in its own right has been debated (Mace et al., 2015), but we argue that, it is, as it represents an emergent property of a set of ecosystem assets that interact and support multiple ecosystem processes. These processes in turn support the resilience of ecosystem service delivery at larger scales (Folke et al., 2004; Oliver et al., 2016). Research efforts need to highlight the dependencies and trade-offs between the use of

ecosystem services and the depreciation of the biodiversity asset base or integrate such depreciation into service values. The motivation behind this approach can be anchored as much to instrumental utilitarian concerns (i.e., long term maintenance of ecosystem service flows – option and insurance values) as to an intrinsic/existence value perspective (King et al., 2021).

In terms of the compositional state of a marine ecosystem for example, it can be argued that improved measurement of ‘species diversity’ would provide a key indicator to characterise the condition of ecosystem assets. The species accounts could also provide information at landscape scales for different countries, if they reflected diversity in species assemblages between ecosystem assets (King et al., 2021). However, currently the SEEA species abundance accounts do not adequately reflect variability in distribution and abundance. King et al. (2021) present some case studies from across the world to demonstrate how work on species accounts could proceed. At the same time, the SEEA-EA guidelines (Section 6.3.4) recognize the possibility to record in the Ecosystem and Species Appreciation (ESA) flows which record data on the non-use values that people hold for ecosystems and species such as absence, abundance or diversity of ecosystems and species. These flows are not ecosystem services but signpost the health and ability of ecosystems to deliver services and preserve the natural functioning and therefore link with ecological value (see Section 2).

The CAN network and nodes represent a systematic approach to record ESA flows via multiple indicator data sets for the measurement of different aspects of ‘ecosystem health’ i.e., resilience, naturalness, sustainable functioning and biodiversity. An additional node is also needed to incorporate ‘societal wellbeing’ proxied by cultural ecosystem services which has been proved difficult to be meaningfully monetized with exchange or welfare prices.

Finally, making the link between SEEA-EA and other auxiliary accounts, such as CAN, to decision support systems should also be a key requirement. The accounts would provide a key tool in scoping baselines and for the monitoring of environmental change and its consequences (Admiral et al., 2013; Turner, 2016). The CAN, while not providing a fully integrated array of information, might still encourage greater ‘buy-in’ from the competing interests/stakeholders that typically arise in complex environmental change contexts.

In the early stages of a CAN construction process it is necessary to work with available data and accept that data gaps will inhibit the development of a holistic dashboard in the first phase. The accounting protocol requires time series data over a consistent time horizon. But where it is not possible to gain complete coverage with the available data, incomplete data series should at least fall between the start and end point of the chosen overall period. It may then be possible to interpolate to get a longer representation of the indicator.

In the next section we provide a simplified arrangement of CAN indicators using a four - cell matrix, see Figure 5.3 below.

	MONETARY		
1	SNA (national/regional) • GDP • Blue economy indicators	EXTENDED SNA/ SEEA-EA (national/regional) • SEEA EA/Monetary Accounts • Genuine Savings	2
4	CULTURAL INDICATORS (national/regional) • Blue flag award	ECOLOGICAL INDICATORS (national/regional) • SEEA EA/Physical Accounts • Ecological Valuation Assessment	3
	NON-MONETARY		

Figure 5.3: CAN matrix.

In the first cell (upper left) we report the SNA compliant indicators: Gross Domestic Product (GDP) over time at the national and regional scale. These indicators are in monetized units. In the second cell (upper right) we present Extended SNA indicators still in monetized units. This cell includes a mix of aggregate indicators e.g., Genuine Savings (GS) and Expanded Genuine Saving (Di Gennaro et al., 2025) as well as SEEA-EA. The GS indicator is a weak sustainability indicator which revised the SNA indicator to include other stocks of natural and social capital. In our example, we include two water-related indicators (flood control and water purification) which we consider can be related to the marine ecological quality context (Grizzetti et al., 2021). Subsequently, the GS was further expanded to proxy societal wellbeing with a poverty measure, pushing the GS towards a stronger sustainability measurement. Furthermore, using Bartolini et al. (2024) and revising and downscaling the work by Addamo et al. (2025), four marine ecosystem services primarily related to the *Posidonia oceanica* were monetized and included as SEEA initial ecosystem services measures. These ecosystem services refer to fish provisioning, carbon sequestration, daily recreation and marine-based tourism. The third cell (bottom right) contains biophysical change indicators (non-monetary) covering stock extent and quality changes for a range of marine ecosystems. The fourth cell (bottom left) contains cultural ecosystem service change indicators (non-monetary). A preliminary approach to CAN for Sardinia, Italy is presented in Box 5.1 below (Turner et al., 2026).

Box 5.1: A preliminary approach to CAN: Italy, Sardinia

Policy question: Given economic growth, how can ecosystem and biodiversity changes over time be more effectively monitored and communicated to support sustainable economic policy decisions?

Methodology: This preliminary study focuses on using the Complementary Accounts Network (CAN) approach as integrative tool to support decision-making at national (Italy), regional (Sardinia) and local (Sardinia BBT) levels. It aims to develop a holistic indicator dashboard designed to provide key information in a clear and visually engaging manner, facilitating informed decision-making.

Results: Figure below reports the main findings of CAN preliminary application.

MONETARY	
SNA (Italy, Sardinia, Sardinia BBT) A similar pathway – increased GDP over time - for both regional and national economies	EXTENDED SNA/ SEEA-EA (Italy, Sardinia, Sardinia BBT) - At the national level, GS and deontological GS rose steadily, whereas they declined regionally in Sardinia. - Ecosystem service values (e.g., recreation) showed stagnation or decline, particularly in Sardinia and BBT.
CULTURAL INDICATORS (Italy, Sardinia, Sardinia BBT) Italy and Sardinia showed increases in Blue Flag certifications, but BBT remained static, highlighting a lack of local progress or engagement.	ECOLOGICAL INDICATORS (Italy, Sardinia, Sardinia BBT) <i>Posidonia oceanica</i> density declined significantly, especially in Sardinia. <i>Loggerhead turtle</i> nests increased nationally but fluctuated locally in Sardinia
NON-MONETARY	

Abbreviations: SNA System of National Accounts, SEEA-EA System of Environmental-Economic Accounting Ecosystem Accounting

Conclusion: The CAN framework successfully provides a multi-dimensional, scalable indicator dashboard that:

- Bridges the gap** between economic activity accounts, cultural and ecosystem/biodiversity indicators.
- Highlights key trends and pressures** such as local biodiversity degradation despite national-level economic growth.
- Supports more inclusive, informed policy-making** by accounting for biophysical and socio-cultural changes that standard economic frameworks do not adequately cover.

The pilot suggests CAN is a pragmatic and flexible solution for integrating ecosystem health into policy processes, with relevance across scales from national to local (Turner et al., 2026).

In summary, the diagram below, Figure 5.4, puts the various accounting approaches into the work package plan for MARBEFES and indicates how the various components of Work Package 4 fit together, with links to other work packages.

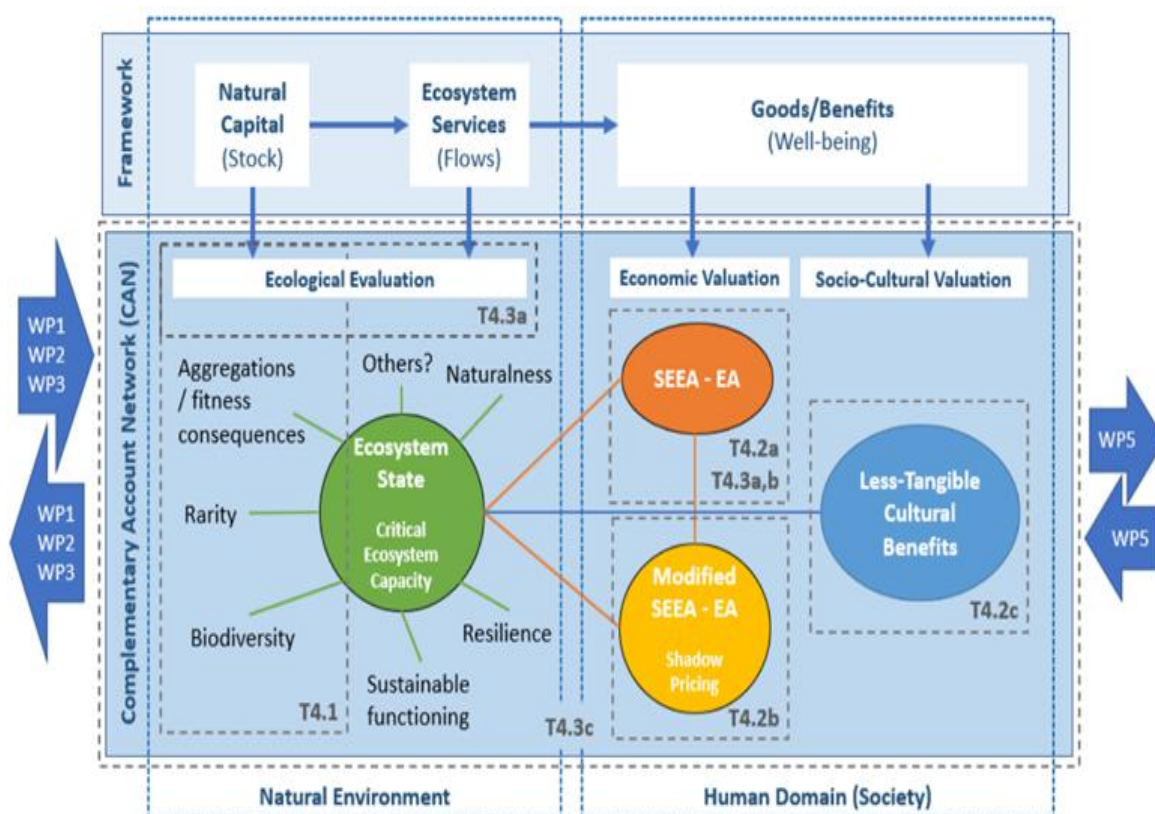


Figure 5.4: Accounting approaches fitting into the work package plan for MARBEFES.

To expand Natural Capital Accounts for MARBEFES BBTs, the reader needs to have in mind what are the basis of the SNA, SEEA-EA and CAN approach. For example, Figure 5.5 summarises the three approaches in relation to the economic value, in particular the conceptual value and methods described in Section 3.7; as a more detailed illustration, Table 5.1 below provides an overview of methods that can be applied to different types of ecosystem services in relation to users, conceptual value and approach. In particular, the SEEA-EA accounts framework is fully compatible with SNA conventions and restrictions, while the CAN accounts can provide complementary information to SNA accounts and hence allow a certain flexibility and fit-for-purpose design.

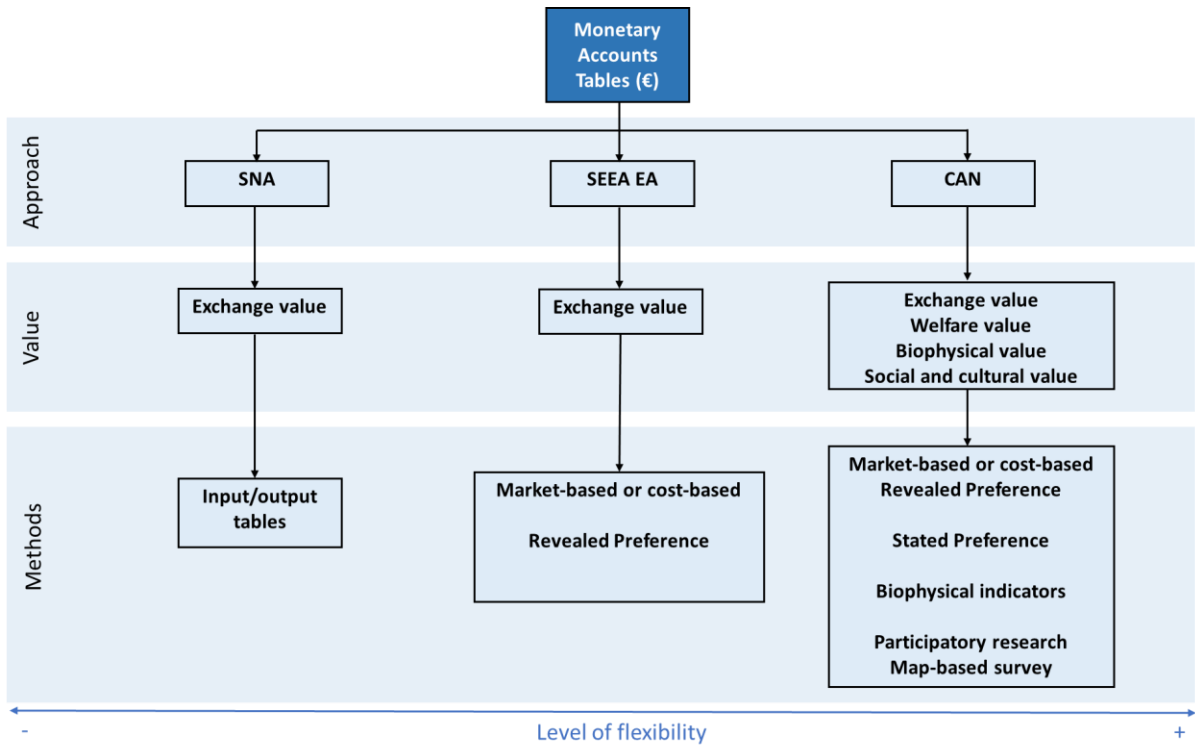


Figure 5.5: Expanding Natural Capital Accounts for MARBEFES BBTs in CAN.

Table 5.1: Example of ecosystem services and applied methods in SEEA-EA and CAN

ECOSYSTEM SERVICES	DESCRIPTION	USERS	APPROACH	
			SEEA-EA	CAN
			Based on Exchange prices	Based on Exchange prices and/or welfare values
MARINE				
Marine fish	Wild fish species that can be harvested and used as food. Harvesting is still needed to produce benefits out of this ESs.	Fisheries sector	Adjusted Market price Residual Resource rent	Adjusted Market price Residual Resource rent Contingent valuation or Choice experiment Travel cost Benefit transfer
Marine fish	Wild recreational fishing	Household	Residual Resource rent	Adjusted Market price Contingent valuation or Choice experiment Travel cost Benefit transfer
Marine recreational visits	Daily visits to the marine space to unwind, relax and enjoy nature	Household	Travel cost	Adjusted Market price Contingent valuation or Choice experiment Travel cost Benefit transfer
Raw material (dead seaweed)	Dead sea weed are currently treated as a general waste and remove for truistical purposes.	Government, businesses (food or cosmetics sector)	Avoided costs Replacement cost	Hedonic price Travel cost Stated preference
Flood control	The natural control of water flow in rainy season.	Global society	Adjusted Market price Avoided cost	Adjusted Market price Avoided cost Contingent valuation or choice experiment
OTHERS				
Global climate regulation	Maintenance of stable and sustainable air gases combination	Global society	Adjusted Market price Damage cost	Adjusted Market price Social cost Payment for ecosystem services Fiscal instruments Contingent valuation or choice experiment

5.2 Application of the MARBEFES Method

This section provides guidance on the framework approach to undertake a full valuation of the marine ecosystems in a case study area (including BBTs and out-scaling sites in MARBEFES), addressing both their natural and human components, as developed in MARBEFES WP4 (Figure 5.6). Visualisation in ARIES ([Artificial Intelligence for Environment & Sustainability](#)) is applied to the EVA data to inform PNCA tables.

The valuation process is delivered in the context of the DAPSI(W)R(M) cause-consequence-response framework (Elliott et al., 2017; see Section 1). The analysis of Drivers, Activities and Pressures will allow ‘scoping question’ (or several) to be identified about the natural environment needs in the case study area that require further investigation. The study of the (environmental) State (and its changes) and Impacts (on Welfare) will provide the information needed to sustainably manage the ecosystem while ensuring its functioning and ability to deliver societal benefits through appropriate Responses (as management Measures).

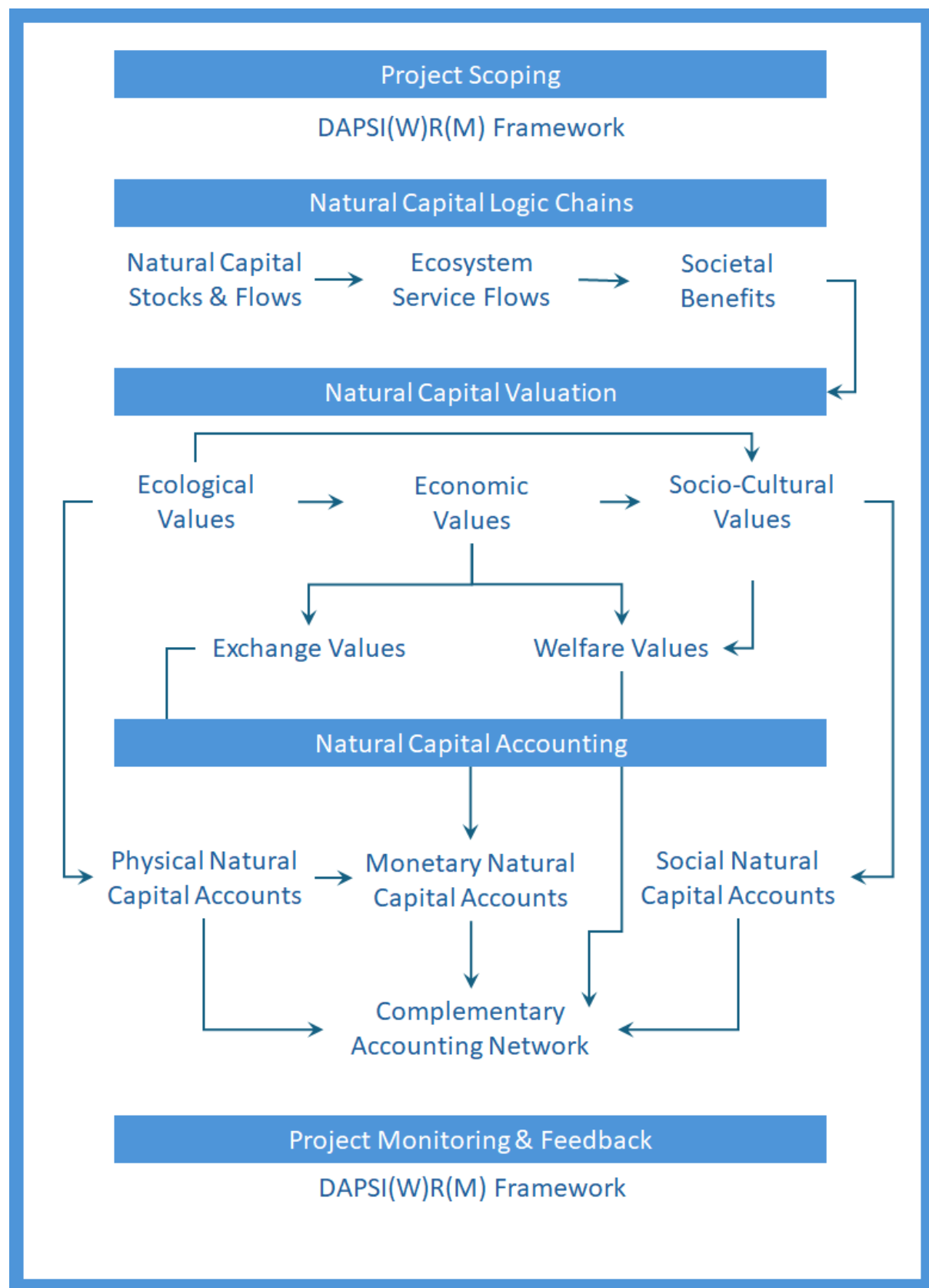


Figure 5.6: Diagram of the ecosystem valuation process using the WP4 valuation and accounting methods in the context of the DAPSI(W)R(M) framework. The arrows show the expected flow of information between the different valuation and accounting approaches.

At a first glance, undertaking the full valuation process (from top to bottom in Figure 5.6) may seem an overwhelming task, as each valuation requires the user to apply different methods (some may be more familiar than others) and to gather an evidence base for their case study area, which may be broad in nature and sourcing. However, sequential links can be identified between the different valuation and accounting methods. These links between methods allow some or all the data gathered for or generated from one valuation method to be used in another valuation method following a series of sequential tasks (Figure 5.6). This means that, although each method can be implemented on its own, the overall valuation process may benefit from building on evidence and outputs shared between methods and Work Package Tasks.

The sequential links in data flow mentioned above and represented in Figure 5.6 are outlined in the following subsections:

- Section 5.2.1 focuses on the data flow between the Ecological Value Assessment (EVA) and the Physical Natural Capital Accounts (PNCA). It includes how EVA could be used as a metric for condition assessment as well as how this information can be visualised in ARIES.
- Section 5.2.2 focuses on the data flow from the PNCA into the economic and socio-cultural valuation methods. It includes how this information may be visualised in ARIES if spatially explicit valuation data are provided.

5.2.1 Data flow from Ecological Value Assessment (EVA) into Physical Natural Capital Accounts (PNCA).

The Ecological Value Assessment (EVA, Guide 1, as updated following the implementation and testing by the BBTs and provided in Annex 1) and the Physical Natural Capital Accounts (PNCA, Section 3.4 of Guide 2) focus on the assessment of the natural components of the ecosystem (i.e., the State element of DAPSI(W)R(M)). EVA assesses the inherent value of biodiversity in an area by considering the multiple ecosystem components present (species and habitats) and their importance (e.g., due to their rarity or their key role in supporting ecosystem functioning). As such, the EVA operates within a purely eco-centric standpoint. In turn, the PNCA considers the ecosystem services provided by nature by quantifying the natural capital relevant to them (stock and flows reported as physical measurements in the PNCA tables). As the ecosystem services have a value in relation to their ability to provide societal benefits, the perspective used in PNCA is an anthropocentric one. Despite these differences in perspectives, the two approaches may partly overlap in their data requirements and indeed outputs from the EVA may also inform the characterisation of the ecosystem condition in PNCA.

A definite framework for PNCA condition tables for marine ecosystems is still in development. Currently, condition tables are completed using evidence gathered from multiple sources, including, for example, the assessments under the Marine Strategy Framework Directive (Good Environmental Status), the Water Framework Directive (Ecological and Chemical Status) or the Habitat and Species Directive (favourable condition). Based on this, multiple variables can be used to measure the state (physical, chemical, structural, functional, etc) of the ecosystem for the PNCA tables, as defined by a standardised methodology suggested to be adopted EU-wide (Vallecillo et al., 2022; as described in detail for the PNCA method in Section 3.4).

In MARBEFES, we have developed an innovative, alternative approach to inform the condition tables in PNCA (Egidazu-de la Parte et al., 2026). This approach integrates the results from the EVA into a standardised habitat classification (EUNIS level 3 habitat classification has been used in MARBEFES) to allocate a condition value to marine habitats in the case study area. Such an approach allows for its visualization within the ARIES platform (Artificial Intelligence for Environment & Sustainability), which can summarise the results into the PNCA tables, maps and/or result reports.

Table 5.2 summarises how the data from EVA should be used to inform the PNCA tables. This is based on the detailed information on the data requirements and outputs of the two methods as reported in Guide 1 (EVA; Section 2) and Guide 2 (PNCA; Section 3.4). In some cases, it may be easier for the user to start populating the PNCA tables (Guide 2) in terms of habitat extents and ecosystem services flows needed (starting from the BBT elaborated Logic Chains, Annex 4), to then move to the EVA (Guide 1), which could then provide the input to inform the extent and condition tables within the PNCA (Section 3.4).

Table 5.2: Data flow between EVA (Guide 1) and PNCA (Guide 2).

Evidence	Role in EVA	Source	Use in PNCA
EUNIS habitat map (at EUNIS Level 3 or higher)	Input data layer to characterise marine benthic and pelagic habitat Ecosystem Components (EC)	EUSeaMap from European Marine Observation and Data Network (EMODnet) https://emodnet.ec.europa.eu/en/seabed-habitat-map-europe	Input data layer to identify ecosystem assets (= EUNIS habitats at minimum Level 3 for all BBTs) used as assessment units in the physical account tables. The measure of the spatial extent of the assets is derived from the map, once clipped over the case study area of interest (e.g., BBT area).
Other habitat map (e.g., Habitat Directive Annex habitats, other)	Input data layer to characterise marine benthic habitat Ecosystem Components (EC) in more detail	EUSeaMap (EMODnet); Regional (e.g., OSPAR) or national databases, local habitat monitoring (at higher habitat type resolution than available in EUNIS)	Although the EUNIS habitat classification (Level 3) was selected for use in MARBEFES to standardise the valuation implementation across BBTs, other habitat classifications may be used, translated into EUNIS classification and integrated into the EUNIS map for the BBT, especially to fill gaps (e.g., areas poorly covered by EUNIS map or with insufficient level of detail)
Data on occurrence or abundance of species of one or more of the following groups: • Seabirds • Marine mammals • Marine reptiles • Fish • Zoobenthos (e.g., macrobenthic invertebrates) • Phytobenthos (e.g., macroalgae, saltmarshes, seagrasses) • Plankton	Input data to characterise marine species as Ecosystem Components (EC). For some groups (e.g., fish) additional data on body size may be included to differentiate life stages and use their distribution as indicator of functional areas (e.g., nursery grounds)	Regional and national monitoring programmes (e.g., for MSFD, Habitats and Birds Directives, WFD) and additional Local monitoring. The data may include a wide variety of measures depending on the EC addressed, the sampling method etc. (see Table 2.3 in Guide 1 - EVA). They may also have variable spatial and temporal coverage (e.g., data from multiple years, seasons)	PNCA focus on ocean assets (habitats/ecosystems) and the spatial distribution of occurrence and abundance of biological communities (e.g. fish) may be used to report on the stocks and flows associated with a specific habitat/ecosystem type, especially where the occurrence/abundance of the species can be used as indicators or proxies for key habitat functions and ES; e.g. distribution of fish and of their spawning or juvenile stages as indicators for all wild fish (overall and as larval production or recruitment potential); distribution of some life stages as a proxy for recruitment potential; distribution of some charismatic species as to

Evidence	Role in EVA	Source	Use in PNCA
			indicate potential for habitat recreation, although additional data may be required to characterise the condition of the landscape and seascape for recreational purposes)
Data-derived products on occurrence or abundance of species of one or more of the groups above (or proxies for them)	Input data to characterise marine species as Ecosystem Components (EC)	These may be already available from EMODnet, Copernicus, or other national or study-specific sources (e.g., maps resulting from the application of species distribution models, oceanographic models) or may be derived from the available monitoring data above (via interpolation, spatial analysis, other modelling) (see Table 2.3 in Guide 1 - EVA)	Species distributions from modelled data may also be overlapped with the distribution of the habitat assets to characterise their condition (see examples above)
Spatial gridded data on presence (as 5)/absence (as 0) or abundance (or proxy for it) for all individual species or habitat types for each assessed EC	Intermediate output of EVA (raw metric values from algorithm calculations, characterising specific subgroups of species or habitat types within each EC; see section 2.3 of Guide 1 - EVA)	EVA process. The gridded data are based on hexagonal grid used in EVA (see section 2.2.2.1 of Guide 1 - EVA)	As above
Spatial gridded data on diversity index (as standardised continuous value 0 to 5) based on occurrence and/or abundance of: • all species or habitat types, by EC • specific groups of species or habitat types identified within each assessed EC as of particular value due to: - their rarity (local, regional or national) - their regular occurrence in the BBT (i.e., not locally rare) - their ecological significance (other than habitat formation; e.g. species with key roles in the food web, species life stages or habitats paying a key function in supporting viable populations, e.g. through juvenile	Intermediate output of EVA (standardised scores (relative value) as per algorithm calculations of the individual metrics (Assessment Questions, AQs), characterising the diversity of specific subgroups of species or habitat types within each EC; see section 2.2.3 of Guide 1 - EVA)	EVA process. The gridded data are based on hexagonal grid used in EVA (see section 2.2.2.1 of Guide 1 - EVA)	This might be a synthetic index that could be used as a proxy for condition of the habitat assets across groups of species, especially where these groups are identified in relation to a key function they have in the ecosystem (e.g., ecologically significant species). The EC-specific maps for the individual AQ values could potentially be used to extract the information relevant to characterise a specific aspect of condition (e.g., ecosystem functioning accounted for by AQ10-11), should a more detailed characterisation than the one at EC-level (see below) be needed.

Evidence	Role in EVA	Source	Use in PNCA
recruitment) - their habitat forming ability of species (or biogenic habitats) - the symbiotic relationship they have with another species			
Spatial gridded data on the ecological value (as standardised continuous value 0 to 5) for: • each assessed EC (EC-specific EV) • across all the assessed ECs (total EV)	Final output of EVA	EVA process. Gridded data are based on hexagonal grid used in EVA (see section 2.2.2.1 of Guide 1 - EVA)	This a high-level index, integrating multiple aspects of the ecological value (occurrence/abundance, rarity, ecological significance etc). The EC-specific EV maps may inform habitat condition, provided that ECs directly linked with the benthic habitat are selected. A new total EV may be derived from such subset of ECs so as to be more relevant to the condition assessment.
Spatial gridded data on the confidence index (as standardised categories 1-Low, 2-Medium, 3-High) for: • each assessed EC (confidence in the EC-specific EV) • across all the assessed ECs (confidence in the total EV)	Final output of EVA	EVA process. Gridded data are based on hexagonal grid used in EVA (see section 2.2.2.1 of Guide 1 - EVA)	As above

A detailed protocol has been developed to prepare the EVA results for use into the PNCA tables. As the EVA results are produced as spatial outputs (maps of Ecological Value and associated confidence, for individual Ecosystem Components (ECs) and as a total value integrated across all ECs; see Section 2.2), instructions have been provided for implementation in a GIS environment (QGIS or ArcGIS Pro). While the detailed protocol is provided in Annex 5, the key aspects of processing the results from EVA for use in PNCA tables are outlined below:

- Selection of the EVA results relevant to evaluate ecosystem condition: the condition reported in the PNCA tables is that relevant to the benthic habitats in the area being assessed; therefore, the EVA results specific to the ecosystem components that can be directly linked with such benthic habitats (e.g., Benthic Habitats, Intertidal Macroalgae, Angiosperms, Benthic macrofauna (infauna and/or epifauna), Demersal fish, (wading) Birds) are most relevant in this case.
- Rescaling of EVA results from EVA spatial grid to habitat types (mapped as polygons): EVA results are reported according to a pre-defined spatial system (regular grid), whereas the assessment units relevant to the PNCA tables are the benthic habitat types (standardised according to the EUNIS habitat classification in MARBEFES). Therefore, the EV results need to be rescaled/allocated from the EVA spatial grid into the habitat vector layer (EUNIS habitat polygons) relevant for the PNCA condition assessment.

An example of the implementation of EVA and use of its results to inform the PNCA condition tables is given in Box 5.2 for the case study of the southern Gulf of Biscay (BBT9).

It is highlighted that the spatially explicit nature of the EVA results allows for an easy transfer of the information into the habitat units relevant to the PNCA tables. This may not always be the case with other approaches, especially when the available evidence relies on assessments undertaken at the broad spatial scale, e.g., providing values referring to regions or broad demarcation areas, which cannot be down-scaled to the habitat level. This limitation was encountered for example in BBT9 (see Box 5.1) while attempting to implement the approach defined in Vallecillo et al. (2022) to derive parameters of habitat condition. Therefore, the approach developed in MARBEFES for informing the PNCA condition tables has an advantage in being spatially explicit, as well as homogeneous and comparable between habitats and case study areas (e.g., BBTs).

The implementation of both EVA and PNCA may also benefit from the integration of the work undertaken on biodiversity and ecosystem tools, such as in the characterisation of some of the assessment questions of the EVA as well as for the visualisation of the PNCA within ARIES, especially with regards to visualizing flows. For example, the metric of habitat function may provide information supporting the selection of ecologically significant species and the associated metric calculation for EVA. It may also indicate how the data on habitat and species distributions may be combined to obtain a measure of condition for PNCA, which is linked with the habitat functioning and its provision of ecosystem services. The biological trait models may also be used to link the structural assessment of biodiversity from EVA to a more functional assessment (see detailed discussion of this in Annex 1), thus making it easier to link the results with marine processes and ecosystem services, also for the benefit of PNCA.

Box 5.1. Example of how to inform PNCA tables: the case study of the Southern Gulf of Biscay (BBT9)

BBT9, the south Gulf of Biscay, is a case study (led by the University of Cantabria/FIHAC) covering the coastal area between the San Martín and the Santona estuaries in the north of Spain (237 km²). It includes a variety of marine environments (sandy beaches, rocky headlands, seagrasses, etc), seven estuaries and several protected areas.

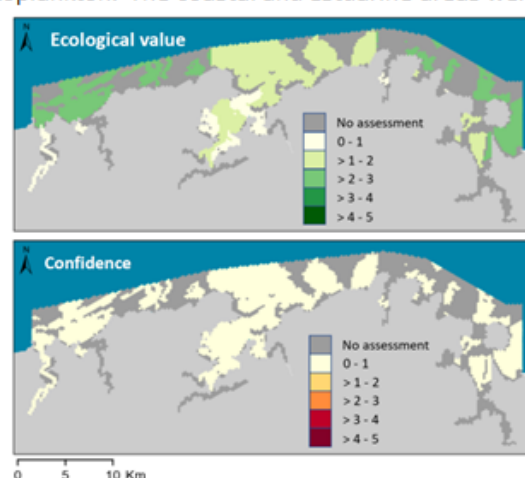
Policy question addressed: What is the ecological value of the coastal and marine biodiversity in the Southern Gulf of Biscay?

The condition in the **PNCA** tables (see section 3.4.2 of M39 report) was assessed using two different methods for each of the 18 habitat types (EUNIS level 3) present in the BBT (9 for estuaries and 9 for the coastal zone).

Method 1, the MARBEFES method: the results from the EVA are used to indicate habitat condition.

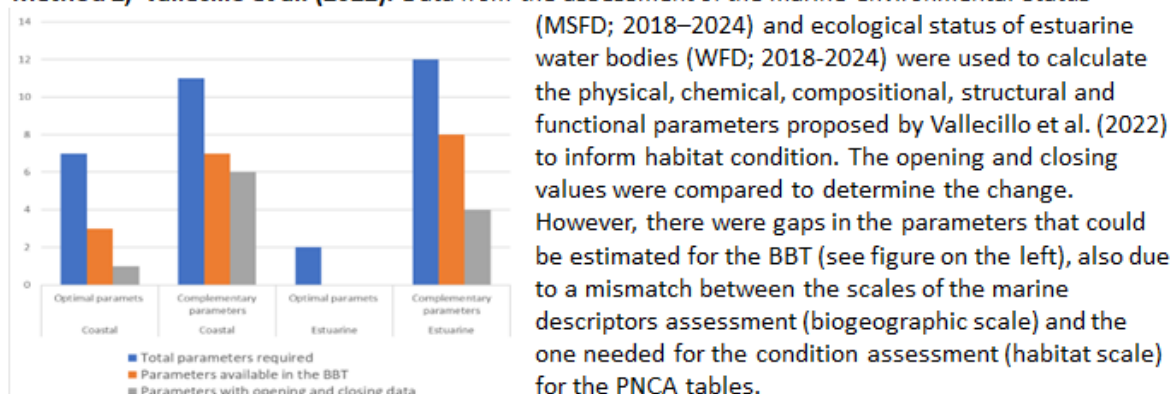
An **EVA** (see M34, Appendix 1) was implemented in BBT9 by including the following Ecosystem Components (ECs): Benthic habitats, Macrobenthos, Macroalgae (intertidal and subtidal), Angiosperms (seagrasses and saltmarshes), Fish, Waterbirds and Phytoplankton. The coastal and estuarine areas were assessed separately and the results combined.

A combination of monitoring data (e.g., WFD intertidal and subtidal benthic monitoring) and model results (e.g., EUNIS habitat Level 3 maps, spatially extrapolated species abundances) was used to inform the assessment. The EV for the benthic habitat EC was based on all the assessment questions (AQs) in the EVA protocol (M34, Appendix 1), while the AQs addressed for the species ECs depended on the specific data availability (e.g., AQ7-AQ11 for macrozoobenthos). The figure on the right shows the resulting maps combining the EV across all ECs and the associated confidence based on the evidence used for the assessment.



The EV spatial results for Benthic habitats, Macroalgae and Angiosperms were selected, rescaled to the habitat types (polygons) in the BBT and combined into a revised total EV. These spatial data were visualised in ARIES (Artificial Intelligence for Environment & Sustainability) and the extent and condition metrics derived for the different habitat types to populate the PNCA tables.

Method 2, Vallecillo et al. (2022): Data from the assessment of the marine environmental status



(MSFD; 2018–2024) and ecological status of estuarine water bodies (WFD; 2018–2024) were used to calculate the physical, chemical, compositional, structural and functional parameters proposed by Vallecillo et al. (2022) to inform habitat condition. The opening and closing values were compared to determine the change. However, there were gaps in the parameters that could be estimated for the BBT (see figure on the left), also due to a mismatch between the scales of the marine descriptors assessment (biogeographic scale) and the one needed for the condition assessment (habitat scale) for the PNCA tables.

Concluding remarks: in the experience of BBT9, compiling PNCA tables using Method 1 was more practical and informative than using Method 2. The EVA has also the advantage of providing spatially explicit, homogeneous and comparable data which can be easily used in Method 1 as a proxy for habitat condition for the PNCA tables and then visualise them in ARIES.

5.2.2 Data flow from Physical Natural Capital Accounts (PNCA) to economic and socio-cultural valuation methods

Once the habitat extent, their condition tables and the ecosystem services flow tables in the PNCA have been completed, this information can be used to estimate the monetary value of the ecosystem services considered (see Section 3.6). In the Monetary Natural Capital Accounts (MNCA) tables, economic values can only be estimated using exchange values (i.e., market prices). Similarly to the data needed to populate the PNCA, the economic data to populate the MNCA tables can be spatially linked to the habitat extent and visualized in ARIES along with the visualization of MNCA tables and the related report. This process is consistent with that used by [ARIES for SEEA](#) for the terrestrial environment.

Welfare and socio-cultural values will also be informed by the data collected in the PNCA. However, as economic data are collected to estimate the welfare values using environmental valuation methods and techniques (see Section 3.6) and the quantitative and qualitative data to estimate the socio-cultural values (see Guide 3, Section 4), those BBT data will be part of the dashboard of indicators informing the Complementary Accounting Network (CAN) framework (Section 5.1). Investigation of the welfare and socio-cultural values may use WP1 results on stakeholder perceptions and needs. If and only if, the collected welfare and social-cultural values can be spatially linked to some spatial feature, MARBEFES will experiment its potential visualization within ARIES as this visualization could be easier to read and interpret for policy makers but faces theoretical challenges.

Another tool that could be used, either in a standalone context or as a means of providing time series economic data to the CAN framework, is the BEACH (Benefit Evaluation through Assessment of Component Habitats) for MARBEFES (B4M) tool. Originally developed for UK marine waters and previously demonstrated within Northern Ireland waters (Burdon et al., 2024), the B4M tool is an innovative spreadsheet tool that uses marine benthic habitat data and the 'Matrix approach' (Potts et al., 2014) to underpin a methodology that takes national-scale estimates of the economic value of societal benefits obtained from the ecosystem services arising from marine habitats to estimate the value of societal benefits obtained from smaller scale (regional or local) areas. Whilst the tool currently uses the EUNIS (European Nature Information System) classification system for marine benthic habitats (European Environment Agency, 2026) to define habitat types and the Turner et al. (2015) coastal and marine framework to define classes of societal benefit, the underlying logic and concepts can be adapted to make use of alternative (habitat and/or societal benefit) classification schema.

The tool follows a two-stage procedure; First, the reported economic values of national-level societal benefits are disaggregated across the different constituent marine benthic habitat types at the national scale to generate habitat-specific estimates of economic value. This process incorporates information not only on the range and extent of habitats occurring at the national level, but also on the relative importance of different habitat types in providing each of the societal benefits under consideration (as provided by the 'Matrix approach'). Subsequently, given the habitat make-up of a specific area (at a regional or local scale), the B4M tool reaggregates the habitat-specific values generated by stage one to derive site-specific estimates of the economic values of societal benefits for the sub-region of interest, which account for and reflect the range and extent of habitats occurring across the new sub-national scale.

The BEACH tool was initially designed to promote stakeholders' understanding of economic aspects of the marine environment and support their engagement in decision-making (Burdon et al., 2024). For example, it can be used to estimate the value of societal benefits secured from a marine area and any change in that value caused by amending its spatial boundary in chosen ways. It also allows the tool's underpinning data – both ecological and economic – to be updated to generate new value estimates for comparative purposes. The tool is therefore relevant within a natural capital accounting context as it is driven by national-level habitat extent data, an output of physical natural capital

accounts (see Section 3.4) and national-level economic valuation data, an output of national-level marine natural capital accounts (ONS, 2025). The B4M tool can generate outputs from a single year or generate outputs from multiple years, as required to generate opening and closing natural capital accounts, therefore providing evidence to support the implementation of a CAN.

The information collected in CAN is required for the final part of the DAPSI(W)R(M) process: the development of (policy and/or management) Responses (as measures) to address the scoping question(s) initially formulated. Finally, as accounts, either as natural capital accounts or CAN, collect temporal data, they allow for monitoring over time and continuous feedback and evaluations.

6. Lessons Learned and Recommendations

6.1 Introduction

This section presents the lessons learned from the ecological, economic and socio-cultural valuation methods applied across the BBTs and forms the output from Task 4.4. A range of valuation methods were developed, refined and applied across the BBTs (WP2) and out-scaling sites (WP6). Gaining feedback throughout the process was an essential part of WP4 to ensure that the methods and their outputs, were understood and fit-for-purpose. The feedback also enabled the project to provide recommendations for further developments and future applications of the valuation methods.

6.2 Methods

Feedback was obtained using a variety of methods throughout the project. The methods ranged from informal *ad hoc* feedback whilst developing and applying the methods, to more structured feedback gathered during workshops, science meetings and questionnaire surveys. The methods used to gain feedback are summarised below:

- Mentimeter feedback following an EVA workshop (June 2024).
- Short questionnaire survey undertaken during a WP4 session at the annual science meeting in Seville (August 2025).
- Questionnaire survey on socio-cultural methods (July 2024 to April 2026).
- Questionnaire survey on all valuation methods (March 2026 to May 2026) - adapted from the socio-cultural survey to include all valuation methods.
- Informal *ad hoc* feedback (September 2022 to June 2026).

A summary of the feedback obtained from each of the BBTs is presented in Table 6.1 below; note this table does not represent where the methods were applied, rather where feedback was received. All cells in dark green include feedback from the full questionnaire survey, cells in light green reflect feedback from the short questionnaire. All cells with n/a reflect that the valuation method was not applied in that BBT. All blank cells reflect no response was received.

Table 6.1: Summary of feedback from each BBT

BBT No.	BBT Name	Ecological	Economic	Socio-Cultural	Physical Accounts	Monetary Accounts	CAN
1	Porsanger fjord-Barents Sea	Yes [^]		Yes [*]			n/a
2	South Spitsbergen			n/a			n/a
3	North Spitsbergen			Yes [*]			n/a
4	Finnish Archipelago	Yes [^]	Yes	Yes [*]	Yes	n/a	n/a
5	Curonian lagoon-Lithuanian coast	Yes	Yes	Yes [*]	Yes	n/a	n/a
6	Gulf of Gdansk	Yes [^]	Yes	Yes [*]			n/a
7	Belgium-Dogger bank	Yes ^{^"}	n/a	Yes [*]	Yes ["]	n/a	n/a
8	Irish Sea (Liverpool-Dublin Bay)	Yes ^{^"}		Yes [*]	Yes ["]	Yes	n/a
9	Southern Gulf of Biscay	Yes	Yes	Yes [*]	Yes	n/a	n/a
10	Balearic	Yes [^]		Yes [*]			n/a
11	Sardinia	Yes ^{^"}	Yes	Yes [*]	Yes ["]	Yes	Yes
12	Gulf of Heraklion	Yes	n/a	Yes [*]	Yes	n/a	n/a
WP6	Martinique	n/a	n/a	n/a	n/a	n/a	n/a
WP6	Black Sea	Yes	n/a	n/a	n/a	n/a	n/a

Key [^] Results from EVA metimeter survey^{*} Results from Socio-Cultural Survey["] Results from short survey

6.3 Lessons Learned

6.3.1 Ecological Value Assessment

Ecological Value Assessment (EVA) was one of the mandatory tools and was applied to all 12 BBTs (WP2), in addition to the Black Sea (WP6). Feedback was received from 10 of these study sites. In general, the researchers undertaking the EVA had marine ecology and/or GIS backgrounds and therefore felt confident in applying the method. Given their academic backgrounds, most researchers had previous experience in habitat and species mapping, with some having experience of using earlier versions of EVA for example in the Belgium-Dogger Bank BBT.

Feedback suggests that the initial EVA guidelines produced were “fairly clear”, but that the initial planning phase which involved gathering the data and understanding the methods took much longer than initially anticipated. This was challenging for some researchers, particularly with respect to data availability and having to adjust the method based on what data and experience were available. For example, whilst most Institutes had expertise to cover the benthic habitats and fish species, some lacked other areas of expertise such as macrophytes and seabirds.

Regarding the implementation phase, data availability was reported as a particular challenge across all the BBTs and therefore multiple researchers identified challenges in tailoring the method to suit local data availability. Challenges identified from the Mentimeter survey included identifying the features of interest within an EC, identifying and gathering the right data, spatialising the data and results in the required grid and calculating the AQs using the algorithms provided. There were also challenges highlighted regarding the acceptability of the EVA approach with some stakeholders in the Black Sea due to its complexity and lack of relevant experience. In addition, scale was identified as a challenge, particularly for larger BBTs. It was recognised, that although the analysis was not complicated in terms of calculation methodology, it did require advanced knowledge of ecosystem components and valuation principles. However, the outputs were easily understood, partly due to the learning process which had been undertaken during the implementation phase.

Despite the challenges regarding lack of data and expertise, the EVA method was well received by the researchers, with all BBTs stating that they would use the method in the future as it was an academically interesting exercise which generated valuable easily understood outputs. New

collaborations were established with stakeholders when identifying data sources and in some BBTs the EVA process resulted in new data layers which were previously not available. Now that the method is understood, it was felt that additional components could be included in the future to improve the assessments if additional data are obtained. Stakeholders identified the EVA approach as being valuable for interdisciplinary research projects relating to environmental decision-making, impact assessments and marine spatial planning. The Mentimeter survey, identified environmental managers, researchers, consultants and technical advisors to be the most likely users of the EVA tool; it is noted that policy implementers and policymakers were also identified as potential users but to a lesser extent, their main role being as beneficiaries of the spatial results originating from EVA to support their decision making. It was also recognised that the EVA process could also be combined with the Socio-Cultural Valuations and Monetary Natural Capital Accounts, but that this would require a multidisciplinary team to deliver.

6.3.2 Economic Valuation

Economic valuation was not a mandatory valuation method and was undertaken at five of the BBTs (WP2); neither of the two out-scaling sites applied economic valuation methods (WP6). Most researchers within WP4 had marine ecology and GIS backgrounds and therefore studies relied on the wider experience in environmental economics within MARBEFES; it is likely that their resource capacity within the project limited the number of sites where economic valuation could be undertaken. A range of methods were applied including travel cost, choice experiments and benefit transfer.

The economic valuation methods are well established (see Section 3.7), however one of the challenges identified in the planning phase was preparing and adapting the methodology for specific case study contexts. This phase involves extensive engagement with stakeholders to fully understand and characterise the ecosystem services under investigation. The preparation phase was also considered as being long, but it was recognised that this time was required to understand and establish the methods that were to be employed. Researchers were heavily reliant on the economic expertise in the wider WP4 MARBEFES team to provide advice and guidance to develop the methods. Data availability was also highlighted as a challenge for the planning phase, including the identification of suitable data sources, data retrieval and adjustment of data to exchange values. Finally, identifying research questions which were both original and policy relevant were identified as challenges during the planning phase.

Several challenges were identified during the implementation phase. Existing data was a real challenge, often restricting which tools and methods could be applied at the BBT level. Primary data collection was particularly demanding, for example for the travel cost modelling in the Lithuanian BBT which required an extensive data set, gathered from public questionnaire surveys. Data related challenges were also identified in the Southern Gulf of Biscay, with specific challenges associated with collecting data and adapting it to the requirements of each evaluation. A second challenge relates to the lack of experience of applying economic valuation methods at the BBT level, having to rely heavily on advice and guidance from experienced environmental economists within the wider MARBEFES project. Another challenge at the implementation phase relates to finding a common language with experts from other disciplines.

Challenges were identified with the data analysis phase of the applications, for example the requirement to make extrapolations about total numbers of visitors in the Curonian Lagoon BBT to run the model as these data were not readily available. In addition, by investigating the economic consequences of algal blooms increased the complexity of tool application, however this was a new development of the method application and therefore generated new data. In some cases, the

analysis was challenging because there were discrepancies between the type of data available and the requirements of the economic models.

The economic valuation methods applied in WP4 are well established, however their novelty lies within the adaptation of them to address issues at the BBT level. A lot of the tools applied here are flexible in their approach and therefore were all successfully implemented. The collection of site-specific primary data will naturally be a time-consuming process to ensure that there is sufficient data to run the economic models or undertake the analysis of the data. For a lot of the BBTs such economic valuation data has not been collected before and therefore MARBEFES has generated pioneering primary data for use in policy and management of the sites. The guidelines provided for use by the BBTs were considered by some researchers to be difficult to understand, given that the researchers had no previous training or expertise in applying the methods; this emphasises the need for a multidisciplinary team to implement such methods. Overall, the researchers saw the benefit of the economic value methods and their use for policy and management in the marine environment, however felt a multidisciplinary team would be valuable as economic expertise was generally lacking at the individual BBT sites.

6.3.3 Socio-Cultural Valuation

Socio-cultural valuation methods (survey and interviews) were identified as a mandatory tool, which was successfully implemented at 11 out of 12 BBTs (WP2), but was not implemented at either of the out-scaling sites (WP6). A bespoke questionnaire was developed to gather feedback directly from the BBTs who implemented the method(s); this questionnaire was subsequently used for the basis of collecting the feedback on all valuation methods. Researchers who undertook the socio-cultural valuation had mixed backgrounds covering humanities, social sciences and marine ecology. Whilst some researchers had previous experience of cultural ecosystem services analysis methods and public attitudes surveys, some researchers did not have previous experience. All researchers worked in collaboration with the socio-cultural lead, to develop and implement the surveys in their respective BBTs.

Comprehensive guidelines were provided which was considered extremely helpful at the planning stage. The survey, which could be implemented online, via email or face-to-face, was standardized across the BBTs and therefore only required minor input to tailor it for local conditions, including translations where required. In some cases, the planning phase took longer than anticipated given the requirement for ethics approval and advertising the survey, in addition to the need for translations. This resulted in some challenges associated with missing peak tourist seasons, for example in Porsanger Fjord and North Spitsbergen, which reduced potential response rates. Some BBTs faced challenges with the participatory mapping elements of the survey given the scales of maps caused some confusion in recognising locations, the lack of computer literacy of respondents and overlaps between terrestrial and aquatic zones when visitors spend most of their time on the terrestrial part of the coast. Other researchers identified challenges in recruiting participants for a focus group and getting online responses given they were located away from the BBT and it took time to build relationships with local people and organisations. Some researchers relied heavily on discussions and guidance from the tool leaders and contributing experts to successfully plan the survey. The feedback also highlighted challenges associated with choosing an appropriate online platform for managing the online questionnaires.

In relation to the implementation phase, challenges were identified relating to the time taken to implement the survey. Specific challenges were highlighted regarding advertising the surveys to get sufficient responses to the online survey. Also, the mapping exercises were challenging when accessing the online survey from a mobile phone or a tablet. When administering the survey in person, the length of time to complete the survey (20-30 mins) was much longer than anticipated (10 mins)

which made it more difficult to convince people to contribute. The face-to-face survey was also challenging given the number of open-ended questions which required completing, getting access to some groups for example kites and sailors and the travel required to complete surveys across a large site. Several researchers mentioned challenges associated with ensuring the respondent profile from survey participants was not skewed. Stakeholder fatigue was raised as an issue in three BBTs (Porsanger fjord-Barents Sea, North Spitsbergen and Balearic).

Coding open-ended questions was often highlighted as a challenge given the time required to undertake this task and several researchers did not have the required training in coding and in some circumstances used AI to assist with this task. Additional time was required when the answers to open-ended questions required translating from native languages into English. Using online surveys was recognised as being a valuable survey vehicle however some researchers recognised challenges, such as sometimes understanding what respondents mean by their open-ended responses. A few researchers felt that the survey was too long and having the scenarios at the end was challenging cognitively for respondents to grasp all the concepts and think about the future at that stage. The participatory mapping elements were considered a great idea which produced insightful results, however the scale of the maps made the results difficult to analyse.

The quality of the guidelines and efficient support from the task lead was regularly highlighted as aspects of the research which worked well. Feedback was very positive on the socio-cultural method challenging natural scientists to learn new aspects about people's perceptions about environmental issues and the drivers behind why people visit specific sites, which can be valuable when managing potentially conflicting activities undertaken in the natural environment. Feedback suggested that the methods enabled researchers to build stronger relationships with local people and organisations within the case study sites. All the researchers who provided feedback would use the socio-cultural valuation methods again, some suggesting that it should be applied to complement other valuation methods. One researcher highlighted the need to change existing regulations and implementation of new environmental regulations could be informed by the results of such societal investigations.

6.3.4 Physical Natural Capital Accounts

The production of Physical Natural Capital Accounts was a mandatory method and was fully completed at 10 out of 12 sites, with partial completion at the other two BBTs (WP2); at the time of writing, this approach had not been applied in either of the out-scaling sites (WP6). Feedback received on this method largely came from the short questionnaire survey undertaken at the MARBEFES annual science meeting (3 responses) and the full WP4 valuation methods questionnaire (4 responses); it is noted that feedback on this method was rather limited. Researchers who undertook this work at the BBTs were largely marine ecologists, GIS experts and data scientists. None of the BBTs had previous experience working within this field and therefore relied on the guidance document and additional support and guidance from the task leads and from BC3 with regards to the Q-GIS guidelines.

The EVA applies similar GIS tools and commands to the Physical Natural Capital Accounts and therefore doing one right after another was very helpful in the process. It was observed that GIS operations which involve intersections for both the EVA and Physical Natural Capital Accounts were resource intensive in terms of time and computational speed. The experience of applying the Physical Natural Capital Accounts in the Irish Sea BBT concluded that it was relatively straight forward to undertake using basic GIS knowledge and therefore can be achieved without computer programming skills.

Three main challenges were identified from the general feedback which included (1) the availability of time series data in the correct form (e.g., habitat extent data from multiple years), (2) application of the Physical Natural Capital Account method which was new to all of the researchers and (3) finding a common language to collaborate with experts from different disciplines. It was reported that

applying and linking scripts worked well however the main challenge was gathering sufficient data. There was interest to use the approach in the future and a suggestion that now that the EVA method has been scripted and the physical stocks use GIS operations that a software package with GIS could be developed.

6.3.5 Monetary Natural Capital Accounts

The generation of Monetary Natural Capital Accounts was not a mandatory method and was only implemented at two BBTs (Irish Sea and Sardinia) in WP2; the approach was not applied at either of the out-scaling sites (WP6). The researchers comprised both marine ecologists and environmental economists. Whilst some of the researchers had prior experience applying Monetary Natural Capital Accounts in a national context, transferring this framework to smaller, site-specific scales required substantial adaptation.

A key challenge was the need to bridge knowledge gaps across a multidisciplinary team, especially among natural scientists unfamiliar with economic accounting concepts. Considerable effort was devoted to explaining the Monetary Natural Capital Accounts framework, clarifying terminology and addressing uncertainties. This highlights the need for early-stage capacity building.

Implementation was also challenging as methodological adjustments were necessary to tailor economic valuation approaches to the specific ecosystem services assessed, including fish provisioning, carbon sequestration and marine recreation. Additionally, applying a framework designed for national-scale environmental-economic statistics to smaller geographic areas introduced issues related to scale, data availability and the need to downscale existing datasets due to the limited availability of primary data. In contrast, the data analysis and interpretation phase was relatively straightforward, benefiting from established accounting output (supply and use tables). However, coordination between biophysical and monetary data remained a limitation, as reliance on existing datasets restricted integration and consistency.

Despite these challenges, the Monetary Natural Capital Accounts tool demonstrated clear value. It provides a structured way to link ecosystems and economic systems, translating environmental information into monetary terms that are more accessible to policymakers to support ecosystem-based management and informed decision-making.

6.3.6 Complementary Accounting Network (CAN)

The generation of Complementary Accounting Networks (CAN) was not a mandatory method and was only implemented at one BBT (Sardinia) in WP2; the approach was not applied at either of the out-scaling sites (WP6). The CAN approach was applied by environmental economists and highlighted both its conceptual relevance and its practical data requirements.

A key lesson learned is that CAN relies fundamentally on consistent time series data, analogous to the opening and closing of accounts, to effectively track changes in ecological, economic and socio-cultural dimensions over time. This temporal continuity is essential for integrating CAN within broader natural capital accounting systems.

This experience of applying CAN underscores the critical importance of aligning sampling strategies and data collection goals across disciplines, as well as ensuring consistency over time. In particular, the effective application of CAN depends on coordinated efforts to generate harmonised datasets that integrate ecological, economic and socio-cultural information within a shared temporal framework. The lack of such alignment in this case limited the tool's implementation, highlighting the need for more integrated and systematically planned data collection efforts to support the operationalisation of CAN.

7. Reflections and Recommendations for Future Research and the Use of the Outputs in Marine Management

Several reflections from the feedback are provided below, these all emphasise the added value of working collaboratively, across both disciplines and geographical locations, for the sustainable management of the marine environment.

“Effective management and decision-making require integrating multiple perspectives and the diverse viewpoints from which ecosystems can be understood.”

“It is enlightening to work in an interdisciplinary group. I hope that, together, social sciences and environmental sciences can raise environmental issues higher on the policy and management agenda.”

“It was a pleasure to collaborate with the BBT representatives. Coming from different backgrounds, it was great to gain fresh perspectives and feedback from other disciplines.”

Based on the feedback and lessons learned from WP4, the following recommendations for future research are proposed:

- Applying different valuation methods ideally requires a site-specific multidisciplinary team in which experts should be directly involved in the implementation of the methodologies rather than just providing guidance and support for implementing the method.
- The choice of valuation tools to apply in a case study should be related to the specific questions and challenges that are faced and the availability of site-specific data.
- Given that some of the tools rely on the same underlying data sets, valuation tools could be combined, for example Ecological Value Assessment with Physical Natural Capital Accounts (as in Egidazu-de la Parte et al., 2026), but also potentially with Monetary Natural Capital Accounts and Socio-Cultural Valuation.
- It is good practice to be cautious when taking decisions based on some exercise of economic valuation of ecosystem services – such studies provide valuable information for policy and management but underlying assumptions and caveats need to be understood.
- It is recommended that economic valuation studies should follow the established guidelines for revealed preferences and benefit transfer studies (e.g. Bateman & Kling, 2000; Johnston & Rosenberger, 2010).
- For the socio-cultural valuation surveys, there is no single engagement method that works well to capture data for all groups and therefore several approaches are needed to capture the breadth of responses that are required (e.g. online survey, face-to-face and focus groups).
- There is a need to bridge knowledge gaps across a multidisciplinary team, especially among natural scientists unfamiliar with economic accounting concepts, highlights the need for early-stage capacity building.
- Future applications of both the Physical and Monetary Natural Capital Accounts would benefit from dedicated interdisciplinary training on the concepts and stronger emphasis on primary data collection for accounting purpose. These improvements would enhance both the robustness and usability of the approach, particularly at local scales.
- There is a need to improve inclusivity of social science and humanities expertise in ecosystem valuation research across geographies to ensure the widespread site-specific multidisciplinary expertise needed for holistic approaches to environmental valuation.

- Rather than seeking a full integration of environmental data into the Standard National Accounting (SNA) framework, we argue pragmatically, for the establishment of protocols as a Complementary Accounting Network (CAN) - a pluralistic, multi-dimensional framework designed to capture the full range of values that society derives from nature expressed through monetary (exchange and welfare) and non-monetary terms - to sit alongside the economic accounts and on the same timescale.

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Annexes

Annex 1 Ecological Value and the Properties of an Ecosystem (Guide 1)

This annex provides definitions and background discussion on the following topics (numbered according to the section of the appendix):

1. **Ecological value and Ecological valuation;**
2. **Ecosystem properties relevant to Ecological valuation;**
3. **Other ecosystem properties;**
4. **Additional terms.**

For each of these sections, definitions of concepts are given, with additional information and considerations provided as resulting from the critical review of the available literature relevant to ecological valuation. Where it was deemed necessary (e.g., for better clarity), definitions of some terms given in the literature have been revised. In this case, the definition is referred to as 'modified from' the relevant literature.

A1.1 Ecological value and Ecological valuation

Definitions of Ecological value and Ecological valuation are given below (in bold) as per literature on this topic.

Synonyms and additional information to clarify the concepts defined are also provided.

ECOLOGICAL VALUE

The intrinsic value of the qualities of an ecosystem, regardless of its anthropogenic use or social interests. It is a property of the ecosystem/biodiversity, and is measured in non-monetary terms or even metaphysical ways (modified from Smith and Theberge 1986, Deros et al., 2007a).

Synonyms: biological value, ecological value, intrinsic value, inherent value (of nature/biodiversity).

Additional information: Ecosystem qualities is the attribute valued, covering all levels of biodiversity, from genetic diversity to ecosystem processes. Ecological value is the inherent value of such qualities hence it is a property of biodiversity that does not depend on or is related to human use (non-utilitarian perspective).

Note that an ecosystem may also have qualities that do not reflect an inherent value of biodiversity, but rather have their value attributed in relation to human use, interactions or associations with the environment. For example, where an ecosystem is valued for its ability to provide goods and benefits to human society (this will be investigated in detail in Tasks 4.2- 4.3 of MARBEFES WP4), or where a conservation value is assigned to an ecosystem or its components due to their sensitivity to anthropogenic disturbance. These ecosystem qualities reflecting non-inherent values ARE NOT part of the ecological value as defined here and in the literature (Deros et al., 2007a, b and derived applications). It is of note however that, although the conservation value is not an inherent value *per se*, ecosystem components which have a conservation value may also have an inherent ecological value.

ECOLOGICAL VALUATION

The process of assigning ecological value to an area through the integration of the intrinsic value of the ecosystem qualities within the area (modified from Deros et al., 2007a, 2007b).

Synonyms: biological valuation.

Additional information: Ecological valuation is a spatial/geographical non-monetary assessment which integrates the inherent value of ecosystem qualities measured across multiple ecosystem components (e.g., habitats, species, ecosystem processes/functions) characterising a geographic area of interest (case study area). The combination of ecosystem components within an area contributes to its ecological valuation against a set of criteria assessing their ecosystem qualities (e.g., rarity, fitness consequences – see Section 2 of this appendix).

It is a tool able to highlight subzones that have particularly high ecological or biological significance and to provide a greater-than-usual degree of risk aversion in spatial planning activities in these subzones (Derous et al., 2007b). It does not consider management criteria and quantitative conservation targets (Derous et al., 2007a).

The case study area is divided into spatial assessment units (called ‘subzones’ in Derous et al., 2007a, 2007b), which can be identified through a regular spatial grid or other ecologically meaningful spatial subdivision. Note that the spatial scale/resolution of the subdivisions may differ depending on the ecological feature assessed individually (e.g., coarser resolution for highly mobile species), before integration is undertaken across ecosystem components (through the overlapping of the related subzones) for the overall ecological value. An ecological value is assigned to each subzone in relative terms, i.e., relative to the other subzones in the case study area.

The ecological valuation process as originally developed in Derous et al., (2007a, 2007b) is graphically summarised in Figure A1.1.

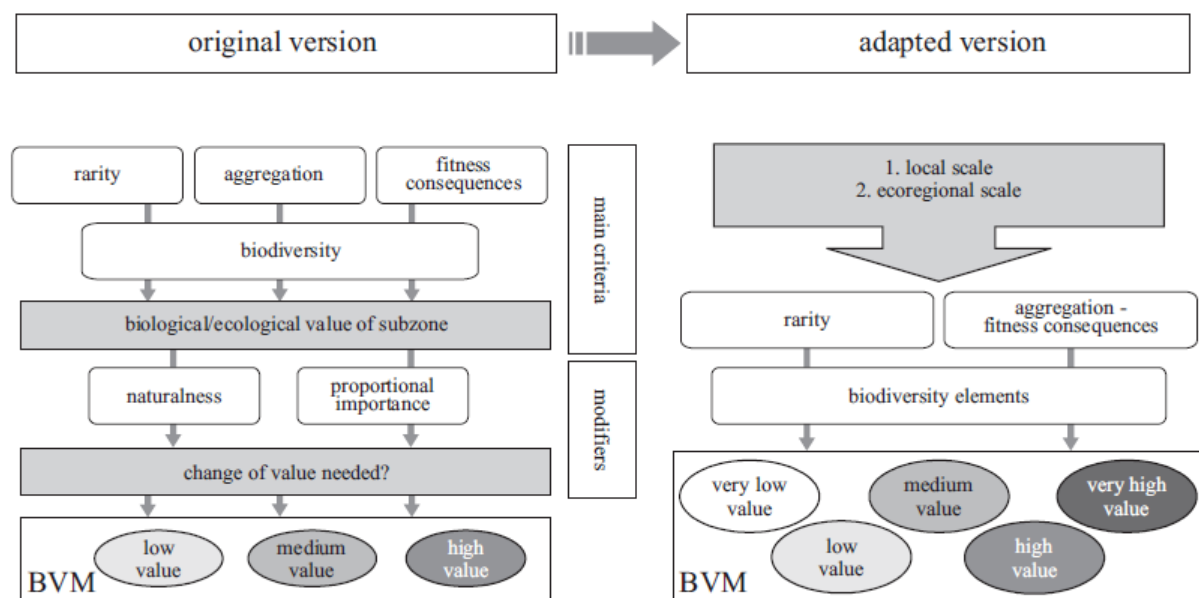


Figure A1.1: Adaptation of the concept for marine biological valuation, from original version (Derous et al., 2007a) to adapted version (Derous et al., 2007b). BVM= biological valuation map. From: Derous et al., (2007b).

A1.2 Ecosystem properties relevant to Ecological valuation

A list of properties that are relevant to the ecological valuation of an ecosystem (i.e., properties of the ecosystem which have an inherent ecological value) is provided in this section, with definitions of the terms and additional information.

BIODIVERSITY

Synonym: Biological diversity

All aspects of variability within the living world, including diversity within and between individuals, populations, species, communities and ecosystems (Convention on Biological Diversity Article 2, United Nations 1992)

Attributes of biodiversity include structures and processes existing on different organisational levels (viz. genes, species, population, community and ecosystem) (Zacharias and Roff 2000; see Figure A1.2).

Surrogate measures for biodiversity have been implicitly included in some form in previous ecological valuation approaches (e.g. Derous et al., 2007a, 2007b), albeit not as a stand-alone valuation criterion but rather as the attribute being measured under other valuation criteria.

The components of marine biodiversity:

GENETIC		SPECIES/ POPULATION		COMMUNITY		ECOSYSTEM	
Structure	Process	Structure	Process	Structure	Process	Structure	Process
Structure	Mutation	Structure	Migration	Structure	Succession	Watermass	Currents
Genotypes	Differentiat.	Abundance	Dispersion	S.Diversity	Predation	Temp	Tides
Fitness	Drift	Distribut	Retention	S. Richness	Competit.	Salinity	Disturban.
Haplotype D	Flow	Focal Spp	Mig/ Drift	S. Evenness	Parasitism	Properties	Gyres
Stocks	Nat. Select	Keystone	Growth	Abundance	Mutualism	Boundaries	Retention
	Inbreeding	Ind. Cond.	Reprod.	Represent.	Disease	Depth/Pres	P-B couple
	Mating	Ind. Comp.	Recruit	Distinctive	Production	Light	Entrain.
	Dir. Select	Umbrella		Biomes	Decomp.	Stratificat.	B-G cycles
	Stab. select	Phenotypes		Biocoenos.		Topograph	Seasonal.
	Dis. select	Fragments		S-A relns.		Substrate	Product.
	Micro. Evol.	Meta-pops		Transitions		Represent.	H-A equil.
	Erosion			Fun.groups		Distinctive	H-L equil.
	Speciation			Heterog.		Anomalies	Turbulence
	Macro. Evol			Endemism		Exposure	Mixing
				Alt. S.Stats		Patchiness	Upwelling
				Symbioses		Nutrients	Divergence
				Biomass		Dis. Gases	Ecol.Integ.
						Anoxia	Erosion
							Desiccation

Adapted from Zacharias and Roff (2000)

Figure A1.2: The attributes of marine biodiversity. Source:

http://www.coastalwiki.org/wiki/Biodiversity_as_a_marine_valuation_concept (Note: number of habitats/biotopes and their extent, could be added under Community or Ecosystem-Structure)

Biodiversity can be measured at different levels of biological organisation from the cell to the ecosystem. The most common measures of species diversity are species richness or diversity indices (for different ecosystem components, i.e., species groups). Similarly, habitat diversity could be measured using the extent of different habitat types (i.e. a mosaic) in the area. In turn, some levels of biodiversity (e.g., genetic) may have very fragmentary data, thus preventing their inclusion as candidates/suitable criteria for ecological valuation. Community or assemblage biodiversity is usually termed diversity; a community may be taken to imply interrelationships between species whereas an assemblage does not make that assumption (Gray and Elliott, 2009).

The practice of reducing biodiversity to a single index or a few dimensions is seen as unjustified by some authors (Derosus et al., 2007a) but, despite this, is universally used, especially in quality assessments (Gray and Elliott, 2009). Assessing biodiversity only on one level (e.g., species richness) can be misleading (not representative of the biodiversity on other levels) (Derosus et al., 2007a) and usually requires to be accompanied by an evenness index, thereby increasing the value of information statistics such as the Shannon-Wiener H' index (Gray and Elliott, 2009).

This argument highlights that biodiversity (syn. species richness), a structural characteristic, alone is not sufficient for ecological valuation and needs to be used in combination with other criteria that represent other ecosystem qualities of an area (e.g., rarity for uniqueness and aggregation or fitness consequence for the functioning of those species).

There is an increasing importance of an understanding of functioning instead of only using structural characteristics. As such, a proxy for functional diversity could be a diversity index created using functional guilds (trophic, ecological, etc) instead of only the taxonomic identity of species (i.e., based on species number or total abundance within each guild similar to using abundance within each species for species diversity/richness). Thrush et al., (2021) details many functional diversity indices, including the ability to compute them in R and their main features.

However, including functional diversity indices might lead to double counting if other criteria included in the ecological valuation provide an assessment of functional diversity (e.g., aggregation and fitness consequence). Therefore, care must be taken when selecting criteria for the ecological valuation. Alternatively, if the overlap between different potential criteria is only partial (because they measure functional diversity in different ways, or attest for different types of functions), weighting could be introduced to reduce the overall contribution of such criteria in the final assessment.

RARITY

Synonym: Uniqueness, Distinctiveness

As a property/quality of an ecosystem component (e.g., habitats, landscapes, species, communities, ecological functions, geomorphological and/or hydrological characteristics), it qualifies being unique (the one and only, without equal, unparalleled, for which no alternatives exist), distinct (remarkable, unusual) or rare (scarce, occurring seldom, not widely distributed, having a low density, uncommon).

As a criterion for ecological valuation, rarity is the degree to which an area is characterized by unique, rare or distinct ecosystem components (landscapes/habitats/communities/species/ecological functions/geomorphological and/or hydrological characteristics) for which no alternatives exist (modified from Derosus et al., 2007a, 2007b).

Rarity is one of the first-order (main) criteria for ecological valuation in Derous et al., (2007a, 2007b and derived applications).

Additional information:

An ecosystem component may be naturally rare (e.g., naturally present in small numbers or extent), or it may be so as a consequence of anthropogenic impacts (e.g. numbers depressed or extend reduced due to human actions). Natural rarity may be due to the species being at the edge of its distribution and/or outside its optimal environmental conditions; climate change migrations may increase or decrease the rarity of a species. Derous et al., (2007a) suggest to include both types in the assessment. A species recently introduced to an area by natural or anthropogenic means will also be rare.

The assessment of rarity of an ecosystem component is scale-dependent (e.g., national, regional, global), which affects data requirements and how this criterion is measured. This can be according to international lists of rare species, habitats or communities on a local, regional or global scale; data on population size at national or regional scale; trends, area of occupancy (as a proxy for number and location of rare/unique/distinctive features in the study area) (Derous et al., 2007a, 2007b). It is therefore essential to first define the scale at which the rarity of an ecosystem component is assessed and then select the appropriate method and data accordingly. Examples of criteria to ascertain the rarity of species and habitats at different spatial scales are given in Table A1.1. It should be noted that a component might be nationally rare or scarce (e.g., because the component occurs in the area of interest at the edge of its range), but not necessarily regionally or globally rare or scarce.

However, a definition of rarity also has to accommodate the condition that a species may naturally not be abundant irrespective of whether its preferred conditions are prevalent, such as species with predominantly solitary individuals (e.g. echinurids). Also, note that rarity may relate to time as well as space in that species may be rare apart from breeding aggregations during the year (e.g. spider crabs). The SACFOR scale (superabundant, abundant, common, frequent, occasional, rare) is commonly used to describe such spatial and temporal patterns (Gray and Elliott, 2009).

Table A1.1: Examples of criteria to define the rarity of species and habitats at different spatial scales (from Derous et al., 2007a, based on UK designations and criteria).

Rare species	Regionally rare (sessile or of restricted mobility) species = species occurring in less than 2% of the 50 × 50 km UTM grid squares of the following bathymetric zones in the region (e.g. North East Atlantic): littoral/sublittoral/bathyal, abyssal	Connor et al. (2002) (only applicable to sessile species; no guidelines available for mobile species); Connor et al. (2004); Lieberknecht et al. (2004a,b)
	Nationally rare species = species occurring in less than 0.5% of the 10 km × 10 km squares within the study area	Sanderson (1996a,1996b); Connor et al. (2004); Lieberknecht et al. (2004a,b)
	Nationally scarce species = species occurring in less than 3.5% of the 10 km x 10 km squares within the study area	
	Nationally rare species = species found in fewer than x km squares in territorial waters	Hiscock et al. (2003); Department for Environment, Food and Rural Affairs (2002)
Rare habitats	Regionally rare habitat = habitat type occurring in less than 2% of the 50 x 50 km UTM grid squares of the following bathymetric zones in the region (e.g. North East Atlantic): littoral/sublittoral/bathyal, abyssal	Connor et al. (2002)
	Nationally rare habitat = habitat type restricted to a limited number of locations in territorial waters	Department for Environment, Food and Rural Affairs (2002)

The scale at which rarity is defined for the ecosystem components may be different from the one at which the ecological valuation is being undertaken. The temporal aspect of rarity also needs to be defined under the methodology as something can now be rare whereas previously it was common (i.e., nearly extinct species) and conversely, species introduced to an area will be rare until they become established.

Amongst the ecosystem components and qualities that can be considered for the assessment of rarity, Derous et al., (2007a, 2007b) include: rare species, keystone species, umbrella species, endemic species, distinctive/unique biomes or communities, complex/diverse topography or seabed morphology, ecological heterogeneity, distinctive/unique ecosystems, critical processes (e.g., nutrient cycling, retention), unique/distinctive oceanographic features (with respect to temperature, salinity, stratification, anoxia, natural boundaries etc) or causing species to aggregate (e.g., natural refugia), unique/distinctive oceanographic processes (e.g., unique tidal systems, gyres, entrainment, natural erosion and deposition, other natural disturbance). However, some of these components (habitat-

forming species, keystone species) are also considered for the assessment of fitness consequences and therefore these should be excluded from rarity to avoid double counting.

According to these authors, accidental recordings or vagrants (termed stragglers or adventitious species in the ichthyological literature) should not be considered for the ecological valuation as they are not inherent to the ecosystem or community under consideration and hence do not contribute to the inherent biological value of the study area. However, recent entrants such as non-indigenous species should be included as they may eventually become established.

FITNESS CONSEQUENCES (or FUNCTION)

As a property/quality of an ecosystem component, it qualifies its ability to make a vital contribution to the fitness (i.e., increased survival or reproduction) of the population or species present (DFO 2004, in Derous et al., 2007a). Hence, fitness-to-survive has become a central concept in determining anthropogenic stress in an area.

As a criterion for ecological valuation, it represents the degree to which an area is a site where the activity(ies)/functions undertaken by the species there have fitness consequences, i.e., contribute significantly to the survival or reproduction of a species or population (DFO 2004). This degree may be measured by the abundance/density of the relevant ecosystem components providing such functions. For example, if the component is at species level, we are looking at the number of / presence of habitat forming species / keystone species etc. This may represent optimal habitat and environmental conditions for a species.

Fitness consequence is one of the first-order (main) criteria for ecological valuation in Derous et al., (2007a), although in later applications this criterion has been combined with the one of Aggregation (into the criterion FITNESS CONSEQUENCE and AGGREGATION) due to the potential overlap between them (Deraus et al., 2007b and derived applications).

Additional information:

In broader terms, this property can be qualified as the ability of an ecosystem component (e.g., whether species or habitat) to provide a key FUNCTION which contributes to the fitness of others (e.g., survival or reproduction of the populations/species, or a specific structural property or ecological process). However, there is the potential for confusion in combining the fitness of the individual or population (to survive) and the fitness of the environment to support those species or populations. This relates to the stress-subsidy continuum in that optimal environmental conditions for one species will create a subsidy (benefit) whereas the same conditions will reduce the fitness of another species (stress) (Elliott and Quintino, 2019).

Fitness consequence accounts for the FUNCTIONAL qualities of the ecosystem components that, through their presence or aggregation (metric), provide value to an area. For example, areas with high diversity of genetic stocks of a species (require genetic data), where individuals stop for a certain amount of time to rest or feed, or where structural habitats or keystone species are present that enhance survival and reproduction of species (e.g., refuge function) (Deraus, 2007). Indicators (of the consequences of environmental fitness) may include habitat-forming species (ecoengineers)/biogenic habitats, keystone species, EFH (e.g., for nursery function).

To measure this functional property, there is the need to (i) first, define the function(s) of interest (e.g., habitat formation, keystone role), then (ii) define/identify the relevant ecological components

(species, habitats, populations, etc) providing that function (e.g., habitat forming benthic species, keystone species) and (iii) define the metric to measure this criterion for an area (e.g. presence of habitat forming benthic species, total number of ...).

It is highlighted that, when operationalising this property as a criterion for ecological valuation, there may be some overlap with other criteria (see above and below), which may depend on which ecosystem components are selected for the assessment. In particular, functional diversity based on traits analysis may be synonymous with fitness as a species suited to its own habitat, modifying its habitat and being in optimal environmental conditions. Therefore, the ecological valuation process should make allowance for this possibility so that double counting is avoided.

AGGREGATIONS

As a property/quality of an ecosystem component, it qualifies specifically species which individuals occur at high density on a temporary (e.g., in a particular season) or permanent basis. Such aggregations may be associated to the performance of key life cycle functions (e.g., wintering, resting, feeding, spawning, breeding, nursery, rearing area or migration routes). In some cases, aggregations may involve specific life stages (e.g., aggregations of juveniles of the species in nursery areas) for which the survival or fitness is enhanced by the function performed. In this sense, this property largely overlaps with fitness consequences.

As a criterion for ecological valuation, it represents the degree to which an area is a site where most individuals of a species are aggregated for some part of the year; or a site which most individuals use for some important function in their life history; or a site where some particular structural property or ecological process occurs to create exceptionally high density (DFO 2004, Derous et al., 2007a, 2007b).

One of the first-order (main) criteria for biological valuation in Derous et al., (2007a), although in later applications this criterion has been combined with the one on Fitness consequence, as mentioned before. This may be the result of aggregation merely showing that a species is in its optimal environmental conditions.

Additional information:

Criterion especially important for highly mobile species (e.g., birds, marine mammals, fish). The distribution of Essential Fish Habitat (EFH) may be an example of the application of this criterion (see EFH definition in Section 4 of this appendix).

The assessment is undertaken for subzones within the study area and the connectivity within the study area can be highlighted by clusters of subzones with similar aggregation value.

As aggregations of mobile species may be temporary (e.g., seasonal), this may set the temporal scale for the criterion. However, it is of note that different seasons or different temporal scales may be relate only to different species, depending on their life history (e.g., for some spawning aggregations may occur in the winter, for others in the spring, for others they may occur over multiple seasons). As the criterion for ecological valuation is likely to integrate multiple components, the temporal scales integrated in the assessment need to be made explicit. The degree to which an area is a site of aggregation may be determined based on the density of one species or the number of aggregating species, which might be weighed by their respective density and/or duration of the aggregation in the year.

The definition in Derous et al., (2007a, 2007b) includes the degree to which an area acts as a site where some structural property or ecological process is exceptionally prevalent. This likely refers to aggregations of components or environmental characteristics (e.g., habitats) other than species, but which may lead to associated aggregations of species (e.g., species associated to that habitat to benefit from its structural property or the ecological process occurring in it).

NATURALNESS

Synonym: pristineness.

The characterization of an area with emphasis on its native/indigenous species (number of native/endemic species as opposed to alien, introduced or cultured species).

The characterization of an area based on the degree to which an area is currently (though not inherently) in a pristine condition and characterised by native species (i.e., absence of perturbation by human activities and number of native/endemic species as opposed to the presence of alien, introduced or cultured species (Derous et al., 2007a)).

This has been included as a second-order (modifying) criterion for biological valuation in Derous et al., (2007a), but it was not taken any further in the development of the ecological valuation protocols and in subsequent applications.

The criterion is assessed by comparison to appropriate pristine areas or reference sites, or with baseline levels or reference conditions. Alternatively, proxies for the degree of naturalness can be used, e.g., information on the presence of native and absence of introduced species in the study area. While the presence or abundance of native species may provide an added value to the ecological value of an ecosystem, considering the absence of introduced species or the comparison with pristine/reference conditions (which are often defined according to the absence of anthropogenic pressures, e.g. sensu Water Framework Directive) implicitly account for the interaction with anthropogenic factors which is outside of the scope of ecological valuation.

IRREPLACEABILITY

Referred to habitats: Habitats which would be technically very difficult (or take a very significant amount of time) to restore, recreate or replace once destroyed, taking into account their age, uniqueness, species diversity or rarity (NPFF Glossary, UK Government 2021). They include for example sand dunes and salt marsh.

If an ecosystem component is irreplaceable, it is often because it is either rare, the richness is high, it possesses a functional quality that is unique for fitness or for aggregation. These are qualities that are already included in other Ecological Valuation criteria (e.g., rarity, fitness consequences/aggregations) and therefore irreplaceability should/would not be included as an additional criterion in Ecological Valuation to avoid double counting.

ENDEMISM

It refers to the confinement of a habitat or species to a territory of variable extent (e.g., present within a localised area). For the study of endemism, the exclusive geographical distribution of the habitat or species is assessed.

This quality is considered the same as Naturalness (endemic or native species) and therefore endemism is not included as an additional criterion in Ecological Valuation to avoid double counting.

PROPORTIONAL IMPORTANCE

The definition is scale-dependent (Deraus et al., 2007a and references therein):

Global importance: proportion of the global extent of a feature (habitat/seascape) or proportion of the global population of a species occurring in a certain subarea within the study area.

Regional importance: proportion of the regional (e.g., NE Atlantic region) extent of a feature (habitat/seascape) or proportion of the regional population of a species occurring in a certain subarea within the study area. For example, supporting 1% of the international seabird population confers international importance.

National importance: proportion of the national extent of a feature (habitat/ seascape) or proportion of the national population of a species occurring in a certain subarea within territorial waters. For example, supporting 1% of the national seabird population confers national importance.

The assessment of this quality requires data on the extent of marine features or population data of individual species. Thresholds for the term ‘high proportion’ are very scale-dependent and should therefore be set for every case study separately (Deraus et al., 2007a). However, as indicated above, there are agreed national and international standards for this attribute in the case of seabirds and wading birds.

This is one of the second-order (modifying) criteria for biological valuation in Deraus et al., (2007a), but it was not taken any further in the development of the ecological valuation protocols and in subsequent applications. It has been suggested that this property is inherently included in the assessment when the scale of the assessment is defined.

A1.3 Other ecosystem properties

Other properties for which an ecosystem and its biodiversity may be valued, but that are not commonly used for ecological valuation as their value is ascribed in relation to human association (e.g., in response to a pressure).

SENSITIVITY

Sensitivity, in the context of environmental change, is the “susceptibility of a defined component of the system to harm, resulting from exposure to stressors” (Cinner et al., 2013).

The likelihood of change when a pressure (which could be chemical, physical, hydrological or biological) is applied to a species or habitat. It is a function of the ability of the habitat or species to tolerate or resist change (resistance or tolerance) and the rate (or time taken) for it to recover from impact (resilience or recovery) (Tillin & Tyler-Walters, 2014).

The intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery (Tyler-Walters & Hiscock 2005).

Sensitivity (as well as other qualities as vulnerability, fragility etc.) can only be considered in relation to a particular pressure or stressor. If the stressor is a natural environmental characteristic then it should be included whereas if an anthropogenic stressor then not so. As such, it is not considered a property with inherent ecological value and, as such, is not considered as a suitable ecological valuation criterion.

VULNERABILITY

A measure of the degree of exposure of a receptor to a pressure to which it is sensitive (based on Hiscock et al., 1999, Oakwood Environmental Ltd., 2002). (Definition used in Marine Evidence based Sensitivity Assessment (MarESA), https://www.marlin.ac.uk/sensitivity/sensitivity_rationale)

See comments above (in relation to sensitivity).

FRAGILITY

Synonyms: attribute associated to being delicate, in a weakened state, tenuous.

A characteristic of a species or habitat identifying its ability to be broken easily.

This can be seen as a particular case of sensitivity (specifically to physical disturbance) hence the considerations above for sensitivity apply here. Fragility and vulnerability will contribute to a species or community being rare and therefore are implicitly included in the ecological value assessment through the criterion of rarity.

CONNECTIVITY

‘Ecological connectivity’ is the unimpeded movement of species and the flow of natural processes that sustain life on Earth (CMS, 2020).

The flow of materials, energy and/or organisms, genes, etc. across space (e.g. among habitat patches or regions of interests). Specifically, ecological connectivity can include propagule dispersal, adult movement, species migrations, species interactions and ontogenetic linkages, with the associated flows of energy and matter (Beger et al., 2022).

Connectivity is a dynamic ecological (flow) process that emphasises the inter-dependence of different areas (e.g. habitats, patches) which underpins the persistence of populations, species, communities and ecosystems in an area. As such, it has a pivotal role in conservation strategies (see for example IUCN guidelines for conserving connectivity; Hilty et al., 2020), although its operationalisation is still limited in spatial conservation planning (Beger et al., 2022). See also connectivity as a network criterion in the design of MPA networks (below).

MPA NETWORK COHERENCE CRITERIA

REPRESENTATIVITY, ADEQUACY, VIABILITY, CONNECTIVITY and REPLICATION are used as criteria for the designation and assessment of the coherence of MPA networks: (the definitions below are derived from HELCOM MPA designation guidance⁷ and are in line with those in the UK MCZ Ecological Network Guidance⁸, as based on guidance agreed by the OSPAR Commission⁹)

- Representativity is captured by a network when the network consists of areas representing the different biogeographical subdivisions of the sea, which in turn reflect the full range of ecosystems, including the biotic and habitat diversity of those marine ecosystems.
- Adequate and viable sites indicate that all sites within a network should have size and protection sufficient to ensure the ecological viability and integrity of the feature(s) for which they were selected.

⁷ <https://helcom.fi/wp-content/uploads/2022/08/HELCOM-MPA-designation-guidelines.pdf>

⁸ <https://data.jncc.gov.uk/data/94f961af-0bfc-4787-92d7-0c3bcf0fd083/MCZ-Ecological-Network-Guidance-2010.pdf>

⁹ http://www.ospar.org/content/content.asp?menu=00700302210000_000000_000000

- Connectivity in the design of a network allows for linkages whereby protected sites benefit from larval and/or species exchanges and functional linkages from other network sites.
- Replication of ecological features (species, habitats and ecological processes) requires that more than one site shall contain examples of a given feature in the given biogeographic area.

These do not represent the ecological value of a study area *per se*, but in relation to the network of MPAs, i.e., a management measure. As such, these are not considered as suitable ecological valuation criteria.

A1.4 Additional Terms

Additional terminology that may be used or related to ecological value assessment is reported below. Please note that this is not an exhaustive list. The terms are provided in alphabetical order.

ECOSYSTEM-BASED MANAGEMENT

The comprehensive integrated management of human activities based upon the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of goods and services and maintenance of ecosystem integrity (ICES 2003). The main underlying philosophy is *Ecosystem Based Management* and the *Ecosystem Based Approach* – as first developed by the UN Convention for Biological Diversity, this is defined as a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. As such, its application will help to reach a balance of these three objectives of the Convention. It is based on applying appropriate scientific methodologies focused on levels of biological organization which encompass the essential processes, functions and interactions among organisms and their environment. Furthermore, it recognizes that humans, with their cultural diversity, are an integral component of ecosystems.

ESSENTIAL FISH HABITAT (EFH)

“...those waters and substrata necessary to fish for spawning, breeding, feeding, or growth to maturity” (U.S. Magnuson-Stevens Fishery Conservation and Management Act 1976).

Additional information: The EFH definition has been operationalised as the habitat performing a specific function (e.g., nursery, refugia, spawning) for key life stages (e.g., juveniles) or processes (e.g., spawning) of fish/shellfish species and that supports aggregations (higher densities) of the associated individuals/life stages of the species (hence having an ‘added value’ compared to other habitats where the species/life stages occur). The degree by which a site provides the EFH function can be measured for example based on the probability of occurrence of aggregations of the relevant species/life stage (Franco et al., 2022).

Habitat is intended in its wider meaning, including habitat types, the combination of environmental and biotic conditions in an area.

EXPOSURE

The action of a pressure on an ecosystem component, with regard to the extent, magnitude and duration of the pressure (Robinson et al., 2008). (Definition used in Marine Evidence based Sensitivity Assessment (MarESA), https://www.marlin.ac.uk/sensitivity/sensitivity_rationale).

PRESSURE

The mechanism through which a human activity has direct or indirect effects on any part of the ecosystem. The nature of the pressure is determined by activity type, intensity and distribution (Robinson et al., 2008). (Definition used in Marine Evidence based Sensitivity Assessment (MarESA), https://www.marlin.ac.uk/sensitivity/sensitivity_rationale).

RESILIENCE

Synonym: Recoverability.

The degree to which an ecosystem or a part/component of it is able to recover from disturbance without major persistent change (Orians, 1974).

The ability of a receptor to recover from disturbance or stress (Holling 1973). (Definition used in Marine Evidence based Sensitivity Assessment (MarESA), https://www.marlin.ac.uk/sensitivity/sensitivity_rationale).

Recovery may be partial or complete with the difference between regarded as hysteresis (Elliott et al., 2007; Tett et al., 2013). However, the move towards another state may be regarded as a tipping point and in which the system will not regain its original form even if the stressor is removed.

RESISTANCE

Synonym: Intolerance.

Resistance characteristics indicate whether a receptor can absorb disturbance or stress without changing character (Holling 1973). (Definition used in Marine Evidence based Sensitivity Assessment (MarESA), https://www.marlin.ac.uk/sensitivity/sensitivity_rationale). The shift from an unimpacted state to an impacted one along a trajectory of degradation will be regarded as a threshold.

Annex 2: Ecological Significant Species Examples (Guide 1)

Examples of Ecological Significant Species included in the literature applying EVA are provided below. Please note that this is not an exhaustive list.

Examples of Ecological Significant Species	Geographical area (Reference)
<u>Macrobenthos</u> <i>Protodorrvillea kefersteini</i>, <i>Pisione remota</i>, <i>Asbjornsenia pygmaea</i>, <i>Magelona johnstoni</i>, <i>Urothoe pulchella</i>, <i>Fabulina fabula</i>, <i>Abra alba</i>, <i>Galathowenia oculata</i>, <i>Lumbrinerides amoureuxi</i>, <i>Euchone rubrocincta</i>, <i>Lysidice unicornis</i>, <i>Sternaspis scutata</i>, <i>Heteromastus filiformis</i>, <i>Psammogammarus caecus</i> <u>Birds</u> <i>Puffinus mauretanicus</i>, <i>Calonectris diomedea</i>, <i>Morus bassanus</i>, <i>Larus michahellis</i> <u>Demersal Fish</u> <i>Abralia (Asteroteuthis) veranyi</i>, <i>Alloteuthis</i>, <i>Illex coindetii</i>, <i>Loligo vulgaris</i>, <i>Sepiolidae</i>, <i>Todaropsis eblanae</i>, <i>Engraulis encrasicolus</i>, <i>Sardina pilchardus</i> <u>Marine Mammals and Sea Turtles</u> <i>Balaenoptera acutorostrata</i>, <i>Phocoena phocoena</i>, <i>Caretta caretta</i>	Portuguese coast (Gomes et al, 2018)
<u>Macroalgae Soft-substratum</u> <i>Bostrychia scorpioides</i>, <i>Caulacanthus ustulatus</i>, <i>Gelidium pusillum</i>, <i>Laminariales</i>, <i>Lyngbya sp.</i>, <i>Porphyra umbilicalis</i>, <i>Ulothrix sp.</i> <u>Macroalgae Hard-substratum</u> <i>Ceramium ciliatum</i>, <i>Ceramium rubrum</i>, <i>Ceramium sp.</i>, <i>Chondrus crispus</i>, <i>Cladophora sp.</i>, <i>Dictyota dichotoma</i>, <i>Pelvetia canaliculata</i>, <i>Peyssonnelia squamaria</i>, <i>Porphyra leucosticta</i> <u>Macrobenthos Soft-substratum</u> <i>Aphrodita aculeata</i>, <i>Branchiostoma lanceolatum</i>, <i>Cerastoderma glaucum</i>, <i>Cerastoderma sp.</i>, <i>Chamelea gallina</i>, <i>Chamelea striatula</i>, <i>Chartella papyracea</i>, <i>Echinocyamus pusillus</i>, <i>Mactra stultorum</i>, <i>Nucula hanleyi</i>, <i>Nucula nitidosa</i>, <i>Nucula nucleus</i>, <i>Nucula sp.</i>, <i>Nucula sulfata</i>, <i>Nucula turgida</i>, <i>Pomatoceros lamarcki</i>, <i>Pomatoceros triqueter</i>, <i>Pontocrates altamarinus</i>, <i>Ruditapes decussatus</i>, <i>Sabella pavonina</i>, <i>Sabella spallanzanii</i>, <i>Sabellaria spinulosa</i>, <i>Sabellidae</i>, <i>Sabellides octocirrata</i>, <i>Serpula concharum</i>, <i>Serpula lobiancoi</i>, <i>Serpulidae</i>, <i>Spisula elliptica</i>, <i>Spisula sp.</i>, <i>Spisula subtruncata</i>, <i>Tellina (Moerella) donacina</i>, <i>Tellina compressa</i>, <i>Tellina fabula</i>, <i>Tellina pseudopusilla</i>, <i>Tellina pygmaea</i>, <i>Tellina serrata</i>, <i>Tellina sp.</i>, <i>Veneridae</i>, <i>Venerupis aurea</i>, <i>Venerupis senegalensis</i>, <i>Venerupis sp.</i>, <i>Venus casina</i>	Basque continental shelf (Bay of Biscay) (Pascual et al., 2011)

Examples of Ecological Significant Species	Geographical area (Reference)
<u>Plants</u> <i>Zostera marina</i> <u>Hyperbenthos</u> <i>Astrorhiza limicola</i> , <i>Halyphysema tumanowiczii</i> , <i>Hippolyte varians</i> , <i>Palaemon serratus</i> , <i>Pandalus propinquus</i> <u>Epibenthos (hard substrata)</u> <i>Alcyonium digitatum</i> , <i>Alcyonium glomeratum</i> , <i>Alcyonium species</i> , <i>Asterias rubens</i> , <i>Asterina gibbosa</i> , <i>Astropecten irregulares</i> , <i>Crossaster papposus</i> , <i>Echinus esculentus</i> , <i>Henricia oculata</i> , <i>Hinia incrassata</i> , <i>Marthasterias glacialis</i> , <i>Monodonta lineata</i> , <i>Psammechinus miliaris</i>	Isles of Scilly archipelago, UK (Vanden Eede, 2007)
<u>Macrobenthos</u> <i>Abra alba</i> ; <i>Asterias rubens</i> ; <i>Bathyporeia pilosa</i> ; <i>Bathyporeia sarsi</i> ; <i>Eumida sanguinea</i> ; <i>Eurydice affinis</i> ; <i>Eurydice pulchra</i> ; <i>Mysella bidentata</i> ; <i>Nephtys hombergii</i> ; <i>Pariambus typicus</i> ; <i>Scolelepis squamata</i> ; <i>Tellina fabula</i> <u>Birds</u> <i>Arenaria interpres</i> ; <i>Larus argentatus</i> ; <i>Larus fuscus</i> ; <i>Sterna albifrons</i> ; <i>Sterna hirundo</i> ; <i>Thalasseus sandvicensis</i>	Belgian coast (Vanden Eede et al, 2014)
<u>Macrobenthos</u> <i>Abra alba</i> ; <i>Spisula subtruncata</i>	Dutch coast (Deraus, 2007)

Annex 3: Extent, Supply and Use and Condition Tables (Guide 2)

This annex gives guidance on the way in which the extent, supply and use and condition tables can be developed depending on data availability. Most likely, data will not be available to complete each table and that will be fine. Despite the experimental phase of Ecosystem Accounts being officially over following publication of the SEEA-EA, many countries are still experimenting with how accounts tables can be developed when dealing with data limitation issues, especially thematic accounts such as the Ocean Accounts (i.e., Marine Natural Capital Accounts). Once gaps in data availability are identified, it becomes easier to get closer to fill in full accounts tables as suggested in the SEEA-EA (United Nations, 2021). Remember first to check with your own national statistical office and/or ministry of the environment if your country is already compiling natural capital accounts as then many data, at the national level at least, will be available already.

A3.1 MARBEFES Ecosystem Extent Account Tables

Opening extent at the beginning of the accounting period (e.g., start of 2022, which has to be equal to the end of 2021).

Table A3.1: Ecosystem Extent Account (units of area*). Modified from United Nation (2021).

Opening extent		Coastal and marine ecosystem assets				
	EUNIS habitat	MB252	MC352	MC3521	MC3517	
	Habitat name	Posidonia Oceanica	Mediterranean coastal detritic bottoms	Association with rhodolithes	Association with Laminaria rodriguezii	Total
		AREA	AREA	AREA	AREA	AREA
Additions to extent						Area
Manage expansions		Area				
Unmanaged expansions		Area				
Reductions in extent						Area
Managed reductions		Area				
Unmanaged reductions		Area				
Net change in extent		Area	Area	Area		Area
Closing extent		Area				Area

*(km²; Ha)

A3.2 MARBEFES Supply and Use Tables

These are the tables that will need to be filled in following the process illustrated in Figure 3.2 and Figure 3.4. The tables report some examples of ecosystem types following the EUNIS classification for the Mediterranean Sea. The quantities reported in Table 3A.2, Table 3A.3 and Table 3A.4 are calculated multiplying the extent (e.g. km²/ha) in Table 3A.1 by the flow of service (e.g. carbon sequestration and storage (tCO₂eq/ha); finfish (tons/km²); seaweed (kg/km²)) provided by that area.

Table A3.2: MARBEFES Supply Table for one year only (e.g. 2022). Modified from United Nations (2021).

ECOSYSTEM SERVICES	UNIT OF MEASURE	COASTAL AND MARINE ECOSYSTEM ASSETS						
		ECOSYSTEM TYPES (EUNIS CLASSIFICATION)						
		MB252	MC352	MC3521	MC3517			
Wild fish	tonnes	Quantity	Quantity	Quantity	Quantity			
Healthy climate (Carbon sequestration and storage)	tCO ₂ eq	Quantity	Quantity	Quantity	Quantity			
Seaweed	Kg	Quantity	Quantity	Quantity	Quantity			

Table A3.3: MARBEFES Supply Table when more than one year is known and all habitats are added together. Modified from Thornton et al., (1029) and United nations (2021).

ECOSYSTEM SERVICES	UNIT OF MEASURE	COASTAL AND MARINE ASSETS	YEARS				
		ECOSYSTEM TYPES (EUNIS CLASSIFICATION)	2019	2020	2021	2022	2023
Wild fish	tonnes	MB252, MC3521, MC3517, MC352	Quantity	Quantity	Quantity	Quantity	Quantity
Healthy climate (Carbon sequestration and storage)	tCO ₂ eq	MC3521, MC3517,	Quantity	Quantity	Quantity	Quantity	Quantity
Seaweed	Kg	MB12	Quantity	Quantity	Quantity	Quantity	Quantity

Table A3.4: MARBEFES Use Table (for each year) disaggregated by industry sectors. Modified from United Nation (2021).

ECOSYSTEM SERVICES	UNIT OF MEASURE	USERS AND BENEFICIARIES					
		SOCIETY	INDUSTRY				
		Households and government	Fish sector	Tourism sector	Cosmetic sector	Other sector	Other sector
Wild fish	tonnes	Quantity	Quantity				
Healthy climate (Carbon sequestration and storage)	tCO ₂ eq	Quantity					
Seaweed	Kg	Quantity			Quantity		

A3.3 MARBEFES Condition Tables

The following modified SEEA-EA table (Table A3.5) shows an example of Condition Table for individual ecosystem type (i.e. individual habitats) within the area considered (see Section 2.2). This table focuses on the variables needed to assess the different states (as specified in the SEEA-EA (United Nations, 2021) and EU-wide methodology (Vallecillo et al., 2022 – See Section 2.2) of the ecosystem type at the beginning of the accounting time and at the end of it so to assess any change.

This are nice to have tables because it is a demanding task in terms of data and information required to fill it in. However, if a BBT would wish to compile them, if their country has not done this yet, some guidance can be found in Vallecillo et al. (2022), or information can also be found in Task 4.1.

Table A3.5: MARBEFES Ecosystem condition variable account per ecosystem type (Modified from United Nation, 2021).

SEEA Ecosystem Condition Typology Class (biotic only)	VARIABLES		ECOSYSTEM TYPE (i.e. habitat)		
	Descriptor	Measurement unit	Opening value	Closing value	Change
Compositional state	Variable 1				
	Variable 2				
Structural state	Variable 3				
	Variable 4				
Functional state	Variable 5				
	Variable 6				
Landscape/seascape	Variable 7				

Table A3.6 summarizes the condition of different ecosystem types in the area of interest. This table highlights any biotic changes within an accounting period for different ecosystem types. Abiotic changes can also be considered, but in MARBEFES we are only focusing on the biotic.

Table A3.6: MARBEFES Ecosystem Condition Account (condition indices). Modified from United Nation, 2021. This table reports some examples of ecosystem types following the EUNIS classification for the Mediterranean Sea.

Accounting entries	Ecosystem type (EUNIS classification)						
	MC352	MB252	MC3521	MC3517			
Opening condition value							
Change in biotic ecosystem characteristics (composition, structure and function)							
Net change in condition							
Closing condition value							

Annex 4: Logic Chain Development (Guide 2)

In this appendix we report Initial Logic Chains developed for some BBTs following the Logic Chain workshop¹⁰: Bay of Biscay (Atlantic); Mallorca (Mediterranean); Lithuanian Coast and Curonian Lagoon (Baltic); Oristano, Sardinia, (Mediterranean). These examples can be followed by other BBTs to develop their own logic chains for all the societal benefits they identify.

¹⁰ MARBEFES Logic Chain workshop, University of East Anglia (UK), 2-3 February 2023.

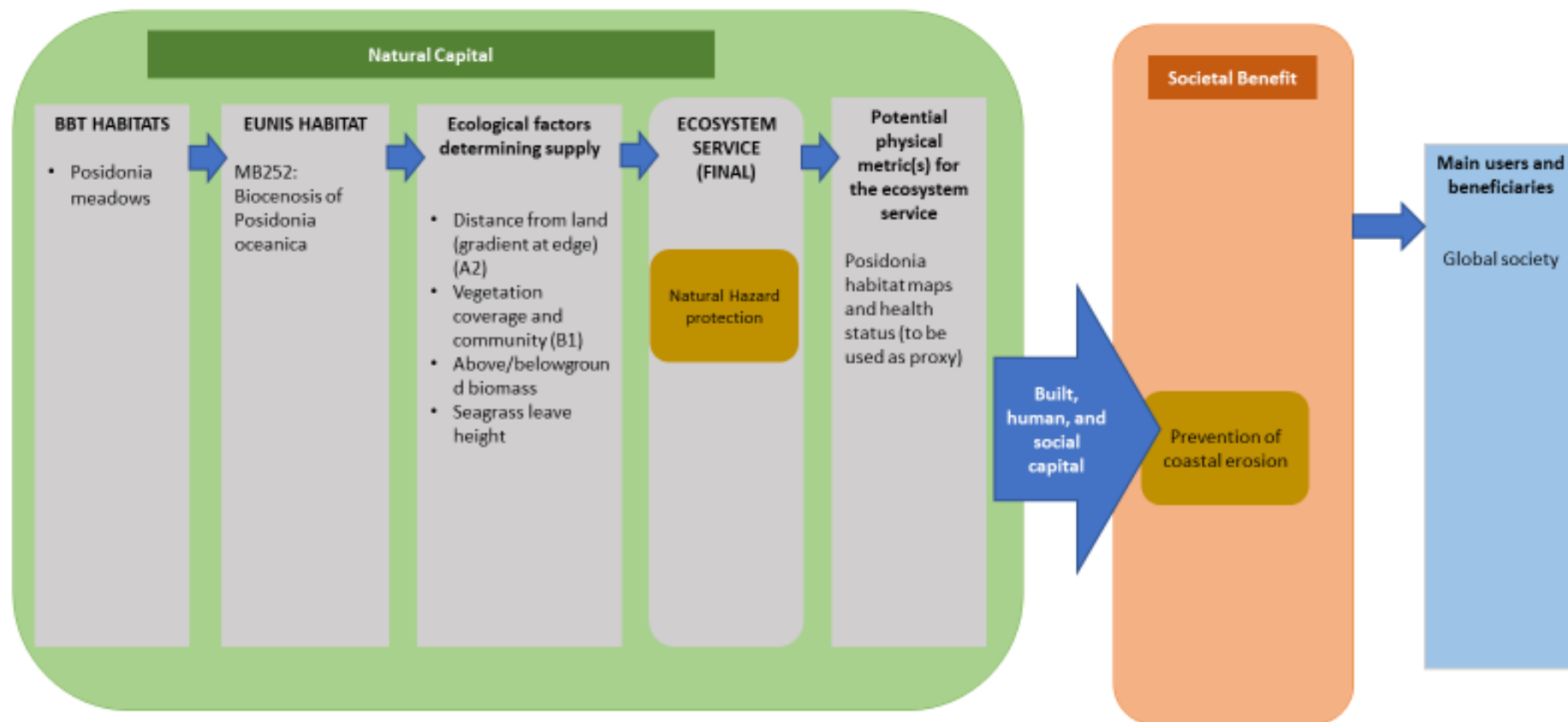


Figure A4.1 Example of Healthy Climate (non-SNA benefit) Logic Chain for the Mallorca BBT.

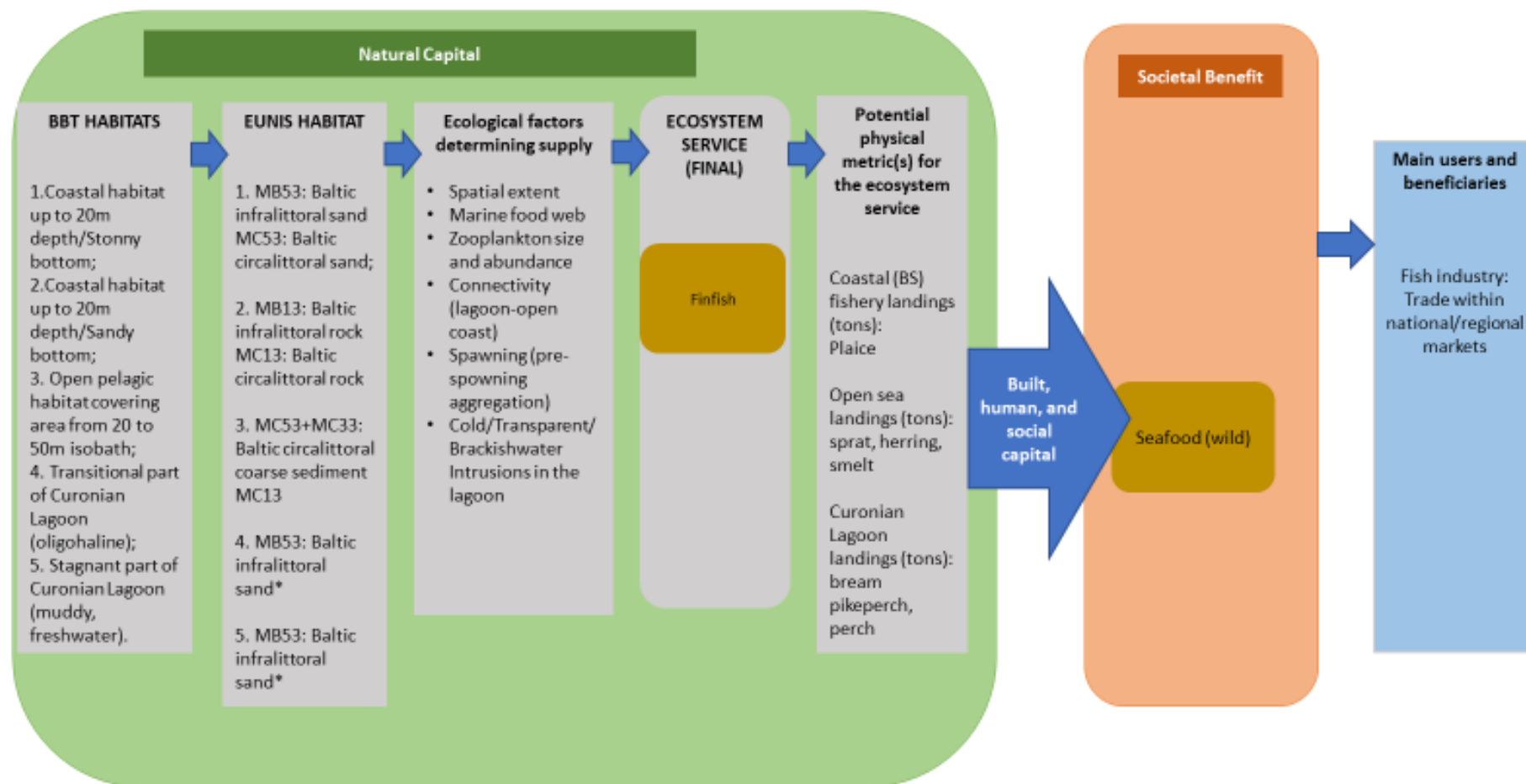


Figure A4.2 Example of Seafood (wild) (SNA benefit) Logic Chain for the Lithuanian BBT.

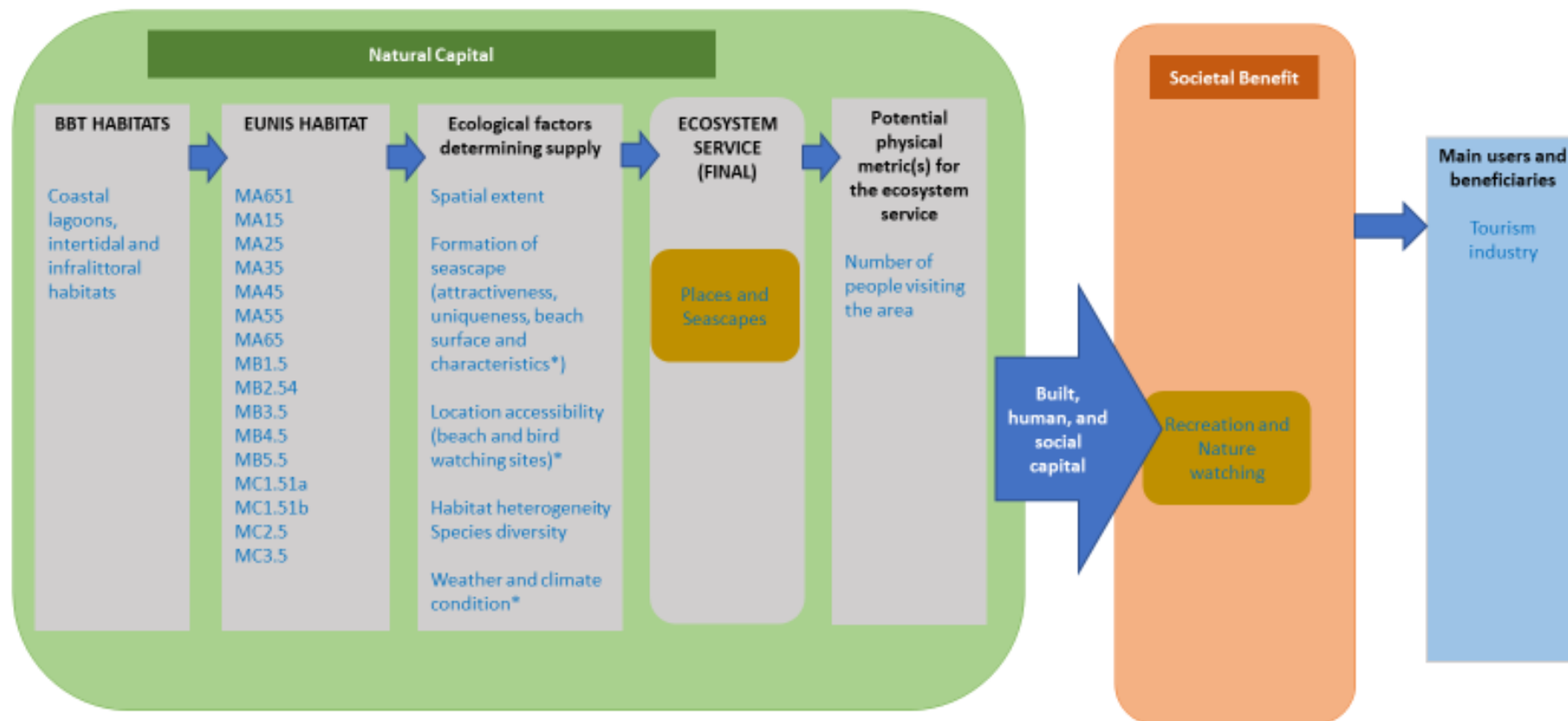


Figure A4.3 Example of Recreation and Nature Watching (SNA benefit) Logic Chain for the Oristano (Sardinia) BBT

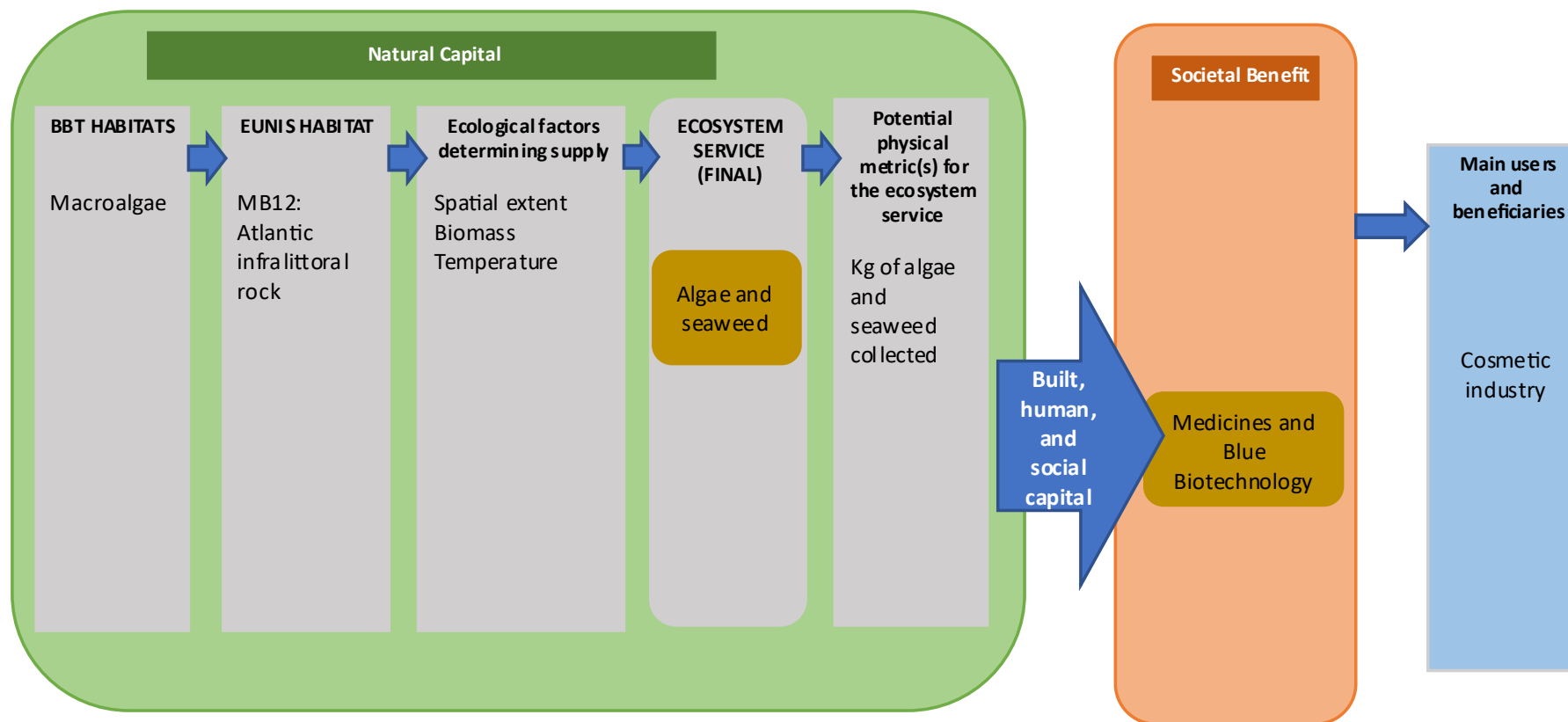


Figure A4.4 Example of Medicines and Blue Biotechnologies (SNA benefit) Logic Chain for the Bay of Biscay BBT.

Annex 5: Steps to Assess Marine Physical Accounts (Guide 2)

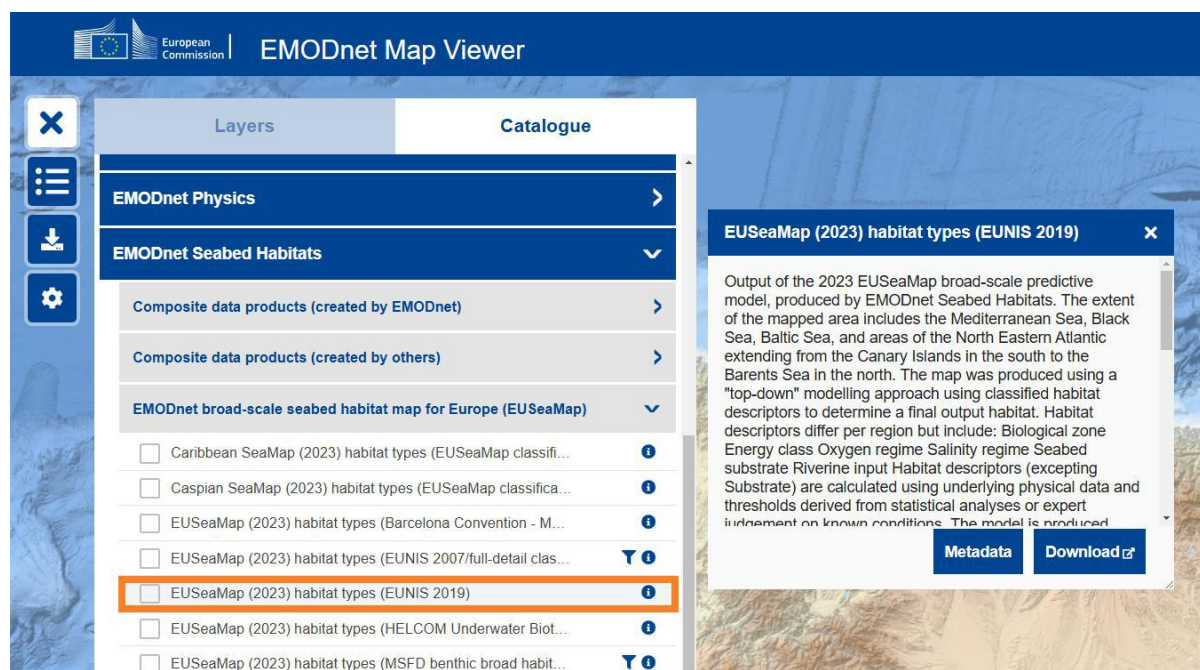
A5.1 Methods

Stock Physical Accounts in the Southern Gulf of Biscay BBT integrates the information of the Ecosystem Valuation Assessment (EVA) into level 3 EUNIS habitat classification (downloadable in EMODnet Map Viewer) to assess both ecosystem extent and condition. EUNIS habitat classification is used to calculate ecosystem extent while EVA is used to evaluate ecosystem condition. In the stock physical accounts, we use those EC (e.g. Benthic Habitats, Intertidal Macroalgae, Angiosperms) that can be directly linked to habitats, avoiding including those EC that cannot be directly linked to habitats (e.g. Pelagic Fish, Seabirds). In this document, the steps that can be followed in QGIS and ArcGIS Pro software to compute Physical Accounts in the BBTs are described.

A5.1.1. QGIS

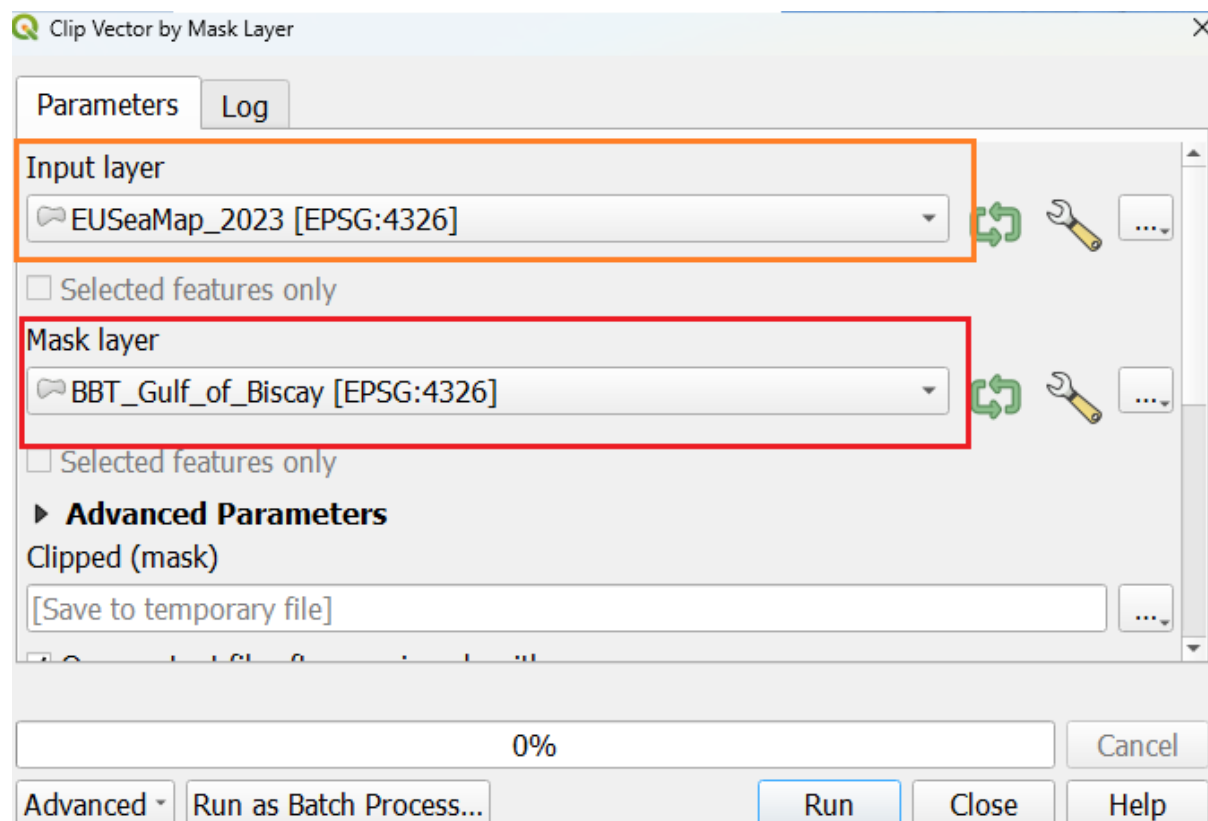
A5.1.1.1. Download EUNIS habitat map

Download level 3 EUNIS habitat classification in EMODnet Map Viewer and unzip it in a folder. We can find it in the 'Catalogue' tab under the 'EMODNet Seabed Habitats' section. We select the third option 'EMODNet broad-scale seabed habitat map for Europe (EUSeaMap)' and we download the fifth option 'EUSeaMap (2023) habitat types (EUNIS 2019)'.



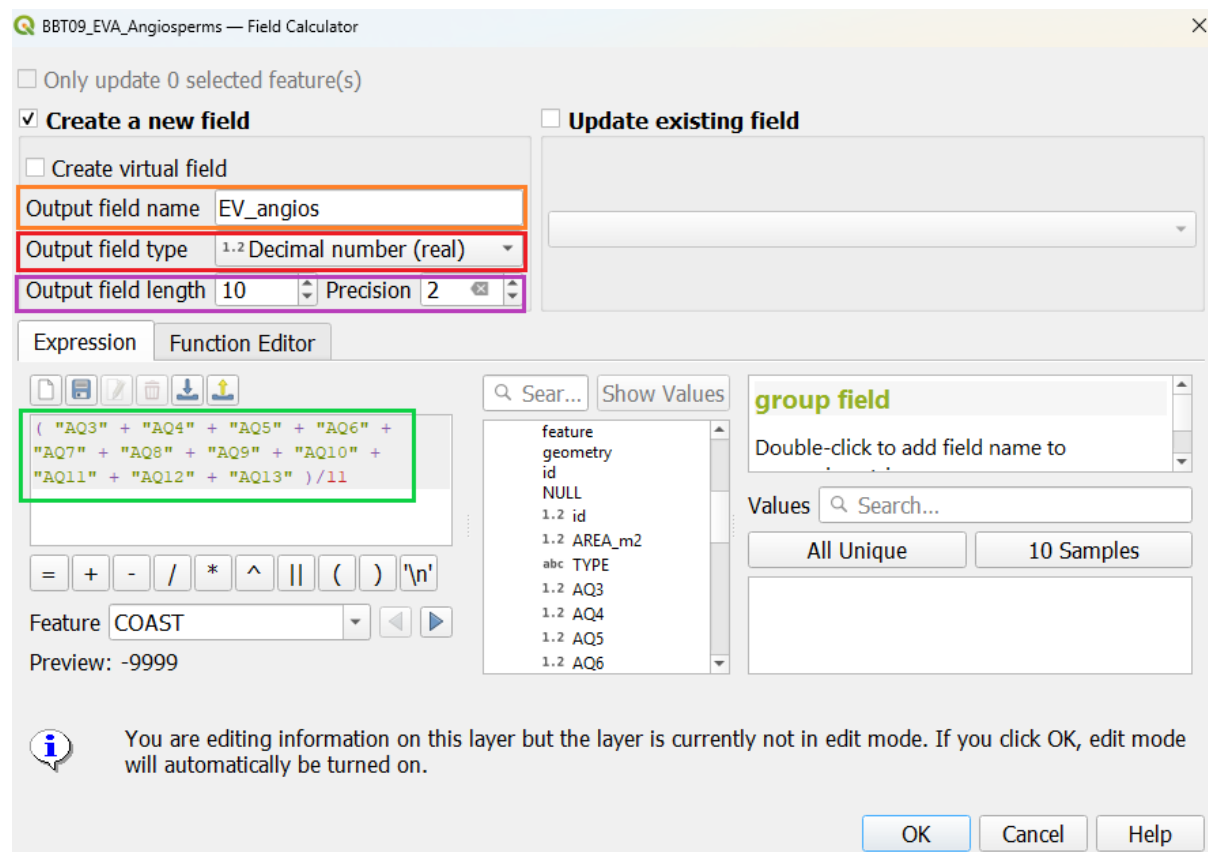
A5.1.1.2. Clip EUNIS habitat to the BBT area

Clip the EUNIS habitat classification into the Southern Gulf of Biscay BBT area. We can use the 'Clip vector by mask layer' tool to clip EUNIS to the BBT area. If you encounter any geometry problem with the EUNIS geometries, we can solve it using the 'Fix Geometries' tool. In the 'Clip vector by mask layer' tool we set the EUNIS habitat map as Input layer and the BBT shapefile as Mask layer.



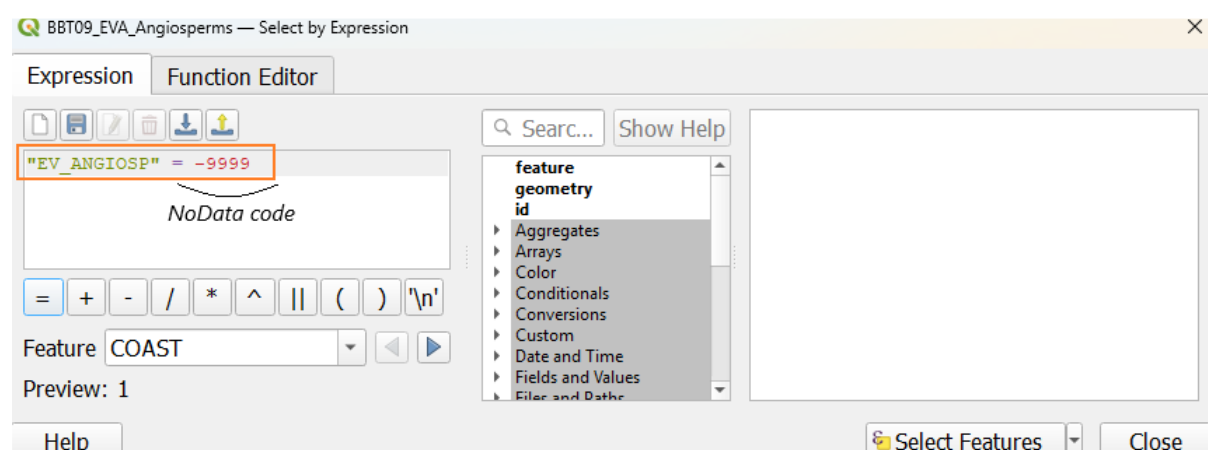
A5.1.1.3. Compute Ecological Values and Confidence of each EC

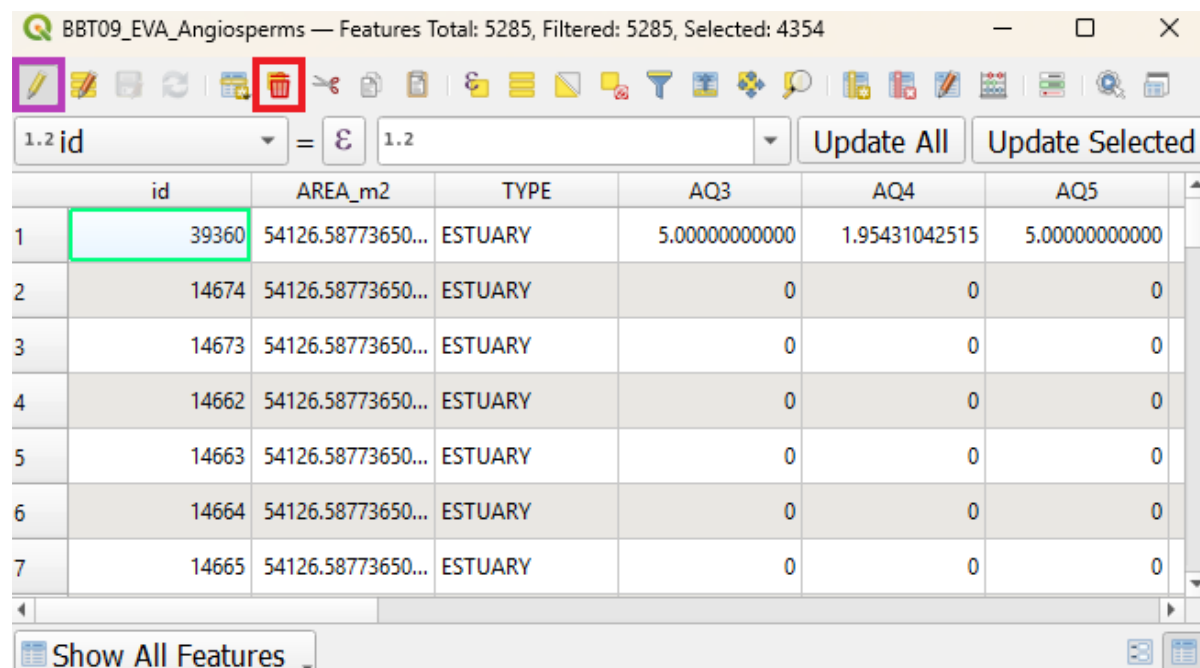
First of all, download the Ecosystem Component (EC) EVAs. Then, create an average/max (depending on the old or new protocol) EV field using the 'Field Calculator' tool. We can access it right-clicking on the EVA vector file, we 'Open Attribute Table' and using 'Open Field Calculator' on the top-right side of the table. We can also use the SearchBar of the Processing Toolbox of QGIS (right panel). We define the new name (Output field name), type (Output field type), length (Output field length) and expression needed to compute the average/max EV.



A5.1.1.4. Clear the EVAs

Delete subzones (hexagons) with *NoData* values to avoid including them in the physical account analysis. We can use the 'Select features using an expression' tool to select *NoData*. Right-click on the EVA and open the attribute table. Find the 'Select features using an expression' tool in the top side of the table. Apply the expression to select *NoData* subzones, enable 'Toggle Editing Mode' and 'Delete Selected Features'.





BBT09_EVA_Angiosperms — Features Total: 5285, Filtered: 5285, Selected: 4354

1.2 id = € 1.2 Update All Update Selected

	id	AREA_m2	TYPE	AQ3	AQ4	AQ5
1	39360	54126.58773650...	ESTUARY	5.000000000000	1.95431042515	5.000000000000
2	14674	54126.58773650...	ESTUARY	0	0	0
3	14673	54126.58773650...	ESTUARY	0	0	0
4	14662	54126.58773650...	ESTUARY	0	0	0
5	14663	54126.58773650...	ESTUARY	0	0	0
6	14664	54126.58773650...	ESTUARY	0	0	0
7	14665	54126.58773650...	ESTUARY	0	0	0

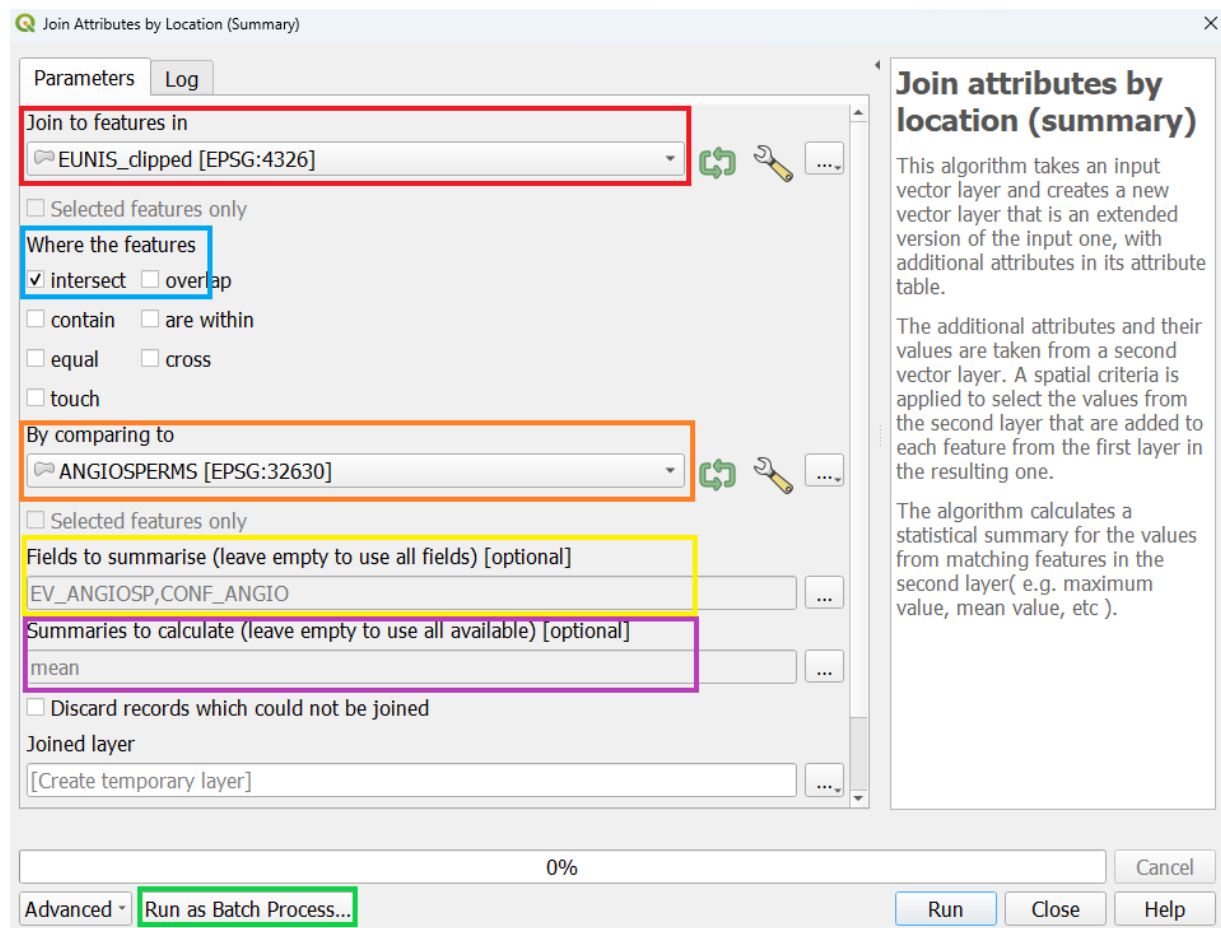
Show All Features

A5.1.1.5. Compute EV in all ECs and clean them

Repeat the steps 1.1.3 and 1.1.4 to compute EV for all ECs and clean *NoData* subzones.

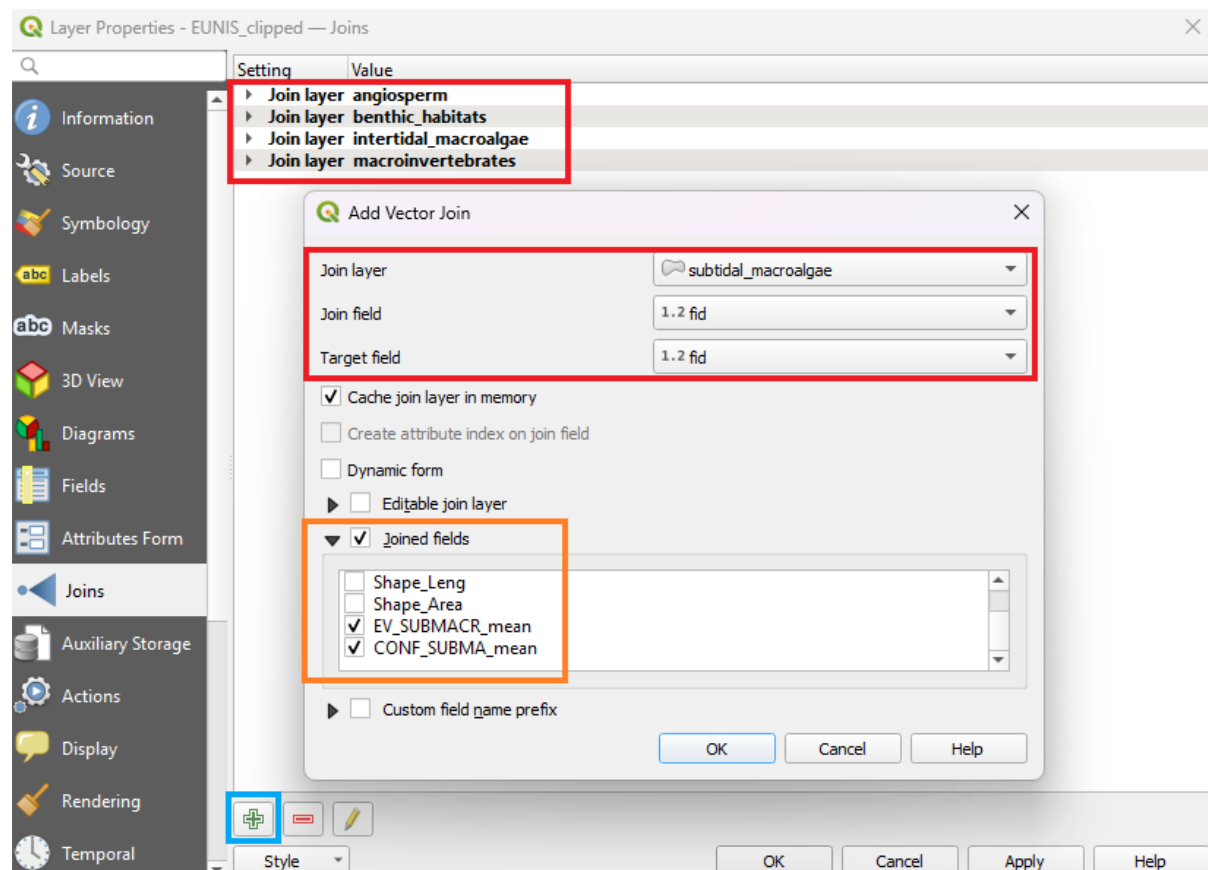
A5.1.1.6. Integrate EVAs into EUNIS habitat maps

Compute the EV and Confidence of each EUNIS habitat polygon using the average EV and Confidence scores of ECs. Each hexagon of the ECs that intersect with each of the EUNIS habitat polygons will be taken into account. We can use the 'JoinAttributes by location (summary)' tool to do so. Repeat the process with all EC EVAs to add mean EV and Confidence scores into the EUNIS habitat file or 'Run as Batch Process' to do so.

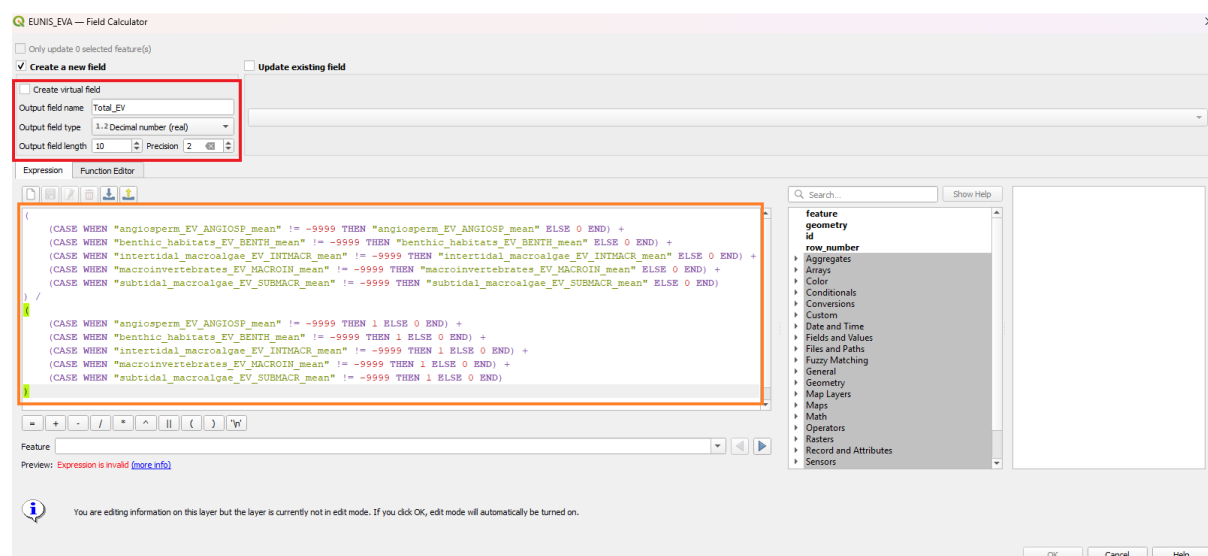


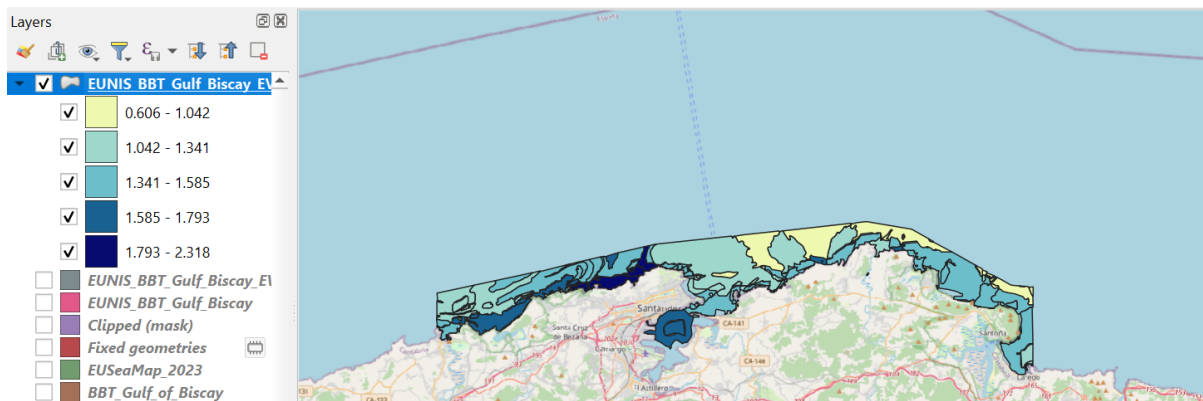
A5.1.1.7. Compute Overall habitat EV and Confidence scores

First of all, we need to join the EV and Confidence scores obtained in step 1.1.6 into a single file. To do so we can do a Join between all – EUNIS habitat-EC EV and Confidence – files. Right-click on the EUNIS clipped file, select 'Properties' and go to 'Joins'. 'Add new joins' to incorporate the EV and Confidence scores of each – EUNIS habitat-EC EV and Confidence – files into the EUNIS clipped file.



Then, we compute overall EV and Confidence scores for each EUNIS habitat polygon. We use the 'Field Calculator' to obtain the Total_EV and Total_CONFIDENCE of each EUNIS habitat polygon using the following expression with the corresponding field names and *NoData* codes. As a result, we obtain the EUNIS habitat map with a field (Total_EV) ranking the areas with the highest EV.





A5.1.1.8. Group EV and Confidence by EUNIS habitat type

Compute area-weighted EV and Confidence scores based on EUNIS habitat types ("AllcombD") and areas ("area") of each polygon. First, open the 'Field Calculator' to create a new field called 'area' and use the expression $\$area$ to compute the area of each EUNIS habitat polygon. Then, use the following expression with the corresponding field names to compute area-weighted EV and Confidence scores for each EUNIS habitat type.

Field Calculator dialog box showing the configuration for creating a new field named 'Habitat_EV' with a decimal number type. The expression being entered is:

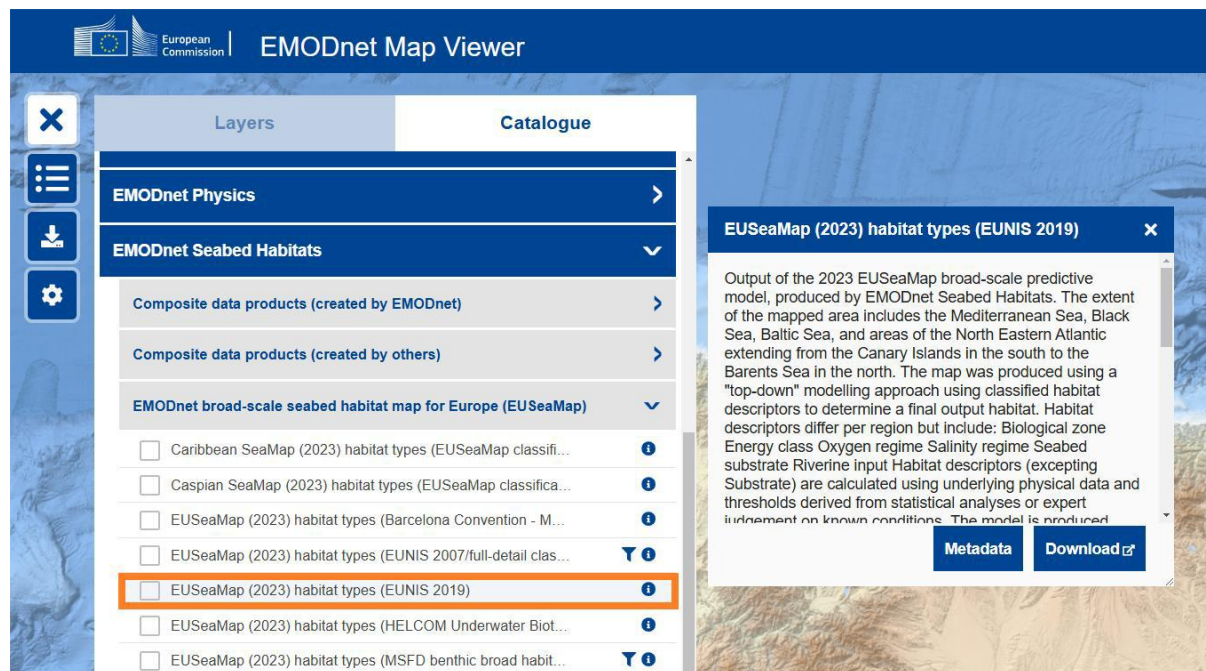
```
sum("Total_EV" * "area", group_by:= "AllcombD") /
sum("area", group_by:= "AllcombD")
```

The 'Preview' section indicates that the expression is invalid.

A5.1.2. ArcGIS Pro

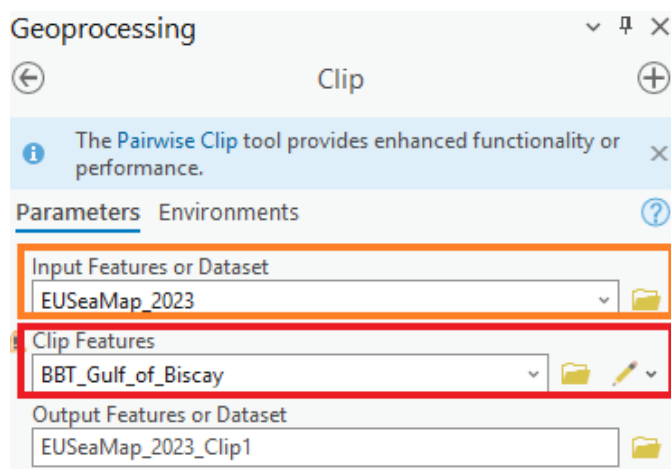
A5.1.2.1. Download EUNIS habitat map

Download level 3 EUNIS habitat classification in EMODnet Map Viewer and unzip it in a folder. We can find it in the 'Catalogue' tab under the 'EMODNet Seabed Habitats' section. We select the third option 'EMODNet broad-scale seabed habitat map for Europe (EUSeaMap)' and we download the fifth option 'EUSeaMap (2023) habitat types (EUNIS 2019)'.



A5.1.2.2 Clip EUNIS habitat to the BBT area

Clip the EUNIS habitat classification into the Southern Gulf of Biscay BBT area. We can use the 'Clip' tool from the Geoprocessing panel to clip EUNIS to the BBT area. In the 'Clip' tool we set the EUNIS habitat map as Input Features or Dataset and the BBT shapefile as Clip Features.



A5.1.2.3 Compute Ecological Values and Confidence of each EC

First of all, download the Ecosystem Component (EC) EVAs. Then, create an average/max (depending on the old or new protocol) EV field using the 'Calculate field' tool. First, we create new fields using the 'Add Field' on the top-left side of the table. Then, we can access the 'Calculate Field' tool by right-clicking on the EVA vector file, 'Attribute Table' and using 'Calculate Field' (select the created new field and right-click). We can also use the SearchBar of the Geoprocessing panel of ArcGIS Pro (right panel). We define the expression needed to compute the average/max EV.

BBT09_EVA_Benthic_Habitat

Fields: BBT09_EVA_Be...Gulf of Biscay)

Field: Add Calculate Selection: Select By Attribute

FID

Shape

Add Field

Add a new field in this table.

AQ1

1	65	Polygon				5
2	252	Polygon	62990	54126.587736	COAST	5
3	334	Polygon	71596	54126.587736	COAST	5
4	340	Polygon	71595	54126.587736	COAST	5
5	406	Polygon	63249	54126.587736	COAST	5
6	423	Polygon	21666	54126.587736	COAST	5
7	505	Polygon	70803	54126.587736	COAST	5
8	745	Polygon	62729	54126.587736	COAST	5
9	757	Polygon	21148	54126.587736	COAST	5
10	759	Polygon	21147	54126.587736	COAST	5
11	859	Polygon	70283	54126.587736	COAST	5
12	919	Polygon	21407	54126.587736	COAST	5

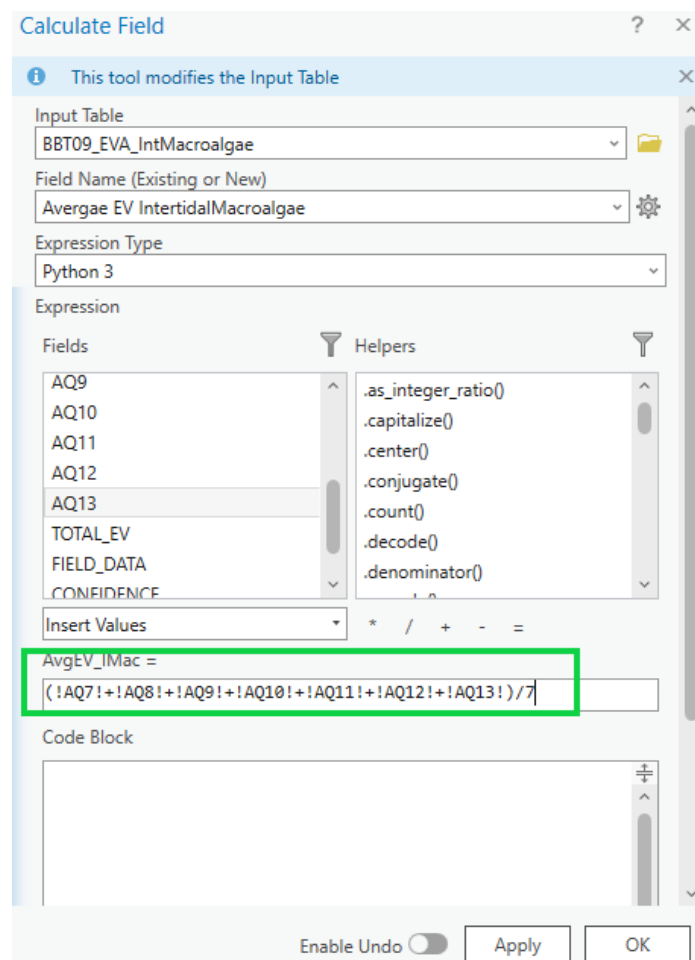
Current Layer

BBT09_EVA_Benthic_Habitat (BBT S

Visible	Read Only	Field Name	Alias	Data Type
☒	☐	AQ6	AQ6	Double
☒	☐	AQ7	AQ7	Double
☒	☐	AQ8	AQ8	Double
☒	☐	AQ9	AQ9	Double
☒	☐	AQ10	AQ10	Double
☒	☐	AQ11	AQ11	Double
☒	☐	AQ12	AQ12	Double
☒	☐	AQ13	AQ13	Double
☒	☐	TOTAL_EV	TOTAL_EV	Double
☒	☐	FIELD_DATA	FIELD_DATA	Long
☒	☐	CONFIDENCE	CONFIDENCE BENTHIC HABITAT	Double
☒	☐	Avg_EV	AVERAGE EV BENTHIC HABITAT	Double

Click here to add a new field.

[illegible]



A5.1.2.4 Clear the EVAs

Delete subzones (hexagons) with *NoData* values to avoid including them in the physical account analysis. We can use the 'Select By Attributes' tool to select *NoData*. Right-click on the EVA and open the attribute table. Find the 'Select fBy Attributes' tool on the top side of the table. Apply the expression to select *NoData* subzones and 'Delete Selected Features'.

Select By Attributes

Input Rows: BBT09_EVA_IntMacroalgae

Selection Type: New selection

Expression: Load Save Remove

Where: TOTAL_EV is equal to -9999

+ Add Clause

☐ Invert Where Clause

Apply OK

BBT09_EVA_Benthic_Habitat

Field: Add Calculate Selection: Select By Attributes Zoom To Switch Clear Delete Copy

	FID	Shape *	id	AREA_m2	TYPE	AQ1	AQ2	AQ3	AQ4	AQ5	AQ6	AQ7	Delete Selection (Delete)			
1	65	Polygon	63770	54126.587736	COAST	5	0.928753	5	0.928753	5	0.928753	0.909	Delete selected rows from the table.			
2	252	Polygon	62990	54126.587736	COAST	5	1.303012	5	1.303012	5	1.303012	0.909091	0.065645	0.065645	0	0
3	334	Polygon	71596	54126.587736	COAST	5	2.803214	5	2.803214	5	2.803214	0.909091	0.09224	0.09224	0	0
4	340	Polygon	71595	54126.587736	COAST	5	0.423965	5	0.423965	5	0.423965	0.909091	0.378552	0.378552	0	0
5	406	Polygon	63249	54126.587736	COAST	5	0.676601	5	0.676601	5	0.676601	1.363636	0.228512	0.228512	1.25	0.032518
6	423	Polygon	21666	54126.587736	COAST	5	1.777044	5	1.777044	5	1.777044	1.363636	0.311631	0.311631	1.25	0.006259
7	505	Polygon	70803	54126.587736	COAST	5	0.243601	5	0.243601	5	0.243601	0.909091	0.04717	0.04717	0	0
8	745	Polygon	62729	54126.587736	COAST	5	2.357177	5	2.357177	5	2.357177	0.909091	0.258061	0.258061	0	0
9	757	Polygon	21148	54126.587736	COAST	5	4.880499	5	4.880499	5	4.880499	0.909091	0.097645	0.097645	0	0
10	759	Polygon	21147	54126.587736	COAST	5	0.04989	5	0.04989	5	0.04989	1.363636	0.488624	0.488624	1.25	0.000313
11	859	Polygon	70283	54126.587736	COAST	5	0.962048	5	0.962048	5	0.962048	0.909091	0.023896	0.023896	0	0
12	919	Polygon	21407	54126.587736	COAST	5	1.579557	5	1.579557	5	1.579557	0.909091	0.228408	0.228408	0	0

A5.1.2.5 Compute EV in all ECs and clean them

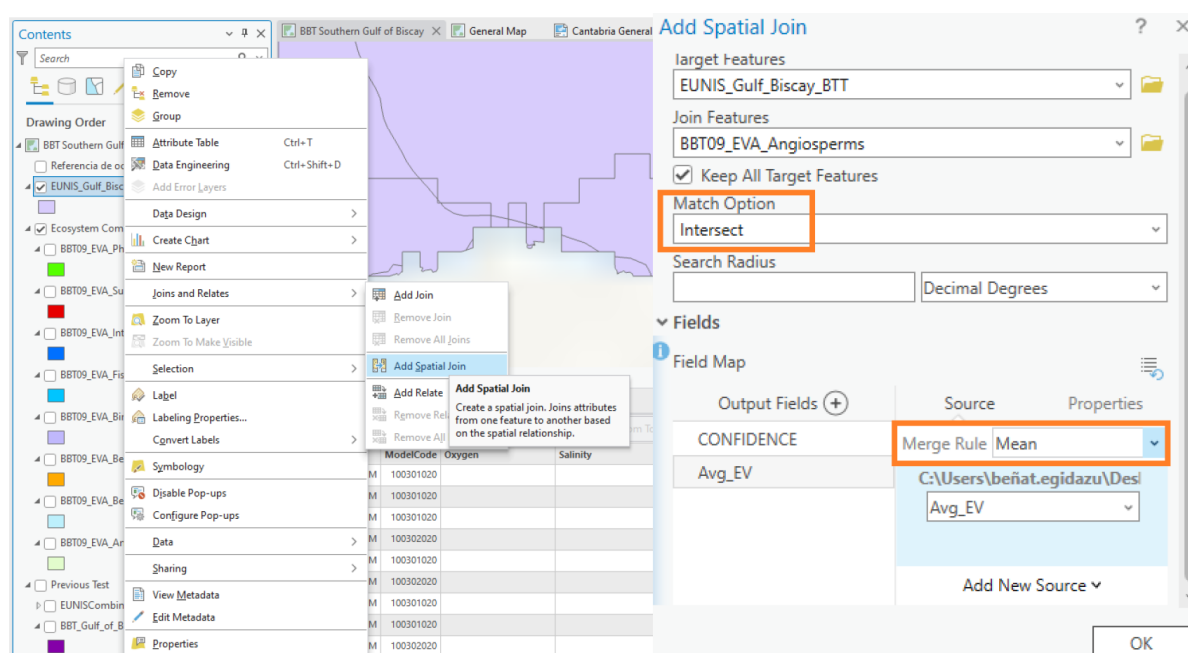
Repeat the steps 1.2.3 and 1.2.4 to compute EV for all ECs and clean *NoData* subzones.

A5.1.2.6 Integrate EVAs into EUNIS habitat maps

First, we create in the EUNIS habitat file new *double* fields where average EV & Confidence scores will be stored. Create one EV & Confidence field for each Ecosystem Component.

Fields: EUNIS_BBT_So...Gulf of Biscay) X						
Current Layer EUNIS_BBT_Southern_Gulf_Biscay (▾						
	Visible	Read Only	Field Name	Alias	Data Type	Allow NULL
☒	☒	☐	EV_Angio	EV Angiosperm	Float	☐
☒	☒	☐	CONF_Angio	CONFIDENCE Angiosperm	Float	☐
☒	☒	☐	EV_Benth	EV Benthic Habitats	Float	☐
☒	☒	☐	CONF_Benth	CONFIDENCE Benthic Habitats	Float	☐
☒	☒	☐	EV_MInv	EV MacroInvertebrates	Float	☐
☒	☒	☐	CONF_MInv	CONFIDENCE MacroInvertebrates	Float	☐
☒	☒	☐	EV_IMacr	EV IntertidalMacroalgae	Float	☐
☒	☒	☐	CONF_IMacr	CONFIDENCE IntertidalMacroalgae	Float	☐
☒	☒	☐	EV_SMacr	EV SubtidalMacroalgae	Float	☐
☒	☒	☐	CONF_SMacr	CONFIDENCE SubtidalMacroalgae	Float	☐

Then, we compute the EV and Confidence of ECs using the average EV and Confidence of all ECs. Each hexagon of the ECs that fall into each of the EUNIS habitat polygons will be taken into account. Do a 'Spatial Join' between the EUNIS habitat file and each EVA shapefiles. Define Match Option as Intersect and Merge Rule of EV and Confidence fields as Mean.



Copy average EV and Confidence results into the fields created in the beginning of this step (If there is no hexagons intersecting an EUNIS habitat polygon ArcGIS Pro defines the value as null and then, when we copy the result into the fields created in *Step g*, this null values are set to 0).

Repeat the steps described in 1.2.6 with all EC EVAs to add average EV and Confidence information to the EUNIS habitat file.

A5.1.2.7 Compute Overall habitat EV and Confidence scores

Compute the overall EV & Confidence scores of each EUNIS polygon using the 'Field Calculator'. Use the following expressions with the corresponding field names to avoid including *NoData* values in the computation. First, create two *double* new fields to store the TOTAL_EV & TOTAL_CONFIDENCE.

Calculate Field ? x

i This tool modifies the Input Table x

Input Table
EUNIS_BBT_Southern_Gulf_Biscay

Field Name (Existing or New)
Overall EV

Expression Type
Python 3

i Expression

Fields

CONFIDENCE Benthic Habitats
EV MacroInvertebrates
CONFIDENCE MacroInvertebrates
EV IntertidalMacroalgae
CONFIDENCE IntertidalMacroalgae
EV SubtidalMacroalgae
CONFIDENCE SubtidalMacroalgae
Overall EV

Helpers

.as_integer_ratio()
.capitalize()
.center()
.conjugate()
.count()
.decode()
.denominator()

Insert Values

Overall_EV =

compute_mean(!EV_Angio!, !EV_Benth!, !EV_MInv!, !EV_IMacr!, !EV_SMacr!)

Code Block

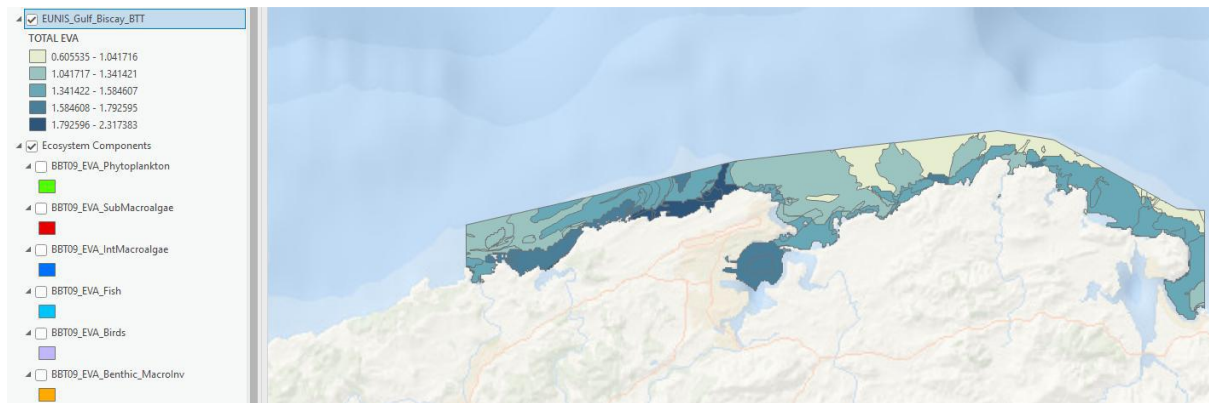
```
def compute_mean(*values, nodata_value = 0):
    # Filter valid values
    valid_values = [v for v in values if v != nodata_value]
    # Computes mean if there are valid values
    if valid_values:
        return sum(valid_values)/ len(valid_values)
    # If there are not valid values return 0
    return 0
```

✓ Expression is valid

☐ Enforce Domains

Enable Undo ☐ Apply OK

As a result, we obtain the EUNIS habitat map with a field (TOTAL_EV) ranking the areas with the highest EV.



A5.1.2.8 Group EV and Confidence by EUNIS habitat type

Compute area-weighted EV and Confidence scores based on EUNIS habitat types (“AllcombD”) and areas (“area”) of each polygon. First, create a new field to store the area of each polygon and use ‘Calculate Geometry’ to compute the area of each EUNIS habitat polygon. Then, create two new fields to store Grouped_EV (equals to Habitat_EV in the QGIS guide) & Grouped_CONFIDENCE (equals to Habitat_CONF in the QGIS Guide). Compute them using ‘Calculate Field’ tool and the following expression.

Annex 6: Monetary Account Tables (Guide 2)

Table A6.1 and Table A6.2 are respectively examples of supply and use tables for only one year.

Table A6.1: MARBEFES Supply Table in monetary terms for one year only (e.g. 2022)

ECOSYSTEM SERVICES	UNIT OF MEASURE	COASTAL AND MARINE ASSETS					
		ECOSYSTEM TYPES (EUNIS CLASSIFICATION)					
		MB252	MC3521	MC3517	MC352	MB12	Others
Wild fish	currency	Money value	Money value	Money value	Money value		
Carbon sequestration and storage	currency		Money value	Money value			
Provisioning (Seaweed)	currency					Money value	

Table A6.2 MARBEFES Use Table in monetary terms for one year only (e.g. 2022)

ECOSYSTEM SERVICES	UNIT OF MEASURE	USERS				
		SOCIETY	INDUSTRY			
		Households and government	Fish sector	Tourism sector	Cosmetic sector	Other sector
Wild fish	currency	Money value	Money value			
Carbon sequestration and storage	currency	Money value				
Provisioning (Seaweed)	currency	Money value			Money value	

Annex 7: Revealed Preference Methods (Guide 2)

A7.1 The single-site travel cost

In practice, estimating a single-site travel cost model and deriving the relevant welfare values takes the following steps.

1) Data Collection. The first step is to collect data on visitors' behaviour and travel costs. Such data can be collected on-site (interviewing visitors directly on the studied site) or off-site, using for example an online survey. Researchers collect information on the frequency of visits, the distance that visitors are traveling to reach the site (which can be calculated by asking about the location of the respondents' homes), the costs associated with traveling to the site (such as transportation costs, parking fees and entrance fees). Researchers typically also collect data on the socioeconomic characteristics of visitors, such as income level or education and on the activities carried out at the site, including the length of their visits. As part of the travel cost it is also important to include the opportunity value of travel time because the time spent traveling is also a resource (remember one of the assumptions of this model is that travel time does not provide utility). Recent research shows that one can set such a value equal to 3/4 of the average gross wage rate (Fezzi et al., 2014). For example, if the average hourly wage rate in the sample is \$10/hour, the value of travel time should be set as \$7.5/hour.

2) Econometric modelling. The typical approach to estimate equation (1) on the data collected at the previous step is to use a Poisson regression. This is the most established method when dealing with a dependent variable which is an integer representing counts (i.e. number of visits). The Poisson distribution can be written as:

$$\Pr(x = n) = \frac{e^{-\lambda} \lambda^n}{n!}, n = 0, 1, 2, \dots \quad (2)$$

Where λ is the mean of the distribution (i.e., the mean number of visits) that can be specified as:

$$\lambda = \exp(\beta_1 TC + \beta_2 Z_1 + \dots + \beta_{k+1} Z_k) \quad (3)$$

where all symbols are defined as before, i.e., $Z_i, \dots, Z_k$ indicate the different socio-economic characteristics and $\beta_1, \dots, \beta_{k+1}$ are the parameters to be estimated using maximum likelihood. This model can be augmented by also including the travel cost to a nearby site, in order to acknowledge possible substitution effects between sites. A weakness of the Poisson specification is that this distribution assumes the variance to be equal to the mean which, typically, leads to significantly underestimating the parameters' standard errors. This assumption can be relaxed by using a quasi-maximum likelihood estimator or by implementing alternative count-data models such as the negative binomial. The negative binomial model can deal, at the same time, with both overdispersion and endogenous stratification generated by on-site sampling and, therefore it is the suggested approach when one needs to deal with both of such issues.

3) Economic valuation. Once the model parameters have been estimated, they can be used to predict the expected number of visits for any combination of explanatory variable values. Using the notation in (3) those will simply be:

$$\lambda^* = \exp(\hat{\beta}_1 TC + \hat{\beta}_2 Z_1 + \dots + \hat{\beta}_{k+1} Z_k) \quad (4)$$

Where the "hats" indicate the parameters' estimates. This can be useful for policy-makers who are interested in evaluating the impact of changes in travel costs or site quality on visitation behaviour. The willingness to pay for access (i.e., the welfare value represented in Figure 4.4), when using both the Poisson and the Negative Binomial distributions, can be calculated as:

$$WTP(access) = -\frac{\lambda^*}{\hat{\beta}_1} \quad (5)$$

This information can be used to make decisions about park management and conservation, as well as to inform policies related to park access and funding. For example, one can calculate how the number of visits and welfare would change with an introduction or a variation of the access fee. However, this approach does not allow us to understand how welfare and visits change following a change in the site's environmental quality.

A7.2 The multi-site travel cost

The random utility model can be written as:

$$U_{nj} = \beta_1 TC_{nj} + \beta_2 Z_{1j} + \dots + \beta_{k+1} Z_{kj} + e_{nj} = V_{nj} + e_{nj} \quad (6)$$

With U_{nj} is the utility for individual n when visiting site j , TC_{nj} is the round-trip travel cost from the house of the respondent n to site j (including the value of time) and $Z_{1j}, \dots, Z_{kj}$ are the different characteristics of site j . The term e_{nj} is the random component, representing the portion of the utility associated with the visit to the site which is unobserved by the researcher, while the deterministic part of the utility (observed by the researcher) is indicated with V_{nj} .

Of course, utility (or well-being) is not observed. However, we can assume that each individual will choose to visit the site characterized by the highest utility. The parameters in (4) can then be estimated within a discrete choice framework. In particular, when e_{nj} is distributed as extreme value type I (Gumbel), we obtain the Conditional Logit Model (CLM) formulation, as shown by McFadden (1974).

Within a CLM, the probability of individual i visiting a site j can be written as:

$$p_{nj} = \frac{\exp(V_{nj})}{\sum_{j=1}^J \exp(V_{nj})} \quad (7)$$

where J indicates the total number of recreational sites included in the analysis.

The multi-site random utility approach can be implemented following the steps below.

1) *Data Collection*. The first step is to choose the area of study and identify all the different outdoor recreation sites included in the analysis. One could decide to include all the beaches of a region, or even all possible outdoor sites including not only coastal sites but also city parks, trails, etc. This type of analysis can include any number of sites from about 10 to more than 200. The second step is to collect data on site characteristics, visitors' behaviour and travel costs. Such behavioural information typically needs to be collected off-site, for example by using an online survey or interviewing tourists and residents in person, at malls, airports, etc. One option is to collect information on the last outdoor site visited in the area. Another (and more informative) option is to ask which sites in the area the respondent has visited during a time interval, such as the previous week or month. Researchers can then calculate the distance from the visitors' homes to all the sites in the analysis and the costs associated with traveling to the different sites, including the value of travel time, as explained in the previous section.

2) *Econometric modelling*. The CLM in equation (5) can be estimated via maximum likelihood. If the data includes information on multiple trips the model needs to be extended and one approach is the repeated discrete-choice travel-cost framework introduced by Morey et al., (1993). In this approach, we assume that the time horizon of interest can be decomposed into a set of choice occasions (typically, one day corresponds to one choice occasion), in which individuals make repeated and independent decisions on whether to partake in outdoor recreation and on which site to visit. The sum of outdoor visits represents the demand function over that time horizon. This approach allows us to understand both outdoor recreation participation and site choice in a behaviorally-consistent manner that permits the calculation of individuals' welfare changes associated with both sites' quality variations and sites' availability. In this framework, we need to include site $J + 1$ to represent the choice of not partaking in outdoor recreation (i.e., "staying at home"). Indicating with N the number of individuals in the survey and with T the total number of choice occasions and with y_{njt} a dummy

variable equal to 1 if respondent n chooses site j in choice occasion t , the log-likelihood of this model becomes:

$$\ln L = \sum_{t=1}^T \sum_{n=0}^N y_{njt} \ln(p_{njt}) \quad (8)$$

Note that in this formulation the order of the visits does not matter, but only the total number of times that a respondent has visited each site during the time period of interest. If the information collected is only about the last visited site, then the likelihood is the same as in equation (8) with $T = 1$.

3) *Economic valuation*. Once the parameters are estimated, Hanemann (1999) shows how to calculate welfare values associated with sites' quality changes. These refer to alterations of ecosystem characteristics (e.g., a decrease in coral cover due to climate change) or anthropogenic factors (e.g., the presence/absence of a lifeguard at the beach) and can happen at one specific site or at multiple sites at the same time. It can also be used to estimate the impact of the introduction of an entrance fee to one or more of the sites. For each choice occasion, the variation in welfare for individual n associated with changes in sites' qualities can be written as:

$$\theta^{-1} \ln \left(\frac{\sum_{j=0}^J \exp(V_{nj}^*)}{\sum_{j=0}^J \exp(V_{nj})} \right) \quad (9)$$

where V_{nj}^* refers to the attributes of the sites after the quality change. Note that this equation refers to one choice occasion only and, therefore, it needs to be multiplied by the total number of choice occasions in the time span of interest. The same formula can be used to calculate the value of access to a site or to more sites. One simply needs to remove the sites from the J sites in the numerator.

This equation can be used to calculate the welfare changes for each individual in a sample or, alternatively, aggregated at a local or regional level by multiplying welfare changes by the number of people resident in a certain area. Such welfare changes refer only to the use values from recreation. It is worth repeating that they can be thought of as the willingness to pay of each individual n associated with the changes in quality from V_{ij} to V_{ij}^* for all possible sites J . As mentioned before, this approach is very flexible since a researcher or a policy maker can simulate variation in any of the variables in model (4) at any of the sites in the region and, also, simulate site closure. In each scenario, the model can be used to forecast the changes in economic benefits in terms of recreation opportunities and, also, the number of visits to each site.

A7.3 The hedonic pricing method

$$P = f(S, N, E) \quad (10)$$

where P is the price of the real estate property, S is a vector that captures its structural characteristics (area, number of bedrooms, number of bathrooms, floor height, existence of an elevator in the premises...), N is a vector that comprises neighborhood characteristics (safety, proximity to schools, public services, shopping centers...) and E is a vector that includes the environmental characteristics appreciated by the market, which are the object of the evaluation. Note that, while it is true that most hedonic pricing applications use real state transactions as data basis, any good whose characteristics are well known, including its environmental attributes and for which there exists a smooth market, might also perform.

The most common specification used for equation (10) is semilogarithmic:

$$\ln P = B_0 + B_{S_1} \cdot S_1 + B_{S_2} \cdot S_2 + \dots + B_{N_1} \cdot N_1 + B_{N_2} \cdot N_2 + \dots + B_{E_1} \cdot E_1 \quad (11)$$

Let's assume that P reflects the price of apartments in a city and that E_1 captures a given environmental quality attribute, be it air quality, silence, or views, measured in a suitable unit. It is expected that, other things equal, the prices of apartments with better environmental quality (larger E_1) will be higher. The statistical estimation of the coefficients of equation (11), from a sufficiently

large sample, makes it possible to quantify the effect on the price of apartments that is attributable to each of their characteristics (area, public services, etc.), including environmental quality from coefficient B_{E_1} .

More specifically, an improvement in environmental quality from E_1^0 to E_1^1 , all other characteristics remaining equal, will result in an increase in the price of apartments from P^0 to P^1 , corresponding to a growth rate g :

$$g = P^1/P^0 - 1 \quad (12)$$

From equation (11), it can be shown that the effect of an improvement in environmental quality considering (12), can be expressed as:

$$B_{E_1} \cdot (E_1^1 - E_1^0) = \ln P^1 - \ln P^0 = \ln (P^1/P^0) = \ln (1 + g) \quad (13)$$

Hence,

$$g = \exp(B_{E_1} \cdot (E_1^1 - E_1^0)) - 1 \quad (14)$$

If E_1 is a dummy variable, for instance, the enjoyment of views over the bay where the town is located, $E_1^1 - E_1^0 = 1$ and (14) can be reduced to a much simpler expression:

$$g = \exp(B_{E_1}) - 1 \quad (15)$$

Once this growth rate g is obtained, it can be used to estimate the economic value of the environmental attribute. The consistency of the model requires that the underlying markets function correctly, mainly that there is mobility in the real estate market and that transaction costs, primarily those related to information on market conditions and the making of sales and purchases, are low. Sampling is generally cross-sectional, taking real estate data from purchase transactions taking place in a specific interval and controlling for relevant changes. Given that the set of characteristics that influence the price of real estate is very broad and that not all of them are orthogonal (independent of each other), it is advisable that the sample size be high in order to reduce the variance of the estimators. It is also advisable that the real estate considered in the sample be homogeneous so that potential hidden variables do not increase the variance of the estimators.

Annex 8: Stated Preference Methods (Guide 2)

A8.1 The Random Utility Theory

When modeling stated preferences, the random utility model is the theoretical approach. Random utility assumes that individuals make rational choices by selecting the option that maximizes their utility. Utility is divided into two components: a deterministic and a random component. The basic idea is that people receive utility that depends on observable and not observable characteristics (Manski, 1977). The unobservables could be characteristics of the individual and/or attributes of the items considered for choice and/or variation in preferences among the population and measurement error or missing data.

In CV, a random utility model of the unobserved willingness to pay can be described as follow:

$$\widehat{WTP}_n = \beta X_n + \varepsilon_n \quad (1)$$

Where β are the parameters to be estimated, X_{int} the explanatory variables (e.g., gender, age, education) and ε the error term, assumed to be distributed normally with mean zero and standard deviation σ .

In DCE, the utility of respondent n for alternative i in the choice situation t can be represented with a linear in the parameter's utility function:

$$U_{int} = \beta X_{int} + \varepsilon_{int} \quad (2)$$

Where β are the parameters, X_{int} the attributes of the alternative i and ε the error term $\sim N(0, \sigma^2)$.

Tests for non-linearity can be used to control for non-linear specifications (e.g., a quadratic form) (Mariel et al., 2021).

A8.2 Contingent Valuation

The data analysis process varies with the question format used for the contingent- valuation question (Boyle, 2017).

Analyses of dichotomous-choice data and payment card require econometric analyses.

The probability of saying “yes” to a given amount is estimated with a probit or logit model.

With a probit model, the probability that a respondent would be willing to pay a given *bid* amount is assumed to follow a normal distribution:

$$\text{Prob}(y_i = 1) = \varphi(\beta X_i) = \int_{-\infty}^{X_i \beta} \varphi(z) dz \quad (3)$$

With a logit model, the probability that a respondent would be willing to pay a given *bid* amount is assumed to follow a logistic distribution:

$$\text{Prob}(y_i = 1) = \frac{\exp(\beta X_i)}{1 + \exp(\beta X_i)} \quad (4)$$

The explanatory variables X are chosen based on economic theory and previous empirical studies.

Maximum Likelihood Estimation is used to estimate the model's parameters.

The mean WTP for dichotomous choice data is calculated by:

$$\text{mean WTP} = \frac{\bar{X}\beta'}{\beta_{bid}} \quad (5)$$

Where $\bar{X}$ represents the mean of the explanatory variables, β' the vector of estimated parameters and β_{bid} the estimated coefficient of the *bid*. The confidence intervals are normally obtained using the Krinsky and Robb method (1986).

Payment card data can be estimated using the Cameron and Huppert model (1989), modeling the interval where respondents have revealed that their values reside and assuming the respondent's WTP is included between the chosen bid and the following one. Bids are sorted from the lower to the higher value and represented by the vector $t = (t_1, t_2, \dots, t_j)$, where t_j is the generic *bid* selected by the i -th respondent. The probability that the generic t_j is selected by the i -th individual is:

$$\Pr(t_j) = \Phi\left(\frac{t_{j+1} - X_i' \beta}{\sigma}\right) - \Phi\left(\frac{t_j - X_i' \beta}{\sigma}\right) \quad (6)$$

Willingness to pay can be derived as:

$$E(WTP) = \exp(\hat{\beta} x_i) \exp\left(\frac{\hat{\sigma}^2}{2}\right) \quad (7)$$

With an open-ended question, a simple mean is computed after having deleted from the sample protest bids and outliers, or to avoid negative estimates of WTP, censored regression models are used, such as the Tobit model (Tobin, 1958).

A8.3 Discrete Choice Experiments

The probability of choosing one alternative is estimated through probit and logit models. The most used models are the multinomial logit model (MNL), the random parameter logit model (RPL) and the latent class model (LC).

Assuming that the error term is i.i.d. Extreme Value Type I distributed, the probability that a respondent n chooses alternative i in a sequence of choices t can be modeled with a conditional logit model (MNL):

$$P_{ni} = \prod_{t=1}^T \frac{e^{\beta' x_{ni}}}{\sum_j e^{\beta' x_{nj}}} \quad (8)$$

The standard MNL has the limitation of providing a point estimate for each coefficient, equivalent to assuming preference homogeneity for the entire sample. Such a condition is not likely to hold; therefore, analysts are often concerned with estimating more flexible models that account for taste heterogeneity.

The RPL assumes that coefficients are individual-specific and follow a random distribution, for which a location and a scale parameter are estimated (Train, 2009):

$$P_{ni} = \int \frac{e^{\beta' x_{ni}}}{\sum_j e^{\beta' x_{nj}}} \varphi(\beta|b, \Omega) d\beta \quad (9)$$

In which $\varphi(\beta|b, \Omega)$ is the probability density function of the distribution of the coefficients.

LC model addresses the issue of individual heterogeneity, assuming a discrete mixing distribution for the parameters β , with individual parameters clustered in classes (Greene & Hensher, 2003).

Given membership of class c , the probability of the respondent n 's sequence of choices is given by:

$$P_{n|c} = \prod_{t=1}^T \frac{\exp(X_{int}' \beta_c)}{\sum_{j=1}^J \exp(X_{jnt}' \beta_c)} \quad (10)$$

The unconditional probability of choosing alternative i is a weighted average of all the parameter estimates β_c for each class c :

$$P_{ni} = \sum_{c=1}^C \pi_c P_{n|c} = \sum_{c=1}^C \pi_c \prod_{t=1}^T \frac{\exp(X_{int}' \beta_c)}{\sum_{j=1}^J \exp(X_{jnt}' \beta_c)} \quad (11)$$

where π_c is the probability of belonging to the class c .

Willingness to pay is calculated as the ratio of non-monetary coefficients and the cost coefficient:

$$WTP_i = -\frac{\beta_i}{\beta_{cost}} \quad (12)$$

It is possible to identify the source of preference heterogeneity, including some socio-economic (e.g., age, gender, income), behavioral and attitudinal variables in the estimation. These variables enter the model by interacting with the attributes. When the variables used to explain the heterogeneity of preferences are endogenous, i.e., they depend on other variables of the model itself (e.g., attitudinal variables) and are collected through Likert-scale, hybrid models which take into account endogeneity and measurement errors should be used (Ben-Akiva et al., 2002).

Annex 9: Benefit Transfer Methods (Guide 2)

There are different approaches (methods) to benefit transfer, which can be classified into four groups:

- Unit value transfer. The policy site is considered to be quite similar (or similar enough) to the study site, so values are assumed to be transferable, be it as point estimates or as measures of central tendency (mean or median), without requiring any further adjustment. If V_j is the value for a certain ecosystem service at a study site j , then

$$VBT_i = V_j \quad (16)$$

where VBT_i is the estimated value by benefit transfer at policy site i .

- Adjusted unit value transfer. The socioeconomic context r at the policy site i is considered to be different from the context s and the study site j , so values are adjusted for differences in income or wealth to be transferred:

$$VBT_{ir} = f(V_{js}) \quad (17)$$

where $f(\cdot)$ is a suitable adjustment function. In its simplest form, this adjustment function might be:

$$f(V_{js}) = P \cdot V_{js} \quad (18)$$

where P is an appropriate price index to account for differences in purchasing power.

- Value function transfer. This approach is based on the existence of validated value functions constructed on ecological and socioeconomic parameters identified for study sites and observable for policy sites. The corresponding parameters at the policy site are then incorporated into the value functions to estimate its economic value. In the simplest form, the value function is estimated from a single primary study, which implies the strong assumption that the implicit value drivers are identical at the study and policy sites. If this assumption cannot be sustained, more complex forms of value functions must be considered, usually combining information from multiple sites applying statistical methods to provide a range of possible values for the policy site. In both cases, the value function $g(\cdot)$ is of the form:

$$V_{js} = g(E_{js}, S_{js}, B) \quad (19)$$

where V_{js} is the estimated value at site j with context s (say WTP estimates from revealed or stated preferences), E_{js} is the corresponding vector of ecological variables, S_{js} is the corresponding vector of socioeconomic variables and B is the vector of value parameters to be estimated by statistical regression. Then, the benefit transfer estimated value VBT for policy site i and socioeconomic context r would result from

$$VBT_{ir} = g(E_{ir}, S_{ir}, B) \quad (20)$$

where E_{ir} and S_{ir} are the corresponding vectors of ecological and socioeconomic variables at site i and context r and B is the vector of value parameters estimated from equation (19).

- Meta-analysis. It can be defined as a quantitative synthesis of evidence on a certain empirical outcome, with evidence gathered from prior primary studies. For benefit transfer, meta-analysis is usually undertaken by applying meta-regression models, using the results from existing primary studies as observations. Dependent variables are the values obtained in primary studies and explanatory variables may include all sorts of relevant attributes from the primary studies suitable to generate influence, including ecological and socioeconomic characteristics as well as, in this case, study characteristics related to the valuation method and its exercise. The value function $h(\cdot)$ is of the form:

$$V_{js} = h(E_{js}, S_{js}, T_{js}, B) \quad (21)$$

where V_{js} is the estimated value at site j with context s , E_{js} is the corresponding vector of ecological variables, S_{js} is the vector of socioeconomic variables, T_{js} is the vector of study method variables and B is the vector of value parameters to be estimated by statistical regression. The benefit transfer estimated value VBT for policy site i and socioeconomic context r would result from

$$VBT_{ir} = h(E_{ir}, S_{ir}, T_v, B) \quad (22)$$

where E_{ir} and S_{ir} are the corresponding vectors of ecological and socioeconomic variables at site i and context r , T_v is a vector of standardized values to compensate for possible method bias and B is the vector of value parameters estimated from equation (21).

Annex 10: Socio-Cultural Valuation Sample Questionnaire (Guide 3)

Information sheet

[Note: you can edit the below to adhere to your institution's ethical guidelines or requirements.]

This research is entitled “The Social and Cultural Benefits of Dublin Bay” and it is a subproject on the larger European Union-funded MARBEFES project. It is being led by [insert researcher names and organisations].

We aim to understand key social and cultural benefits of marine biodiversity and ecosystems. These are benefits such as recreation, social bonding, local identity, spirituality, art, tradition and education. We want to understand how to better protect these benefits through environmental policy. We also want to understand how these benefits are (or are not) tied to healthy ecosystems.

The questionnaire should take approximately 20 minutes to complete.

Your anonymous answers will be stored securely on the ??? online platform and robust data analysis softwares. The answers will be analysed to understand the cultural benefits that people derive from Dublin Bay.

If at any stage you have concerns about your rights as a participant in this study, you can contact the Data Protection Officer at [name of organisation] by email at [email address]. If you are not satisfied with [name of organisations]’s response, or feel we are not processing your data in accordance with the law, you can complain to the Irish Data Protection Commission [give website link].

You can see the results when they are gathered on the project website: www.marbefes.eu. You may email [provide name and contact details] if you have any questions about this questionnaire or how to view the results.

Please sign here [or tick] to confirm that you are happy to participate in the questionnaire.

Interview ID:	Interviewer:	Site:
Entered in Database:	Date:	Time:

Marine Biodiversity and Ecosystem Functioning Leading to Ecosystem Services
(MARBEFES)

The Social and Cultural Benefits of Dublin Bay

Age: ☐ 0-25 ☐ 26-45 ☐ 46-65 ☐ 66+ **Gender:** ☐ Male ☐ Female ☐ Prefer not to say

Where do you live? ☐ Dublin City ☐ Dublin County ☐ Elsewhere in Ireland ☐ Abroad

YOU AND THE BAY

1. How often do you visit the Dublin Bay coastline or sea?

☐ Daily ☐ At least once a week ☐ At least once a month ☐ At least once a year ☐ This is my first time
☐ Other [Please describe] _____

2. Why do you visit? [Please tick all that apply]

[Note: you can edit the options here and in the next question to reflect or include key social and cultural activities in your BBT.]

☐ Swimming ☐ Relaxation ☐ Enjoying the views ☐ Water Sports
☐ Cycling ☐ Jogging ☐ Visiting a place or amenity ☐ Wildlife viewing
☐ Dog walking ☐ Photography ☐ Walking ☐ Other (s) [Please describe]:

3. What benefits do you associate with the Dublin Bay coastline or sea? [Please tick all that apply]

[Note: you should check the stakeholder results from the 2023 WP1 interviews to make sure these options include socio-cultural benefits indicated as important by MARBEFES stakeholders in your area. You can contact Herman Hummel and team from WP1, or Ashley Cahillane from WP4.]

☐ Contact with nature ☐ Leisure ☐ Awe
☐ Traditions ☐ Home ☐ Cultural Heritage
☐ Educational ☐ Physical exercise ☐ Scenic

☐ Mental Health ☐ Sense of Community ☐ Tourism

☐ Other(s) [Please describe]: _____

4. Please give a bit more detail about the benefits you have ticked above:

5. What is special about the Dublin Bay coast or sea?

[If possible, provide an image of a grid-marked map of the targeted location here. Ask the participant to mark (e.g. A2, H3) (i) three places they visit most and why and (ii) two places they visit least and why]

ENVIRONMENTAL CHANGE

6. Do you think the health of the coastal and marine environment in Dublin Bay is important for the way you use the Bay? [1 to 5]

☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Very unimportant	Quite unimportant	Neither important or unimportant	Quite Important	Very Important

7. How healthy do you think the coastal and marine environment is in Dublin Bay? [1 to 5]

☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Very Unhealthy	Quite Unhealthy	Neither Unhealthy or Healthy	Quite Healthy	Very Healthy

8. What influences the health of the coastal and marine environment in Dublin Bay? Why is this happening?

9. Over the past 10 years, have you noticed any changes in the coastal environment? If so, when and why did this happen?

10. Have any of these changes affected the way you use Dublin Bay? (*about question 9*)

11. Select the 3 future scenarios below which would most affect the way you engage with Dublin Bay?

- | | | |
|--|---|---|
| ☐ Changes in biodiversity | ☐ Land reclamation | ☐ Oil spill |
| ☐ Climate Change | ☐ Flooding | ☐ Sea level rise |
| ☐ Coastal erosion | ☐ Increase in recreational use | ☐ Unplanned development |
| ☐ Drilling | ☐ Litter | ☐ Other[please describe]:_____ |

MANAGEMENT

12. Who do you think is responsible for looking after the coastal and marine environment in Dublin Bay?

13. What do you currently do to protect the bay?

14. How do you think the bay can be protected in future?

Thank you for completing the survey “The Social and Cultural Benefits of Dublin Bay”. Your views will help us to understand key perceptions and uses of the bay.

You can see the results when they are gathered on the project website www.marbefes.eu. You can keep up with the project via the website and social media pages [provide links].

You may email [provide name and contact details] if you have any questions about this questionnaire or how to view the results.

Annex 11: Socio-Cultural Valuation Data Analysis (Guide 3)

Data analysis will depend on the research and/or policy questions you seek to address, plus the expertise, resources and time on your team. Below is a broad overview of the types of possible analyses, which you can tailor to your research plan.

A11.1 Participatory findings

For the quantitative data relating to spatial distribution, researchers can first code all the locations where CES are identified. Coding involves creating labels for small elements of data to identify patterns such as frequency of mention or relationship between elements. Researchers can then note numerical patterns in terms of (i) quantity of CES identified in particular places and (ii) different types of CES identified in different places. Researchers can also code mentions of visitations to and use of particular places, in order to determine links between direct engagement and CES.

Import the coded place names into a geographic information systems software such as QGIS or ArcGIS. You can overlay them with layers such as statutory zones (e.g. Natura 200) or socio-economic considerations (e.g. employment, deprivation). This creates “hotspots” of cultural value instantly visible to planners and highlights potential symbioses or mismatches between valued places and their current use patterns or legal protection. Researchers can look at where there are discrepancies in how different areas may be being prioritised and the implications of this. Researchers can add an intensity index, weighting each site by (a) number of CES types and (b) frequency of visitation codes; then produce a graduated symbol map so policymakers can see where relatively small areas deliver outsized cultural benefits.

See Figure A11.1 for a vertical flowchart of potential steps involved in analysing the spatial data.

Analysing the Spatial Data

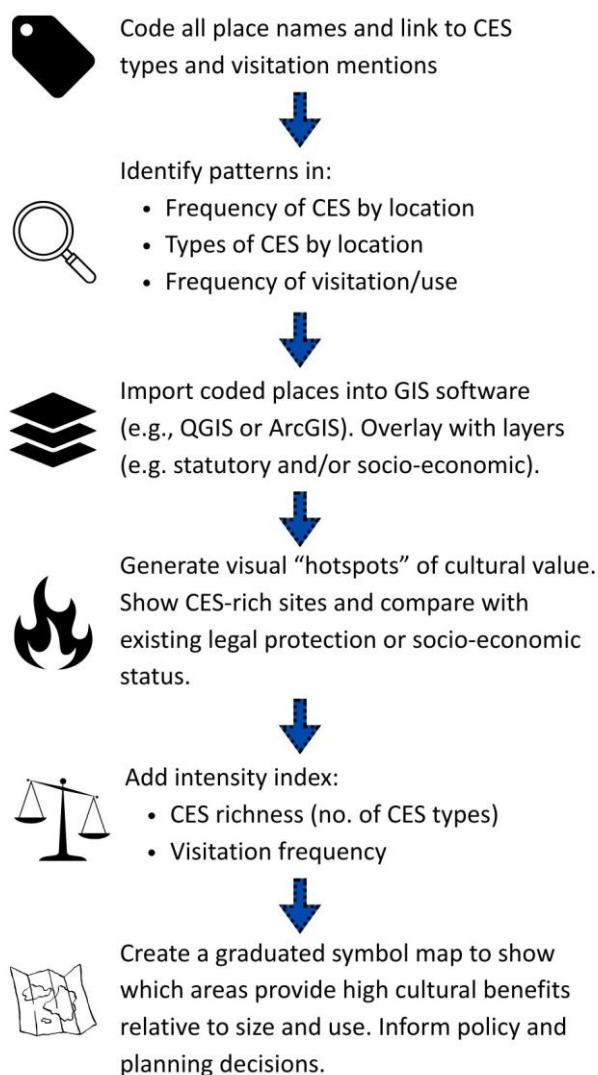


Figure A11.1: A vertical flowchart of potential steps involved in analysing the spatial data.

The data from all the forms of participatory research—interviews, focus groups, surveys and arts-based workshops—should be transcribed and coded, perhaps using Nvivo 12 software (QSR International, 2018).

Researchers can take an inductive (i.e. allowing the code labels to emerge from the data rather than applying a pre-existing framework) or deductive (i.e., using a pre-existing framework to determine code labels) approach to coding, or a combination of both. For example, an inductive approach can first be used to discern the CES that arise in a particular place. Perceived benefits/disbenefits from or threats to CES should also be coded. A second round of coding could then commence, possibly using a deductive approach based on frameworks familiar to policymakers, such as IPBES or the Common International Classification of Ecosystem Services (CICES). Where appropriate, map CES codes onto the SEEA-EA cultural-services account headings, so the results can be picked up by agencies compiling

natural-capital accounts (UN, 2021). Results can then be organised into categories to demonstrate links between CES and particular wellbeing benefits.

Researchers should consider and analyse questions of social diversity, equity and power within their results. For example, researchers can code stakeholder type and plot “influence over ecosystem decisions” on one axis and “degree of cultural benefit” on the other. Highlight groups in the low-influence/high-benefit section to justify their inclusion in policy drafts. Flag whose stories are absent altogether.

The data from the advisory meetings and the arts-based creative workshops may need more interpretive work as the data has been subjected to crafting and revision by participants and researchers. In these cases, environmental humanities analysis methods and theoretical frameworks (such as close reading methods in literary studies (Belsey 2013; Brannigan 2015; Gladwin 2018) may also need to be applied to navigate the symbolic connotations and aesthetic affectations that will be layered onto the data. An environmental humanities theoretical framework can help to narrow focus by directing attention to key thematic or aesthetic concerns to target.

From close-reading, extract recurrent symbols (e.g. “working harbour”, “sacred well”) and treat them as intangible heritage assets—policymakers handling cultural or tourism development can act on them directly. Coded CES and benefits can also be fed into small, follow on workshops with stakeholders to devise future scenarios or storylines for nature management. Creative scenarios can be combined with a “policy laboratory”, cross-referencing which current regulations help or hinder which storylines.

See Figure A11.2 for a panel flowchart of potential steps involved in analysing the qualitative data.

Analysing the Qualitative Data

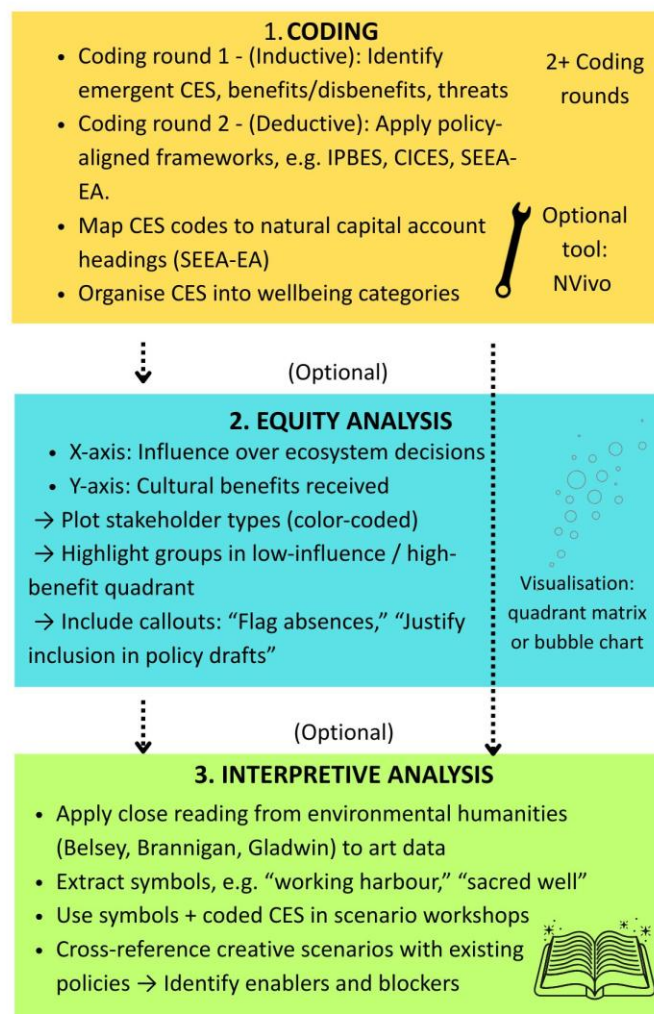


Figure A11.2. A panel flowchart of potential steps involved in analysing the qualitative data.

A11.2 Cultural Representations findings

Relevant cultural representations should initially be logged in tables or databases (Ryfield et al., 2019). Adopt a lightweight Dublin-Core or Text Encoding Initiative (TEI) header for each item (identifier, creator, date, place, rights, CES codes). Using heritage-sector metadata keeps the collection accessible and compatible with UNESCO Intangible Cultural Heritage (ICH) registers, making it easier for culture ministries to absorb or cross-reference. Store "cultural-impact" evidence (e.g. awards, sale numbers, social-media metrics) in a separate field—this helps later when flagging high-impact works to policymakers.

Then, researchers will need to read or analyse the cultural products. Cultural products may initially be read or analysed with an instrumental eye seeking to gain basic information rather than perform aesthetic analysis. Researchers can deploy digital reading techniques to gain this surface information, yet these may miss cultural, linguistic, or aesthetic nuances. Relevant digital reading techniques include named-entity recognition (NER) [e.g. using SpaCy] to pull geographical locations automatically. Voyant Tools can be used for term frequency or topic modelling to spot likely CES clusters before close reading. Any digital reading methods employed must be used with care to ensure the technology is

up to standard and that any key information (e.g. implicit references or Indigenous place names) are not being missed.

Readers should look to identify key CES represented and **indicate** the frequency, location and potency of their mention, as well as attendant benefits/disbenefits or threats (it is sufficient to indicate, because it may be impossible and redundant to be comprehensive about how often a CES is mentioned and all its attendant benefits or threats, particularly with larger works).

To determine the potency, employ a 0–3 scale for explicitness (e.g. direct mention = 3, metaphorical hint = 1) and a 0–3 scale for beneficial or harmful connotation. These can feed into quantitative analysis later, potentially to build heat maps that speak the same language as quantitative CES layers.

Arts and humanities approaches (Belsey 2013; Brannigan 2015; Gladwin 2018) will help guide this type of analysis. An environmental humanities theoretical framework can help to narrow focus by directing attention to key thematic or aesthetic concerns to target.

Results can be organised into categories to show trends in CES over time in a particular place. Researchers may also seek to identify particular cultural products for closer aesthetic analysis to provide more qualitative evidence of the perceptions of CES in particular areas and their links to wellbeing benefits. This provides rich case studies for policy briefs and compelling qualitative vignettes that decision-makers can quote. Researchers may choose works scoring high in categories such as (a) CES diversity, (b) policy relevance (e.g. depicts an underregulated site), or (c) cultural impact. For this analysis, CES data should be coded. First, use an inductive coding method and then a deductive method in line with a framework such as IPBES or CICES.

See Figure A11.3 for a vertical flowchart of potential steps involved in analysing the cultural representations data.

Analysing Cultural Representations

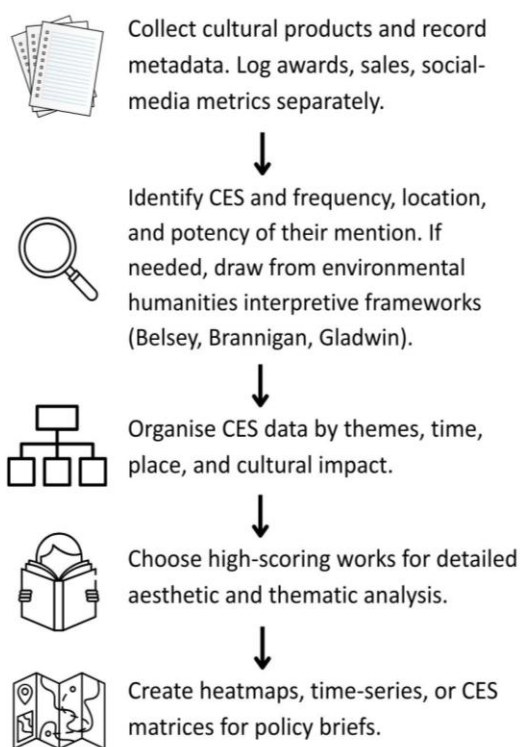


Figure A11.3: A vertical flowchart of potential steps involved in analysing the cultural representations data.

A11.3 Integrating the data

The results across all the methods used will need to be integrated. Researchers can first consult with team members to ensure the data analysis process has been robust. Researchers can then work with their team to devise a strategy for integrating the data. Team consultation is important to ensure quality and accuracy of analysis.

To integrate the coded data collected from all the research methods, researchers should match the quantitative information about spatial distribution to the qualitative data about feelings and perceptions. Seek to identify which locations have high, medium, or low concentrations of CES. Seek to identify patterns between direct access to/use of particular locations and CES. Note also the *frequency* of direct use of locations and connect these to identified CES and their associated perceptions and feelings. Seek to identify which locations are associated with which CES and the volume and quality of these associations. Microsoft Excel may be used for basic integration, although more advanced tools such as Microsoft Power BI may be necessary for complex integration. Researchers may use Multiple Correspondence Analysis (MCA), wherein data are represented as points on spatial graphs to analyse underlying patterns in datasets.

Invite participants back for a checking/interpretation workshop. This step both validates the findings and creates early-stage buy-in among future implementers.