D5.3 Monetary valuation in SEEA EA

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1 Preface

The importance of biodiversity, natural capital and healthy ecosystems and the services they supply has increasingly been acknowledged in diverse policy initiatives (e.g., the EU nature restoration and amending Regulation from 2024, EU Biodiversity Strategies 2020 and 2030, Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES), UN's Natural Capital and Ecosystem Services Accounting (SEEA EA), Intergovernmental Panel on Climate Change (IPCC) and Convention on Biological Diversity (CBD)).

The EU Horizon Research and Innovation Action "Science for Evidence-based and sustainabLe decIsions about NAtural capital" (SELINA) aims to provide robust information and guidance that can be harnessed by different stakeholder groups to support transformative change in the EU, to halt biodiversity decline, to support ecosystem restoration and to secure the sustainable supply and use of essential Ecosystem Services (ES) in the EU by 2030.

SELINA builds upon the Mapping and Assessment of Ecosystems and their Services (MAES) initiative that has provided the conceptual, methodological, data and knowledge base for comprehensive assessments on different spatial scales, including the EU-wide assessment (Maes et al. 2020) and assessments in EU member states. Knowledge and data for different ecosystem types are increasingly available.

The overall objective of Work Package (WP) 5 "Ecosystem Accounting" is to integrate insights from ecosystem conditions and ecosystem services into the UN System of Environmental Economic Accounting Ecosystem Accounting (SEEA EA) framework. It addresses key challenges such as incorporating externalities, improving the accounts' spatial and temporal resolution, and exploring how different valuation methods influence ecosystem services and asset values.

The Deliverable D5.3 "Monetary Valuation in SEEA EA" presents an evaluation of monetary valuation methods, aiming to clarify the implications of choosing different valuation methods for the creation of SEEA ecosystem accounts. We collected primary data from Ireland, Lithuania, Portugal, Greece, the Czech Republic and La Reunion. Additionally, we performed an exercise in value function transfer based on data from the Ecosystem Services Valuation Database. This allowed us to perform a qualitative evaluation of valuation methods framed around their timeliness, salience, credibility and cost, as well as a quantitative evaluation on the variability in monetary value estimates across valuation methods.

2 Summary

SELINA Deliverable 5.3 evaluates monetary valuation methods for ecosystem services within the UN SEEA EA framework, focusing on how methodological choices influence ecosystem accounts and their utility for decision support. As EU Member States implement mandated ecosystem accounting under Regulation (EU) 2024/3024, understanding the implications of valuation choices, particularly the distinction between SEEA EA's preferred exchange values and broader welfare values, is crucial.

We empirically tested valuation methods across Demonstration Projects and Test Sites in the Czech Republic, Greece, Ireland, La Réunion, Lithuania, and Portugal, covering provisioning, regulating, and cultural services. We employed both primary data collection and value transfer techniques. We extensively explored meta-analytic value function transfer using the Ecosystem Services Valuation Database (ESVD) as a scalable alternative. We developed meta-analytic value functions for key biomes (agricultural land, wetlands, grassland, temperate forests) by modelling existing primary valuation data against site and context characteristics (ecosystem condition, population density, income). This approach enables rapid, spatially explicit estimation of exchange and welfare values and regular updates for accounting cycles.

Our testing found that strict adherence solely to exchange values presents limitations for policy. By excluding consumer surplus and non-use values, exchange values can significantly underestimate the societal importance of services lacking markets. Our quantitative evaluation using value function transfer often showed welfare-based estimates (including consumer surplus) exceeding exchange values, typically by 30–70% for services with strong non-market components like regulating and cultural services, reflecting the inclusion of these broader benefits. Decisions informed only by exchange values risk prioritizing market returns over protecting ecosystems providing substantial non-marketed benefits, potentially leading to suboptimal investments. Welfare-based valuations offer a more comprehensive measure essential for secondary and tertiary uses like cost-benefit analysis, conservation prioritization, and impact evaluation.

We recommend a layered approach: utilize efficient value transfer methods for estimating exchange values for core national accounts and primary uses. Complement this with welfare-based valuations, ideally from primary studies or carefully validated local transfers that ideally include information on ecosystem condition and the local institutional context, for comprehensive policy analysis and secondary/tertiary uses. Acknowledging the limitations and uncertainties of each approach is crucial.

This work will feed into research on externalities, disservices, remote sensing, and links between ecosystem condition and monetary valuation to inform SEEA EA recommendations in Europe and globally (Deliverable 5.4). Further development of meta-analytic transfer will incorporate ecosystem condition indicators to explain value variation. The insights and methods here will also support SELINA’s integrated ecosystem assessment framework (Task 6.4) and the project’s Compendium of Guidance on ecosystem accounting and valuation.

3 List of abbreviations

CAP	Common Agricultural Policy
CVM	Contingent Valuation Method
EFTA	European Free Trade Association
ESVD	Ecosystem Services Valuation Database
EU	European Union
EU ETS	European Union Emissions Trading System
DCEM	Discrete Choice Experiment Method
DP	Demonstration Project
GDP	Gross Domestic Product
MS	Member States
SEEA EA	System of Environmental-Economic Accounting, Ecosystem Accounting
SIMeF	Simplified Forest Product Quotation System
SNA	System of National Accounts
TCM	Travel Cost Method
TEV	Total Economic Value
TS	Test Site
WP	Work Package
WTA	Willingness To Accept
WTP	Willingness To Pay

4 Introduction

4.1. Background

The System of Environmental-Economic Accounting, Ecosystem Accounting (SEEA EA) was designed as a statistical framework to integrate ecosystems and their services into national economic and environmental statistics (United Nations 2024). It provides a standardized approach to measuring ecosystems in terms of their extent, condition, and the services they provide to society, and aims to do so using theory and methodologies that align with the System of National Accounts (SNA). By doing so, it aims to facilitate the inclusion of information on ecosystem extent, condition and services in decision-making processes alongside traditional economic indicators. To fulfil that purpose, a key feature of SEEA EA is its use of monetary valuation to express the contributions of ecosystems in economic terms, allowing for direct comparisons with standard measures such as Gross Domestic Product (GDP), national income, and sectoral productivity, as measured by SNA. This can potentially help policymakers and analysts assess trade-offs between economic activities and environmental sustainability, making the case for better management of natural capital.

However, key issues concerning the methods for monetary valuation of ecosystem services in SEEA EA need further study. SEEA EA employs monetary valuation in a specific and constrained way, to ensure alignment with the principles of the SNA. Rather than attempting to capture the full societal or intrinsic value of ecosystems, SEEA EA focuses on exchange values—the prices that would be observed if ecosystem services were transacted in markets. This approach ensures consistency with existing economic frameworks and allows ecosystem accounts to be integrated with national accounts without distorting economic aggregates. While this method provides useful insights for policy and planning, it also comes with recognized limitations. Exchange values do not capture total welfare effects of using a good or service, since they do not take into account changes in consumer and producer surplus (see Chapter 5 for a more detailed description of this issue). Using exchange values to generate monetary accounts of ecosystem services creates two broad types of issues:

1. Issues with the relevance of information provided by exchange value-based estimates in various decision contexts.

2. Methodological issues with accurate measurement of exchange values of ecosystem services.

4.1.1 Information relevance

One of the key challenges of using exchange values for ecosystem services in SEEA EA is ensuring that the information remains relevant and useful across different decision-making contexts. Exchange values, by design, only reflect the price that would be observed if ecosystem services were transacted in markets. However, many ecosystem services—such as flood protection or recreational value—do not have well-defined markets or direct transactions. As a result, the estimated exchange values may only provide a partial picture of the true benefits that ecosystems provide to society (NCAVES-MAIA 2022).

This limitation becomes particularly important when ecosystem accounts are used to inform public policy, environmental management, and cost-benefit analyses. Policymakers often need to evaluate trade-offs between economic activities and environmental conservation, which requires a comprehensive understanding of both market and non-market values. Since exchange values do not capture welfare effects, such as consumer surplus or broader societal benefits, they may undervalue ecosystem services that generate public goods or indirect benefits. For example, water purification by wetlands provides significant benefits to communities, but if these benefits are not fully captured in observable or simulated transactions based on exchange values, their economic significance might be understated in national accounts. This could lead to misaligned priorities in decision-making, where ecosystem degradation is not adequately reflected as a loss in economic terms.

Another issue is the use of exchange value-based estimates in business and private sector decision-making. While firms and investors typically rely on market-based measures, ecosystem services often operate outside traditional market structures. If businesses and financial institutions only consider exchange values, they may underestimate environmental risks and fail to incorporate the long-term benefits of ecosystem preservation into their strategies (Capitals Coalition 2024).

More generally, studies on the valuation of ecosystem services historically had limited success in their uptake in decision processes. Barton et al. (2022) found that the vast majority of valuation studies showed no documented uptake of their results into decision processes, and suggested that barriers to uptake can derive from a lack of timeliness of results, lacking salience, credibility, legitimacy and excessive cost and capacity requirements.

4.1.2 Measurement of exchange values

Beyond issues with their relevance in decision-making, the accurate measurement of exchange values presents several methodological challenges. Since many ecosystem services are not directly traded in markets, estimating their exchange values often requires the use of proxies, imputed prices, and modelling techniques involving simulated exchange values. These methods come with uncertainties and limitations.

One major methodological issue is the identification of appropriate price signals for non-market ecosystem services. For example, when valuing recreational benefits of forests, analysts might use entry fees to national parks or value estimates from travel cost studies. However, due to institutional incompatibility these values may not accurately capture the economic significance of forests, especially for populations that access these areas freely (Scheufele and Pascoe 2023, Barton 2020). Similarly, when estimating exchange values for carbon sequestration, economists might use carbon credit prices, but these can be highly volatile and dependent on policy-driven market mechanisms (Dong, Gao et al. 2022, Liu, Wojewodzki et al. 2023).

Another challenge is ensuring consistency in valuation methods across different ecosystem types and regions. Since ecosystems provide multiple services that interact in complex ways, double counting or undervaluation can occur if methodologies are not carefully designed. For example, the same forest might be valued for both timber production and carbon sequestration, but if the valuation methods are not properly aligned to the same theory of value, there could be an overestimation or underestimation of the total ecosystem value.

Additionally, the spatial and temporal dimensions of ecosystem services pose difficulties in measurement. Ecosystem values can vary significantly depending on local ecosystem conditions, land use, and socio-economic and institutional contexts. A wetland near an urban area may provide high-value flood protection due to dense infrastructure, while a similar wetland in a remote area might be assigned a much lower exchange value, even though both provide critical ecological functions. Moreover, ecosystem services change over time, and static exchange value estimates may not fully capture dynamic ecosystem processes or long-term degradation risks.

4.2. Scoping and aims

Given the issues described above, this deliverable aims to further clarify how different valuation assumptions and approaches determine the monetary values of SEEA ecosystem services and ecosystem asset accounts, and what the practical and policy implications are of choosing one type of valuation method over another.

The theoretical and methodological differences between value concepts have already been discussed in detail previously (NCAVES and MAIA 2022, Barton et al. 2019) and will only be summarised in this report. Challenges remain when it comes to the use of accounting data in applications and decision-making processes, the need to build links to the discussions of diverse value perspectives, and the need for further research on measurement and valuation of exchange values (IPBES 2022). The main aim of this deliverable is therefore to clarify the implications of choosing specific value concepts, and valuation methods linked to these, for ecosystem accounting.

On November 24, 2024, Regulation (EU) 2024/3024[1], which amends Regulation (EU) No 691/2011 and introduces new environmental economic accounts modules, entered into force. The regulation mandates the EU-27 Member States (MS) and the EFTA states to generate ecosystem accounts, including accounts of ecosystem services, at the national level, and report these to EUROSTAT. MS are required to resubmit data starting from 2024 onwards. The initial submission by Member States is expected to take place by the end of 2026, with official data becoming available from 2027 onwards.

While monetary accounts are not currently mandatory for ecosystem service accounts, the regulation emphasizes the significance of assigning monetary values to ecosystem services. This is intended to raise awareness about the costs of inaction and help the European Union meet its environmental goals. As a step towards potential future integration, the regulation

[1] http://data.europa.eu/eli/reg/2024/3024/oj

calls for pilot studies that adhere to SEEA EA standards, which will pave the way for the introduction of monetary account reporting in future amendments.

To this end, we focused on generating valuation data using a variety of methods in collaboration with SELINA Demonstration Projects (DP) and Test Sites (TS). We tested valuation methods in two DPs and four TS to generate new primary data for a range of valuation methods across ecosystem services. In addition, we used newly developed value transfer functions to estimate values for the same ecosystem services in the same locations. These data then allowed for a comparative analysis of the strengths and weaknesses of the different valuation methods, given their information costs and the characteristics of their outputs. For more details on the methods, see Chapter 5.

This analysis aims to allow a more informed understanding of the implications of choosing to use SEEA EA compatible exchange values, compared to welfare-based valuation methods. Comparing valuation methods, both using primary data and value transfer techniques, will also allow more informed decision making when it comes to choice of valuation method, given available data and resources, as well as information need, of a decision context.

4.3. Connections to other parts of SELINA

This Deliverable is a building block in SELINA's Strand B (Understanding ecosystems and their services) that further develops our knowledge on monetary valuation in SEEA EA. The recommendations on how to choose the most appropriate monetary valuation method in SEEA EA will feed into the Framework for Integrated Ecosystem Assessment under development in Task 6.4 as methods for designing an integrated ecosystem assessment. The findings in this deliverable will also form part of SELINA's Compendium of Guidance as it relates to ecosystem accounting. Finally, it will provide input to Deliverable 5.4 - Recommendations for SEEA EA implementation in the EU and globally. Further discussion on how this deliverable contributes to knowledge building in SELINA can be found in Chapter 10.

4.4. Report structure

- Chapter 5 provides a theoretical overview of the purpose of monetary valuation, the various monetary valuation methods available for valuing ecosystem services, and the approach SEEA EA uses.
- Chapter 6 describes the evaluation framework we developed for comparing monetary valuation methods.
- Chapter 7 gives an overview of the DPs and TSs we used to generate valuation data.
- Chapter 8 describes the methods for valuation data collection we used to generate monetary value estimates, both for primary data collection in the DPs and TSs, and for the value transfer methods we applied.
- Chapter 9 presents the results of the evaluation of monetary valuation methods.
- Chapter 10 discusses the evaluation results and gives recommendations for choosing appropriate valuation methods, given the decision context.

5 Monetary valuation of ecosystem services

5.1 The purpose of monetary valuation

The primary objective of monetary valuation is to provide policymakers and private actors with a common metric to assess and compare the benefits of ecosystem services alongside conventional economic activities. More specifically, NCAVES and MAIA (2022) have defined a list of what exchange value based monetary accounts can be used for:

- Comparing the values of environmental assets (including ecosystems) with other asset types (e.g., produced assets) as part of extended measures of national wealth.
- Highlighting the relevance of non-market ecosystem services (e.g., air filtration).
- Assessing the contribution of ecosystem inputs to production in specific industries and their supply chains.
- Comparing the trade-offs between different ecosystem services through consideration of relative prices.
- Deriving complementary aggregates such as degradation adjusted measures of national income.
- Evaluating trends in measures of income and wealth.
- Improving accountability and transparency around the public expenditures on the environment by recognising expenditure as an investment rather than a cost.
- Providing baseline data to support scenario modelling and broader economic modelling.
- Assessing financial risks associated with the environment; and calibrating the application of monetary environmental policy instruments such as environmental markets and environmental taxes and subsidies.

Monetary value as a common metric facilitates the evaluation of trade-offs associated with land use and resource management decisions. For example, by assigning monetary values to services that are often excluded from market transactions, such as flood protection provided by wetlands or the recreational value of urban green spaces, decision-makers can better justify investments in conservation and sustainable management practices (Caparrós et al., 2017; Scheufele and Pascoe, 2023).

In addition to its specific objectives, the purpose of monetary valuation of ecosystem services should be viewed in the context of the various uses of ecosystem service values, as described by Brander et al. (2018). These values have multiple applications, which can be categorized into primary, secondary, and tertiary uses, as illustrated in Fig. 1.

Primary uses of ES values aim to address the following questions:
- What is the relative importance of ecosystem service contributions to the economy?
- Are there trends in their monetary value over time?
- How do different economic sectors, jurisdictions, or management areas compare in terms of their annual contributions to the economic product through ecosystem services?

Secondary uses of ES values involve more advanced analyses, including:

- Scenario analysis: How will the exchange values of ecosystem services change in response to global drivers such as climate change, species loss, and population growth?
- Trade-offs: Using ecosystem accounting data to inform financial and social cost-benefit analyses, as well as multi-criteria decision-making. This involves evaluating the exchange value of ecosystem services relative to alternative land uses and considering their importance in relation to economic welfare values and other non-monetary values.

Tertiary uses of ES values provide insights to inform decision-making, including:
- Impact evaluation and attribution: Assessing the impact of policy instruments or management measures on the exchange value of ecosystem services over a specific time period and in a specific area or population. This requires the use of before-after-control-impact approaches.
- Policy design: Determining the regulatory standards and economic incentives needed to achieve policy objectives for the exchange value of ecosystem services.

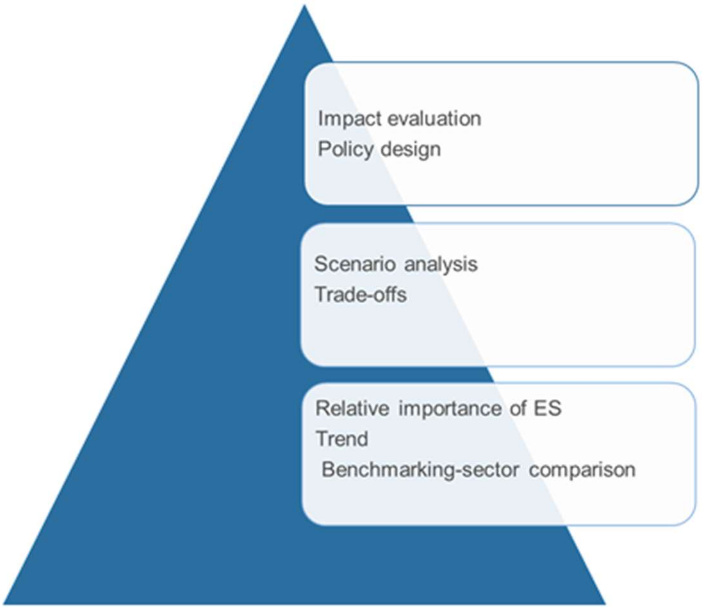

Figure 1: Purpose of ecosystem service values.

5.2 Value concepts

A spectrum of value concepts exists for the valuation of ecosystem services. Fig. 2, taken from Barton et al. (2019), illustrates this by presenting a central core of exchange value of goods and services produced by economic units and measured by the SNA, around which expanded concepts of value reside: extending the concept of value into ecosystem services can be done by valuing their contribution to a service within the SNA, but itself being outside of it. Moving further from the SNA, valuation of complementary goods related to an ecosystem service can also be measured using exchange values. These concepts are all compatible with SEEA EA.

When also including consumer surplus, this can still be measured in monetary values but is no longer compatible with the concept of exchange value as used in SNA. Moving even further from the SNA, valuation of ecosystem services can also include measures of well-being and health, which are not economically valued. Finally, ecosystem condition indicators can be used as indicators of value not directly linked to human interaction with the ecosystem. These value concepts can lead to different measures of value, even if measured on the same (monetary) scale (Scheufele and Pascoe 2023). In the following sections we go into more detail on the concepts of exchange value and welfare value, how SEEA EA applies value and what the implications of that choice are.

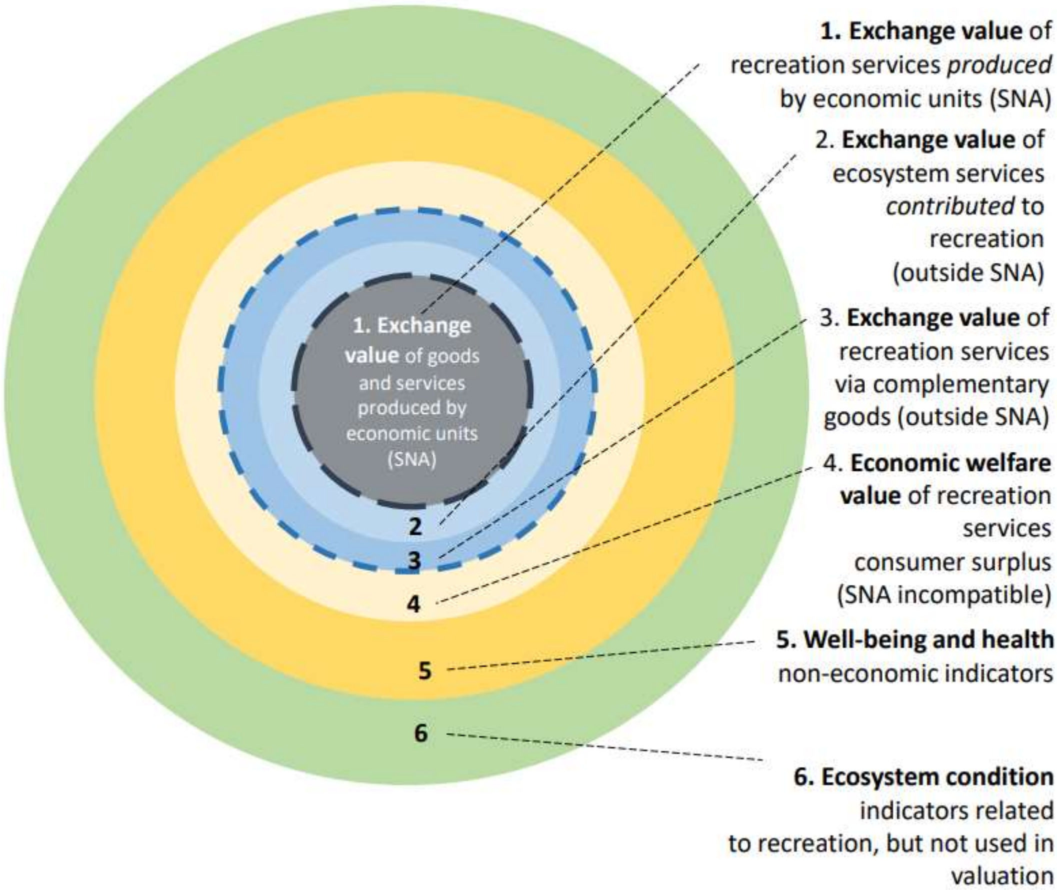

Figure 2. A conceptual diagram of value concepts with increasingly broad definitions of value. Taken from Barton et al. (2019).

5.2.1 Exchange-based valuation

Exchange value is a foundational concept in monetary valuation, originally rooted in neoclassical economics, and is defined as the price at which a good, service, or asset is exchanged between willing buyers and sellers under specified market conditions (European Commission et al., 2009). In essence, it represents the amount of money that would be received or paid in a competitive market transaction. This is the value concept used in the System of National Accounts. Within the context of ecosystem accounting, exchange value is adapted to estimate the monetary worth of ecosystem services—such as carbon sequestration, water purification, and recreational opportunities—even when these services are not actively traded in a formal market.

Because many ecosystem services are provided freely by nature, there is often no observable market transaction from which to derive a price. Therefore, the exchange value in ecosystem accounting is typically *simulated* - it is an imputed value that reflects what the service would be worth if it were exchanged under market conditions. This simulation relies on various valuation techniques, including estimating replacement or maintenance costs, travel cost methods, and hedonic pricing. These techniques operate under the assumption that a hypothetical market exists, allowing ecosystem contributions to be expressed in monetary terms and compared with conventional economic activities. This process of imputation facilitates the integration of natural capital into national accounts and policy decision-making frameworks.

The foundation of values in SNA, and therefore in SEEA EA, is the transaction as a statistical unit (Barton et al. 2019). A key implication of this is that supply and use have to be equal. Due to this definition in (hypothetical) market transactions, (simulated) exchange values do not capture all dimensions of human welfare. For example, exchange values generally reflect only the transactional component of value—excluding elements such as consumer surplus, cultural significance, and intrinsic benefits, which are often addressed by complementary welfare valuation methods.

5.2.2 Welfare-based valuation

Welfare-based valuation centres on the measurement of the total benefits that ecosystem services confer on human well-being. Unlike exchange value—which is confined to observed or imputed market transactions—welfare-based valuation seeks to capture the full spectrum of benefits, including those that extend beyond direct payments. This approach incorporates consumer surplus (the difference between what individuals are willing to pay for an ecosystem service and what they actually pay), non-use values (such as existence and bequest values), and intangible benefits that individuals derive from the continued provision of ecosystem services (Caparrós et al., 2017).

At its core, welfare-based valuation is anchored in the economic concept of utility. It reflects the monetary equivalent of the incremental well-being or utility gains that result from changes in the provision or quality of ecosystem services. In practice, methods such as contingent valuation, discrete choice experiments and travel cost models can be employed to estimate individuals' willingness to pay (WTP) for improvements in ecosystem services, thereby quantifying the additional welfare that these services provide. By encompassing both market and non-market benefits, welfare-based measures offer a more comprehensive assessment of ecosystem contributions to societal welfare (European Commission et al., 2009 ; Haines-Young & Potschin, 2017). This approach is particularly important for ecosystem services that are provided free of charge or are undervalued in market terms, such as air quality regulation, recreational opportunities, and biodiversity conservation.

5.3 Data collection methods

5.3.1 Primary data

Primary data collection involves gathering original, site-specific information directly from the source rather than relying on secondary or published data. Primary data collection is meant to accurately capture the unique characteristics of ecosystems and society's interaction with them in a study site, thereby ensuring that valuation estimates are both contextually relevant and robust. A large number of valuation methods based on primary data exists.

5.3.2 Value transfer

Value transfer (also known as benefit transfer) involves the use of research results from existing primary valuation studies at one or more locations ("study sites") to predict ES values for other sites or policy contexts ("policy sites") (Brander, 2013; 2022). Value transfer methods have been employed widely in national and global ecosystem assessments, value mapping applications and policy appraisals (Costanza et al., 1997; Bateman et al., 2013; Schägner et al., 2013). The use of value transfer is widespread but requires careful application to minimise transfer errors (Johnston et al., 2021) and the potential application of these methods for ecosystem accounting remains at a nascent stage (Grammatikopoulou et al., 2023; 2022). Four alternative approaches for conducting value transfer are described here.

Unit value transfer uses values for ecosystem services at a study site, expressed as a value per unit (usually per unit of area or per beneficiary), combined with information on the quantity of units at the policy site to estimate policy site values. Unit values from the study site are multiplied by the number of units at the policy site. Unit values can be adjusted to reflect differences between the study and policy sites (e.g., income and price levels).

Value function transfer uses a value function estimated for an individual study site in conjunction with information on parameter values for the policy site to calculate the value of an ecosystem service at the policy site. A value function is an equation that relates the value of an ecosystem service to the characteristics of the ecosystem and the beneficiaries of the ecosystem service. Value functions can be estimated from a number of primary valuation methods including hedonic pricing, travel cost, production function, contingent valuation and choice experiments (Rosenberger and Loomis, 2017).

Meta-analytic function transfer uses a value function estimated from the results of multiple primary studies representing multiple study sites in conjunction with information on parameter values for the policy site to calculate the value of an ecosystem service at the policy site. Since the value function is estimated from the results of multiple studies it is able to represent and control for greater variation in the characteristics of ecosystems, beneficiaries and other contextual characteristics (Rosenberger and Phipps, 2007; Schmidt et al., 2016).

Value generalization. Rosenberg and Stanley (2006) referred to 'generalization' sources of error in benefit transfer, United Nations (2021) refers to "value generalization techniques' (p.351) and NCAVES-MAIA (2022) to 'value generalization' as an ecosystem accounting specific value transfer method. Value generalization aims to account for all the spatial variation in provision of ecosystem services from all accounting units of ecosystem assets within the entire accounting area. Value generalization' is a spatially sensitive value transfer method interpolating from a sample of ecosystem assets to all ecosystem assets of the accounting area (Barton 2023). See Grammatikopoulou et al. (2023) for a discussion of 'value generalization'. Barton (2023) demonstrates an empirical application to urban ecosystem accounting.

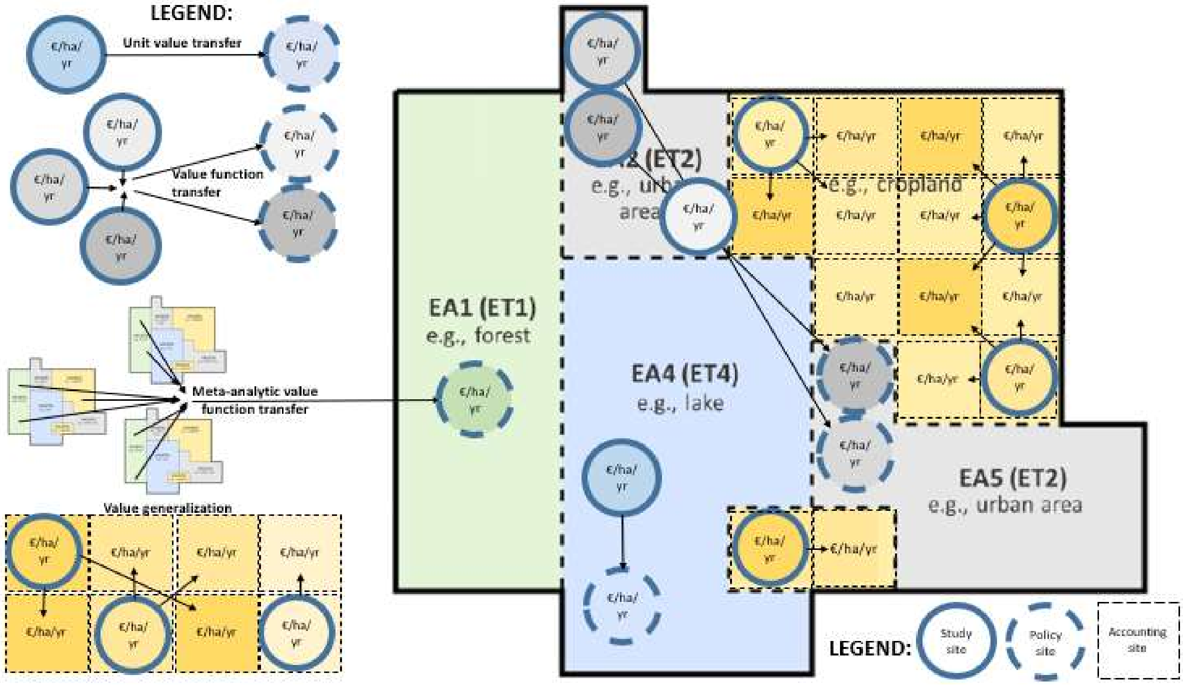

Figure 3. Four value transfer approaches. Source: NCAVES-MAIA (2022), p.104.

The choice of which value transfer method to use to provide information for a specific policy context is largely dependent on the availability of primary valuation estimates and the degree of similarity between the study and policy sites. However, there is no consensus yet on which value transfer method works best in a given circumstance (Johnston et al., 2018). In cases where value information is available for a highly similar study site, unit value transfer may provide the most straightforward and reliable means of conducting value transfer (Ghermandi et al., 2016). On the other hand, when study sites and policy sites are different, value function or meta-analytic function transfer offers a means to systematically adjust transferred values to reflect those differences (Kaul et al., 2013). Similarly, in the case that value information is required for multiple different policy sites, value function or meta-analytic function transfer may be a more accurate, practical and consistent means for transferring values. Value generalization may use one or a combination of transfer methods, depending on which have lowest transfer errors in the accounting context (e.g. thematic local, regional or national accounts) (Barton 2023).

In the SEEA EA framework, ecosystem extent and condition accounts provide the biophysical foundation for service-flow accounts: extent accounts map the area of each ecosystem type, condition accounts record its health via key indicators, and together they determine the physical supply and use of services (Q) (United Nations 2024). SEEA EA then requires that these physical flows be monetized using exchange values (P) to populate both service-flow valuation accounts and ecosystem-asset accounts, so that opening- and closing-stock values of natural capital reflect $V = P \cdot Q$ (Grammatikopoulou et al. 2023). However, when unit values derived from one location or time are generalized across heterogeneous areas or periods, a frequent practice in benefit transfer, value generalization errors arise. These errors (including uniformity, sampling and regionalization components) can distort both spatial comparisons and trend detection in asset stocks if contextual variation in ecological productivity or willingness to pay is ignored (NCAVES and MAIA 2022). Meta-analytic benefit-transfer functions, especially those estimated within a Bayesian hierarchical framework, explicitly model study-to-study heterogeneity and thus can quantify and reduce transfer error, yielding more robust valuation functions for accounting purposes (Moeltner et al. 2007). Explicitly acknowledging and characterizing value generalization errors up front is therefore crucial when comparing monetary valuation methods, as it frames how meta-analytic outcomes and transfer uncertainties propagate into ecosystem-asset accounts.

5.4 Value assumptions under SEEA EA

Monetary valuation methods under SEEA EA are based on specific value assumptions designed to ensure consistency with national accounting principles. These assumptions determine how ecosystem services are valued and how their economic contributions are represented within the SNA. In this section, we describe key value assumptions applied in SEEA EA, and what the implications of following these assumptions are for its use in decision-support.

5.4.1 Key assumptions

Exchange values

SEEA EA uses exchange values, meaning that ecosystem services are valued at the price that would be observed if they were traded in markets (United Nations, 2021). This approach aligns with the SNA, which also relies on observable market transactions to measure economic activity (European Commission et al., 2009). In practice, this extends the production boundary of SNA to include ecosystem contributions (see Fig. 2). The rationale for this assumption is to allow for direct integration of ecosystem services into national accounts without distorting established economic aggregates such as GDP. By using exchange values rather than broader welfare measures, SEEA EA ensures comparability between ecosystem services and other economic goods and services (Obst et al., 2016).

Market price equivalence and cost-based approaches

Since many ecosystem services do not have direct market prices and can therefore not use directly measured exchange values, SEEA EA applies methods that approximate exchange values (Hein et al., 2020). These include:

- Market price equivalence: Using prices from comparable markets where ecosystem services are traded, such as carbon credits as a proxy for carbon sequestration (Edens and Hein, 2013).
- Cost-based approaches: Estimating values based on the costs of maintaining or replacing an ecosystem service, such as the cost of building water treatment facilities as a proxy for wetland filtration services (United Nations, 2021).

These approaches are chosen because they provide policy-relevant estimates that align with economic decision-making, while avoiding speculative or subjective valuations (Remme et al., 2015).

Exclusion of consumer and producer surplus

SEEA EA does not include consumer and producer surplus in its valuation of ecosystem services (United Nations, 2021). This means that the total economic value (including willingness to pay beyond observed market transactions) is not captured. The rationale for this choice is that surplus-based measures reflect broader welfare effects, which are not included in the SNA and would create inconsistencies in economic reporting if SEEA EA would include consumer and producer surplus (Bateman et al., 2011). Instead, SEEA EA focuses strictly on the portion of value that could theoretically be transacted in a market (Edens and Hein, 2013).

Marginal valuation and avoidance of non-market welfare estimates

SEEA EA assumes that ecosystem services are valued at the margin, meaning that their value is based on incremental changes in service provision rather than total ecosystem benefits (Obst et al., 2016). This avoids large-scale, hypothetical estimates that could introduce uncertainty and subjectivity into national accounting (United Nations, 2021). The rationale for this is to keep ecosystem accounts policy-relevant and compatible with economic decision-making frameworks, which typically operate on marginal cost-benefit analyses (Dasgupta, 2021).

5.4.2 Typology of valuation methods in SEEA EA

Following the value assumptions as described in the previous section led to the development of a typology of agreed upon valuation methods for monetary valuation in SEEA EA (NCAVES and MAIA, 2022). These valuation methods are ranked for their proximity to the preferred method of market prices (see Table 1).

Table 1. Ranked order of preferred valuation methods in SEEA EA. Adapted from NCAVES and MAIA (2022).

SEEA EA order	Valuation method
1	Market prices
2	Prices from similar markets
3	Residual value; resource rent
	Hedonic pricing
	Productivity change
4	Averting behaviour
	Travel expenditure
5	Replacement cost
	Avoided damage cost
	Simulated exchange cost

5.4.3 Implications for decision-support

The choice to base the valuation typology for monetary valuation of ecosystem services in SEEA EA on exchange values, market price equivalence, and cost-based approaches, has significant implications for decision-making in environmental policy, economic planning, and sustainability strategies. While these assumptions ensure compatibility with the System of National Accounts (SNA) and facilitate integration into market-based decision-making, they also introduce limitations in how ecosystem values are represented, particularly in relation to broader societal and ecological considerations.

Integration with macroeconomic and policy frameworks

One of the key advantages of SEEA EA's valuation approach is that it enables ecosystem services to be reflected in national accounts, allowing policymakers to track ecosystem contributions to economic activity in a way that is consistent with traditional economic indicators (United Nations, 2021). This can help governments design policies that incorporate natural capital into economic growth strategies (Obst et al., 2016). By maintaining consistency with GDP and national wealth accounts, SEEA EA facilitates comparability between environmental and economic policies, which can support the design of environmental taxes, subsidies, and green growth strategies (Hein et al., 2020).

However, consistency with traditional economic indicators, such as GDP, also makes it easier to force metrics of environmental policy to be compatible with an economic growth strategy, thereby potentially stimulating further nature loss. Additionally, since exchange values exclude non-market benefits and broader welfare effects, decisions informed by SEEA EA risk underestimating the true economic significance of ecosystem services, especially those providing public goods or non-excludable benefits, such as climate regulation, biodiversity, and cultural values (Bateman et al., 2011; Dasgupta, 2021). This may lead to suboptimal investment in ecosystem protection, as policymakers may prioritize sectors with higher observable market returns over those with significant but less visible societal benefits.

Risk of underestimating long-term sustainability Impacts

By focusing on market-compatible valuation methods, SEEA EA can miss critical long-term sustainability risks that arise from ecosystem degradation. Ecosystem services often operate on long time scales, and their benefits may not be immediately reflected in market-equivalent values (Dasgupta, 2021). For instance, the loss of wetlands may be undervalued if their flood protection services are not fully accounted for in exchange values, leading to short-sighted land-use decisions that increase future disaster risks (Remme et al., 2015).

Moreover, cost-based approaches (such as restoration costs or avoided damage costs) are used to approximate values without legal requirements for enforcement of ecological compensation. This makes them sensitive to assumptions about technological alternatives and institutional settings (Obst et al., 2016), which can lead to inconsistencies in valuation across institutional/governance regimes (reflecting plural values), making it harder to establish universal policy guidelines for ecosystem protection (Hein et al., 2020).

Limitations for local and social decision-making

At a local level, decision-makers often require a more comprehensive assessment of ecosystem services, including cultural, social, and non-market values, which are not fully captured by SEEA EA's approach (Edens and Hein, 2013). For example, indigenous communities often rely on ecosystems for subsistence, spiritual, and cultural practices, which cannot be adequately represented using market-equivalent valuation methods (Schröter et al., 2014). This can lead to policies that prioritize economic activities over traditional ecological knowledge and local conservation priorities (Chan et al., 2012).

Additionally, the methods chosen for national level accounting to achieve "comparable markets" very easily ignore local institutional contexts governing nature values. The IPBES VA Summary for Policy Makers (2022) found that “valuation methods for ecosystem services should recognise variation in local rights regimes in order to improve the representation of local use values and how they may vary depending on communities' management of their local ecosystems", and that "spatial variation in institutional context should also be of general concern for valuation practitioners supporting national ecosystem accounting".

Furthermore, because SEEA EA does not include consumer and producer surplus, it fails to account for distributional effects—that is, how ecosystem benefits and costs are shared among different social groups (Bateman et al., 2011). This could result in inequitable decision-making, where policies favour industries and sectors that generate direct market values, while overlooking vulnerable communities that rely on ecosystem services for livelihoods and well-being (Dasgupta, 2021).

Impacts on private sector and financial decision-making

Businesses and financial institutions increasingly recognize the importance of natural capital accounting for risk management and investment planning (Dasgupta, 2021). By using exchange values, SEEA EA aims to allow firms to incorporate ecosystem services into corporate balance sheets in a way that aligns with standard economic and financial reporting frameworks (Obst et al., 2016). This can help guide private sector investment in sustainability initiatives and ecosystem restoration projects (Hein et al., 2020).

However, since exchange values do not fully capture welfare effects or long-term environmental risks, companies may underestimate their exposure to ecosystem-related disruptions, such as biodiversity loss or declining natural resource availability (Schröter et al., 2014). Capital Coalition's Governance for Valuation (2024) discusses exchange values as reflecting the importance or worth of the change in capitals to people's wellbeing only in exceptional cases. The CSRD ESRS E4 standards[2] also do not provide norms for using exchange values for incorporating ecosystem services values into corporate reporting. This highlights the need for complementary valuation methods, such as shadow pricing or risk-adjusted accounting approaches, to enhance corporate decision-making beyond what SEEA EA alone can provide (Bateman et al., 2011).

[2] ESRS E4 Delegated-act-2023-5303-annex-1_en.pdf

6 Evaluation framework

The implications of choosing to focus on exchange values in SEEA EA, as described in the previous section, need further clarification. For this purpose, we developed an evaluation framework, which we then applied to various data collection methods and valuation data outputs generated in collaboration with SELINA DPs and TSs (see Chapters 7 and 8). This evaluation consists of both a **quantitative analysis** of variation in value estimates, and a **qualitative evaluation**.

The qualitative part of the evaluation framework is based on work presented in the IPBES Values Assessment, which identified elements in ecosystem services valuation studies that contribute to effective uptake in policy processes (IPBES 2022, Barton et al. 2022). The premise of this framework is that there are key factors that influence uptake of assessments. This assumption is based on work done in previous studies on the empirics of successful uptake of ecosystem services studies in decision-making processes (Posner et al. 2016, Bagstad et al. 2013, Laurans et al. 2013). In the subsections below, we describe the factors we adopted into our evaluation framework specifically for evaluating monetary valuation methods within the context of SEEA EA. This does not include all factors that have been proposed in previous studies, but the ones we deem most relevant to performing a comparative evaluation of monetary valuation methods under SEEA EA.

For each of the factors, we defined subfactors, describing elements specifically relevant to SEEA EA's capacity to support policy processes. Based on the value concepts behind each valuation method and the data collection we performed (both using primary data and value transfer methods), we then used expert judgment to evaluate each of these subfactors as having either low, medium or high potential for effective use in SEEA EA-related policy decisions. This allowed for a more structured discussion of the implications of choosing one valuation method or value concept over another for diverse types of ecosystem services and in different contexts. Since the goal is to compare monetary valuation methods, we only included the part of the method involved with generating the biophysical ecosystem service account if different valuation methods used different methods for biophysical estimation.

The following sections describe each of the qualitative evaluation factors.

6.1 Timeliness

The definition used for this factor in the IPBES Values Assessment is: *Delivering quality information when and where it is needed to assist progress in the policy cycle.*

For our purpose, timeliness refers to the ability of a valuation method to produce results that reflect current ecosystem extent and condition, and are available within a timeframe that aligns with policy and management needs. In the SEEA EA context, valuation estimates must be updated regularly to track changes in ecosystem health, service provision, and environmental degradation. Methods that deliver timely data enable decision-makers to respond to dynamic environmental challenges and adjust management practices accordingly.

This factor is particularly important when rapid policy responses are required—for instance, during emerging threats from climate change or land-use alterations.

We divided this factor into the subfactors described in Table 2.

Table 2. Description of timeliness subfactors.

Timeliness subfactor	**Description**
Availability of recent data	Whether biophysical data, market prices, cost data, or survey results used for valuation that reflect current economic conditions (e.g., timber prices from last year vs. a decade ago) are available.
Valuation model processing time and potential for automation	The time required to apply a valuation model, from data collection to final monetary estimates, and its potential for automation. Some methods (e.g., avoided cost) can be computed quickly, while others (e.g., survey-based valuation) take longer.
Integration with SEEA accounting periods	Whether the valuation method can generate results at the required reporting frequency (e.g., annually or at multi-year intervals) to fit SEEA EA accounting needs and/or periodic accounts to comply with law mandates.

6.2 Salience

The definition used for this factor in the IPBES Values Assessment is: *Addressing the options in the decision arena, including budgetary and legal consequences.*

For our purpose, salience pertains to the relevance and usefulness of the valuation outputs to the intended users, such as policymakers, environmental managers, and stakeholders. For monetary valuation methods under SEEA EA, salience is evaluated by how well the results capture the aspects of ecosystem services that matter most to society. A salient valuation method directly informs policy decisions by clearly linking ecosystem changes to economic outcomes and societal well-being, thereby ensuring that the values provided are meaningful for setting priorities in conservation and sustainable development.

We divided this factor into the subfactors described in Table 3.

Table 3. Description of salience subfactors.

Salience subfactor	Description
Alignment with SEEA EA principles	The degree to which values are able to reflect ecosystem service contributions to economic activity without double-counting or misrepresenting benefits.
Able to inform SEEA EA relevant policy decisions	The extent to which the valuation method aligns with the impacts of key policy questions, environmental regulations, and economic planning priorities that can be observed with SEEA EA accounts. A method that produces results not directly useful for decision-making may have limited impact.
Suitability for spatially explicit valuation	Whether the method produces values that can be mapped at relevant spatial scales for ecosystem accounting.
Temporal consistency of monetary values	Whether the valuation method accounts for long-term changes in ecosystem services (e.g., discounting future values, integrating degradation effects). Methods that only provide short-term values may be less useful for long-term planning.
Relevance to the benefits provided by the ecosystem service and which are being valued	Whether the method is appropriate for the type of ecosystem service being assessed (e.g., replacement cost for water purification benefits vs. hedonic pricing for urban green space benefits).
Ease of communicating results	The extent to which valuation results can be clearly explained to policymakers, businesses, and the public. Complex models with abstract results (e.g., contingent valuation) may be harder to use in decision-making.

6.3 Credibility

The definition used for this factor in the IPBES Values Assessment is: *Building on a shared understanding of how things work, conditions and trends and prospects of consequences, through transparent methods with explicit assumptions and documented uncertainty.*

In the context of selecting a monetary valuation method for ecosystem services in ecosystem accounting, credibility refers to the degree to which the chosen method produces reliable, scientifically sound, and widely accepted estimates. It ensures that the monetary values assigned to ecosystem services are defensible and useful for policy and decision-making. SEEA EA standards attempt to achieve credibility in several ways. By aligning closely with the SNA's valuation principles—particularly its emphasis on exchange values—it ensures that valuations fit within established national accounting standards. The tiered approach to valuation methods reinforces this by prioritizing those monetary valuation methods that best approximate exchange values. Another core consideration for maximising credibility is the role of biophysical modelling as a foundation for valuation, ensuring that economic values are based on solid ecological understanding. While transparency is not explicitly required, clearly communicating methodological choices and assumptions is essential for building trust in any valuation effort.

We divided this factor into the subfactors described in Table 4.

Table 4. Description of credibility subfactors.

Credibility subfactor	**Description**
Robustness of valuation approaches	Whether market-based methods (e.g., market price, avoided cost, replacement cost) or non-market valuation data rely on well-documented and transparent value data.
Consistency and quality of input data sources	Whether biophysical and economic data used in valuation are collected using standardized methods across different time periods and regions.
Transparency in model assumptions	Whether the method clearly documents all data sources, assumptions (e.g., shadow pricing, hedonic model specifications), and uncertainties, so that results can be scrutinized and reproduced.
Validation against observed economic behaviour	Whether the valuation method allows for testing against real-world market behaviour or expenditure patterns (e.g., comparing estimated recreation values with actual tourism revenue).

6.4 Cost proportionality

The definition used for this factor in the IPBES Values Assessment is:
The cost of information is less than and proportional to benefits of the decision under consideration (Barton 2007).

For our purpose, cost proportionality refers to the economic and resource efficiency of the valuation methods, encompassing both the financial outlay and the time and expertise required to implement the approach. In SEEA EA, valuation methods must be feasible within the constraints of available resources while still delivering high-quality data. Methods that are cost-effective enable more frequent updates and wider application across different ecosystems and regions. This is particularly important for large-scale ecosystem accounting initiatives where budget constraints can limit the scope and frequency of data collection and analysis. Efficient methods help ensure that the overall benefits of incorporating ecosystem service values into national accounts justify the investment.

We divided this factor into the subfactors described in Table 5.

Table 5. Description of cost proportionality subfactors.

Cost proportionality subfactor	**Description**
Data collection cost	The expense of gathering necessary data, whether through primary sources (e.g., surveys, field measurements) or secondary sources (e.g., literature, remote sensing, government databases). High data costs may limit feasibility in resource-constrained settings.
Processing and analysis cost	The level of software, modelling expertise, and processing power needed to implement the method (e.g., input-output models require specialized tools, while simple market price methods do not).
Technical expertise requirements	Whether the valuation method requires specialized economic, ecological, or statistical knowledge to apply correctly (e.g., econometric analysis for hedonic pricing).
Feasibility under budget constraints	Whether the valuation method is affordable and implementable given available financial and institutional resources. A method that is too costly to scale or replicate may not be suitable for routine ecosystem accounting.
Scalability and replicability	The extent to which the method can be applied consistently, and potentially automated, across different geographic areas or time periods without excessive costs (e.g., benefit transfer methods offer cost-effective scaling) so as to produce consistent and comparable estimates across time periods.

6.5 Legitimacy

The definition used for this factor in the IPBES Values Assessment is:
Representing the interests of all legitimate stakeholders through composition of the team and transparency of the process followed.

In the context of monetary valuation methods for SEEA EA, legitimacy refers to the extent to which the valuation process and its outputs are perceived as fair, inclusive and procedurally just by the full range of stakeholders affected by ecosystem accounting. A legitimate valuation method will ensure that diverse perspectives (e.g. local communities, indigenous groups, industry, government) are acknowledged, and that the assumptions, data sources and modelling choices are transparently documented. This helps secure buy-in, reduces conflict, and supports the durability of policy decisions that draw on SEEA monetary accounts.

We divide legitimacy into the sub-factors described in Table 6.

Table 6. Description of legitimacy subfactors.

Legitimacy subfactor	**Description**
Stakeholder representation	The degree to which the valuation design and implementation allows the active involvement all relevant stakeholder groups (e.g. local residents, resource users, regulators, NGOs) in scoping, data-gathering and interpretation of results.
Inclusivity of value perspectives	The extent to which the method accommodates or at least acknowledges the value systems of all stakeholders involved.
Institutional acceptance	The level of formal or informal endorsement and uptake of the method by key institutions (e.g. national statistical offices, environment ministries, regional authorities), indicating procedural legitimacy within existing governance frameworks.

7 Cases

In this chapter, we present the sites in which we collected primary data for valuing ecosystem services. We describe the general size, geography, ecosystems, ecosystem services that were valued and the relevance of generating SEEA EA ecosystem services accounts to the management of these areas.

In Chapter 8, we go into detail on the data collection and analysis performed in each of these sites to generate accounts of monetary values of ecosystem services.

7.1 Czech Republic

General description

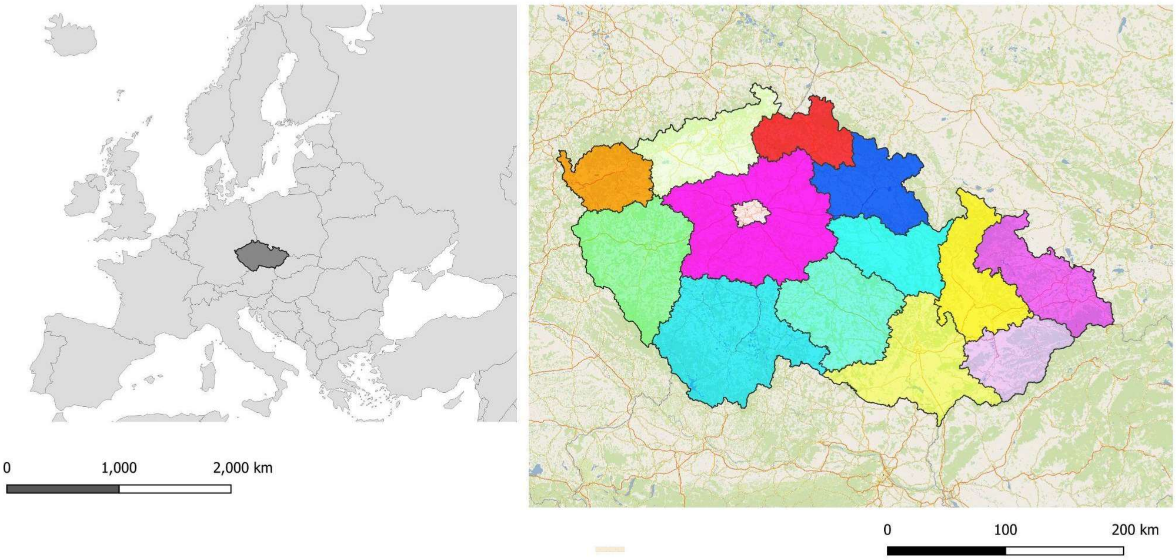

Figure 4: Czech Republic test site: national and regional breakdown (14 regions).

The test site covers the entire national area of the Czech Republic. This covers an area of 78,866 square kilometres. With a population of 10.7 million, it has an average population density of 135 inhabitants per square kilometre.

Its landscape is characterized by uplands and highlands, framed by mountains that descend into lowlands along major rivers like the Labe, Vltava, and Morava. Situated on the European watershed, the country receives an average annual precipitation of 693 mm, draining into three different seas.

Agricultural land constitutes the largest portion of the Czech Republic's land cover at 56.8%. Forests and semi-natural areas cover 36.5%. Urbanized areas account for 6.7%, water bodies for 0.7%, and wetlands for the remaining 0.1% (Grešlová et al. 2021).

Ecosystem Service of Hunting and Game Management

In accordance with both the SEEA EA (United Nations et al. 2021) and CICES v5.1 (Haines-Young and Potschin 2018) classifications, hunting is considered to provide both provisioning and cultural ecosystem services.

The ecosystem service of providing wild game (game meat) is a provisioning service, while hunting is generally classified as a cultural ecosystem service. However, these two services are closely intertwined, as the availability of game is essential for hunting. Human activities often involve both the utilization of game meat (provisioning) and the recreational pursuit of hunting (cultural), making it difficult to separate and quantify these distinct ecosystem services clearly.

Hunting is also intertwined with game and forest management as it is crucial in regulating wildlife populations and maintaining ecosystem balance. Over the centuries, hunting has evolved from a subsistence-based practice to the modern discipline of game management. Nowadays, game management encompasses activities to manage wildlife populations while preserving hunting traditions sustainably. It plays a crucial economic role, balancing wildlife conservation with mitigating potential negative impacts caused by certain species. Game management interacts with various sectors, including agriculture, industry, tourism, and research, contributing to landscape formation, rural development, and job creation. Given the inherent interrelationship between hunting and game management, we utilize the term "hunting" to collectively refer to both activities.

Hunting and Game Management in the Czech Republic

Game management in the Czech Republic is governed by the Hunting Act (Zákon o myslivosti), which establishes the legal structure for hunting activities, species protection, and population control.

The country is divided into designated hunting grounds, each managed by local hunting associations, private owners, or state agencies. A hunting ground is a contiguous area comprising one or more hunting plots. Key requirements for recognition include a minimum area of 500 hectares of uninterrupted hunting plots and compliance with other criteria specified in the Game Management Act regarding the hunting ground's shape and boundaries.

A hunter must possess the following documents while hunting: an identity card, hunting license, hunting permit, and compulsory insurance certificate. The hunter must also carry a firearm license and firearm certificate if using a firearm. If hunting with a falconry bird, its registration card is required. Obtaining a hunting license requires passing a hunting exam. Additional exams are necessary for specialized activities, such as game management and falconry.

Legally hunted species in the Czech Republic fall into two main categories: game mammals and game birds. Important game mammals include Red Deer (*Cervus elaphus*), Roe Deer (*Capreolus capreolus*), European Fallow Deer (*Dama dama*), Mouflon (*Ovis musimon*), Wild Boar (*Sus scrofa*) and Hare (*Lepus europaeus*). Pheasant (*Phasianus colchicus*) and Duck Species (Mallard - *Anas platyrhynchos*) are among the most popular game birds.

Management and Policy Issues in the Monetary Valuation of Hunting Ecosystem Services

Monetary valuation of hunting ecosystem services underscores the multifaceted importance of hunting, providing a solid foundation for evidence-based policymaking, stakeholder collaboration, and ensuring the sustainable management of ecological and economic resources. The monetary valuation of ecosystem services associated with hunting, encompassing both provisioning and recreational aspects, provides valuable insights for:

- Guiding sustainable wildlife management practices.
- Informing the design and evaluation of relevant policies, including wildlife damage compensation schemes and Payment for Ecosystem Services programs.
- Conducting comprehensive cost-benefit analyses to assess the economic implications of hunting-related ecosystem services.
- Mitigating stakeholder conflicts by quantifying the value of hunting in comparison to alternative land use options.

7.2 Greece

The selected Test Site in Greece is located at the west part of Peloponnesus, the largest peninsula of Greece, being the southernmost part of the Greek mainland. It hosts one National Park at the north (Strofylia and Kotychi wetlands National Park) and an extensive Area for Nature Protection at the south, both including Natura 2000 sites. The TS lies just a few kilometres from the metropolitan area of Patras, the third largest Greek city and ca. 260 Km from Athens. It hosts a variety of ecosystems ranging from coastal dunes and thermophilous Aleppo pine forests and phrygana, coastal lagoons, rivers and wetlands. The TS is famous among tourists for the extensive sand dunes and beaches, as well as for the world-famous site of Ancient Olympia.

The area has been severely affected by numerous wildfires (including a catastrophic megafire) as well by severe storms during the last two decades. Each year the region attracts visitors from Greece as well as from abroad, mainly during the summer months. Tourism infrastructure development is ongoing in the area; however, the area is considered not overcrowded. In addition, the strategic plan for recreation in the area includes year-round activities inside and outside protected areas, nature and biodiversity friendly infrastructure in protected areas, as well as restrictions and specific measures for vulnerable habitats (e.g. priority habitats) and species (e.g. endemics, endangered).

The area provides diverse types of ecosystem services, just indicative ranging from food production to biodiversity maintenance, coastal protection, and recreation. However, cultural ecosystem services are the main policy related debate issue, that all decisions and measures for the area refer to, as a management priority. The herein proposed assessment for monetary valuation will support local stakeholders and decision makers to clarify current priorities and needs at the local scale, and provide suggestions for the TS sustainable development and protected area management.

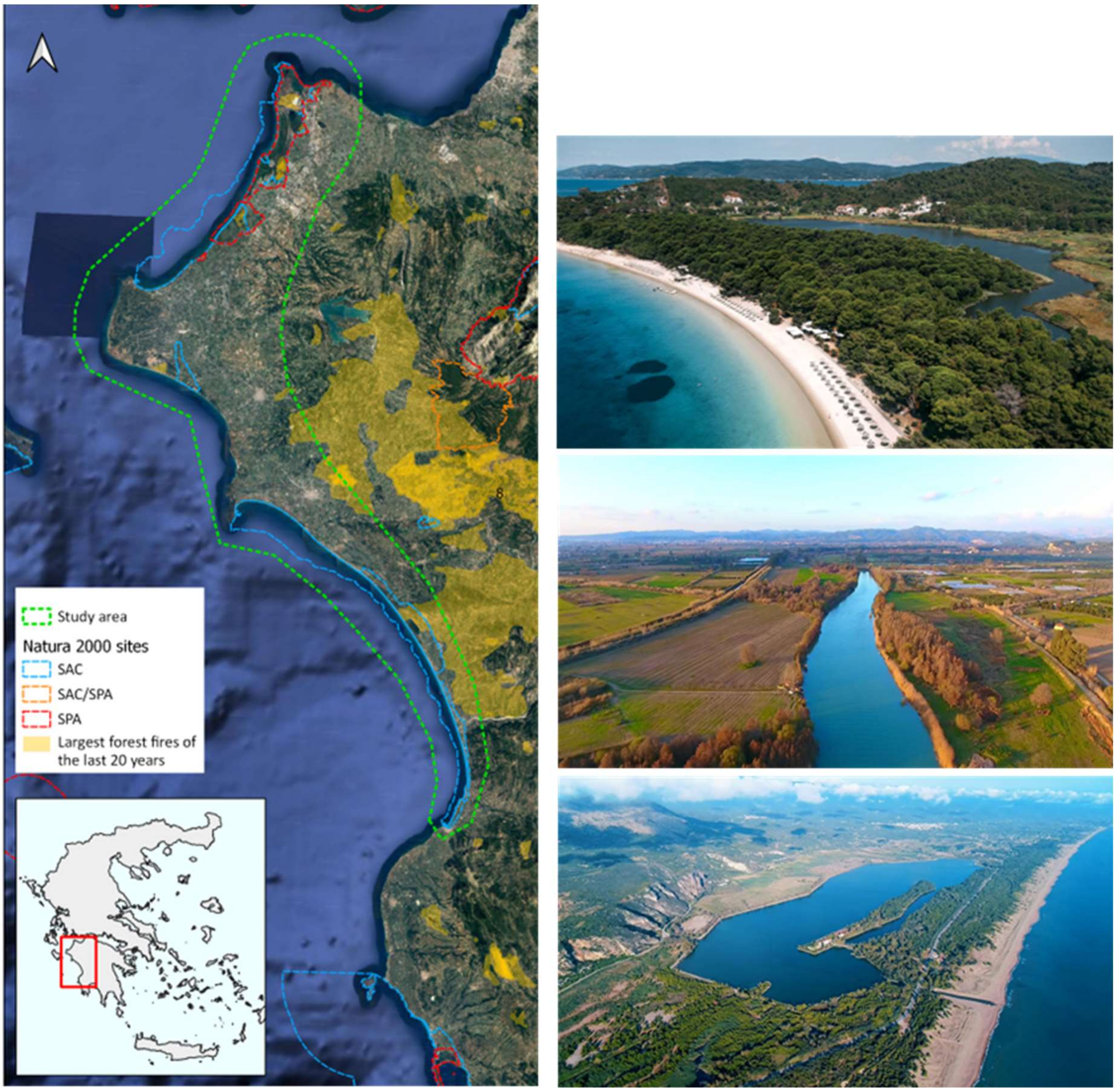

Figure 5: Peloponnesus test site. Left panel: study area. Right panel (top and bottom): Characteristic example of agricultural activities affecting ecosystems and their services. In the upper right panel, the severe modification of the river route and its riparian ecosystems is observed, alongside the loss of the river deltaic system. Former wetlands and riparian areas are now covered by intensive agricultural activities (lower, right panel).

7.3 Ireland

Introduction

The two Irish sites are both owned by Ireland's semi-state forestry company, Coillte[3] (the Irish term for forests). Coillte's forestry operation encompasses approximately 11% of Ireland's land area, and is primarily focused on timber production. Many of Coillte's sites around Ireland are open to the public and provide multiple recreational opportunities, with woodland trails, visitor car parks, viewing points, sculpture and interpretive signage provided in several of the company's forests.

We tested valuation methods for recreation-related services. For SELINA, the surveys on values of recreational woodland sites were carried out in partnership with an ongoing research project on the production of ecosystem services by Irish forestry called FOR-ES[4], led by University College Dublin and Trinity College Dublin, in partnership with Coillte and funded by the Irish Department of Agriculture, Food and the Marine. The two sites were selected as being good examples of popular recreational woodlands with good visitor access and amenities (e.g. designated trails, picnic sites, car parking etc.) and with dedicated nature conservation and restoration activities recently completed or underway.

In line with EU and Irish environmental regulation, in the past 20 years Coillte has focused increasingly on sustainable forestry practices, with a commitment to restoring woodland ecosystems and enhancing the value of Coillte's operational woodlands and recreational sites for biodiversity[5], with a long-term aim of managing 50% of its forest estate for biodiversity in the long term. The two sites selected - Hazelwood and Devil's Glen - have been selected for nature restoration activity; a 5-year biodiversity project at Hazelwood was completed at the end of 2024, whilst restoration works in Devil's Glen are ongoing.

[3] https://www.coillte.ie/

[4] https://www.for-es.ie/

[5] https://www.coillte.ie/our-forests/public-goods/biodiversity-2/

Hazelwood Forest, Co. Sligo

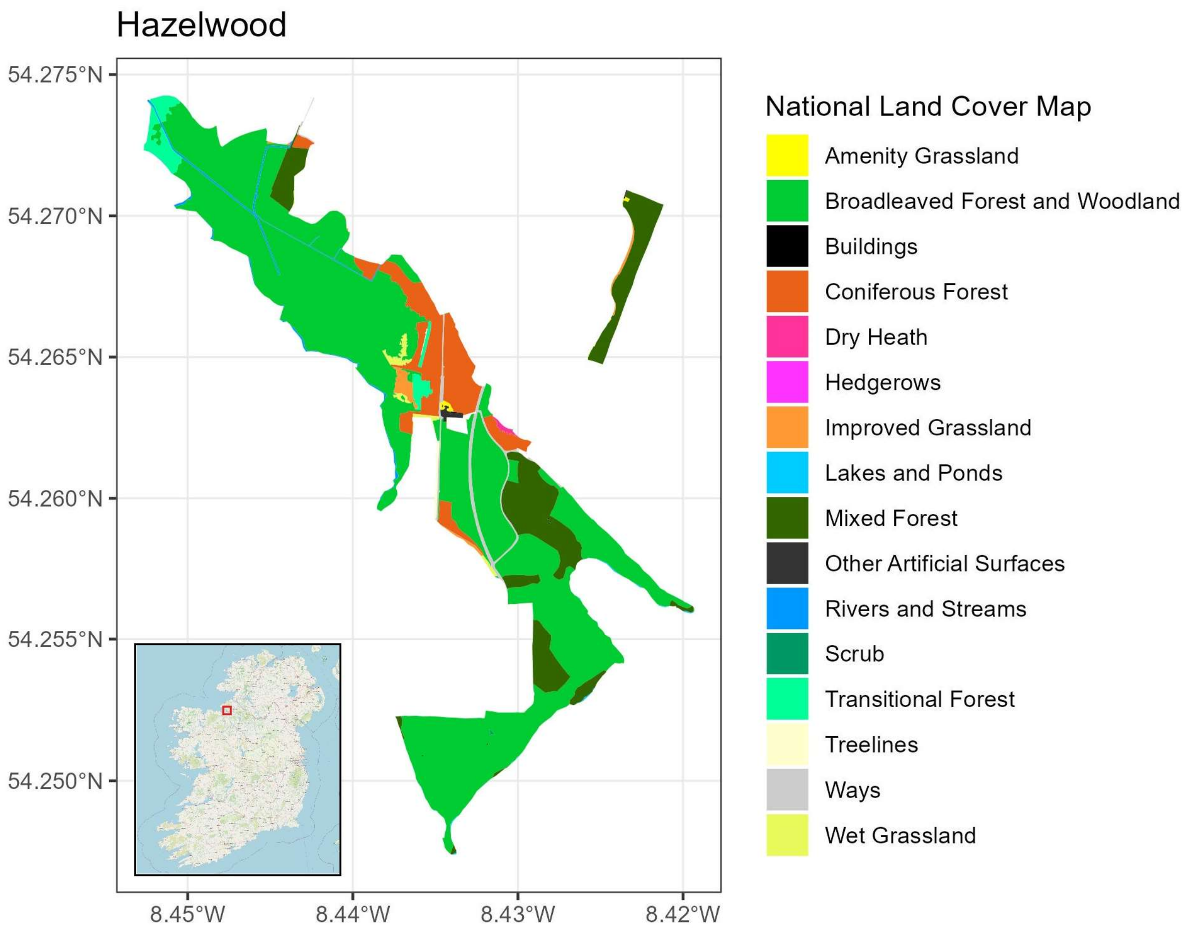

Figure 6. Map of Hazelwood Forest and its land cover (based on Ireland's National Land Cover Map 2018, published by Tailte Éireann[6]).

Hazelwood Forest[7] is located approximately 5.6 km by road from Sligo town, the principal town in county Sligo in the province of Connaught. Sligo is a coastal port town on Ireland's western coast, being the largest urban centre in Connaught and the 24th largest in Ireland. Sligo sits between Lough Gill, a large freshwater lake (surface area 12.8 km^2), and the estuary of the River Garravogue which drains from the lake. Lough Gill is associated with the Lough Gill Special Area of Conservation (SAC) designated under the EU Habitats and Species Directive. The qualifying interests of the site (as listed under Annex I of the EU Directive) that are of relevance to the recreational woodland areas include "Natural eutrophic lakes with Magnopotamion or Hydrocharition - type vegetation", "Semi-natural dry grasslands and scrubland facies on calcareous substrates (Festuco-Brometalia) (* important orchid sites), "Old sessile oak woods with Ilex and Blechnum in the British Isles", and "Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae)".

[6] https://tailte.ie/map-shop/professional-map-products/national-land-cover-map/

[7] https://www.coillte.ie/site/hazelwood/

The 130 hectares of woodland area overlaps with Hazelwood Demesne, a historic house and parkland dating to the 18th century. The house is of significant architectural heritage value, and the area has strong links with Irish history, the arts and Irish mythology. Although the house is not currently open to the public, the woodland trails through Hazelwood Demesne offer views of the house and surrounding land. The county of Sligo is also known as “Yeat’s Country,” indicating strong links with the artistic Yeats family including the poet Willian Butler Yeats and his artist brother Jack B. Yeats. The main visitor car park at Hazelwood Forest sits adjacent to the shore of Lough Gill, with a wide view out to Inisfree Island, made famous in W.B. Yeats’ 1888 poem “The Lae Isle of Innisfree”. The Hazelwood site therefore has significant cultural heritage value, in addition to its biodiversity interest.

The commercial forest area of Hazelwood consists mostly of non-native species dominated by Sitka spruce (*Picea sitchensis*). Other non-native species which are abundant in the area include sycamore (*Acer pseudoplatanus*), horse chestnut (*Aesculus hippocastanum*) and beech (*Fagus sylvatica*). Common native species include oaks (*Quercus robur* and *Q. petraeus*), ash (*Fraxinus excelsior*), holly (*Ilex aquifolium*), hazel (*Corylus avellana*), Scots pine (*Pinus sylvestris*) yew (*Taxus bacatta*), and silver and downy birch (*Betula pendula* and *B. pubescens* respectively). The woodland trails offer relatively flat walking routes, with picnic areas and a wide lake shore with views of Inisfree, where duck-feeding is popular and from which some visitors engage in wild-swimming and boating activities. Hazelwood is therefore very popular with families and groups of young people including school groups.

Figure 7. View of Lough Gill from Hazelwood. Source: www.coillte.ie.

The Devil's Glen Forest, Co. Wicklow

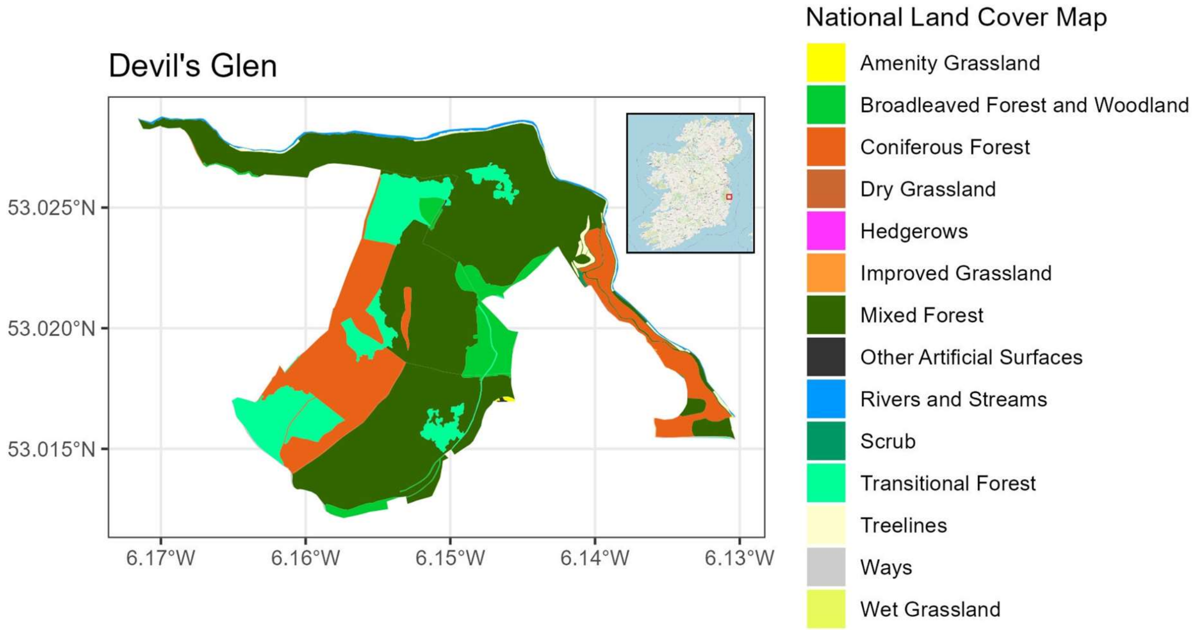

Figure 8. Map of The Devil's Glen Forest and its land cover (based on Ireland's National Land Cover Map 2018, published by Tailte Éireann[8]).

The Devil's Glen Forest[9] is located in County Wicklow on Ireland's eastern coast, in the province of Leinster. The woodland sits alongside a steep glacial channel which forms part of the Vartry River system. The Glen gets its name from a waterfall at the point where the Vartry river enters the Glen - prior to the construction of a reservoir on the river in the 1860s, the flow of water over the falls was significantly greater than at present, the sound of which was associated in local legend with the underworld. Although the waterfall is much reduced in size and power, it remains an important feature for visitors to the Glen.

The Devil's Glen is designated as a proposed Natural Heritage Area (NHA) under the Irish Wildlife Acts. NHAs are sites deemed important at a national level for the conservation of particular habitats and species that do not necessarily meet the requirements for designation under the EU Natura 2000 directives. The designation for the Glen recognises its value as an example of old stand woodland on acid slopes, with most oak trees in excess of 100 years old, which has a diverse ground flora and a high diversity of bird species. The active plantation area is dominated by Sitka spruce, with oaks, birch, hazel, holly and ash being the predominate native species, with non-native species Spanish chestnut (*Castanea sativa*) and European larch (*Larix decidua*) being common. Other woodland areas in the surrounding landscape, though less important in terms of biodiversity, provide some refuge for fauna as a continuation of the forest in Devil's Glen.

The site is approximately 8 km drive from the village of Ashford in Co. Wicklow, and 11km from the town of Rathnew. And 12km from the village of Newcastle – the three closest built-

[8] https://tailte.ie/map-shop/professional-map-products/national-land-cover-map/
[9] https://www.coillte.ie/site/the-devils-glen/

up areas and the starting point for most of the visitors surveyed at this site. The site includes a trail named after Irish poet Seamas Heaney and contains several wooden sculptures commissioned from Irish artists by Coillte, adding to the cultural heritage interest of the site.

Figure 9. Information point at Devil's Glen. Source: Conor Kretsch - Cohab.

7.4 La Réunion

Ecosystems and biodiversity

Réunion Island, a French tropical volcanic island in the Indian Ocean, spans 2,512 km² and reaches a maximum elevation of 3,070 m. Due to its position in the trade wind zone, especially on the eastern side, it receives over 3,000 mm of rainfall annually, resulting in significant seasonal runoff. The surrounding ocean is influenced by the westward South Equatorial Current, creating warm and nutrient-poor marine conditions.

The island harbours globally important biodiversity, with one-third of its surface still covered by native vegetation and many endemic plant species. Since human settlement 350 years ago, Réunion has experienced severe biodiversity loss, including the extinction of 30 out of 45 vertebrate species. Marine biodiversity includes iconic species such as migrating whales, and the island's coral reefs cover approximately 25 km of its western coast. Protected areas now include a National Park (43% of the land) and a marine reserve protecting 80% of the fringing reefs on the west and south coasts.

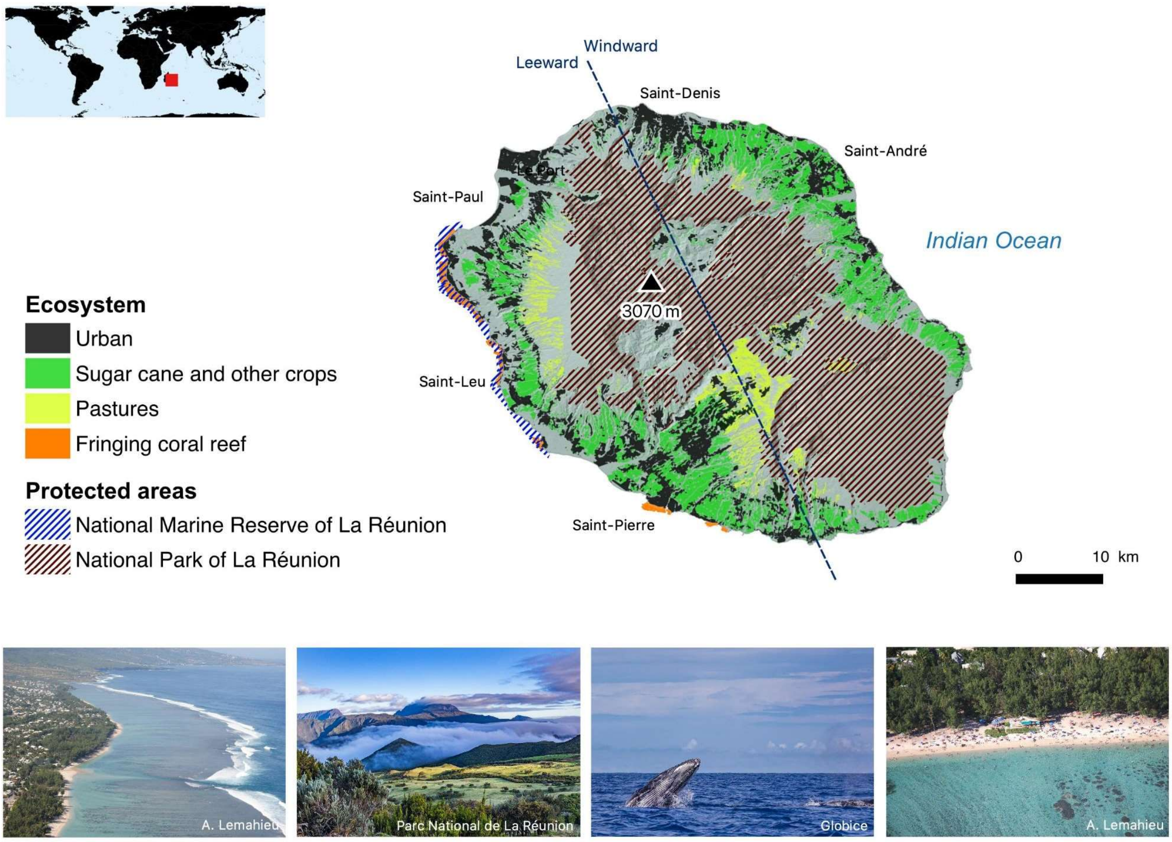

Figure 10: Réunion Island test site. Location map and ecosystem map. Lower panel shows picture (from left to right): fringing coral reef on the West coast, Piton des Neiges and pasture landscape in the uplands, jumping whale (Megaptera novaeangliae) on the West coast and beach users in the back reef depression on the West coast.

Land use, demography, economy and research

Most of Réunion's population lives along the coast, especially in the north and west. Rapid growth has expanded the population from 500,000 in 1980 to 870,000 in 2023, with urban areas increasing from 59 km² to 300 km². Agriculture, mainly sugarcane, occupies 15% of the island. Urban land use accounts for 10%.

As an EU outermost region, Réunion receives substantial development support, with €1.795 billion allocated from the FEDER-FSE+ program for 2021–2027. Despite this, the economy faces structural challenges—tourism contributes only 3% to GDP, and 38% of the population lives below the poverty line. Tourism, concentrated on the west coast, has grown significantly, supporting over 50 diving operators.

Research and innovation are vital to sustainable development on the island. Approximately 500 full-time researchers, including 340 permanent academics, work on issues like biodiversity, renewable energy, climate resilience, and socio-economic development. The University of La Réunion and research institutions such as CIRAD, IRD, BRGM, and IFREMER play key roles.

Planning, governance and challenges

Réunion's long-term sustainability depends on managing land conversion, protecting ecosystems, reducing urban sprawl, and mitigating natural hazards. The limited protection of coastal and lowland areas poses risks to ecosystem service delivery. The island's governance follows French and EU regulations and international conventions (CBD, UNFCCC, UNCLOS). Strategic frameworks include the **Schéma d'Aménagement Régional (SAR)**, **Schéma de Mise en Valeur de la Mer (SMVM)**, **Document Stratégique de Bassin Maritime (DSBM)**, and the **Stratégie Réunionnaise pour la Biodiversité (SRB)**. These ensure integrated planning across land and marine environments.

However, adaptation to climate change remains insufficient. Current policies favour short-term protective infrastructure ("holding the line"), increasing coastal exposure and vulnerability. Rising sea levels, underused regulatory tools, and low public awareness—over 80% of residents are unaware of risk prevention plans—limit adaptive capacity. More ambitious, coordinated action is needed to align development with environmental integrity and resilience.

Marine ecosystem recreation service: immersed sea uses in the open sea

Swimming in Réunion Island began as a modest and local activity, mostly practiced in the calm, shallow waters of the western lagoon. From the 1950s to the 1980s, as road access improved and urban populations grew, beaches became popular recreational spaces for local families. The emergence of a beach culture was closely tied to weekend leisure, public holidays, and early coastal tourism. During this period, shark attacks were extremely rare and did not influence public perception or behaviour. Over the following decades, from the 1980s to the early 2000s, swimming and other water-based activities such as surfing, diving, and kayaking expanded rapidly, supported by the development of tourist infrastructure and lifeguard services on the west coast. Coastal waters were widely seen as safe and accessible. Today, 8% of the local population practice an activity linked to the water (either freshwater or sea water). There is limited publicly available data specifically detailing tourist participation in sea-related activities in Réunion Island. However, according to the 2023 report from the Regional Tourism Observatory (IRT), the island welcomed 556,089 international visitors, marking a 12.2% increase compared to 2022. While exact figures for marine activity participation (such as swimming, surfing, diving, or boat excursions) are not consistently reported, these activities remain central to Réunion's coastal tourism offer, especially along the west coast. The lagoon areas and marine reserves are particularly popular among visitors for their perceived safety and biodiversity.

Marine ecosystem recreation service and shark risk

The marine ecosystem recreation service refers to the benefits people derive from engaging in recreational activities in coastal and marine environments, such as swimming, surfing, snorkelling, diving, and wildlife observation. These services contribute significantly to human well-being, local identity, and the tourism economy—particularly in island territories like Réunion, where coral lagoons and fringing reefs attract both residents and visitors. However, the emergence of shark-related risks, especially since 2011 in Réunion, has deeply disrupted access to these services. A sharp increase in shark bite incidents led to the closure of several popular sites and the implementation of costly safety measures, including shark nets, shark patrols, and a controversial shark fishing programme combined with a widely rejected ban on immersed sea uses (out of the lagoon areas). As a result, the perception of safety has become a key determinant of recreational use, modifying behaviour and spatial patterns of coastal activities. These changes transformed the relationship between people and the sea, disrupting everyday practices and the image of Réunion as a safe island for coastal tourism. Although no shark bites on humans have been recorded since 2019, access to open-water swimming remains restricted or heavily supervised outside the lagoon areas, and public confidence is still recovering. The shark risk illustrates how changes in ecosystem dynamics and human-wildlife interactions can lead to a partial or total loss of cultural ecosystem services, with significant social, economic, and governance implications. Managing this risk requires balancing biodiversity conservation, public safety, and the long-term sustainability of ocean-based recreation.

Between 1940 and 1979, Réunion Island recorded one confirmed fatal shark bite incident on a living human (in 1972) and one case of a shark bite on a motorized inflatable boat (in 1979). Additionally, eight other incidents involved shark bites on human remains where it was not possible to determine whether the bites occurred before or after death. During the same period, four cases of disappearances at sea were reported, all involving individuals whose bodies were never recovered, with simultaneous sightings of sharks nearby. The early 1980s saw two non-fatal shark bite incidents on living humans, recorded in 1980 and 1981. From 1980 to 2022, a total of 62 unprovoked incidents involving physical contact between a shark and a living human (or their personal equipment) were recorded on the island. Of these, 55 were shark bites on living individuals (an average of 1.3 cases per year), including 46 bites on human bodies and 9 bites on personal gear (such as surfboards). Seven additional cases involved physical contact without biting. Fifteen other cases involved either post-mortem bites or incidents where the timing could not be determined. One disappearance at sea with simultaneous shark presence was also documented. No shark bite incidents on humans have been reported in Réunion since May 2019. The 2010s marked a significant increase in the frequency of shark bite incidents. Between 2011 and 2019 alone, 31 unprovoked incidents were reported, including 19 bites on human bodies, 7 on personal equipment, and 5 cases of non-biting contact. The frequency during this period averaged 2.9 cases per year—more than twice the average annual rate observed between 1980 and 2022.

7.5 Lithuania

The Demonstration Project in Lithuania covers all the agricultural area of the country. Lithuania is located in eastern Europe, near the Baltic Sea. The country has an area of 65,300 km2. According to the Koppen classification, the climate is Dfb (humid continental climate). The average annual precipitation is 675 mm, and the temperature is 6.9 °C (Gomes et al., 2021). In 2024, the country had almost 3 million residents who lived in the three major cities, Vilnius (Capital), Kaunas and Klaipeda. In 2023, the country had a GDP of 73.8 billion EUR.

The agriculture sector is an important economic sector in the national economy. In 2024, the Gross agricultural production was 3,808.6 million Euros. In total, the agriculturally used area in 2023 was 2,872,407 ha. In 2024, the Yield of crops was 4.09 tons per 1 hectare (https://osp.stat.gov.lt/). In this context, it is important to understand the impacts of agricultural activities on ecosystem services, especially because agriculture in Lithuania is based on a high intensification. In this DP, we will assess the impacts of agriculture activities on crop supply, carbon sequestration and erosion regulation. In that context, monetary valuation of these ecosystem services can support decision processes.

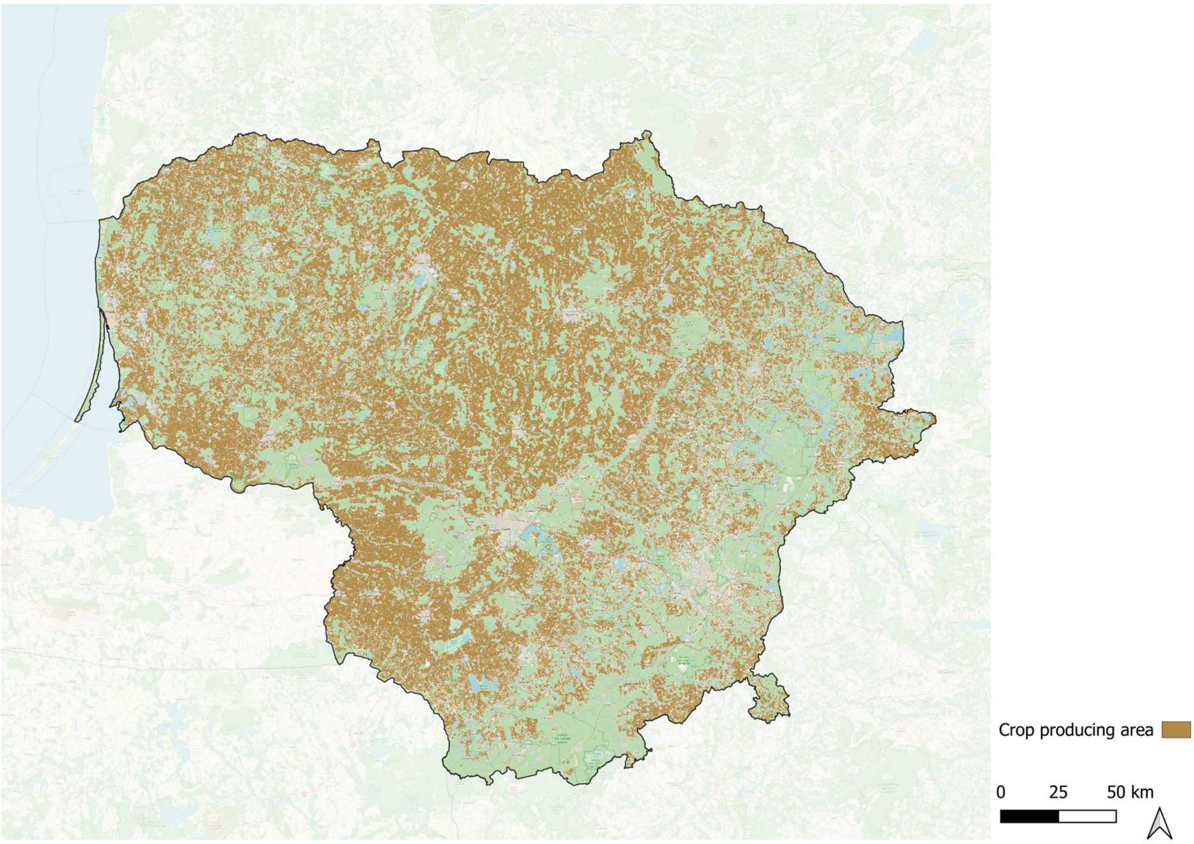

Figure 11. Map of the Lithuanian DP, showing the national area, with the crop producing area in brown.

7.6 Portugal

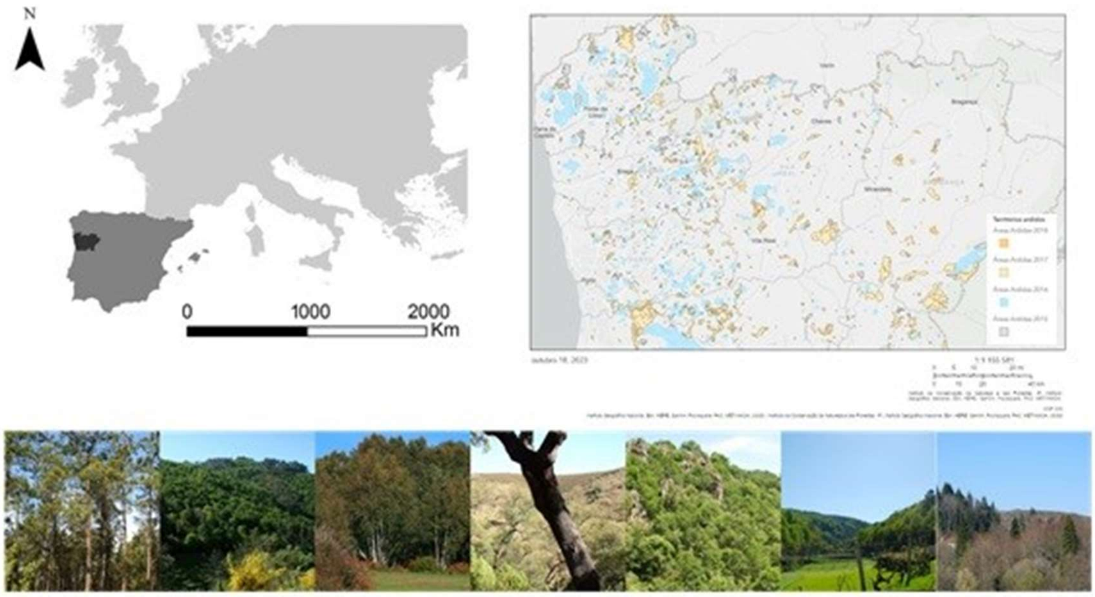

Figure 12: Northern Portugal test site. Left panel: Northern Portugal (NUTS-II EU) test site. Right panel: area burned in Northern Portugal between 2015 and 2018. Bottom panel: forested landscapes in Northern Portugal.

The test site is the Northern Portugal - NUTS-II EU administrative region, covering ca. 21,515 km^2 (Fig. 12). This region has ca. 3.6 million inhabitants, of which 90% live in coastal areas. The employed population (43% of the total inhabitants) primarily works in the tertiary (64%) and secondary (34%) economic sectors, while only 2% are currently employed in the primary sector (INE, 2021). Northern Portugal is predominantly mountainous, with elevation ranging from 0 to 1545 m. It comprises two biogeographical regions, the Atlantic (west) and the Mediterranean (centre and east) (EEA, 2016). In the western coastal areas, the climate is characterised by temperate summers and mild winters, whereas the eastern inland region experiences hot summers and cold winters (Beck et al. 2018). Mean annual precipitation varies from ca. 540 to 1500 mm, while mean annual temperature ranges from ca. 10 to 15 ºC (IPMA, 2011).

The main land use and land cover (LULC) types at the test site are forests (37%), agriculture (29%), and shrublands (22%) (DGT, 2018). Protected areas account for 25% of the total surface area (ICNF, 2023). Forests and seminatural areas in the region provide a wide range of provisioning (e.g., timber, firewood, wild mushrooms), regulating (e.g., carbon storage, soil retention, water regulation), and cultural (e.g., nature-based tourism) ecosystem services (Ribeiro et al., 2011). Fire activity in Northern Portugal is very high, with 11972 fires/year burning 48828 ha/year, primarily forests (56%) and shrublands (40%) (INE, 2024). The 2017 fire season was the most devastating nationwide and among the worst in Northern Portugal, resulting in ca. 90 Mha of burned forest and shrublands (INE, 2024). The intensification of fire regimes in Portugal, mainly driven by human-induced changes in landscape flammability and ignition patterns coupled with more severe climate conditions (Fernandes et al., 2014), has resulted in substantial ecological and socioeconomic damage in recent decades (Mateus & Fernandes, 2014; Mendes et al., 2021).

In this context, assessing the negative impacts of forest fires on ecosystem services is crucial for formulating effective policies regarding fire risk management and the integration of natural capital into decision-making. This test site aims to demonstrate how forest fires that occurred in 2017 negatively impacted a set of key ecosystem services in Northern Portugal and how these negative externalities can be integrated into ecosystem accounts (see Deliverable 5.1). To this end, valuing ecosystem services provided by forests and shrublands is essential to establish a baseline for subsequent assessment of forest fire impacts. Thus, this test site applied a modelling framework to assess timber and cork provisioning services and carbon storage and soil retention regulating services in the pre-fire environment, i.e., within the area corresponding to the fire perimeters prior to fire occurrence, in biophysical and monetary terms.

8 Data collection

This chapter opens with an overview of primary data collected for testing monetary valuation methods in SELINA DPs and TSs (a detailed overview of data collection and valuation methods applied per DP/TS can be found in **Annex 1**). The last section provides an overview of value transfer methods we applied to estimate monetary values for the same ecosystem services as those tested using primary data collection, generating both exchange values and welfare values for comparative analysis.

8.1 Primary data collection

This subsection describes per ES which primary data based valuation methods were tested and which data was collected from it. The exercise of testing valuation methods using primary data in DPs and TSs had the purpose of evaluating the more practical aspects of these valuation methods, specifically related to data availability and information costs. Table 7 gives an overview of the DPs and TSs, the ecosystem services that were valued in each, using which method and data source, as well as the tier this valuation method is placed in. These tiers are defined in NCAVES and MAIA (2022): the higher the tier, the more accurate valuation data are expected to be, but at the same time more demanding to generate. For a detailed overview of the valuation methods used in each of the DPs and TS, see **Annex 1**.

Note that the valuation methods tested through primary data collection do not cover the full spectrum of monetary valuation methods, recommended by SEEA EA or otherwise. This is due to the availability of data and capacity in the DPs and TSs.

Table 7. Overview of valuation methods tested using primary data in partnered DPs and TSs, including information on the sources of valuation data and the SEEA EA Tier of the method.

DP/TS	Ecosystem service	Valuation method	Valuation data source	SEEA EA Tier
Czech Republic	Game meat provision	Market prices	Game meat prices	1-2
	Recreational value of hunting	Consumer expenditure method	Hunter expenditures	1
		Land rental values	Hunting ground rental costs	1
Greece	Crop provision	Market prices	Crop prices	1
	Global climate regulation	Market prices	Carbon credit prices	3
	Biodiversity maintenance	Avoided damages	Biodiversity protection cost	1

	Recreation	SEV+Random utility model	Travel cost	3
		Willingness-to-pay	Contingent valuation	Untiered
Ireland	Recreation	SEV+Random utility model	Travel cost	3
		Willingness-to-pay	Contingent valuation	Untiered
La Réunion	Recreation	Consumer expenditure	Travel cost	1
		Willingness-to-pay	Contingent valuation	Untiered
		Avoided damage cost	Shark risk mitigation cost	1
Lithuania	Crop provision	Resource rent	Production factors	3
Portugal	Wood provision	Market prices	Standing biomass prices	1
	Cork provision	Market prices	Producer prices	1
	Global climate regulation	Market prices	Carbon credit prices	3
		Avoided damage cost	Social cost of carbon	3
	Erosion regulation	Replacement cost	Soil replacement cost	1
		Avoided damage cost	Post-fire soil erosion mitigation cost	1

8.2 Value function transfer

This section describes the work done to create value functions based on the ESVD, and how we applied them to test sites to compare to primary data.

Considerable research effort over the past 40 years has attempted to estimate the economic value of ES provided by all forms of natural capital (MA, 2005; TEEB, 2010; IPBES, 2018). The ESVD provides a global collection of the results of economic valuation studies with details on the type of ecosystem, ES, location, valuation method, and beneficiaries (Brander et al, 2024). The objective of the ESVD is to make the (standardized) economic values openly available to inform public and private decision making on the sustainable use, management and conservation and restoration of nature.

The available stock of information from primary valuation studies is not, however, readily usable for conducting value transfers to inform policy making for two reasons:

1. The data are not globally representative but instead represent the study sites assessed in the literature, which is determined by past research funding and interests. Simple summaries of data from primary valuation studies cannot therefore be treated as representative of the distribution of ecosystem service values.
2. Ecosystem service values are inherently highly spatially variable and influenced by site and context characteristics that determine the supply and demand for ecosystem services. Direct transfer of summary values from available primary valuation studies are therefore likely to result in substantial transfer errors (over- or under-estimation of values).

These challenges can be addressed to some extent through the use of meta-analysis to estimate value functions that can in turn be used to estimate site and context specific values. The aim of the analysis described in this section to develop and test meta-analytic value functions for ecosystem services from four biomes (agricultural land, grassland, inland wetlands, and temperate forests); and to include a number of spatially defined geo-socio-economic parameters to enable the prediction of ecosystem service values that reflects different site and context characteristics (e.g., population density, income per capita, biodiversity intactness, etc.).

8.2.1 Conceptual model for ecosystem service value functions

The conceptual model for ecosystem service value functions is represented in Fig. 13. The dependent variable that is explained by the model is ecosystem service value measured in international dollars per hectare per year at 2020 price levels. Based on the specifications of published meta-analyses of ecosystem service values and on theoretical expectations we define four groups of explanatory variables that represent different determinants of variation in value, these are:

1. Characteristics of the study site (e.g., the area, elevation, ecosystem type, protected status, ecosystem condition).

2. The biophysical and socio-economic context of the study site (e.g., surrounding land cover, income per capita, population density).
3. The ecosystem service valued. Binary (dummy) variables are used to indicate the ecosystem service using the SEEA EA ecosystem service reference list (e.g., biomass provisioning, global climate regulation, pollination, recreation).
4. The methodological characteristics of the valuation application (e.g., valuation of total flow, valuation method, value concept).

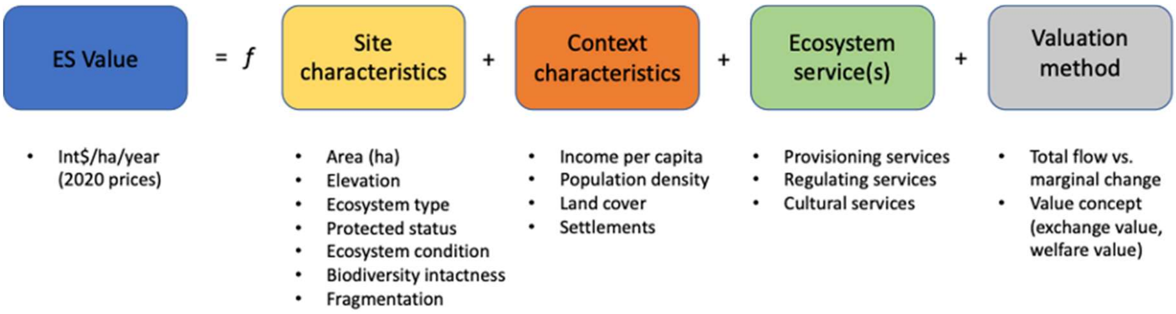

Figure 13. Conceptual specification of an ecosystem service value function.

8.2.2 Description of ESVD and supplementary data

The conceptual value function outlined above is implemented using data from the ESVD, which is supplemented with additional data. Here we provide a brief description of these data.

The data development process for the ESVD is described in detail in Brander et al., (2024). For the present analysis, we used the most recent update of the ESVD, which contains both additional value observations (rows) and additional variables (columns) that are of relevance to this application. Currently, the ESVD contains over 12,000 records derived from more than 1,400 studies. Approximately 70% of these value estimates have been standardised to the common set of units described above; and approximately 50% have undergone expert review. In the ESVD, values that are reported in different units (currencies, spatial scale, temporal units, beneficiary units, price levels) are standardised to a common set of units, namely international dollars per hectare per year at 2020 price levels for all relevant beneficiaries. We use this standardised value as the dependent variable in the meta-regression analyses. The key additional variables that have been included in the ESVD for SELINA are: 1. A variable to indicate whether a primary valuation is for the total flow of an ecosystem or some marginal change in the total flow. The ESVD also contains a text description, obtained from the underlying paper/report, of the ES flow or change in flow. These descriptions are highly diverse in terms of units and types of changes (e.g., extent, condition, and use) and so it is only feasible to specify broad categorical variables; 2. A set of variables that indicate the value concept that is measured by a primary valuation (i.e., exchange value, consumer surplus, producer surplus, avoided cost, or other value concepts).

A number of the variables used to describe the study site and its biophysical and socio-economic context are obtained from spatially referenced datasets. To link spatially defined variables to each valuation study site, we use information on the location and size of each study site recorded in the ESVD. The location of each study site is recorded as a point location using geographic coordinates (decimal degrees) and the spatial extent (area) of each study site is recorded in hectares. Ideally, spatial defined information for each study site would be extracted for its specific location and shape. Point locations for study sites are generally available but information on areal dimensions (e.g., polygons for study sites) are not. In consequence, previous analyses have used point locations to define spatial variables without reference to the shape or extent of the study site (e.g., Brander et al., 2006; 2013). This, however, is a simplification that potentially mis-specifies the characteristics of study sites and their surroundings (Ghermandi and Nunes, 2013). A proposed improvement developed in this paper, is to use the area of each study site to define a polygon (e.g., circle) around its point location as an approximation of its areal dimension. A conceptual representation of this approach is provided in Fig. 14.

A set of potentially relevant spatially defined explanatory variables for inclusion in value functions has been identified through a review of the literature on meta-analytic value transfer, consultation with colleagues working on ecosystem service valuation and discussion with experts at the FAO Office of Climate Change Biodiversity and Environment (OCB). The list of potential variables is provided in the Annex 2 together with the statistic for each parameter and the data source.

These spatially defined variables contain information on ecosystem condition, human pressures, and socio-economic context. The data have varying resolutions ranging from national level to 30-meter resolution. Ideally, the spatially defined variables would also match the time period of the underlying primary valuations. This, however, is challenging given the limited temporal dimension of many datasets, and the large temporal variation in the valuation literature.

The selected valuation study sites were uploaded to Google Earth Engine as a CSV file containing one row per site with information on the year of the study, the point location in decimal degrees longitude and latitude, and the site area in hectares. In Google Earth Engine, study sites were approximated as circular areas defined by the areal extent around each site location. To capture information on the influence of the surrounding area on study site values, three buffer areas of 10, 30 and 50 kilometers were defined. Since *a priori* we do not know at what scale these potential determinants of ecosystem service value operate, we test alternative scales of measurement in the meta-regression to empirically determine the appropriate scale for each variable. Spatially defined variables were aggregated for each of the study sites for four polygons representing the study site itself and the three buffers. Aggregations were made by calculating an average value over the four areas for continuous variables or by calculating the percentage of the area per category for categorical variables. Aggregations were calculated at a scale of 30 meters. For spatially defined variables for which data for several dates are available, only the data with a date closest to the study year was taken into account.

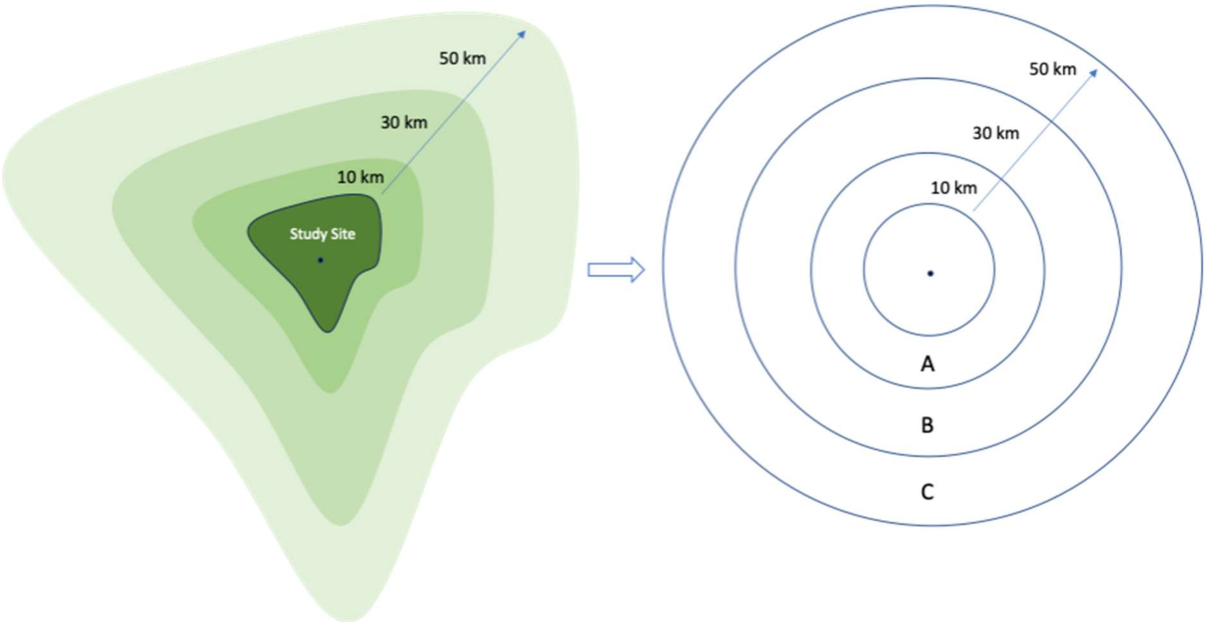

Figure 14. Conceptual representation of study site and buffer areas for extracting spatially defined variables.

8.2.3 Meta-analytic value functions

In this section we present the estimated value functions for five biomes/ecosystems: agricultural land, inland wetlands, grassland, temperate forest and urban green-blue. These value functions are estimated using Ordinary Least Squares (OLS) regression models to explain variation in the dependent variable with a set of explanatory variables.

The modelling process involves the following steps:

- Selection of the sub-set of data from the ESVD for each target biome, ecozone or ecosystem.
- Exclusion of outlier observations for which the ecosystem service value falls outside 1.5 times the interquartile range of log of transformed values.
- For each biome, a number of alternative model specifications were investigated to identify a combination of explanatory variables that are theoretically consistent, (mostly) statistically significant, and explain a high proportion of variation in the dependent variable.

The estimated value functions are provided in Table 8-12 below. The explanatory power of the estimated models varies considerably, ranging from low (grassland model adjusted R2 = 0.35) to high (agricultural model adjusted R2 = 0.7). This is notably high compared with similar meta-analyses of the ecosystem service valuation literature (e.g., Brander et al., 2006; Ghermandi et al., 2010; Taye et al 2021; Brouwer et al., 2022).

In general, the estimated coefficients on the explanatory variables conform to theoretical expectations. For example, the coefficients on the area of the valued site are negative (indicating diminishing returns to scale); the coefficients on population density are positive (indicating higher values with larger numbers of beneficiaries); the coefficients on biodiversity intactness are positive (indicating higher values with higher ecosystem condition).

Regarding the methodological variables, in three of the five models (agriculture, temperate forest, and urban green-blue) we find a statistically significant positive coefficient on the variable indicating that the valuation is for the total flow (as opposed to a marginal change in the flow). In the other two models (grassland and inland wetlands) we do not find a statistically significant effect. We also obtain mostly statistically significant coefficients on the variables indicating whether the value concept that is estimated is an exchange value or consumer surplus (the omitted category is other value concepts such as avoided costs and producer surplus). In almost all cases, exchange values are found to be higher than consumer surplus.

Table 8. Agricultural land value function (dependent variable is the value of ecosystem services in Int.$/ha/year, 2020 prices, natural log).

	B	Std. Error	t	Sig.
Constant	2.8	1.535	1.824	0.069
Area (hectares; ln)	-0.145	0.035	-4.094	<0.001
Elevation (metres; ln)	0.375	0.088	4.268	<0.001
EPI (NPP/maxNPP)	3.487	0.738	4.725	<0.001
Nighttime light	-0.044	0.014	-3.159	0.002
Annual cropland	-0.451	0.277	-1.629	0.104
Perennial monoculture	0.326	0.47	0.693	0.489
Rice paddy	2.964	0.561	5.279	<0.001
Income per capita (Int$; ln)	-0.294	0.126	-2.334	0.02
Population density (10km buffer)	0.003	0.001	3.3	0.001
Crop provisioning	1.499	0.435	3.447	<0.001
Wood provisioning	-1.388	0.538	-2.581	0.01
Grazed biomass	-1.249	1.969	-0.634	0.526
Livestock	-2.403	0.754	-3.188	0.002
Pest control	1.423	0.414	3.438	<0.001
Pollination	2.548	0.719	3.544	<0.001
Soil erosion regulation	0.85	0.499	1.705	0.089
Soil quality regulation	0.73	0.529	1.379	0.169
Water flow regulation	4.008	2.103	1.906	0.057
Recreation	0.355	0.467	0.76	0.448

Visual amenity	-0.594	0.478	-1.242	0.215
Existence and bequest value	2.631	0.539	4.883	<0.001
Total flow	1.493	0.311	4.808	<0.001
Exchange value	1.086	0.42	2.587	0.01
Consumer surplus	-0.476	0.395	-1.206	0.228
N	511			
Adjusted R^2	0.702			

Table 9. Inland wetland value function (dependent variable is the value of ecosystem services in Int.$/ha/year, 2020 prices, natural log).

	B	Std. Error	t	Sig.
Constant	-2.108	2.85	-0.74	0.46
Area (hectares; ln)	-0.368	0.046	-7.942	<0.001
Elevation (metres; ln)	0.582	0.085	6.836	<0.001
Biodiversity intactness (ln)	6.548	3.036	2.157	0.032
Land cover % water (30km buffer)	0.021	0.009	2.36	0.019
Income per capita (PPP; ln)	0.279	0.148	1.89	0.059
Population density (30km buffer)	0.535	0.095	5.61	<0.001
Wood provisioning	-1.877	0.701	-2.676	0.008
Biomass provisioning	-4.518	1.893	-2.387	0.017
Water supply	-1.65	0.605	-2.726	0.007
Baseline flow regulation	2.272	1.341	1.694	0.091
Nursery and habitat	3.226	0.653	4.938	<0.001
Nutrient retention	-1.491	0.495	-3.012	0.003
Soil quality regulation	-3.666	1.046	-3.504	<0.001
Recreation	1.995	0.46	4.337	<0.001
Existence and bequest value	2.915	0.554	5.264	<0.001
Total flow	-0.396	0.304	-1.301	0.194
Exchange value	-0.093	0.406	-0.23	0.818

Consumer surplus	-0.8	0.392	-2.043	0.042
N	409			
Adjusted R^2	0.391			

Table 10. Grassland value function (dependent variable is the value of ecosystem services in Int.$/ha/year, 2020 prices, natural log).

	B	Std. Error	t	Sig.
Constant	2.076	2.842	0.73	0.466
Area (hectares; ln)	0.18	0.067	2.697	0.008
Elevation (metres; ln)	-0.335	0.191	-1.749	0.082
Protected Area % (ln)	0.089	0.127	0.698	0.486
EPI (NPP/maxNPP; ln)	1.946	2.18	0.892	0.373
Nighttime light (ln)	-1.191	0.455	-2.616	0.009
Road density (ln)	-0.635	0.225	-2.817	0.005
Land cover % grassland (10km buffer)	-0.031	0.009	-3.381	<0.001
Population density (30km buffer)	1.257	0.323	3.889	<0.001
Income per capita (PPP; ln)	0.27	0.324	0.835	0.404
Wild animal provisioning	-1.711	0.883	-1.938	0.054
Water supply	-2.579	1.054	-2.447	0.015
Pollination	2.665	1.849	1.441	0.151
Air filtration	-3.087	1.053	-2.932	0.004
Nutrient retention	-3.189	1.52	-2.098	0.037
River flood regulation	-2.063	1.495	-1.38	0.169
Soil erosion regulation	-4.622	1.84	-2.512	0.013
Recreation	-1.333	0.813	-1.639	0.102
Existence and bequest value	0.296	0.745	0.397	0.691
Total flow	0.204	0.64	0.319	0.75
Exchange value	1.003	0.531	1.888	0.06
Consumer surplus	-0.973	0.637	-1.528	0.128

N	248
Adjusted R^2	0.348

Table 11. Temperate forest value function (dependent variable is the value of ecosystem services in Int.$/ha/year, 2020 prices, natural log).

	B	Std. Error	t	Sig.
Constant	2.689	1.012	2.657	0.008
Area (hectares; ln)	-0.239	0.031	-7.748	<0.001
Elevation (metres; ln)	0.073	0.048	1.505	0.133
Protected Area % (ln)	-0.165	0.046	-3.606	<0.001
Temperate evergreen forest	1.527	0.246	6.204	<0.001
Road density (10km buffer; ln)	0.254	0.132	1.923	0.055
Land cover % forest (50km buffer)	0.011	0.003	3.66	<0.001
Population density (10km buffer; ln)	0.272	0.095	2.857	0.004
Income per capita (Int$; ln)	0.245	0.094	2.604	0.009
Crop provisioning	1.07	0.647	1.653	0.099
Wild animal provisioning	-2.006	0.454	-4.415	<0.001
Wood provisioning	-0.808	0.36	-2.248	0.025
Water supply	-2.568	0.371	-6.92	<0.001
Pollination	0.14	0.781	0.18	0.858
Air filtration	-1.177	0.265	-4.438	<0.001
Nutrient retention	-3.47	0.562	-6.179	<0.001
Rainfall regulation	-1.546	0.894	-1.73	0.084
River flood regulation	0.613	1.536	0.399	0.69
Soil erosion regulation	-0.592	0.362	-1.637	0.102
Soil quality regulation	-2.069	0.44	-4.705	<0.001
Storm mitigation	-1.448	0.78	-1.857	0.064
Recreation	-1.238	0.376	-3.291	0.001
Existence and bequest value	0.88	0.476	1.847	0.065
Total flow	0.829	0.246	3.364	<0.001
Exchange value	-0.986	0.297	-3.319	<0.001

Consumer surplus	-0.356	0.364	-0.978	0.328
N	621			
Adjusted R^2	0.448			

Table 12. Urban green-blue value function (dependent variable is the value of ecosystem services in Int.$/ha/year, 2020 prices, natural log).

	B	Std. Error	t	Sig.
Constant	19.903	2.245	8.864	<0.001
Area (hectares; ln)	-0.105	0.058	-1.801	0.073
Air pollution (ln)	-1.181	0.386	-3.064	0.002
Road density (30km buffer; ln)	1.006	0.273	3.68	<0.001
Population density (10km buffer; ln)	-0.233	0.166	-1.402	0.162
Income per capita (PPP; ln)	-1.7	0.169	-10.087	<0.001
Global climate regulation	0.942	0.446	2.113	0.036
Recreation	0.513	0.517	0.991	0.322
Visual amenity	1.797	0.807	2.228	0.027
Total flow	1.072	0.469	2.288	0.023
Exchange value	1.736	0.471	3.69	<0.001
Consumer surplus	2.403	0.586	4.101	<0.001
N	240			
Adjusted R^2	0.392			

8.2.4 Quantification of prediction performance and uncertainty

To explore how accurate the estimated value functions are in predicting values across ecosystem sites, we conduct an in-sample value transfer exercise in which we use the estimated models to predict the primary value of each observation in the underlying data. We then use the results of this exercise to compute the mean and median absolute percentage error (i.e., the percentage difference between the observed and predicted values). The results are presented in Table 13 and show that the mean and median absolute percentage error are approximately 20-57% and 12-44% respectively, which is in line with similar analyses reported in the literature (Rosenberger and Stanley, 2006; Thomassin and Johnston, 2011). Table 14 provides information on the lower and upper bound 95% confidence intervals for predicted values, expressed as the mean percentage adjustment factor to the predicted value. The confidence intervals are wide, indicating the high level of uncertainty in the predicted values.

Table 13. Mean and median absolute percentage transfer error.

	Mean	Median
Agriculture	0.40	0.22
Inland wetlands	0.44	0.32
Grassland	0.43	0.29
Temperate forest	0.20	0.12
Urban green-blue	0.20	0.12

Table 14. Confidence intervals (mean adjustment factor to predicted value).

	Low CI	High CI
Agriculture	0.29	1.38
Inland wetlands	0.01	1.99
Grassland	0.00	2.02
Temperate forest	0.46	1.55
Urban green-blue	0.47	1.53

9 Evaluation results

In this chapter, we present the results of our evaluation of monetary valuation methods. As described in Chapter 6, this evaluation consists of both a quantitative part and a qualitative part. The first section in this chapter describes the quantitative variation in monetary value estimates, showing value transfer results, as well as summary data of the value estimates for primary data collection and value transfer methods, using (extended) SNA-conform methods and welfare based methods using consumer surplus. The second section describes the evaluation using our qualitative framework.

9.1 Variation in value estimates

9.1.1 Value transfer results for SELINA test sites

Here we describe the results of the meta-analytic value transfer application for estimating ecosystem service values for the SELINA test sites. Each country included in this application contains a number of 'policy sites' (in the nomenclature of the value transfer literature) or ecosystem assets (in the nomenclature of ecosystem accounting) for which ecosystem service values are estimated. Table 15 provides information on the number of ecosystem assets per country. The Greek case is for a single ecosystem asset, whereas the Lithuanian and Portuguese cases are for 622 and 737 ecosystem assets, respectively. This provides an opportunity to test the applicability of the value transfer approach to estimate ES values for large numbers of ecosystem assets, controlling for variation in site and context characteristics. It should be noted that we do not have primary valuations for all of these ecosystem assets and the services they provide, and therefore cannot compute transfer errors (i.e., the difference between meta-analytic transfers and primary valuations) to gauge the accuracy of transferred values. The spatial extents of ecosystem assets are diverse, ranging from very small (less than 1 hectare) to very large (over 0.5 million hectares in the Greek and Czech cases). The ecosystem assets are also diverse in terms of ecosystem type. Table 16 provides an overview of the number of ecosystem assets per ecosystem type and country. Note that we currently do not have value functions for two of the listed ecosystem types (rivers and streams and shrubland) and therefore cannot estimate values for these ecosystem assets.

Table 15. Number of ecosystem assets per country with descriptive statistics on asset area (hectares).

	N	Mean	Median	Minimum	Maximum
Czech Republic	188	39,153	6188	<1	547,162
Greece	1	589,261	589,261	589,261	589,261
Ireland	2	316	316	300	333
Lithuania	622	1,907	219	<1	26,504
Portugal	737	89	27	10	3,524
Total	1,550	5,937	62	<1	589,261

Table 16. Number of ecosystem assets per country and ecosystem type.

	Rivers and streams	Wetlands	Shrubland	Temperate forest	Grassland	Agriculture
Czech Republic	8	34		46	27	73
Greece						1
Ireland				2		
Lithuania	77		80	162	80	223
Portugal			230	187		320
Total	85	34	310	397	107	617

To illustrate how the value transfer functions are applied, we provide a worked example for a single ecosystem asset. For this purpose, we selected one of the two ecosystem assets in Ireland (Devils Glen, County Wicklow). This is an area of forest covering 332.6 hectares. The spatially defined variables (e.g., elevation, road density, population density) are obtained using Google Earth Engine following the approach described in Section 8. In order to predict the value of the recreation ecosystem service at this site, the binary parameter for recreation is set to 1. Similarly, the binary parameter for exchange value is set to 1 in order to approximate this value concept. The predicted value for this site is computed by multiplying the estimated coefficients in the temperate forest value function (Table 17, Column 2) by the parameter values for this specific site (Table 17, Column 3). The sum of these products (Table 17, Column 4) gives the natural log of the predicted value, and the value in Int$/ha/year is subsequently computed as the exponent of this number. In this case, the predicted value is 2,023 Int$/ha/year.

This example can be extended to illustrate how predicted values change with different site or context characteristics. For example, by changing the parameter value for population density within a 10km distance of the ecosystem asset from 114 to 200 people per km^2, the predicted value of recreation increases from 2,023 Int$/ha/year to 2,354 Int$/ha/year.

Table 17. Example value transfer application to a single ecosystem asset (recreational value of Devil's Glen forest, Country Wicklow, Ireland).

	Value function coefficients (V)	Site parameter values (S)	V*S
Constant	2.689	1	2.69
Area (hectares; ln)	-0.239	332.6	-1.39
Elevation (metres; ln)	0.073	199	0.39
Protected Area % (ln)	-0.165	0	0.00
Temperate evergreen forest	1.527	1	1.53
Road density (10km buffer; ln)	0.254	994	1.75
Land cover % forest (50km buffer)	0.011	12.7	0.14
Population density (10km buffer; ln)	0.272	114.4	1.29
Income per capita (Int$; ln)	0.245	42,426	2.61
Crop provisioning	1.07	0	0.00

Wild animal provisioning	-2.006	0	0.00
Wood provisioning	-0.808	0	0.00
Water supply	-2.568	0	0.00
Pollination	0.14	0	0.00
Air filtration	-1.177	0	0.00
Nutrient retention	-3.47	0	0.00
Rainfall regulation	-1.546	0	0.00
River flood regulation	0.613	0	0.00
Soil erosion regulation	-0.592	0	0.00
Soil quality regulation	-2.069	0	0.00
Storm mitigation	-1.448	0	0.00
Recreation	-1.238	1	-1.24
Existence and bequest value	0.88	0	0.00
Total flow	0.829	1	0.83
Exchange value	-0.986	1	-0.99
Consumer surplus	-0.356	0	0.00
Int$/ha/year (ln)			7.61
Int$/ha/year			2,023

The application of value functions to all ecosystem assets and services is performed in a similar way but using a database and code developed in SPSS version 29. In total we estimate 3,987 values for all relevant combinations of ecosystem assets and ecosystem services. Table 18 provides mean and median values per hectare for each country. As expected, there is a high degree of variation in the estimated values, reflecting diversity in ecosystem services, ecosystem type, and site and context characteristics. Mean values are substantially higher than median values indicating a skewed distribution with a tail of high values.

Table 19 presents summary statistics on the means and distribution of estimated values by ecosystem service, which highlights the variation in values between and within ES. On average, we estimate high values for recreation, existence values, crop provisioning, and flood regulation, and low values for livestock, grazed biomass, and wild animal provisioning. We note, however, that within these ecosystem services there is substantial variation in estimated values. These summary values can be disaggregated further by country (see Table 20) and ecosystem type (see Table 21) but there remains substantial variation in values within each disaggregated group of ecosystem assets. This emphasizes the key point of using meta-analytic value functions, that ecosystem service values are inherently highly variable reflecting multiple determinants of supply and demand that vary spatially, and it is necessary to account for this in value estimation.

Table 18. Mean and median ecosystem service values by country (Int.$/ha/year; 2020 prices).

Country	N	Mean	Median	Std. Deviation
Czech Republic	180	11,228	445	31,745
Greece	1	203	203	.
Ireland	2	832	832	496
Lithuania	465	6,352	3,414	11461
Portugal	507	2,402	2,153	3153
Total	1,155	5,363	2,348	14,939

Table 19. Mean and distribution of estimated values per ecosystem service (Int.$/ha/year; 2020 prices).

	N	Mean	S.E.	S.D.	Minimum	Maximum
Crop provisioning	223	3086	136	2031	213.23	8715
Wood provisioning	375	402	24	460	60.16	5804
Grazed biomass	223	198	9	130	13.66	558
Livestock provisioning	223	62	3	41	4.31	176
Wild animals provisioning	315	217	21	381	0.86	3606
Water supply	242	126	13	203	0.36	1514
Pollination	162	1939	221	2807	240.95	18423
Soil erosion control	749	1628	93	2544	0.05	54413
Soil quality regulation	162	213	24	308	26.46	2023
Air filtration	242	372	42	655	0.22	4936
River flood mitigation	242	2150	253	3936	0.61	29566
Storm mitigation	162	396	45	574	49.24	3765
Recreation	425	5004	1034	21313	1.26	218343
Ecosystem and species appreciation	242	3425	349	5434	6.42	38614

Table 20. Mean estimated values per ecosystem service and country (Int.$/ha/year; 2020 prices).

	Czech Republic	Greece	Ireland	Lithuania	Portugal
Crop provisioning				3,086	
Wood provisioning					402
Grazed biomass				198	
Livestock provisioning				62	
Wild animals provisioning	119			246	
Water supply				126	
Pollination				1,939	

Soil erosion control				630	2,105
Soil quality regulation				213	
Air filtration				372	
River flood mitigation				2,150	
Storm mitigation				396	
Recreation	11,180	203	832	465	
Ecosystem and species appreciation				3,425	

Table 21. Mean estimated values per ecosystem service and ecosystem type (Int.$/ha/year; 2020 prices).

	Wetlands	Temperate forest	Grassland	Agriculture
Crop provisioning				3,086
Wood provisioning		879		277
Grazed biomass				198
Livestock provisioning				62
Wild animals provisioning		214	222	
Water supply		129	120	
Pollination		1,939		
Soil erosion control		1,091	16	2,618
Soil quality regulation		213		
Air filtration		520	72	
River flood mitigation		3,112	201	
Storm mitigation		396		
Recreation	41,801	466	324	7,746
Ecosystem and species appreciation		4,065	2,130	

Regarding the value concept that is approximated in the value transfer application, the results presented above are for exchange values. To illustrate the potential for using the estimated meta-analytic value functions for predicting welfare values, we repeated the analysis to approximate consumer surplus values by setting the parameter value for consumer surplus to 1 (and the parameter value for exchange value to 0). Fig. 15 provides a comparison of the mean predicted exchange and consumer surplus values for ecosystem services from temperate forests. Exchange values are estimated to be approximately 50% lower than consumer surplus values across all ecosystem services. The inclusion of binary variables for exchange values and consumer surplus in the value function provides a simple empirically determined adjustment in the value transfer application. This approach, however, does not capture potentially more nuanced differences between value concepts across different ecosystem services. Further analysis could attempt to quantify differences in value concepts for specific ecosystem services.

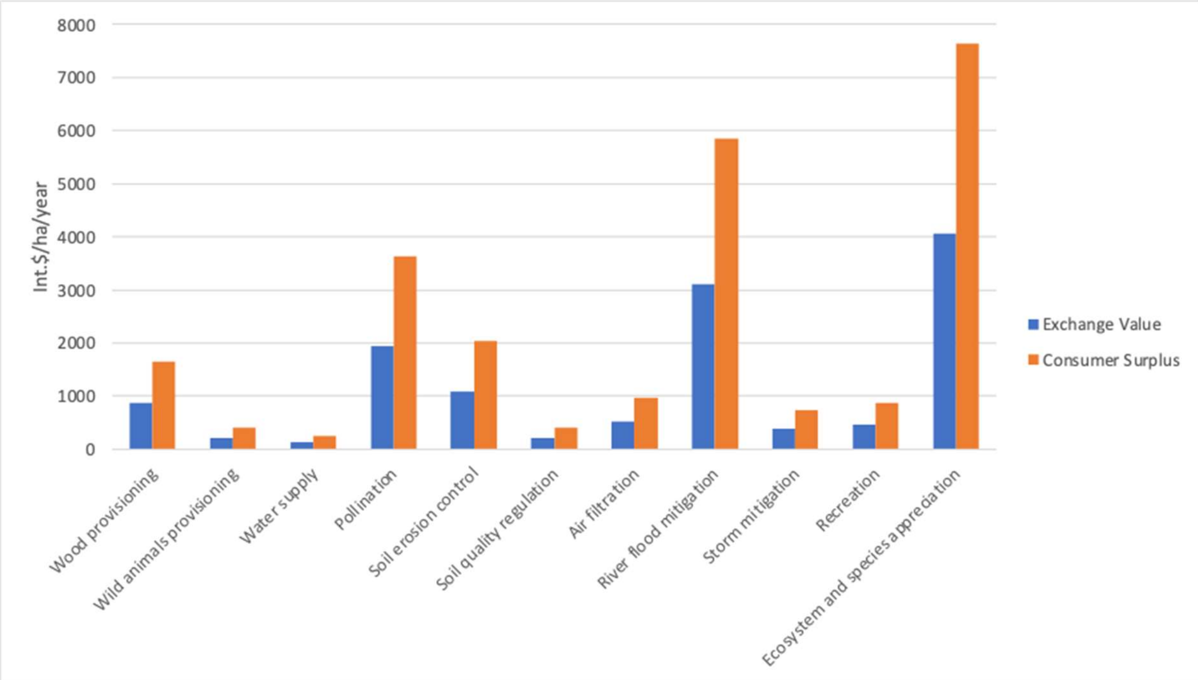

Figure 15. Mean predicted exchange and consumer surplus values for ecosystem services provided by temperate forests (Int.$/ha/year).

The analysis presented above explores and attempts to address some of the limitations to using meta-analytic value transfer methods for monetary valuation in ecosystem accounting, namely the estimation of exchange values and accounting for ecosystem condition and marginal valuations. Other limitations remain and are well described in the literature (Brander et al., 2012; Johnston et al., 2020). We do not revisit these limitations in full here except to note that generalisation error, which occurs when values for study sites are transferred to policy sites that are different without fully accounting for those differences (Rosenberger and Stanley, 2006), is likely in accounting applications for ecosystem assets that are not well represented in the underlying valuation literature and/or have extreme characteristics. It is therefore not advisable to apply value transfer functions to ecosystem assets with parameter values that fall outside of the ranges found in the underlying data.

9.1.2 Summary data of value estimates

The values presented in Table 22 summarise all value estimates standardised to Int.$ per hectare per year in 2020 prices. These values were sourced from the tests run in the DPs and TSs using primary data collection and modelling (see Chapter 8 for details), as well the values generated from the value function transfer test (see Chapter 8 and section 9.1.1 for details).

Table 22. Mean estimated values per ecosystem service and country (Int.$/ha/year; 2020 prices), for both primary data and value transfer data. Welfare based values are marked in *italics*. All values are taken from either the primary data collection or the value function transfer exercise, converted first to 2020 prices and then to Int.$.

Ecosystem service	**Data source**	**Valuation method**	**Czech Republic**	**Greece**	**Ireland**	**Lithuania**	**Portugal**
Crop provisioning	Primary data	(Extended) SNA-conform (Crop prices)		3,891			
		(Extended) SNA-conform (Resource rent)				465	
	Value function transfer	(Extended) SNA-conform				3,086	
	Value function transfer	*Consumer surplus*				*647*	
Wood provisioning	Primary data	(Extended) SNA-conform					2804
	Value function transfer	(Extended) SNA-conform					402
	Value function transfer	*Consumer surplus*					*389*
Cork provisioning	Primary data	(Extended) SNA-conform					41,856
Grazed biomass	Value function transfer	(Extended) SNA-conform				198	
	Value function transfer	*Consumer surplus*				*41*	
Livestock provisioning	Value function transfer	(Extended) SNA-conform				62	
	Value function transfer	*Consumer surplus*				*13*	
Wild animals provisioning	Primary data	(Extended) SNA-conform	3				
	Value function transfer	(Extended) SNA-conform	119			246	
	Value function transfer	*Consumer surplus*	*203*			*298*	
Water supply	Value function transfer	(Extended) SNA-conform				126	
	Value function transfer	*Consumer surplus*				*168*	
Global climate regulation	Primary data	(Extended) SNA-conform (carbon prices)		33			5,348

	Primary data	(Extended) SNA-conform (SCC - global 2017)					3,140
Pollination	Value function transfer	(Extended) SNA-conform				1,939	
	Value function transfer	*Consumer surplus*				*3,642*	
Soil erosion control	Primary data	(Extended) SNA-conform					51,953
	Value function transfer	(Extended) SNA-conform				630	2,105
	Value function transfer	*Consumer surplus*				*1,173*	
Soil quality regulation	Value function transfer	(Extended) SNA-conform				213	
	Value function transfer	*Consumer surplus*				*400*	
Air filtration	Value function transfer	(Extended) SNA-conform				372	
	Value function transfer	*Consumer surplus*				*656*	
River flood mitigation	Value function transfer	(Extended) SNA-conform				2,150	
	Value function transfer	*Consumer surplus*				*3,921*	
Storm mitigation	Value function transfer	(Extended) SNA-conform				396	
	Value function transfer	*Consumer surplus*				*744*	
Recreation	Primary data	(Extended) SNA-conform	55				
	Primary data	*Consumer surplus (TCM)*		*151*			
	Value function transfer	(Extended) SNA-conform	11,180	203	832	465	
	Value function transfer	*Consumer surplus*	*4,738*	*43*	*1563*	*634*	
Ecosystem and species appreciation	Unit value transfer	(Extended) SNA-conform		120			
	Unit value transfer	*Consumer surplus*		*484*			
	Value function transfer	(Extended) SNA-conform				3,425	
	Value function transfer	*Consumer surplus*				*5,207*	

9.2 Qualitative evaluation

In this section we describe the results of our qualitative evaluation framework. The first sections focus on using primary data collection, grouped into three types: SNA-conform approaches, extended SNA-conform approaches (using simulated exchange values), and welfare based valuation approaches. We structured these results per SEEA EA ecosystem service type (provisioning, regulating and cultural): for each service type, we provide our evaluation for each of the valuation approaches along each of the evaluation criteria. In a separate section, we evaluate the use of value transfer methods for generating monetary value accounts using the same framework. Where we did not have access to data from SELINA TSs and DPs, we based our evaluation on literature study and expert judgement.

9.2.1 Primary data - Provisioning services

Our evaluation of monetary valuation methods for biomass provision covered the following provisioning services:

- Crop provisioning services (Lithuania, Greece)
- Wood provisioning services (Greece, Portugal)
- Cork provisioning services (Portugal)
- Wild animals provisioning services (Czech Republic)

For biomass provisioning services, SNA-conform approaches include the direct use of market prices for biomass that is traded on markets, resource rent based valuation by subtracting all human-made costs from market prices, land rent values for agricultural biomass, and productivity change methods using production functions. It should be noted that for provisioning services contributing to products that require high inputs of human capital and labour, using pure market prices is not appropriate as an equivalent value of the ecosystem service, since it also captures the value of those human inputs.

Extended SNA-conform approaches using simulated exchange values are typically less relevant for provisioning services than for regulating and cultural services, since for most provisioning services, markets are available for the products that ecosystems contribute to. In our test cases, this is the case for crop provisioning, wood provisioning and cork provisioning services. Wild animals provisioning services are not always traded on markets in practice, but typically, markets for these or equivalent products are available (game meat, mushrooms and berries are examples). Therefore, extended SNA-conform approaches for provisioning services are typically only relevant when market data are not available.

The welfare-based approach to valuation focuses on measuring changes in overall social well-being or utility derived from changes in the flow of provisioning ecosystem services. This approach typically aims to capture the total economic value, which includes not only the exchange value (reflected in market prices) but also the consumer surplus (the difference between what consumers are willing to pay and what they actually pay) and potentially non-use values (values that individuals hold for an ecosystem service even if they do not directly use it). For biomass provisioning services, this data is typically less common to generate than exchange-based values using SNA-conform approaches. Stated preference methods, such as contingent valuation or discrete choice experiment methods, can be used to estimate willingness-to-pay for renting land used for provisioning services, as was done for example for farmland (Buchholz et al. 2022). However, this is not typical and the SEEA EA Statistical Paper, when discussing the inclusion of welfare values, assumes it is reasonable that for provisioning services there is good alignment between exchange values and welfare values (UN 2024).

Table 23. Summary overview of evaluation for provisioning services using primary data.

Valuation method	**Timeliness**	**Salience**	**Credibility**	**Cost proportionality**	**Legitimacy**
SNA-conform approaches	Medium to high	High	Medium	Low to high	Medium
Extended SNA-conform approaches	Medium	Medium to high	Medium	Low	Medium
Welfare based approaches	Low to medium	Medium to high	Low to medium	Low	High

9.2.1.1 Timeliness

SNA-conform approaches

During our testing, we found that for valuation based directly on market prices approaches for timber and major agricultural crops, recent biomass and price data is typically readily available through statistical reporting on the national level, since this is already part of the SNA. Using resource rent, land rent and productivity change methods, which are better approximations of the ecosystem's contribution to the value of the product, yielded lower availability of recent data: obtaining comprehensive and up-to-date data on all production costs, including capital costs and labour compensation, can be more challenging and might involve some time lags. For instance, accurate data on farm profits for agricultural biomass may not always be easily accessible at the regional or local level. Though we have not had data available to test the resource rent, land rent and productivity change methods in any of our cases, these methods require data that is often updated less periodically and is harder to aggregate than using less accurate pure market price methods.

When using market prices, valuation model processing time is generally low, as it primarily involves collating existing statistics with price data without the need for complex modelling. Using the resource rent and land rent methods requires slightly more time, since it involves collecting more micro-economic data at the farm level (if available), though it also does not require complex modelling. More demanding to generate, time-wise, is using production functions to estimate productivity change. This requires econometric estimation of production functions to determine the relationship between various inputs (including ecosystem contributions) and resulting biomass output.

The market price, resource rent, land rent and production function methods are all relatively easy to integrate with SEEA accounting periods, because the required economic input data reflect economic transactions that occur within specific timeframes that can be aligned with SEEA EA reporting periods (Grammatikopoulou et al. 2024).

Extended SNA-conform approaches using simulated exchange values

Based on our data collection, we assess the availability of recent data for the SEV method to be medium for biomass provisioning services. However, this depends on the specific data requirements of the chosen model, which could include historical prices, production costs, demand parameters, and other relevant economic and biophysical data. The timeliness of the valuation will be contingent upon the availability and currency of these underlying data inputs. In our cases, not enough input data was available to generate simulated exchange values in parallel to using other SNA-conform exchange value estimates.

The valuation model processing time for the SEV method can range from medium to high, depending on the complexity of the model being used. Simpler models might involve statistical regressions or basic supply-demand simulations, while more sophisticated models could incorporate dynamic elements, spatial considerations, or agent-based simulations, leading to longer processing times. The ability to generate these values also depends on the ability to generate meaningful demand and cost functions in the absence of market data (Barton et al. 2019).

Simulated exchange values can be highly integrated with SEEA EA reporting periods for integration into ecosystem accounts, as long as the model is using a compatible temporal resolution.

Welfare-based approaches

The availability of recent data for stated preference studies are likely to be low. Conducting original CV or DCE surveys to gather new data can be time-consuming, involving survey design, sampling, data collection, and analysis.

The valuation model processing time for stated preference studies is medium. It involves statistical analysis of the survey responses to estimate WTP or WTA values, which can range from simple descriptive statistics to more complex econometric modelling depending on the survey design and research questions.

Stated preference methods have relatively low potential for integration with SEEA accounting periods. In theory, surveys could be deployed to coincide with accounting periods, but this would require substantial data collection costs compared to SNA-conform valuation methods.

9.2.1.2 Salience

SNA-conform approaches

Accepted SNA-conform approaches for valuation of biomass provisioning services, specifically the resource rent method, land rent method (for crop and grazing biomass) and the productivity change method, all provide information which is highly relevant to policy questions that ecosystem accounting can inform. These methods hold high relevance for policy questions related to the sustainable management of natural resources, resource taxation, and the allocation of returns from biomass extraction or harvesting. By isolating the contribution of the ecosystem to exchange values of produced biomass, policymakers can for example align subsidies with the actual ecosystem productivity rather than just production volume of the final product. In that way, if marginal lands produce lower-value crops but degrade faster, subsidies or other support mechanisms could prioritize high-yield sustainable lands over expansion into fragile ecosystems. When using pure market prices for valuation, which is in most cases not recommended in SEEA EA, this is not possible since it does not clarify the contribution of the ecosystem explicitly.

We estimate that the ease of understanding of using the resource rent method and land rent method (for crop and grazing biomass) are higher than for the productivity change method for most policymakers. The method relies on the technical concepts of production functions and econometric modelling, which might require specialized knowledge to fully grasp the underlying mechanisms and assumptions and might make this harder to communicate. The use of market prices (where appropriate) is the easiest SNA-conform valuation method to communicate.

The ability of these methods to capture spatial variation depend for a large part on the biophysical data that shows where biomass production happens. While prices can vary regionally due to transportation costs, local supply and demand conditions, and other factors, these variations might not always reflect fine-scale spatial differences in ecosystem service provision. For example, spatial variation in timber prices is influenced by distance to markets and resource stocks, and fine scale data on these variations are typically not available. For capturing national or regional variation, such statistics may be available, for instance collected by national statistics bureaus. Land rent values used in the land rent method are better able to distinguish spatial variation, as long as data on land rent values are available. In our study cases, the best available data was found in Eurostat's data browser[10]. If input data is available, the productivity change method is best able to capture spatial variation: By utilizing spatially explicit data on inputs (e.g., soil maps, climate data, land cover) and outputs (yield maps), the production function can be estimated to reflect local ecological and agricultural conditions (NCAVES and MAIA 2022).

The resource rent method, direct market price method and productivity change method have a high ability to capture temporal variation as it relies on market prices and cost data, both of which can fluctuate over time in response to changing economic and market conditions, and are typically reported at multiple administrative levels at sub-annual time periods. The land

[10] Agricultural land prices and rents - statistics | Eurostat

rent method has a slightly lower ability to capture temporal variation. While rental rates can change over time in response to shifts in agricultural markets and land values, they might not always reflect short-term fluctuations in biomass prices. The highest temporal resolution we found was at the annual level, which for most ecosystem accounting purposes is adequate.

Extended SNA-conform approaches using simulated exchange values

The SEV method can be relevant for supporting policy development, particularly in cases where market failures, such as the absence of well-defined property rights or externalities, prevent the direct observation of market prices for biomass provision. By simulating market conditions, the method can provide valuable insights into the potential exchange value of biomass and inform policy interventions aimed at improving market efficiency or addressing externalities. However, this method is typically more useful for those ecosystem services where markets do not exist, making this method less relevant for biomass provisioning services than for regulating and cultural services. Exceptions might be specific types of wild biomass that are not traded on markets, though for those types of biomass, markets for comparable products can be used.

We expect that the SEV method is relatively easy to understand for users of ecosystem accounts. However, while the underlying goal of estimating an exchange value is consistent with market-based policy thinking, the specific modelling techniques and assumptions employed in the SEV method might require some level of technical explanation to ensure policymakers can interpret the results appropriately.

The SEV method has a high ability to capture spatial variation. Economic models can be developed to incorporate spatially explicit data on biomass supply, demand, transportation costs, and other regional factors, allowing for the estimation of exchange values that reflect geographical differences.

Similarly, the SEV method can be designed to capture temporal variation. Dynamic models can incorporate changes in market conditions, technological advancements, or policy shifts over time, providing insights into how the exchange value of biomass provision might evolve.

Welfare-based approaches

Welfare-based approaches potentially hold medium to high relevance for policy questions that require understanding the total economic value of biomass provision, including non-marketed aspects and societal preferences for sustainability or environmental impacts associated with biomass production, which cannot be captured by exchange values. This is less relevant for services generated on private land. For wild biomass provision coming from free access ecosystems, the difference in values between accounting and welfare-based valuations may provide relevant information on unrealized values, which could be obtained with changes in policy or management regimes for ecosystem assets.

The concept of willingness to pay is generally understandable by policymakers, although the technical details of stated preference methodology and the potential for biases might require careful communication.

Stated preference studies can capture spatial variation by targeting surveys to specific geographic areas and incorporating location-specific scenarios or characteristics into the valuation questions.

Stated preference studies can be repeated over time to assess changes in preferences and values for biomass provision, allowing for the analysis of temporal trends, though consistency in sampling is key to creating consistent outputs.

9.2.1.3 Credibility

SNA-conform approaches

The resource rent method relies on observed market prices for the biomass and reported cost data, lending it a relatively high degree of credibility. However, the allocation of certain costs, and the determination of a "normal" return on capital to establish the ecosystem's contribution to the good's production might involve assumptions that reduce this method's ability to base itself purely on observed data. Similarly, the productivity change method relies on observed data regarding agricultural inputs and outputs, as well as potentially observed data on ecosystem service provision, lending it a reasonable degree of credibility. The land rent method (for crop and grazing biomass production) is less dependent on assumptions, as long as data on land rent prices are available.

When it comes to the transparency of the assumptions behind the valuation methods based on observed prices, assumptions about market efficiency, the absence of significant distortions (such as subsidies or taxes), and the representativeness of the observed prices for the specific biomass in question need to be considered. Higher transparency needs are required for the productivity change method, since the creation of production functions can come with uncertain assumptions. The specific functional form chosen for the production function, the selection of relevant input variables, and the assumptions made about the relationships between these variables and biomass output can significantly influence the results and require careful justification for this method to be credible. The robustness of the method depends heavily on the correct specification of the model and the availability of high-quality data on all relevant inputs, including ecosystem services.

The scientific justification for the use of SNA-compatible exchange values for valuing provisioning ecosystem services is relatively strong, though in all applications of ecosystem accounts, it should be made clear that these values do not represent all possible values society places in the ecosystem service, but only the contribution of the service to economy, without taking into consideration consumer surplus. The use of resource rent, land rent and productivity change methods are accepted by the scientific community as appropriate for estimating exchange values of ecosystem services.

Extended SNA-conform approaches using simulated exchange values

The SEV method itself involves simulation, it relies on underlying observed data for the estimation of model parameters, the calibration of model behaviour, and the validation of model outputs against real-world data where available. The quality and representativeness of this observed data are crucial for the credibility of the simulated exchange values. This makes this method potentially similarly credible as SNA-conform methods directly based on observed data.

The transparency of assumptions in the SEV method can range from low to medium. The complexity of the economic models used can sometimes make it challenging to fully articulate and understand all the underlying assumptions. Clear documentation of the model structure, the data sources used, the behavioural assumptions made, and the sensitivity of the results to these assumptions is essential for enhancing transparency.

The SEV method is accepted within the scientific community as a valid method for monetary valuation of ecosystem services for use in SEEA EA (Barton et al. 2019). However, its validity depends on its ability to generate appropriate demand and cost functions, which requires assumptions that require clear justification. The SEV method is considered mostly valid in the absence of real markets, such as for regulating and cultural services. One issue with using simulated markets is that the simulated value is based on behaviour following the logic of a free good: if harvesting wild biomass, such as foraging of berries and mushrooms, would come at a cost for the consumer, they would likely not demand as much of it because it comes with an opportunity cost. Therefore, though the price estimates per unit as generated by SEV might be accurate, the physical quantity of the consumed service might be higher than would realistically be, leading to an overestimation of the value of ecosystem asset accounts (Barton et al. 2019).

Welfare-based approaches

Stated preference methods have a low use of observed data as it relies on stated preferences in hypothetical scenarios, which might not always reflect actual behaviour or market transactions, limiting their credibility when compared to using exchange values for provisioning services.

The transparency of assumptions in stated preference studies can be medium. Researchers need to clearly document the survey design, the hypothetical scenarios presented, the sampling methods, and the statistical analysis techniques used to ensure transparency and replicability (Johnston et al 2017).

In general, stated preference studies have a medium level of acceptance within the scientific community. However, its use is mainly considered appropriate for valuing non-market goods and services, in which it should still be carefully interpreted due to its hypothetical nature and the potential for various biases in survey responses. For biomass provisioning services, such as crop provision, wood provision and wild biomass provision, it is typically less accepted as a method for valuation and mainly used on an experimental basis (Buchholz et al. 2022).

9.2.1.4 Cost proportionality

SNA-conform approaches

Our test cases focused on relatively low-cost methods, which were typically exchange value-based SNA-conform methods. The lowest cost for collecting the necessary data is to use direct market prices, which only requires quantities of the ecosystem service provided combined with price data. For this method, the data collection cost is low to medium, as price data for many biomass commodities are often readily available from market reports, government statistics, and industry publications. However, this method is not recommended for provisioning services that are part of production functions involving significant human input to produce outputs, such as crop or wood production. The resource rent method has a higher data collection cost, as it necessitates gathering detailed information on both the revenues from biomass and all the associated costs of production. An example of the required information necessary to calculate resource rent-based methods can be found in Oras et al. (2023):

Output
- intermediate consumption
- compensation of employees
- other taxes on production
+ other subsidies on production
 = Gross operating surplus
Gross operating surplus
- consumption of fixed capital (depreciation)
- return to produced assets
- labour of self-employed persons
 = Resource rent = Depletion + net return to environmental assets

Farm structure surveys, such as collected by Eurostat, can provide some of this information. This was the method used in Lithuania for example. However, for local accounting the information cost for collecting this data consistently can be relatively high. The land rent method has a lower cost for collecting the required data. In our cases, we were able to use data collected by Eurostat at the NUTS-2 level[11]. Using the productivity change method requires the highest cost of information, though in Europe it is feasible to collect the required data from a combination of sources, such as Eurostat, FAO-stat and data collected by the Joint Research Centre (European Commission et al. 2024).

The level of expertise required for applying the market price method is generally low to medium, primarily requiring a basic understanding of economic principles and market analysis. Similarly, the land rent method requires a fairly low level of expertise, primarily a basic understanding of land economics and agricultural practices. Applying the resource rent method requires a higher level of expertise in economics and accounting to properly identify, allocate, and analyse the relevant cost and revenue data. The highest level of expertise is

[11] Agricultural land prices and rents - statistics | Eurostat

required for applying the productivity change function. This requires not just information on marginal productivity and elasticities of production factors, but also econometric modelling experience.

Processing and analysis cost follow the same logic as the cost of collecting data and the level of expertise required: generally, using direct market prices and lent rent methods takes least time, using the resource rent method requires more time to analyse the data and generate values, while the productivity change method using production functions has the highest processing and analysis cost. In our cases, we have for this reason not been able to estimate production functions and base our evaluation on previous work (European Commission et al. 2024).

Extended SNA-conform approaches using simulated exchange values

The data collection cost for the SEV method can be similar to the productivity change method when using real markets. For provisioning ecosystem services, it would be most appropriate to base the SEV method on production functions for similar goods in case the studied good is not priced on a market. Building and calibrating economic models for these requires a significant amount of data, potentially including information on a realistic institutional context, market prices of related goods, production costs, consumer preferences, and biophysical characteristics of biomass resources.

Applying the SEV method necessitates a high level of expertise in economic modelling, simulation techniques, and statistical analysis. The development and application of these models typically require skilled economists and modelers with a strong understanding of market dynamics and the specific characteristics of biomass provision.

The processing and analysis costs for the SEV method are high. Building, calibrating, running, and interpreting the results of economic simulation models can be computationally intensive and require specialized software and technical expertise.

Welfare-based approaches

The data collection cost for using stated preference methods is high, as conducting well-designed and statistically robust surveys can be expensive and time-consuming, involving significant resources for sampling, survey administration, and data entry.

Designing and implementing a credible CV study requires a medium level of expertise in survey design, questionnaire development, sampling theory, and statistical analysis (Johnston et al. 2017).

The processing and analysis costs for CV are medium, involving statistical analysis of survey data, which might require specialized software and analytical skills (Johnston et al. 2017).

9.2.1.5 Legitimacy

SNA-conform approaches

These methods (market prices, resource rent) generally have high institutional acceptance due to alignment with SEEA EA and national accounting principles in Europe (NCAVES and MAIA 2022). However, they typically score low on stakeholder representation beyond technical experts, as the process focuses on economic data analysis rather than broad engagement in design or interpretation. For instance, while market prices for crops were used in test cases, the process of collecting and applying this data for ecosystem accounting did not directly involve farmers in defining the valuation parameters. They also have low inclusivity of value perspectives, primarily capturing market exchange values and not non-market values important to all stakeholders (IPBES 2022). The medium overall legitimacy reflects the trade-off between high institutional acceptance and limited representation and value inclusivity.

Extended SNA-conform approaches using simulated exchange values

Extended SNA-conform approaches using SEV have medium institutional acceptance, aiming to provide values consistent with SEEA EA though not as well accepted as values based on real market transactions. Stakeholder representation is typically low in the technical model development and simulation process, though data sources (e.g., from surveys for demand functions) might involve some stakeholder input (IPBES 2022). Our work in the test sites, where limited input data hindered the application of SEV for provisioning services, highlighted that data gaps can impact the transparency of the process. Inclusivity of value perspectives is medium, as they attempt to simulate exchange values based on estimated demand, which can reflect some user preferences, but they do not fully capture non-market or passive use values since they are per definition (hypothetical) market-based. The technical complexity also poses challenges for process transparency.

Welfare-based approaches

Welfare-based methods (stated preference) have relatively high potential for stakeholder representation by directly involving diverse groups through surveys in design and data collection (IPBES, 2022; MDPI, 2020), though these are uncommon methods for provisioning services. Test site experience with these methods for cultural services (e.g., cultural in Greece and La Réunion DPs), where surveys were conducted with visitors and residents, demonstrates this direct engagement. They offer higher inclusiveness of value perspectives, capturing a wide range of values including non-use values not reflected in market data. Their institutional acceptance for formal SEEA EA accounting is lower compared to SNA-conform methods due to their hypothetical nature and focus on total value rather than exchange value (SEEA, 2022), though for other uses, such as benefit-cost analysis, welfare values are accepted.

9.2.2 Primary data - Regulating and maintenance services

Our evaluation of monetary valuation methods for regulating services covered the following services:

- Global climate regulation services (Greece, Portugal)
- Soil retention services (Portugal)

Though this is a limited number of regulating ecosystem services, the valuation methods for other services use similar methodologies, so we assess that we are able to evaluate the relevant valuation approaches for those services without having to test their modelling in our DPs/TSs.

For the regulating services we evaluated, SNA-conform approaches include the direct use of market prices for global climate regulation through the use of carbon pricing mechanisms like the European Union Emissions Trading System (EU ETS). For soil retention, we evaluated the avoided cost method, an SNA-conform approach which uses the costs of marketed goods and services needed to replace soil itself or its nutrients (Panagos et al. 2018). Alternative methods are to use the productivity change method (discussed in section 9.2) to isolate the contribution of soil nutrients to the production function, or to estimate avoided cost of dredging sediment from hydropower dams, which is only relevant in specific catchments and was not possible in our test cases.

Extended SNA-conform approaches using simulated exchange values can be used for valuing global climate regulation by using simulated carbon prices, derived from models that estimate the marginal abatement costs of greenhouse gas emissions or from integrated assessment models, represent an extended SNA-conform approach to valuing global climate regulation in Europe (Remeur 2020). Demand functions for regulating services can also be estimated by using stated preference studies using the CVM or DCE methods, though this is not a common approach.

The welfare-based approach to valuation of regulating services focuses on measuring changes in overall social well-being or utility derived from changes in the flow of services. This approach typically aims to capture the total economic value, which includes not only the exchange value (reflected in market prices) but also the consumer surplus (the difference between what consumers are willing to pay and what they actually pay) and potentially non-use values (values that individuals hold for an ecosystem service even if they do not directly use it). For regulating services, as for provisioning services, this data is typically harder to generate than exchange-based values using SNA-conform approaches, and it is less common to do so than for cultural services. Welfare based valuation is more common for the nursery population and habitat maintenance service, which is a supporting service classified as a regulating service under SEEA EA (UN 2024).

Table 24. Summary overview of evaluation for regulating services using primary data.

Valuation method	Timeliness	Salience	Credibility	Cost proportionality	Legitimacy
SNA-conform approaches	Medium to high	Medium to high	Medium	Low to medium	Medium
Extended SNA-conform approaches	Low to medium	Medium to high	Low to medium	Low to medium	Medium
Welfare based approaches	Low to medium	Low to high	Low to medium	Low	High

9.2.2.1 Timeliness

SNA-conform approaches

The availability of recent European data for carbon prices under the EU ETS is high (Remeur 2020). The EU ETS, a cornerstone of the EU's climate policy, operates as a cap-and-trade system, creating a market for carbon emission allowances. This market generates readily accessible and frequently updated data on the price of emitting carbon dioxide (CO_2) and other greenhouse gases. Platforms like the European Energy Exchange (EEX) provide real-time and historical data on EUA (European Union Allowance) prices, which are denominated in Euros per tonne of CO_2 equivalent[12]. Dedicated carbon market tracking services and reports from financial institutions also offer comprehensive information on price trends and market dynamics.

Biophysical estimation of soil erosion control often employs RUSLE models, which can then be used to relate vegetation to a decrease in soil erosion (Natural Capital Project 2025, Buchhorn et al. 2022). This data can be generated relatively quickly, as long as up to date input data on soil conditions are available. In our test cases, this was the case, but this is not the case for all of Europe. While agricultural statistics, soil surveys, and market prices for fertilizers provide some of the necessary information, especially at the national level, comprehensive and spatially explicit data on the actual costs of replacing soil lost to erosion across Europe may be less readily available and consistently reported. Studies at the EU level and in specific countries have estimated the economic losses due to soil erosion, which can provide a basis for inferring replacement costs. However, the direct costs of physically replacing eroded soil or the specific nutrient losses at a granular spatial scale across Europe might require additional data collection or estimation. In sum, the availability of information

[12] https://www.eex.com/en/market-data/market-data-hub/environmentals/spot

to accurately value soil erosion control using avoided cost methods can potentially limit the timeliness of this method, depending on local data conditions.

The valuation model processing time for utilizing EU ETS carbon prices is low. Because the value is derived directly from observed market transactions, minimal processing is required to apply these prices in valuation exercises. Analysts can readily use the prevailing market price to estimate the economic value of avoided carbon emissions or the cost of emitting them.
The processing time of using replacement costs for soil retention is relatively high: if it is based on fertilizer prices, it involves relatively simple calculations once data on nutrient loss is available. However, modelling avoided costs of erosion might require more extensive hydrological modelling to estimate how much sediment runoff drains into water courses.

The ability for carbon prices-based valuation methods to integrate with SEEA accounting periods is high. Carbon price data from the EU ETS is typically available on a daily or weekly basis, allowing for straightforward aggregation to annual or other accounting periods as defined within the SEEA framework. Furthermore, the development of carbon accounts is an active area within SEEA EA, with efforts underway to integrate data on carbon emissions, sequestration, and related economic activities into a coherent accounting framework. The existence of EU-wide carbon pricing data facilitates the compilation of monetary accounts for global climate regulation at the national and supranational levels in Europe.

For soil retention, agricultural statistics, and market prices for inputs like fertilizers are usually available on an annual basis. Data on soil condition and erosion might be collected less frequently but can often be used to support annual estimations or interpolations.

Extended SNA-conform approaches using simulated exchange values

The availability of recent European data for using simulated values based on marginal abatement costs of greenhouse gas emissions or from integrated assessment models is high. Numerous models exist that estimate the cost of reducing greenhouse gas emissions in Europe, considering various technological options and policy scenarios, allowing for estimation of a demand function. Integrated assessment models, which link climate science with economic factors, also provide projections of future carbon prices under different emission pathways. These models are often developed and updated by European research institutions, government agencies, and international organizations[13].

Simulated prices for soil retention, based on demand functions that estimate the avoided costs of soil erosion, offer another extended SNA-conform valuation approach for this ecosystem service in Europe. The availability of recent European data for this method is medium. Various models are available that estimate soil erosion rates across Europe, taking into account factors like rainfall, topography, soil type, and land cover. These models can also be used to assess the impacts of soil erosion on agricultural productivity, water quality, and infrastructure, providing a basis for estimating the economic damages that are avoided when

[13] https://www.europarl.europa.eu/RegData/etudes/BRIE/2020/649352/EPRS_BRI(2020)649352_EN.pdf

soil retention services are maintained (Panagos et al. 2018). However, data availability for estimating an appropriate demand function can be limited, especially at sub-national level.

The valuation model processing time for simulated carbon prices and soil erosion models can range from medium to high. Running complex climate-economic models, especially integrated assessment models, can be computationally intensive and time-consuming, often requiring specialized software and expertise. While some simplified models might have lower processing times, those that aim to provide detailed and scenario-based carbon price estimates typically involve significant computational resources and analytical effort. Running soil erosion models and subsequently estimating the associated avoided costs involves data processing, spatial analysis, and potentially the integration of biophysical and economic models (Panagos et al. 2018). The complexity and time required for this processing will depend on the scale of the analysis and the sophistication of the models used.

Integration with SEEA accounting periods is high. The outputs from climate-economic models, including simulated carbon prices, can generally be generated for different time periods, allowing for their integration with the annual or other accounting periods used in the SEEA framework. These models can provide carbon price trajectories that span several decades, enabling the valuation of climate regulation benefits over relevant time horizons for accounting purposes. The outputs from soil erosion models and avoided cost estimations can typically be aligned with the annual or other periodic accounting cycles of SEEA EA. However, the consistency and frequency of these modelling efforts across different regions and for different accounting periods might vary, potentially requiring some level of temporal interpolation or extrapolation (Panagos et al. 2018).

Welfare-based approaches

Welfare-based valuation methods for global climate regulation and soil retention are similar, since they are both based on stated preference methods. Similarly to provisioning services, regulating services are typically not valued using these methods.

For stated preference methods like CV and choice modelling, the timeliness of the data depends on the recency of the surveys conducted to elicit willingness to pay for the service. For regulating services, we assess this data to be unavailable and would need to be collected for each accounting period.

Stated preference methods like CV and choice modelling involve complex econometric analysis of survey responses, which can be time-consuming, especially with large datasets and sophisticated models.

Integrating the welfare-based approach for regulating services with SEEA accounting periods presents challenges due to the fundamental difference in valuation principles: if survey data can be collected and analysed consistently with accounting periods, nothing inhibits this method from following accounting periods, though the cost of doing so might be high.

9.2.2.2 Salience

SNA-conform approaches

The salience of using EU ETS carbon prices for valuing global climate regulation is high within the European policy context. Carbon pricing, through the EU ETS, is a central instrument in the EU's strategy to achieve its climate targets, including the long-term goal of climate neutrality. As such, the carbon price serves as a direct indicator of the economic value placed on reducing greenhouse gas emissions by European policymakers and market participants. This makes it a highly salient metric for informing policy decisions, assessing the costs and benefits of climate mitigation measures, and evaluating progress towards emission reduction commitments.

The salience of the replacement cost method for soil retention is also high within the European policy context (Panagos et al. 2018). Soil health and the prevention of soil erosion are recognized as critical for sustainable agriculture and environmental protection in Europe. The EU's Common Agricultural Policy (CAP) and various environmental strategies include measures aimed at addressing soil degradation and promoting soil conservation. Valuing soil retention using a cost-based approach directly relates to the economic implications of soil loss, making it a salient issue for policymakers (Panagos et al. 2018).

The suitability of the SNA-conform approach for generating spatially explicit valuations in Europe varies depending on the specific ecosystem service and the availability of spatially disaggregated market data or biophysical data that can be linked to market values. For some services, such as agricultural output (relevant to soil retention) or timber production (related to habitat maintenance and climate regulation), market data might be available at regional or even smaller scales, allowing for the creation of value maps.

The potential for achieving temporal consistency of monetary values for regulating services is high, but requires using the same types of market data or proxies across different accounting periods and applying appropriate discounting rates when valuing ecosystem assets based on future service flows. Integrating the effects of long-term changes in ecosystem services, such as degradation due to climate change or pollution, into SNA-conform valuation typically involves assessing the impacts of these changes on market-valued goods or services. For example, soil degradation could be linked to reduced agricultural yields, which can be valued using market prices. Maintaining temporal consistency also necessitates adjusting monetary values for inflation to ensure comparability across different time periods.

The SNA-conform approach demonstrates varying degrees of relevance to regulating services. For global climate regulation, the approach is relevant through the valuation of carbon sequestration using EU-ETS prices, which directly reflects a market mechanism for this aspect of the service. It can also be relevant through avoided cost methods that assess the economic damages prevented by climate regulation, provided these damages are linked to market sectors. For soil retention, the approach is relevant by valuing its role in supporting agricultural productivity, using methods like replacement cost of nutrients or avoided costs of erosion-related damages to agriculture or infrastructure.

The results generated by the SNA-conform approach can be relatively easy to communicate to policymakers, businesses, and the public in Europe, particularly those familiar with standard economic indicators and national accounts. Expressing the value of ecosystem services in terms of market prices, costs, or contributions to GDP allows for direct comparison with other economic activities. For example, stating that soil retention supports a certain value of agricultural output or that carbon sequestration has a market value based on EU ETS prices can be readily understood by economic actors. However, communicating the value of regulating services like habitat maintenance, which might not have direct market values under this approach, can be more challenging, and the exclusion of non-market values might lead to perceived underestimation by environmental stakeholders.

Extended SNA-conform approaches using simulated exchange values

The extended SNA-conform approach using simulated exchange values aims to improve the alignment of monetary valuation with SEEA EA principles for ecosystem services that lack direct markets, which is typically the case for regulating services. By focusing on simulating exchange values based on estimated demand, this approach attempts to adhere to the SNA's emphasis on exchange rather than welfare values like consumer surplus. While the underlying demand estimation might use techniques that elicit willingness to pay, the last step of simulating a market seeks to derive a value that represents a potential exchange price if the service were internalized in the market. This is considered to have a higher potential for use within the SEEA EA concept compared to approaches that directly include consumer surplus.

The extended SNA-conform approach can provide valuable information for policy decisions relevant to SEEA EA in Europe. By assigning monetary values to previously unpriced ecosystem services, such as the recreational benefits of forests or the value of biodiversity, this approach can make the economic contribution of these services more visible to policymakers. For global climate regulation for example, it can help understand the societal value of climate action. For soil retention, it can highlight the benefits of maintaining soil health beyond agricultural productivity.

Non-market valuation techniques like TCM and CV can be designed to capture the spatial aspects of demand for ecosystem services. For example, TCM relies on the travel costs incurred by individuals from distinct locations to access a site, allowing for the estimation of distance-decay functions and the spatial mapping of recreational values. CV and choice experiments can also incorporate spatial attributes of ecosystem services or the characteristics of the respondents' locations to generate spatially differentiated willingness to pay estimates. These spatially explicit values can then be integrated into ecosystem accounts to provide a geographically detailed understanding of the distribution of benefits.

We evaluate the potential for extended SNA-conform approaches to maintain temporal consistency to be relatively high. Maintaining temporal consistency in the extended SNA-conform approach requires careful consideration of how demand for ecosystem services might change over time and the consistent application of simulation methods (NCAVES and

MAIA 2022). To account for long-term changes, such as population growth, income changes, or shifts in environmental conditions, demand models might need to be updated, or projections made based on relevant socio-economic and environmental scenarios. Consistent application of the market simulation methodology across different time periods is also crucial for ensuring temporal comparability of the results.

The extended SNA-conform approach can be particularly relevant for valuing ecosystem services that lack direct markets, such as the non-use values associated with global climate regulation, soil retention, and habitat maintenance. For global climate regulation, it can capture the public's willingness to pay for a stable climate. For soil retention, it can value the benefits of healthy soils beyond agricultural productivity, such as water filtration or carbon storage. For habitat maintenance, it is highly relevant for estimating the value of biodiversity conservation and the preservation of natural habitats, which are often not traded in markets. By simulating exchange values, this approach attempts to bring these non-marketed benefits into the accounting framework in a way that is more aligned with SNA principles than directly using welfare values.

The ease of communicating results from the extended SNA-conform approach can vary. While the final output is a monetary value (the simulated exchange value), the underlying methods of demand estimation and market simulation can be complex and might be harder to explain to policymakers, businesses, and the public compared to valuations based on direct market prices. Communicating the hypothetical nature of simulated markets and the assumptions involved in demand modelling requires careful framing to ensure that the results are understood and accepted by a diverse audience. However, the ability to provide a monetary value for previously unpriced benefits can enhance the salience of these services in economic discussions and policy debates.

Welfare-based approaches

As discussed previously, use of the welfare-based approach for valuing regulating services has a low degree of alignment with the core principles of SEEA EA. Despite this limited alignment with SEEA EA accounting principles, the welfare-based approach can provide highly relevant information for a wide range of environmental policy decisions in Europe: For global climate regulation for example, it can reveal the overall societal value of mitigating climate change, avoiding issues with price variation in carbon markets caused by short term market dynamics. For soil retention, it can highlight the importance of soil health for various aspects of well-being beyond agriculture, which are not captured by using exchange values linked to agricultural markets.

The welfare-based approach is well-suited for generating spatially explicit valuations in Europe. Non-market valuation techniques can incorporate spatial variations in environmental attributes, socio-economic characteristics, and preferences.

Maintaining temporal consistency in the welfare-based approach requires careful consideration of how preferences and values might evolve over time. Willingness to pay for ecosystem services can be influenced by several factors, including changes in income, environmental awareness, and the availability of substitutes. To ensure temporal consistency, valuation studies might need to be repeated periodically or adjusted based on projections of these influencing factors. For asset valuation using a welfare-based approach, the discount rates applied to future willingness to pay might also need to be considered consistently over time (NCAVES and MAIA 2022).

The welfare-based approach is highly relevant for valuing the full range of benefits provided by global climate regulation, soil retention, and habitat maintenance. For example, for global climate regulation, it can capture not only the avoided market damages but also the non-use values associated with a stable climate, which is not possible when only considering exchange values.

As for provisioning services, the ease of communicating results from the welfare-based approach can vary. While expressing values in monetary terms can be readily understood, the underlying concepts of willingness to pay, consumer surplus, and non-use values might be less familiar to some audiences compared to market prices. Communicating the hypothetical nature of stated preference methods like CV and choice modelling also requires careful attention to ensure that the results are interpreted correctly. However, the ability of this approach to capture the broader societal value of ecosystem services can make the results highly salient for public awareness and policy debates.

9.2.2.3 Credibility

SNA-conform approaches

The robustness of valuation approaches within the SNA-conform method depends heavily on the availability and quality of market data. For global climate regulation, the use of EU-ETS prices for carbon sequestration is considered robust as these prices are determined by a functioning market. However, using avoided cost methods for broader climate impacts relies on complex modelling and assumptions about damage functions, which can introduce uncertainties. For soil retention, replacement costs based on fertilizer prices are relatively robust as these are based on market prices for inputs. Avoided cost estimates for erosion might be less robust if they rely on indirect links to market sectors or involve significant assumptions. Alternative methods for SNA-conform valuation of soil retention are also available, for example using the exergy approach (Palacino et al. 2024), in which the inputs into crop production, including soil nutrients, are calculated in terms of their energy contribution, which can then be converted to monetary values based on crop prices. This method is not yet accepted as valid under SEEA EA and needs further study (NCAVES and MAIA 2022).

The consistency and quality of input data for SNA-conform valuation in Europe can vary. Economic data, such as market prices, production statistics, and cost data, are generally collected using standardized methods by national statistical offices and Eurostat, ensuring a relatively high level of consistency and quality across countries and over time. However, biophysical data related to ecosystem services, such as soil condition, habitat extent, or climate variables, might be collected using different methodologies and frequencies across

European regions and countries, potentially affecting the consistency and quality of the input data used in SNA-conform valuation, particularly for avoided cost or replacement cost methods that rely on both economic and biophysical information.

The potential for transparency in data and assumptions is high. Ideally, all data sources, assumptions made in the valuation process (e.g., when using proxies or cost-based methods), and any uncertainties associated with the estimates should be clearly documented to allow for scrutiny and reproduction of the results. For global climate regulation, transparency would involve detailing the source of carbon prices, the climate models used in avoided cost estimations, and the assumptions underlying damage functions. For soil retention, transparency would require clear documentation of fertilizer prices, data on nutrient loss, and the methodology used to link these to the value of soil retention.

Validation of SNA-conform valuation methods against observed economic behaviour is more feasible for ecosystem services with direct market linkages, which can be challenging for regulating services. For global climate regulation, the market price of carbon in the EU-ETS provides a direct measure of the exchange value for carbon sequestration in covered sectors. For soil retention, the impact of soil quality on agricultural yields and the market value of agricultural products can offer a basis for validation.

Extended SNA-conform approaches using simulated exchange values

The robustness of the extended SNA-conform approach depends on the reliability of the demand estimation and the validity of the market simulation. The robustness of the simulated exchange value also depends on the assumptions made about the market structure (e.g., perfect competition, monopolistic competition) and the relationship between demand and supply. As discussed in Barton et al. (2019), a realistic market assumption should be the basis for simulated exchange, to give policy-makers insights into additional value that could be created/appropriated within current institutional regimes and technology.

The consistency and quality of input data for the extended SNA-conform approach are crucial for its credibility. The biophysical data on the ecosystem services themselves (e.g., climate variables, soil condition, habitat extent) also needs to be of sufficient quality and consistency to link demand to the provision of these services.

Transparency is essential for the credibility of the extended SNA-conform approach, and is potentially high for stated preference methods for simulating exchange values. This includes clearly documenting the survey design, the econometric models used to estimate demand functions, the assumptions made in simulating the market (e.g., regarding supply and market structure), and any limitations of the data or methods. For global climate regulation, transparency would involve detailing how willingness to pay for climate benefits was elicited and how this was used to simulate an exchange value. For soil retention, similar transparency is needed regarding the valuation of non-market benefits. For habitat maintenance, the design of choice experiments for biodiversity valuation and the assumptions in the market simulation should be clearly stated. This level of transparency allows for scrutiny and assessment of the validity of the simulated exchange values.

Validation of the extended SNA-conform approach can be challenging as it deals with non-market goods and services. Direct comparison with observed market behaviour is not possible by definition. However, indirect validation might involve comparing the simulated exchange values with values obtained using other methods or assessing their consistency with economic theory and expert judgment. For global climate regulation, comparing willingness to pay for climate action with actual expenditures on green technologies might offer some insights. For soil retention, the simulated values of non-market benefits could be compared with observed land management practices or property values.

Welfare-based approaches

The robustness of valuation approaches within the welfare-based method depends strongly on the study design and quality of the data sampling. Stated preference methods like CV and choice modelling are susceptible to various biases, such as hypothetical bias, framing effects, and strategic behaviour by respondents.

The consistency and quality of input data for the welfare-based approach are critical for its credibility. For stated preference methods, the design of the survey instrument, the sampling strategy, and the administration of the survey need to be rigorous to ensure representative and reliable data on willingness to pay.

Transparency in the data and assumptions is potentially high, but needs to be comprehensive. This includes clearly documenting the survey design and administration for stated preference methods, the econometric models and variables used in hedonic pricing, and all underlying assumptions made in the valuation process. For stated preference methods, reporting response rates, addressing potential biases, and providing details on the elicitation format are important for transparency.

Validation of the welfare-based approach can be challenging, particularly for non-use values that do not have direct counterparts in market transactions, as is typically the case with regulating services. While some studies attempt to validate stated preference results by comparing them to revealed preference data or through calibration exercises, direct validation is often not feasible.

9.2.2.4 Cost proportionality

SNA-conform approaches

The cost of data collection for SNA-conform valuation in Europe varies depending on the ecosystem service and the chosen valuation method, but is generally relatively low. For global climate regulation, data on EU-ETS prices is readily available at low cost. However, obtaining detailed data for avoided cost estimations might require access to complex climate models and economic impact assessments, potentially increasing costs. For soil retention, data on agricultural statistics and fertilizer prices (secondary sources) is generally accessible at a relatively low cost. However, collecting detailed data on soil erosion rates and their impact on specific agricultural systems might necessitate more resource-intensive primary data collection or specialized modelling. Overall, the cost of data collection tends to be lower when relying on existing secondary data sources and higher when primary data collection or complex modelling is required.

The processing and analysis costs associated with the SNA-conform approach also depend on the complexity of the valuation method, but is generally relatively low. Using market prices for carbon sequestration involves minimal processing. However, employing avoided cost methods for global climate regulation or soil retention would require significant processing power and expertise in climate modelling, economic modelling, or hydrological modelling. Simpler methods relying on readily available market data would generally have lower processing and analysis costs compared to those involving extensive modelling or complex calculations.

Implementing the SNA-conform approach effectively in Europe requires a range of technical expertise. This includes a strong understanding of national accounting principles, particularly the SNA and SEEA EA frameworks, as well as specialized knowledge in environmental economics and valuation techniques that focus on exchange values. For valuing global climate regulation, expertise in climate science and economics might be needed for avoided cost methods. For soil retention, knowledge of agricultural economics, soil science, and potentially hydrological modelling could be required.

The feasibility of implementing the SNA-conform approach under budget constraints in Europe varies depending on the chosen methods and the availability of existing resources, but can potentially be relatively high (Schenau et al. 2022). Utilizing readily available market data, such as EU-ETS prices, is likely highly feasible. However, implementing complex avoided cost or replacement cost methods across all three ecosystem services at a European scale could be resource-intensive, requiring significant financial investment in data collection, modelling, and expert personnel. The feasibility also depends on the institutional capacity within European statistical agencies and environmental organizations to undertake these valuations on a regular basis.

The scalability and replicability of the SNA-conform approach across different geographic areas and time periods in Europe depend on the standardization of data collection and valuation methodologies, but is generally medium. Methods that rely on widely available and consistently collected data, such as national accounts statistics, agricultural prices, and EU-wide market prices like those in the EU-ETS, tend to be more scalable and replicable. However, methods that require localized data or complex, context-specific modelling for each region or time period might face challenges in scalability and replicability without substantial increases in cost and effort. Ensuring the consistent application of valuation methodologies across diverse European contexts is also crucial for achieving scalability and replicability.

Extended SNA-conform approaches using simulated exchange values

The data collection cost for the extended SNA-conform approach can be substantial. Estimating demand for non-marketed ecosystem services requires collecting data for estimating demand functions, which can involve conducting primary research through surveys. These can be expensive in terms of questionnaire design, sampling, interviewer training (if applicable), survey administration (online or in-person), and data entry. For global climate regulation, large-scale surveys to assess public preferences for climate benefits can be costly. For soil retention, similarly, dedicated surveys might be needed to value non-market benefits.

The processing and analysis costs associated with the extended SNA-conform approach can also be high. Analysing survey data to estimate demand functions often requires specialized statistical software and expertise in econometrics. Simulating market exchange values based on these demand functions adds further analytical complexity. For global climate regulation, modelling willingness to pay and simulating a climate benefits market can be resource-intensive. For soil retention, the econometric analysis of stated preference data and the simulation of exchange values also require significant processing time and specialized skills.

Applying the extended SNA-conform approach correctly for valuing regulating services requires a high level of technical expertise (Caparrós et al. 2017). This includes knowledge of environmental economics, non-market valuation techniques (such as TCM, CV, and choice modelling), econometrics, survey design, and market simulation. For global climate regulation, expertise in behavioural economics and climate policy might be beneficial. For soil retention, knowledge of agricultural economics and environmental management could be needed.

The feasibility of implementing the extended SNA-conform approach under budget constraints in Europe can be a concern due to the potentially high information costs associated with primary data collection and complex analysis. Conducting large-scale surveys across multiple European countries to value global climate regulation, soil retention, or habitat maintenance can be expensive. The need for specialized software and econometric expertise also adds to the budgetary requirements. Feasibility might be improved by using existing datasets where available or by employing simpler valuation techniques, but this could potentially compromise the accuracy and robustness of the results.

Welfare-based approaches

The data collection cost for the welfare-based approach can be substantial. Stated preference methods like CV and choice modelling typically require conducting primary surveys, which can be expensive due to the costs associated with survey design, sampling, participant recruitment, and data collection. The cost of data collection can be a significant factor limiting the feasibility of welfare-based valuations, especially for large-scale studies across multiple regions or countries.

9.2.2.5 Legitimacy

SNA-conform approaches

SNA-conform methods (avoided costs, replacement costs) have high institutional acceptance due to their link to economic expenditures or market prices (e.g., carbon) and alignment with SEEA EA (SEEA, 2022). Stakeholder representation is often low, limited to expert consultations for technical modelling and cost calculations rather than broader public engagement (MDPI, 2020). For example, the use of avoided damage costs for climate regulation in the Greece and Portugal TSs primarily involved expert input for the modelling. Inclusivity of value perspectives is evaluated as low, as they primarily capture costs or market prices related to the service, not the full range of values stakeholders may hold (e.g., non-use values for climate regulation or biodiversity).

Extended SNA-conform approaches using simulated exchange values

Extended SNA-conform approaches simulating exchange values have medium institutional acceptance. Stakeholder representation is typically low in the complex modelling process required to simulate exchange values (IPBES 2022). Inclusivity of value perspectives is medium, as the simulated values are based on estimated demand or abatement costs which can reflect some economic aspects but not all values. Transparency of these complex methods can also be a challenge for non-experts.

Welfare-based approaches

Welfare-based approaches (stated preference) score high on stakeholder representation, allowing direct elicitation of values from beneficiaries and concerned citizens through surveys (IPBES 2022). Our experience in test sites (e.g., coastal protection perceptions) underscores the importance of engaging beneficiaries to enhance legitimacy. They offer high inclusivity of value perspectives, capturing a broad range of values including passive use values relevant for some regulating services, such as habitat maintenance services. However, their institutional acceptance for formal SEEA EA accounting is typically lower compared to SNA-conform methods (SEEA 2022), and welfare-based valuation methods are not common to use for regulating services. Similarly to provisioning services, their high overall legitimacy stems from their strength in involving stakeholders and capturing diverse values.

9.2.3 Primary data - Cultural services

Our evaluation of monetary valuation methods for cultural services covered the following services:

- Recreational services (Greece, Ireland, La Réunion)
- Nature based tourism (Greece, La Réunion)

For cultural services, we evaluated the resource rent method and the consumer expenditure method as SNA-conform approaches. The resource rent method, traditionally applied to provisioning services (see section 9.2.1), can be adapted for valuing certain cultural ecosystem services, particularly nature-based tourism (Bronzes et al. 2025). This approach considers the surplus value or profit generated by economic activities that directly rely on the cultural ecosystem service, such as tourism-related businesses operating within or near natural areas. The consumer expenditure method can serve as a proxy for the exchange value of cultural ecosystem services by considering the actual spending by individuals to access these services. This includes expenditures on travel, accommodation, entrance fees, and other related expenses incurred during recreational visits to natural and cultural sites.

For extended SNA-conform methods for cultural services, we evaluated the SEV method. For recreational services, this typically involves using demand functions derived from non-market valuation techniques like TC or CV to determine a revenue-maximizing price, such as an optimal entrance fee for a park.

Similarly, the welfare-based approach to valuation of cultural services can apply stated preference methods, such as the DCE, TC and CV methods, but uses them to estimate a welfare value that includes consumer surplus. While SEEA EA has specific requirements for TCM to be SNA-conform (excluding consumer surplus and cost of time), the inclusion of consumer surplus provides a more comprehensive measure of welfare benefit, which also changes their potential for application in policy processes.

Table 25. Summary overview of evaluation for cultural services using primary data.

Valuation method	**Timeliness**	**Salience**	**Credibility**	**Cost proportionality**	**Legitimacy**
SNA-conform approaches	Medium to high	Medium	Medium to high	Medium to high	Medium
Extended SNA-conform approaches	Medium	High	Medium to high	Low to medium	Medium to high
Welfare based approaches	Medium	High	Medium	Low to medium	High

9.2.3.1 Timeliness

SNA-conform approaches

Availability of recent data for the resource rent method can be low to medium, as it requires specific economic data from tourism sectors that might not always be readily available or disaggregated to the level of specific cultural ecosystem services. Valuation model processing time is generally low, as the calculation involves straightforward subtraction once the data is collected. Integration with SEEA accounting periods is high, as resource rent aligns well with the annual accounting periods used in national accounts.

Availability of recent data for the consumer expenditure method is medium to high, as data on tourism expenditures are often collected regularly by national tourism organizations or statistical agencies across Europe. Valuation model processing time is low, involving the aggregation of expenditure data. Integration with SEEA accounting periods is high, as it aligns with standard annual accounting periods.

Extended SNA-conform approaches using simulated exchange values

Availability of recent data for SEV can be medium, depending on the existence of recent and relevant TCM or CV studies in the European context (Bronzes et al. 2025). The timeliness of the valuation methods we applied in Greece, i.e., TCM and CVM, depend on their ability to deliver actionable data quickly enough to align with the reporting commitments to the SEEA EA framework. The online data collection mechanism that was adopted in the Greek case study resulted in faster data collection and has the potential to be very successful if it is well designed and carefully delivered to respondents. For the travel cost there is no need to employ hypothetical questions and thus it is very straightforward to communicate. Travel cost is a good method for regular (conventional) monitoring of the use value of the resource. Travel cost questionnaire surveys do not have to repeat every year because they are based on behavioural characteristics which do not change if no abrupt changes occur in other key determinants of the travel (e.g., price of petrol, sudden imposition of tolls, etc.). However, if there is an incremental change, or the need to quickly value the results of a proposal that will affect the status of the area I would carry out a contingent valuation with the aim of quickly valuing the effects of specific scenario(s). Thus, for policies needing evidence-based updates (e.g., park fee adjustments, local tourism planning) or for large infrastructure that will change the time and cost for visiting the area, contingent valuation is better. For this reason, and from the perspective of timeliness, contingent valuation can be very useful for cost-benefit analysis in early policy stages. Both methods applied in the Greek case study need careful design, otherwise there is a significant risk of failure.

Welfare-based approaches

The Travel Cost Method (TCM), when including consumer surplus, estimates the total willingness to pay for recreational experiences at cultural and natural sites across Europe. It analyses the costs incurred by visitors to access these sites (including travel expenses and the opportunity cost of time) to infer the value they place on the recreational benefits. While SEEA EA has specific requirements for TCM to be SNA-conform (excluding consumer surplus and cost of time), the inclusion of consumer surplus provides a more comprehensive measure of welfare benefits. Otherwise, the methods are applied similarly to the TCM for estimating simulated exchange values, so the evaluation of this as described in the previous subsection applies to the welfare-based approach as well.

9.2.3.2 Salience

SNA-conform approaches

The resource rent method has high relevance to policy questions, as it can inform the economic contributions of nature-based tourism and potentially justify investments in the conservation of culturally significant natural areas. It also considers subsidisation: in sectors with high subsidies, resource rents can be negative. This information is highly relevant for debating the use of public resources for subsidies (Greaker and Lindholdt 2021). However, it might not capture the full spectrum of cultural values beyond direct economic benefits. The ease of communication to policymakers is medium; while monetary values are easily understood, the limited scope of the method might not fully represent the importance of cultural ecosystem services. Its applicability to different decision contexts is medium, being more suited for decisions related to tourism management and revenue generation than for broader cultural heritage or conservation policies.

The consumer expenditure method has medium relevance to policy questions, providing insights into the economic impact of nature-based cultural activities and the scale of economic sectors benefiting from them in Europe. Ease of communication to policymakers is high, as expenditure data is readily understandable. Applicability to different decision contexts is medium, being useful for tourism planning, regional economic development strategies, and assessing the economic contribution of natural sites for tourism across Europe.

Extended SNA-conform approaches using simulated exchange values

The SEV method has high relevance to policy questions, providing a theoretically grounded estimate of the economic value of cultural services across Europe, which can inform decisions on pricing, investment, and resource management. Ease of communication to policymakers is medium, as the concept of simulated exchange value might require clear explanation, especially regarding the underlying assumptions about market behaviour. Applicability to different decision contexts is medium, as it can potentially be applied to various cultural services, including recreation, tourism, and aesthetic values, supporting a range of policy and management decisions at different scales. However, SEV may not conform with institutional settings locally, so monetary valuation of SEV or exchange values used for accounting purposes should not be used for local policy/project assessment without a local validation being carried out (Barton 2022).

Welfare-based approaches

TCM is salient for immediate, localized decisions but CVM for broader, long-term policies but requires careful communication. As we applied the TCM in Greece it avoids double-counting by using observed behaviour (e.g., travel expenses) to value recreational services, directly linking to economic activity. It aligns with policy questions like park management, and provides spatially explicit data via visitor origins. However, it fails for non-use values and lacks long-term dynamics. On the other hand, the CVM captures policy-relevant WTP and non-market services, including future benefits. Yet, it risks misrepresentation if poorly designed, struggles with spatial mapping, and can be opaque to policymakers due to hypothetical bias.

9.2.3.3 Credibility

SNA-conform approaches

The use of the resource rent method is evaluated as high for its use of established valuation techniques. Consistency with economic theory is also high, as it directly relates to exchange value and returns to assets, aligning with SEEA EA principles. Data quality and availability can be medium, as obtaining accurate and disaggregated data on revenues and costs specifically for cultural ecosystem services can be challenging. Transparency of assumptions is high, as the assumptions about which economic activities are linked to the cultural service and the calculation of costs and revenue are usually explicit. Robustness of results can be low to medium, as it is highly dependent on the specific context and the potential exclusion of non-market values. The level of technical expertise required is low, as the calculation is generally simple.

For the consumer expenditure method, the use of established valuation techniques is high, as it is based on observed economic behaviour. Consistency with economic theory is medium; while it reflects actual spending, it might not fully capture the intrinsic value or willingness to pay for the cultural service itself. Data quality and availability are high, as tourism expenditure data is often collected with reasonable accuracy. Transparency of assumptions is high, with assumptions mainly about which expenditure categories are included. Robustness of results is medium, as it can be influenced by various factors affecting tourism patterns. The level of technical expertise required is low, involving simple data aggregation. When using the TCM to estimate exchange value, methodological issues can arise around endogeneity, measurement errors, model specification issues and sample selection bias.

Extended SNA-conform approaches using simulated exchange values

The use of established valuation techniques is medium to high, as it builds upon recognized non-market valuation methods like TCM and CV. Consistency with economic theory is high, as it directly aims to estimate exchange values consistent with SEEA EA principles and SNA. Data quality and availability are medium, relying on the quality and availability of data from TCM or CV studies. Transparency of assumptions is medium, with assumptions about the demand function, market structure, and the treatment of protest responses needing clear documentation and validation with the local institutional context. Robustness of results can be medium, as the simulated values can be sensitive to the specification of the demand model and market assumptions. The level of technical expertise required is high, requiring significant expertise in econometrics and non-market valuation.

In our work in the Greek test site, we found that since TCM relies on observed behaviour (travel expenses, visitation rates), it is robust and consistent with real-world data. Inputs like distance and costs are often standardized. However, assumptions (e.g., opportunity cost of time) require transparency. Extreme values can occur, and data cleaning and preparation should be a very careful and time consuming stage, if we want high credibility. The CVM, while it documented the hypothetical scenarios transparently, relied on stated preferences which may be considered as a factor that reduces robustness and complicates validation. Sensitivity to survey design (e.g., bid amounts) can undermine consistency. TCM is more credible for tangible services (recreation), CVM, though flexible, risks subjectivity without careful calibration.

Welfare-based approaches

The use of established valuation techniques is medium to high; while widely used, CVM and CE have known limitations regarding their hypothetical nature and potential biases. Addressing these requires careful survey design, robust econometric models, and cross-validation with real-world data. These include (Johnston et al. 2017):

- Hypothetical Bias or Overstating WTP in Surveys
- Strategic Bias or Gaming the System
- Starting Point Bias also known as Anchoring Effect in Bidding Games
- Embedding Effect or Scope Insensitivity
- Protest Votes or Rejecting the Payment Vehicle
- Distributional and Econometric Issues
- Model Selection Issues

Consistency with economic theory is high, based on welfare economic theory. Data quality and availability are medium, depending on the design and implementation of surveys across diverse European populations. Transparency of assumptions is medium, with assumptions about hypothetical scenarios needing clear justification. Robustness of results can be low to medium, susceptible to various biases. The level of technical expertise required is high, needing expertise in survey design and econometric analysis of stated preference data.

9.2.3.4 Cost proportionality

SNA-conform approaches

For both the resource rent and consumer expenditure method, the cost proportionality of data collection is medium to high, requiring the gathering of economic data from tourism operators, park authorities, or national statistics. The cost proportionality of model processing and analysis is high, as the calculation is straightforward. The need for specialized expertise is low, with a basic understanding of economic principles being sufficient.

Extended SNA-conform approaches using simulated exchange values

The cost proportionality of data collection is low to medium, potentially involving conducting surveys for TC or CV across Europe. The cost proportionality of model processing and analysis is also low to medium, requiring specialized software and expertise in econometric modelling and market simulation. The need for specialized expertise is high, requiring economists with advanced skills in non-market valuation and potentially market modelling.

Experience from the application of both the TCM and the CVM in Greece showed moderate but scalable cost. The internet-based questionnaires are cheaper but still require primary data collection and data cleaning. For well executed surveys, processing costs are low (basic regression models), and expertise is moderate as concerns econometrics. However, if the survey has design issues econometrics may become very involved. For the CVM, while flexible, its costs limit scalability, making it less suitable for routine accounting but valuable for unique policy questions.

In general, we found TCM to be more cost-effective for frequent, localized valuation; CVM is reserved for high-stakes, non-market cases.

Welfare-based approaches

For using the TCM as well as for using the CVM and CEM, the cost proportionality of data collection is low to medium, especially when using surveys for large accounting areas. Cost proportionality of model processing and analysis is medium, requiring statistical software and expertise. Need for specialized expertise is medium, requiring economists with experience in recreational demand modelling and stated preference methods. Otherwise, the methods are applied similarly to the TCM for estimating simulated exchange values, so the evaluation based on our empirical work in Greece, as described in the previous subsection, applies to the welfare-based approach as well.

9.2.3.5 Legitimacy

SNA-conform approaches

SNA-conform methods (consumer expenditure, resource rent for tourism/recreation) have medium institutional acceptance, being based on observed economic activity and alignment with accounting principles (NCAVES and MAIA, 2022). Stakeholder representation is often low in the valuation process itself, focusing on analysing existing economic data rather than direct engagement with users about their values. For example, the consumer expenditure analysis in the Greek TS and La Réunion DP utilized tourism data but did not involve tourists in the valuation design. Inclusivity of value perspectives is low, primarily capturing market expenditures and not the diverse, often non-monetary, values associated with cultural sites and experiences (IPBES, 2022).

Extended SNA-conform approaches using simulated exchange values

Extended SNA-conform approaches using SEV based on demand functions (from TCM/CVM) have medium institutional acceptance, aiming to align with SEEA EA. Stakeholder representation can be medium to high if based on data collected directly from users (e.g., surveys in Greece, La Réunion DPs), providing insight into their preferences that allows for analysis of explanatory variables for the value estimates. Inclusivity of value perspectives is evaluated as medium, as they aim to estimate exchange values based on demand but do not capture non-use values. However, transparency of the modelling process is key to realizing this legitimacy potential (IPBES 2022).

Welfare-based approaches

Welfare-based approaches (CVM, DCE, TCM) score high on stakeholder representation, directly involving diverse users and citizens through surveys, as seen in our test sites (Greece, La Réunion DPs) (IPBES, 2022; TEEB for Local and Regional Policy Makers). These surveys provided a platform for visitors and residents to express their values. They offer high inclusivity of value perspectives, capturing a wide range of values, including non-use and passive use values highly relevant for cultural services (Parliament UK). However, their institutional acceptance for formal SEEA EA accounting remains lower (SEEA, 2022). Their strong performance in representing stakeholders and values contributes to their high overall legitimacy.

9.2.4 Value transfer

Our evaluation of monetary valuation methods for value transfer does not distinguish between ecosystem service types, nor between exchange value based methods or welfare value based methods, because the method functions similarly for all of these variations. Instead, we evaluate unit value transfer and value function transfer separately. The unit value transfer evaluation is based on the few tests in our cases, combined with expert judgment and literature review. The value function transfer is based on the development and application of value functions using the ESVD (see section 8.2.2).

Table 26. Summary overview of evaluation for using value transfer methods.

	Timeliness	**Salience**	**Credibility**	**Cost proportionality**	**Legitimacy**
Unit value transfer	High	Medium	Low to medium	Medium to high	Low
Value function/meta -analytic transfer	High	Medium to high	Medium	Medium	Low to medium

9.2.4.1 Timeliness

Unit value transfer

Value transfer offers high timeliness as it utilizes results from existing studies, significantly reducing the time and resources needed compared to conducting primary valuation studies (Grammatikopoulou et al. 2023).

Value function/meta-analytic transfer

The development of meta-analytic value functions and application to SELINA test sites provides a number of insights into the timeliness of this approach. Regarding the availability of data to support this approach, the ESVD is found to contain sufficient data, extracted from existing primary valuation studies, to support the estimation of meta-analytic value functions for a number of ecosystems (agricultural land, inland wetlands, grasslands, and temperate forests). With continuing development of the ESVD and the addition of results from recently published primary valuation studies, it is expected that there will be sufficient data to estimate value functions for other relevant biomes (e.g., rivers and lakes, shrubland, coastal wetlands).

Regarding the valuation model processing time, we note three separate stages in the application of meta-analytic value functions: 1. the development of data for estimating value functions extracted from primary valuation studies and supplemented with spatially referenced secondary data; 2. the statistical estimation of value functions; 3. the application of value functions to predict values for ecosystem assets. The construction of a database of primary valuation results such as the ESVD is a hugely laborious and time-consuming process. It involves collecting, reading, coding, and reviewing information from individual primary valuation studies. The potential to (partially) automate this process with the use of AI tools is being explored but has not yet proved viable or accurate. The ESVD is, however, fully publicly available so it is not necessary to develop this database again for new accounting applications. The addition of supplementary spatially referenced data (as described in section 8.2.2) is a relatively quick process that can be automated to a large extent through tools such as Google Earth Engine. The second step to estimate value functions using meta-regression analysis is also relatively quick, in the order of weeks rather than months. Again, it is potentially possible to use existing published value functions rather than develop models for new accounting applications. The third step of applying value functions to predict values for ecosystem assets in the accounts is also relatively quick and involves defining parameter values for each asset (including spatial analysis for spatially defined variables) and inputting these into the relevant value functions. This process initially requires some time to set up but can subsequently be run quickly.

One of the main advantages of the value function transfer approach is the ease with which it can be applied to fit with the requirements of regular accounting periods. Once value transfer functions have been set up in a modelling framework (i.e., as a script that operates on a database of ecosystem assets), the time involved to estimate ecosystem service values for a new reporting period is very short. Moreover, the specification of the value function potentially enables the estimation of values that reflect multiple determinants that change over time (e.g., ecosystem extent, condition, service provision, income, population etc.). In other words, the estimated values are dynamic rather than static.

9.2.4.2 Salience

Unit value transfer

The salience of unit value transfer is medium, as the relevance of the transferred values depend heavily on the similarity between the original study site and the policy site in terms of similarity of institutional context of the policy being assessed, ecological conditions, socio-economic characteristics, and the specific ecosystem service being valued (Brander et al. 2022).

Value function/meta-analytic transfer

The salience of value function transfer is considered to be medium-high. The alignment of value function transfer results with SEEA EA principles is justifiably questioned, particularly as the underlying primary valuation estimates on which value functions are based represent a mix of different value concepts including welfare values and costs, as well as exchange values. In the application developed in this report, we attempt to address this issue by explicitly modelling the differences between exchange values and other value concepts, and subsequently use this to approximate exchange values in the transferred values.

In terms of relevance for informing SEEA EA related policy decisions, we consider value function transfers to perform reasonably well. Particularly if the specification of the value function enables estimated values to reflect policy relevant characteristics (e.g., ecosystem extent, condition, abundance, population density).

One of the main advantages of the value transfer function approach trialled in this report is the suitability for spatially explicit valuation. By including spatially defined explanatory variables in the value function, it is possible to estimate ecosystem service values for ecosystem assets that reflect multiple spatially variable determinants (e.g., ecosystem condition, population density, fragmentation, accessibility, abundance etc.). The results of the application of value transfer functions to SELINA test sites show the high degree to which ecosystem service values are spatially variable. For ecosystem services for which the unit value is not spatially variable, notably in the case of global climate regulation through carbon sequestration, it is arguably better not to apply a value transfer function that could potentially result in spurious spatial variation.

The use of value function transfers also enables temporal consistency in the estimation of monetary values. Value functions can potentially be applied to estimate past, current and future values of ecosystem assets, controlling for determinants of value that vary over time. The use of value transfer functions ensures a degree of methodological consistency in repeated applications, which might be an advantage in comparison to the use of primary valuations that are conducted at different points in time.

In principle, the value function transfer approach can be applied to estimate values for any ecosystem service for which there are sufficient primary valuation results available as a basis to derive a value function. In practice, some ecosystem services are better represented in the valuation literature than others. Fig. 16 provides an overview of the number of value estimates in the ESVD per ecosystem service. The most prominent ecosystem services include recreation and tourism (19% of total value estimates), food production (17%), raw materials (11%), existence and bequest values (10%), climate regulation (6%), air quality regulation (6%), and moderation of extreme events (4%). The ecosystem services for which there are relatively few value estimates are maintenance of life cycles, biological control, genetic resources, ornamental resources, and spiritual experience.

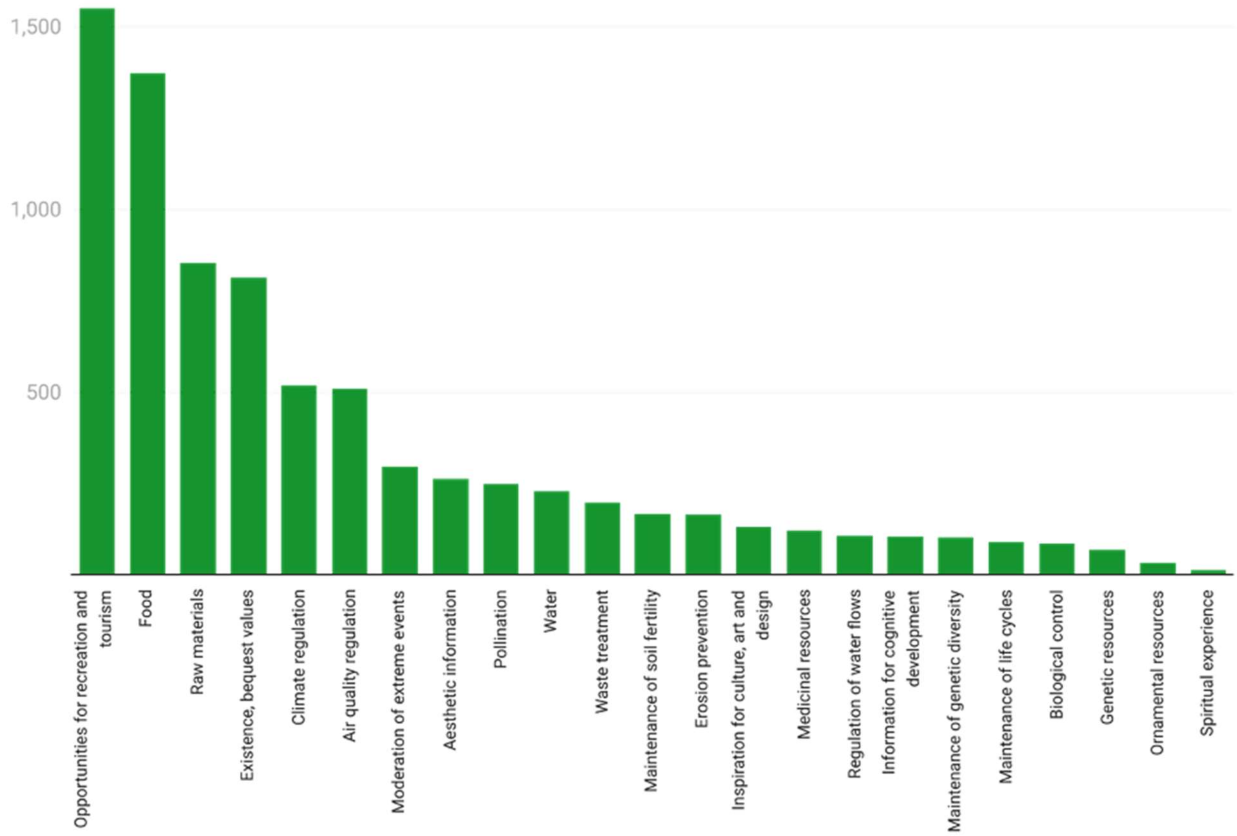

Figure 16. Number of value estimates in the ESVD by ecosystem service (source: Brander et al., 2024).

9.2.4.3 Credibility

Unit value transfer

The accuracy of the transferred values is inherently limited by the quality and methodology of the original studies, as well as the degree of similarity between the study and policy sites. The reliability of benefit transfer is highest when the original and study sites are very similar, and the original valuation study was carefully conducted (Brander et al. 2022).

Value function/meta-analytic transfer

We rate the credibility of value function transfer to be medium due to the presence of a number of potential sources of transfer error (over- or under-prediction of values).

The quality and reliability of the primary value data on which value transfer functions are based is inherently variable due to variation in the underlying primary valuation studies in terms of valuation methods, implementation and reporting; and due to possible analyst errors in the process of interpreting the studies and coding the data. In the case of the ESVD, the data is subject to a two-stage quality control process: an automated check for valid data entries and a review process by invited expert reviewers. Not all data in the ESVD, however, has been fully reviewed (approximately 50%) and so it is likely that there remains instances of miscoding and/or misinterpretation. Quantifying the quality of the underlying primary valuation results remains a challenge and there is no simple metric that is applicable to all relevant valuation methods.

Regarding the quantification of predictive performance of value functions themselves, it is possible to produce quantitative indicators to communicate uncertainty in the predicted values. For the value functions presented in this report we used an in-sample transfer exercise (see section 8.2.2), and find that the mean and median absolute percentage errors for the four value functions are in the range 20-57% and 12-44% respectively. The effect of such errors on credibility will depend on the policy process that the values are used for, with exploratory valuations being less reliant on small errors than for example policy prioritisation studies. A comparison of value estimates between primary data and value function transfer data in Greece (see Table 22) shows that the primary data based value using TCM for consumer surplus (151 int.$ ha^{-1} y^{-1}) falls in between the estimates of exchange value (203 int.$ ha^{-1} y^{-1}) and consumer surplus value (43 int.$ ha^{-1} y^{-1}) generated by the value function transfer.

The development and application of value transfer functions can be reported in a fully transparent way to enable scrutiny and reproducibility. For the application presented in this report for SELINA test sites, the underlying primary valuation data from the ESVD is publicly available, the spatially referenced data are also publicly available, and the estimated value functions are provided. If useful, the code for running the value transfer model in SPSS can be made available.

9.2.4.4 Cost proportionality

Unit value transfer

Unit value transfer generally has a medium to high information cost proportionality compared to conducting original valuation studies, as it primarily involves identifying, evaluating, and potentially adjusting existing value estimates. However, the cost can increase if significant adjustments or supplementary data collection are required to ensure the transferred values are applicable to the policy site.

Value function/meta-analytic transfer

We consider the cost proportionality of using value function transfer to be medium. We note, however, that the fixed costs of initially developing the required data and value functions are high but that the subsequent costs of estimating spatially explicit ecosystem service values for multiple ecosystem assets across large scales, and for multiple reporting periods, are very low.

The cost of constructing a database of primary valuation results from which to derive value transfer functions is substantial. It requires a large input of time from researchers with expertise in ecosystem service valuation methods to collect, analyse and code relevant literature into a database. This costly process can be circumvented, however, by using existing databases such as the ESVD.

The process of developing and applying value transfer functions requires expertise in a number of areas. Developing a database of primary valuation study results requires expertise in ecosystem service valuation methods. Supplementing valuation data with spatially referenced secondary data requires expertise in spatial analysis and use of specialised software such as GIS or Google Earth Engine. The statistical analysis to estimate meta-analytic value functions requires expertise in statistics/econometrics and specialised software such as SPSS, R, or Stata. Implementing value function transfers for large numbers of ecosystem assets requires some database and coding expertise and use of specialised software (e.g. SPSS, R, Python).

The use of value function transfers for accounting applications is likely to be feasible under most budget constraints in the case that existing databases and/or value functions are used. The main constraint is likely to be related to the availability of the required expertise.

Value function transfers perform well in terms of scalability and replicability. This approach can be applied consistently, and potentially automated, across multiple ecosystem assets in multiple geographic areas, and for multiple time periods without high costs. As demonstrated by the application to SELINA test sites, it was feasible to estimate spatially explicit ecosystem service values for over 1,100 ecosystem assets in five countries covering four different biomes and 14 ecosystem services (in total almost 4,000 site specific value estimates).

9.2.4.5 Legitimacy

Unit value transfer

Unit value transfer has low stakeholder representation, as it lacks direct input from stakeholders at the policy site (IPBES 2022). Inclusivity of value perspectives is low, limited to the values captured in the original study sites, which may not reflect the specific context of the policy site. Institutional acceptance can be medium for initial assessments due to low cost and speed. However, the lack of site-specific relevance significantly limits its legitimacy for detailed decision-making, particularly in the context of diverse European local conditions.

Value function/meta-analytic transfer

Value function transfer has low stakeholder representation at the policy site, relying on models derived from other studies (IPBES 2022). Inclusivity of value perspectives is low to medium, depending on the value concepts included in the meta-analysis and the model's ability to capture contextual differences. Institutional acceptance can be medium as it provides more nuanced estimates than unit transfer. However, the technical complexity and lack of local input limit its legitimacy compared to primary data methods (IPBES 2022). Quantifying and communicating uncertainty is vital for credibility and indirectly supports legitimacy by being transparent about limitations, a practice relevant for European applications of value transfer (NCAVES and MAIA 2022).

10 Discussion

By collaborating with SELINA DPs and TSs and by developing value functions based on the ESVD, we were able to evaluate a broad spectrum of monetary valuation methods under the SEEA EA framework. We emphasised the methodological and policy implications of using exchange-value based methods to integrate ecosystem services into national accounts. Our results, obtained through both primary data collection and value transfer exercises across two DPs (Lithuania and Le Réunion) and four TSs (the Czech Republic, Greece, Ireland, and Portugal, aimed to offer insights into the trade-offs between methodological rigour, cost efficiency, and policy relevance among exchange values and welfare values, as well as when choosing primary data collection or value transfer methods.

10.1 Evaluation outcomes

Quantitative differences between exchange and welfare-based estimates

The evaluation we presented indicates that the choice of valuation method—whether based on exchange or welfare concepts—can yield significantly different monetary estimates. Here, we detail the quantitative differences and discuss the factors driving these variations as well as their implications for decision-making.

As predicted by theory, welfare-based valuations—proxied via consumer surplus estimates—should yield consistently higher values than exchange-value-based approaches when comparable studies. We used Lithuania as a case for testing a wide range of ecosystem services using the value function transfer method, to compare welfare-based values to exchange-value based values. Below follow some examples of the variation:

- For **crop provisioning** in Lithuania**,** we estimated an exchange value of **Int.$ 3,086/ha/year** using value function transfer, while the welfare estimate for consumer surplus using value transfer was **Int.$ 647/ha/year**, indicating a *lower* welfare estimate in this case, which is an exception to the overall pattern.
- For **grazed biomass** in Lithuania, we estimated an exchange value of **Int.$ 198/ha/year** versus a consumer surplus value transfer of only **Int.$ 41/ha/year**.
- **Wild animals provisioning**, however, displayed the more typical pattern: the exchange value was **Int.$ 246/ha/year**, while the consumer surplus was **Int.$ 298/ha/year**. In the Czech Republic, this gap was even larger, with consumer surplus (**Int.$ 203/ha/year**) almost doubling the exchange value (**Int.$ 119/ha/year**).
- **Water supply** in Lithuania showed an exchange value of **Int.$ 126/ha/year** and a welfare value of **Int.$ 168/ha/year**.
- For regulating services, we typically found consumer surplus values of roughly double the size of exchange values in Lithuania. **Soil erosion control** for example showed an exchange value of **Int.$ 630/ha/year** and a welfare value of **Int.$ 1,173/ha/year**.
- For cultural services, the differences were slightly smaller but still significant. For **recreational services**, the estimate for exchange values was **Int.$ 832/ha/year**, while the welfare value was **Int.$ 1,563/ha/year**.

In most cases—particularly for services with a strong non-market component—the welfare-based estimates exceed exchange values by 30–70%, reflecting the inclusion of indirect benefits such as consumer surplus and non-use values. However, when service benefits are primarily captured by the actual cost or market proxy (as with some provisioning services), these differences can sometimes reverse. Such quantitative variations are driven by the underlying assumption inherent in each valuation concept: exchange values capture the price that would occur in a competitive market, whereas welfare values attempt to reflect the total societal benefit, including intangible and indirect components. Additionally, the institutional context in which values are assigned plays a role by influencing expectations in a free market (Barton 2022).

Qualitative evaluation of (extended) SNA-conform valuation versus welfare-based valuation

Section 9.2 of the report presents a structured expert evaluation of ecosystem service valuation methods using five evaluation criteria: **timeliness**, **salience**, **credibility**, **cost proportionality** and **legitimacy**. These criteria were applied to the two principal value concepts considered in the report: **exchange values** (in line with SEEA EA) and **welfare values** (reflecting total economic value, including consumer surplus and non-use values). Our qualitative assessment reveals distinct strengths and weaknesses for each value concept, with important implications for their use in ecosystem accounting and policy.

- **Timeliness**
    - **Exchange values** score **high** on timeliness. Because they are based on market proxies or cost-based methods that are often available in secondary data sources (e.g., national statistics, existing accounts), they can be produced relatively quickly and are easier to update over time. This aligns well with the regular cycles of national accounting and facilitates integration with official statistics.
    - **Welfare values**, by contrast, score **lower** in timeliness. They typically require original data collection (e.g., stated preference surveys or benefit transfer with careful calibration), which increases the time needed for analysis. Moreover, welfare values may be more sensitive to social and cultural dynamics, requiring periodic re-assessment to remain relevant. In policy settings requiring rapid responses or frequent updates, this presents a limitation.
- **Salience**
    - **Exchange values**, while useful for aligning with economic aggregates, are often **less salient** in these contexts, since they omit consumer surplus and do not capture the full extent of value derived from non-market services. In stakeholder consultations or participatory planning processes, exchange values may be perceived as underrepresenting the importance of certain ecosystem services, especially those without direct market analogues.
    - **Welfare values** are assessed to be **more salient**, particularly in policy contexts involving **public goods**, **non-market values**, and broader **societal well-being**. Examples could be decision-support where the goal is to inform decisions that weigh full societal costs and benefits—such as land use trade-offs, green

infrastructure investment, or conservation prioritization. Because welfare valuation explicitly captures individual preferences, including willingness to pay for ecosystem improvements or to avoid losses, it resonates more strongly with stakeholders' perceptions of value. For example, services such as recreation, biodiversity conservation, or cultural heritage are better represented by welfare-based estimates.

- **Credibility**
    - **Exchange values** score **higher on credibility** in institutional settings involving links to the SNA. Their alignment with the SNA, use of observable or imputed market prices, and compatibility with existing economic metrics make them easier to explain and justify to policy makers with experience in decision-making based on SNA-conform metrics of economic activity. This formal consistency lends them a degree of legitimacy and acceptability in governmental and international reporting systems.
    - **Welfare values**, although methodologically rigorous, are sometimes perceived as **less credible**, especially among users unfamiliar with economic valuation techniques. Stated preference methods (e.g., contingent valuation) can be criticized for their hypothetical nature, susceptibility to bias, or lack of grounding in real-world behaviour. Nevertheless, among environmental economists and for applications where societal preferences are central (e.g., conservation funding decisions), welfare values are typically seen as more credible than exchange values.
- **Cost Proportionality**
    - **Exchange values** generally offer **better cost proportionality**, particularly when used in conjunction with value transfer methods. Because they rely on existing data or standardized methods (e.g., cost-based or market-price approximations), the costs of implementing them are relatively low compared to their utility in producing comparable, scalable outputs. This is a major advantage for national statistical agencies or large-scale ecosystem accounting.
    - **Welfare values** involve **higher costs**, both in terms of data requirements and analytical complexity. Even when using benefit transfer, the need for careful site-to-site adjustments and validation increases the resource intensity. Where primary studies are needed—especially for culturally specific or non-use values—the costs can become prohibitive for regular use in national accounting.
- **Legitimacy**
    - **Exchange values** SNA-conform methods have **high institutional acceptance** but **lower potential for broad stakeholder representation and value inclusivity in practice**, as illustrated by the data-driven focus in our test sites. Extended SNA-conform approaches have medium scores across these factors, with transparency being a key challenge. Value transfer methods generally score **low on representation and inclusivity at the policy site**.
    - **Welfare values** methods score **high on representation and inclusivity** but **lower on institutional acceptance for accounting**, as seen in the varied uptake across European contexts.

10.2 Methodological implications

Exchange values versus welfare values

The divergence in evaluation results between exchange values and welfare values reflects not only conceptual differences, but also differences in methodology and application, which we discuss in this section.

Exchange values, being compatible with the System of National Accounts (SNA), are well-suited for integration into macroeconomic aggregates. They provide consistency, comparability, and policy utility in economic planning contexts—especially where ecosystem services act as inputs to production (e.g., timber, crop provisioning). However, they systematically exclude non-market values such as consumer surplus and existence values, thus underrepresenting total social welfare.

Welfare-based values, by incorporating willingness-to-pay (WTP) and non-use values, offer a broader measure of societal benefit. This is especially relevant in evaluating public goods, cultural services, or conservation policy, where market transactions are non-existent or non-representative.

Based on our evaluation, we found several factors that contribute to the observed differences:

1. **Value concept assumptions**: Exchange values reflect market transactions and are constrained by the need for equivalence in supply and use, in line with the SNA. In contrast, welfare values are based on utility gains, including those without market expression, such as non-use values.
2. **Service type**: The discrepancy between exchange and welfare values is more pronounced for services with high non-use or indirect benefits—e.g., recreation, climate regulation, or cultural values. For provisioning services like crops or timber, the gap may be narrower, or even reversed, due to the larger fraction of the total benefits being captured in the exchange value.
3. **Valuation method used**: Value function transfer provided a scalable method for applying both exchange and welfare assumptions, but transfer errors remained substantial. The average absolute percentage error ranged from **20% (temperate forest)** to **44% (inland wetlands)**, indicating significant predictive uncertainty regardless of the value concept.

Primary data collection versus value transfer methods

Value transfer methods can provide a less costly alternative to valuation when compared to primary data collection. Therefore, an important aspect of our work was the evaluation of value transfer methods as compared to primary data collection.

- **Primary data collection:** Our empirical tests at SELINA DPs and TSs, such as the detailed recreational surveys in Irish forests and the direct biophysical measurements in Lithuanian agricultural areas, demonstrated that primary data collection can result in high-resolution, context-specific estimates. These are tailored to local ecological conditions and socio-economic factors, reducing uncertainty related to heterogeneity. However, our testing shows that accurate primary data collection tends to be both resource-intensive and time-consuming. Its high cost and longer turnaround times can be challenging under rapid policy update cycles.

- **Value transfer methods:** In contrast, value transfer techniques—including unit value transfers, value function transfers, and meta-analytic transfers—offer a scalable, potentially cost-efficient (depending on transfer error) means to estimate ecosystem service values across diverse regions. While our analyses reveal that value transfer methods can approximate the magnitude of local estimates reasonably well, they inevitably introduce additional uncertainty. This uncertainty arises due to inherent differences between study and policy sites, potential misalignment in biophysical characteristics, and the averaging effects inherent in meta-analytic approaches. Even though our value functions included variables aimed to capture these differences, our results indicate that the average absolute percentage error for transferred values could range between 20% for temperate forests and as high as 44% for inland wetlands. Whether these errors are acceptable will depend on the real change in ecosystem extent and condition, as well as the stage of the decision process in which the values are used: for exploratory assessments these can be acceptable, for planning and prioritisation of measures, these need to be carefully considered.

Trade-offs to consider when choosing value transfer methods stem from the fact that there are likely biases in the selected sample of primary studies. The implicit assumption in selecting primary studies for value transfer that the underlying body of literature provides an unbiased sample of the population of empirical estimates (i.e., no selection biases) and that these estimates provide an unbiased representation of true values (i.e., no measurement error) should be critically considered (Grammatikopoulou et al. 2023). Examples of issues that can be addressed are:

Reliability and precision: Transferred values may not accurately reflect local conditions or the full range of ecosystem service benefits. Differences in ecological, social, and economic contexts can undermine the precision of the estimated values.

Commodity inconsistency: Studies often measure similar ecosystem services using different methodologies or definitions. This mismatch can lead to errors when applying values from one context (donor site) to another (recipient site).

Spatial scale and heterogeneity: Ecosystem services vary significantly over space. Value estimates derived from one location might not be appropriate for another due to differences in environmental conditions, land use, or biodiversity. Such spatial discrepancies can result in transfer errors. This can be partially avoided by using value functions based on large enough datasets that include these parameters.

Differences in institutional context: Local institutional contexts vary and influence the value articulation. Additionally, when simulating markets, for example using stated preference methods, researchers also create simulated value articulating institutions. It is important to recognise institutional contexts in simulating and generalising exchange values for monetary ecosystem accounts (Barton 2022).

Aggregation and scope challenges: When scaling up or "adding up" values across ecosystems or services, the lack of uniformity in how services are defined and measured can lead to aggregation errors.

Temporal transfer errors: Over time, changes in ecosystems or shifts in market conditions can render transferred values outdated. This temporal mismatch can affect the relevance of the valuation to current or future conditions.

10.3 Policy implications and recommendations

Implications of valuation concepts for policy

The SEEA EA framework's deliberate prioritization of exchange values is driven by the need for compatibility with the System of National Accounts (SNA). This alignment is intended to facilitate the integration of ecosystem information with traditional economic indicators (GDP, national wealth), thereby supporting a range of policy objectives outlined in the NCAVES and MAIA typology (2022). These objectives include:

- Comparing the values of environmental assets (including ecosystems) with other asset types (e.g., produced assets) as part of extended measures of national wealth.
- Highlighting the relevance of non-market ecosystem services (e.g., air filtration) by assigning them a monetary value, even if simulated.
- Assessing the contribution of ecosystem inputs to production in specific industries and their supply chains.
- Comparing the trade-offs between different ecosystem services through consideration of relative prices (where exchange values can be estimated).
- Deriving complementary aggregates such as degradation adjusted measures of national income.
- Evaluating trends in measures of income and wealth.
- Improving accountability and transparency around the public expenditures on the environment by recognising expenditure as an investment rather than a cost.
- Providing baseline data to support scenario modelling and broader economic modelling.
- Assessing financial risks associated with the environment; and
- Calibrating the application of monetary environmental policy instruments such as environmental markets and environmental taxes and subsidies.

However, the strict adherence to exchange values, while ensuring SNA compatibility, presents critical limitations for other vital policy applications. As highlighted by our evaluation and consistent with broader critiques (Bateman et al., 2011; Dasgupta, 2021), exchange values inherently exclude non-market benefits and broader welfare effects (consumer surplus, non-use values). This can lead to a significant underestimation of the true economic significance of ecosystem services, particularly those providing public goods or non-excludable benefits like climate regulation, biodiversity, and many cultural values.

The policy implication is that decisions informed *solely* by SEEA EA exchange values risk prioritizing economic activities with clear market returns over the protection of ecosystems providing substantial, but non-marketed, societal benefits. This can lead to suboptimal investment in ecosystem protection and potentially stimulate further nature loss, as consistency with traditional economic growth metrics through using exchange values makes it easier to align environmental policy with growth strategies, potentially overlooking the full costs of environmental degradation.

Welfare-based valuations, by capturing consumer surplus and non-use values, offer a more comprehensive measure of the total benefits society derives from ecosystem services. This makes them highly salient for policy decisions explicitly aimed at maximizing societal well-being, such as cost-benefit analysis for large infrastructure projects, conservation prioritization, and the evaluation of policies related to public goods. Our quantitative comparisons generally showed welfare values exceeding exchange values, particularly for regulating and cultural services with significant non-market components, underscoring the difference in the scope of value captured. The tension between methodological consistency for national accounts (favouring exchange values) and the local specificity required for effective policy analysis (often requiring welfare values) is a key challenge, as noted by the IPBES VA Summary for Policy Makers B7.

The mandated ecosystem accounting under Regulation (EU) 2024/3024 underscores the growing policy demand for monetary valuation to highlight the costs of inaction and support environmental goals. While monetary accounts for ecosystem services are not yet mandatory, the call for pilot studies adhering to SEEA EA standards highlights the ongoing tension between the need for standardized, SNA-compatible accounts and the desire for information that fully reflects the value of nature for detailed policy analysis.

Implications of valuation methods for policy

The choice between primary data collection and value transfer methods also carries significant policy implications, primarily concerning the trade-offs between timeliness, cost, accuracy, and the ability to capture spatial and temporal variations relevant to policy.

Primary data collection, as demonstrated in our test sites, can provide highly credible, site-specific estimates tailored to local conditions. This is invaluable for local or regional policy decisions requiring detailed, context-relevant information, such as specific land-use planning, protected area management, or project appraisal. The detailed insights gained from primary studies can directly inform the design and evaluation of localized interventions. However, the high cost and time requirements of primary data collection limit its feasibility for regular, large-scale national accounting or rapid policy assessments across numerous ecosystem assets.

Value transfer methods, particularly meta-analytic value function transfer, offer a scalable and more timely approach for populating ecosystem accounts across large areas and for regular reporting cycles. This aligns well with the mandate for national-level ecosystem accounting in the EU, where comprehensive coverage is required. Value function transfer allows for the estimation of values that reflect spatially variable determinants (e.g., population density, ecosystem condition), providing a more nuanced picture than simple unit transfer.

However, our evaluation highlights policy implications arising from the uncertainties and transfer errors associated with value transfer (ranging from 20-44% in our value function application). These errors can limit their credibility and salience for local-level, high-stakes policy decisions where precision is critical. A key limitation for policy support is that the value concept in the meta-analytical function may not explicitly identify the underlying "change in" ecosystem extent, condition, or service flow that the original studies valued. This can lead to a mismatch with policy analysis needs, particularly when assessing the value of specific interventions designed to alter these factors. While low-cost methods are attractive for regular accounting, their information value for "change detection" and "contribution of ES to the economy" can be disputed if they do not adequately capture the links between ecosystem extent, condition, and service flows, which are crucial for understanding the impact of policies.

Recommendations for policy

Based on these implications, a layered and context-dependent approach to monetary valuation in support of policy would be most appropriate. This should consider the specific objectives for using monetary values, as described in the NCAVES and MAIA typology, grouped in primary, secondary and tertiary uses. Below follow our recommendations for each.

Primary Uses of Ecosystem Service Values: These uses focus on generating foundational accounts to understand the contribution of ecosystem services to the economy and track trends over time. Policy relevant questions include:

- What is the relative importance of ecosystem service contributions to the economy?
- Are there trends in their monetary value over time?
- How do different economic sectors, jurisdictions, or management areas compare in terms of their annual contributions to the economic product through ecosystem services?

For these primary uses, exchange values, as prioritized by SEEA EA, are the most relevant concept for ensuring compatibility with national economic accounting frameworks and facilitating integration with SNA aggregates. They provide a standardized metric for comparing the economic contribution of ecosystem services alongside traditional economic sectors.

Value function transfer is a highly valuable and efficient method for generating the spatially explicit monetary estimates required for these comprehensive national accounts. Its scalability and relative timeliness make it a practical tool for populating accounts across large areas and meeting reporting requirements, directly supporting objectives like evaluating trends in measures of income and wealth and providing baseline data for scenario modelling. While uncertainties associated with transfer errors exist, for broad-scale assessments and trend analysis at the national level, value function transfer offers a pragmatic approach to achieve comprehensive coverage. However, it is crucial that the limitations in capturing the full value (welfare) are acknowledged when interpreting these results for policy.

Secondary Uses of Ecosystem Service Values: These uses involve more advanced analyses that utilize ecosystem accounting data to inform complex decision-making processes. Objectives include:

- Scenario analysis: How will the exchange values of ecosystem services change in response to global drivers such as climate change, species loss, and population growth?

- Trade-offs: Using ecosystem accounting data to inform financial and social cost-benefit analyses, as well as multi-criteria decision-making. This involves evaluating the exchange value of ecosystem services relative to alternative land uses, and considering their importance in relation to economic welfare values and other non-monetary values.

For scenario analysis focused on changes in exchange values, the exchange value concept remains relevant, particularly when assessing impacts on market-based ecosystem services or those with clear production inputs. Value function transfer can play a role here by providing spatially explicit baseline data and allowing for the modelling of how changes in biophysical drivers might affect transferred values. However, the ability of current value functions to accurately capture the impact of significant changes in ecosystem condition or the emergence of novel conditions under future scenarios needs further investigation.

For trade-off analysis, particularly in cost-benefit analysis, relying solely on exchange values is insufficient and can lead to suboptimal decisions. Welfare values are essential here to capture the full societal benefits and costs, including non-market impacts. While SEEA EA provides a valuable starting point with exchange values, supplementing this with welfare-based valuations (derived from primary studies or carefully calibrated value transfer) is critical for a comprehensive assessment of trade-offs. Value function transfer of welfare values, where robust meta-analyses exist, could contribute to this, but the uncertainties and the need for careful validation against local context are heightened in these applications. The objective of comparing trade-offs between different ecosystem services is best served by considering both exchange and welfare values.

Tertiary Uses of Ecosystem Service Values: These uses provide insights to directly inform policy design and evaluate the effectiveness of interventions. Objectives include:

- Impact evaluation and attribution: Assessing the impact of policy instruments or management measures on the exchange value of ecosystem services over a specific time period and in a specific area or population. This requires the use of before-after-control-impact approaches.
- Policy design: Determining the regulatory standards and economic incentives needed to achieve policy objectives for the exchange value of ecosystem services.

For impact evaluation and attribution, while exchange values can provide insights into the economic outcomes of policies on market-relevant services, they may not fully capture the broader societal impact. Welfare values are often more appropriate for evaluating the overall success of policies aimed at enhancing societal well-being through ecosystem services. Primary data collection is often the most credible method for rigorous impact evaluation due to the need for site-specific baseline and follow-up data. Value function transfer is generally less suitable for precise impact evaluation at a local scale due to transfer errors and the difficulty in ensuring the transferred function accurately reflects the specific intervention and context.

For policy design, particularly when considering economic instruments like taxes or subsidies (calibrating the application of monetary environmental policy instruments), exchange values can provide some relevant information, especially for services with established markets.

However, setting optimal policy instruments often requires understanding the full marginal costs and benefits, which necessitates welfare-based approaches. Value function transfer could potentially provide initial estimates for broad policy design considerations at a national level, but site-specific primary data or highly validated local transfers would be necessary for calibrating instruments for specific regions or contexts. Objectives related to improving accountability around public expenditures or assessing financial risks can utilize both exchange values (e.g., assessing the economic assets at risk) and, ideally, welfare values to understand the full potential impact of environmental changes.

Finally, we present some additional general recommendations:

- Acknowledge the limitations of relying solely on exchange values for policy decisions that extend beyond SNA compatibility and market-based contributions. Encourage the complementary use of welfare-based valuations for comprehensive policy analysis aimed at maximizing societal well-being.

- Leverage value function transfer as a valuable tool for generating comprehensive, spatially explicit monetary estimates for national-level ecosystem accounting and primary uses, supporting objectives like evaluating trends and providing baseline data. Transparently communicate the uncertainties associated with transfer errors and the limitations in capturing the full value concept for certain policy applications.

- For local and regional decision-making, and for secondary and tertiary uses requiring high precision or capturing non-market values, prioritize targeted primary data collection or highly localized and validated value transfer applications. Recognize the tension between national-level standardization and the need for local specificity in valuation for effective policy support.

- Invest in improving the availability and consistency of biophysical and economic data across Europe to enhance the robustness of all valuation methods, including value function transfer. Further research is needed to refine value transfer methodologies, particularly in their ability to reflect changes in ecosystem condition and service flows accurately, thereby improving their utility for change detection and assessing the contribution of ES to the economy.

- Ultimately, the policy relevance of monetary valuation in ecosystem accounting hinges on clearly defining the specific policy question being addressed, choosing the valuation concept and method most appropriate for that question and its context, and transparently communicating the strengths, limitations, and uncertainties of the resulting values to ensure they are used appropriately and effectively in decision-making processes.

10.4 Limitations to the study

While this study represents a comprehensive evaluation of monetary valuation methods within the SEEA EA framework, it was limited by some practical constraints. These relate to methodological issues, data availability, comparability, model performance, and the broader applicability of our results across policy contexts.

1. **Limited empirical coverage of ecosystem services and biomes**

The scope of ecosystem services and ecosystem types included in the empirical evaluations was necessarily selective. The services analysed, such as crop and grazed biomass provisioning, wild animals, global climate regulation, and recreation, were chosen based on feasibility in a subset of test sites, data availability, and alignment with SEEA EA priorities. As such, important services like pollination, air purification, coastal protection, or cultural non-use values were not included in the empirical comparison of valuation methods.

Similarly, the test sites cover a limited range of biomes, mostly temperate forest, agricultural, freshwater, and coastal systems. There is little coverage of open marine ecosystems, arid zones, or boreal and alpine systems, limiting the generalizability of results to other ecological contexts.

2. **Constraints in comparing welfare and exchange values**

While the report includes several comparisons between exchange and welfare valuation results, particularly through the application of a flexible meta-analytic value function, the number of cases where both value concepts could be empirically applied to the same service at the same site was limited. This restricts the robustness of conclusions drawn about systematic quantitative differences between valuation approaches.

3. **Transfer error and uncertainty in value function applications**

While the value function transfer method used in the study allows for consistent estimation of both exchange and welfare values across a large number of sites, its predictive performance varies significantly by ecosystem service. Reported average absolute percentage errors ranged from 20% to 44%, with particularly high uncertainty for wetland services. This degree of transfer error limits the precision and reliability of monetary values when applied at finer spatial scales or for local decision-making. However, such errors are typical for value transfer exercises (NCAVES and MAIA 2022), and are not worse than typical uncertainties in cost-estimates for benefit-cost analyses.

Additionally, while the meta-analytic function includes a variable for “value concept” to toggle between exchange and welfare assumptions, this operationalization assumes that other model parameters hold constant across these conceptual frameworks. In practice, however, the underlying drivers of value (e.g., demand elasticity, substitutability, cultural preferences) may differ significantly depending on the valuation approach.

4. **Incomplete representation of uncertainty and sensitivity**

Although the study acknowledges the existence of uncertainty in both primary and transferred values, and provides some indicators of prediction error, there is limited systematic treatment of uncertainty propagation through the valuation and accounting process (Walther et al. 2025). Due to the nature of the data available to us, we were not able to uniformly apply sensitivity analysis of key assumptions (e.g., discount rates, spatial resolution, price proxies) across case studies. As a result, users of the resulting values may underestimate the range of plausible estimates and the implications of methodological choices.

5. **Limited integration with SEEA biophysical accounts**

The valuation exercises were not fully embedded within complete SEEA EA asset or service accounts. The aim of the study was to focus on differences among valuation methods rather than full account compilation, which limits the demonstration of how valuation outputs would interface with biophysical indicators, condition accounts, or supply-use tables. Consequently, the operational integration of monetary valuation into SEEA-based ecosystem accounts remains partially conceptual rather than fully implemented in this study.

6. **Resource and institutional constraints in primary data collection**

Several test sites reported challenges in collecting new, high-quality primary data, particularly for services such as recreation or cultural use values. As a result, the reliance on value transfer methods was greater than originally planned in some cases. This highlights the practical challenge of implementing welfare-based approaches where institutional capacity, time, or funding are constrained—a point that is especially relevant for national statistical offices considering the operationalization of ecosystem accounting.

7. **Policy relevance not empirically tested**

While the report aims to provide a robust theoretical and expert-based assessment of the policy relevance of different valuation methods, it does not empirically evaluate how end users (e.g. decision-makers, planners) interpret, trust, or apply the valuation results. So far, no stakeholder validation or user testing was undertaken to assess whether different value concepts change perceptions of ecosystem importance or influence decisions (Barton et al. 2024). This limits the conclusions that can be drawn about the real-world salience or utility of the monetary estimates produced.

10.5 Further steps

SELINA deliverable 5.4 will integrate the findings of deliverables 5.5, 5.2 and 5.3 to provide recommendations for SEEA EA implementation in the EU and globally. This will allow us to further link our findings with work done in SELINA on the integration of externalities and disservices, and the use of remote sensing techniques for improving ecosystem accounts. This will also allow us to further address the limitations raised in the previous section, for example by inviting stakeholders in the SELINA DPs to give feedback on our findings, providing a stronger empirical basis for our recommendations, and to further address the links between ecosystem condition indicators and monetary value estimates.

We will also provide input to Work Package 6, especially deliverable 6.6, on the best use of methods and data for evidence-based decision-making. Finally, we will provide input to the parts of the SELINA Compendium of Guidance relating to valuation of ecosystem services.

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12 References

Albanis, K., Xanthopoulos, G., Skouteri, A., Theodoridis, N., Christodoulou, A., Palaskas, D. (2015). Methodology for estimating the value of forest land in Greece - Detailed Manual. ELGO - "DIMITRA", Institute of Mediterranean Forest Ecosystems, Athens, pp. 201.

Aneseyee, A. B., Soromessa, T., & Elias, E. (2020). The effect of land use/land cover changes on ecosystem services valuation of Winike watershed, Omo Gibe basin, Ethiopia. Human and Ecological Risk Assessment: An International Journal, 26(10), 2608-2627.https://doi.org/10.1080/10807039.2019.1675139.

Bagstad, K. J., D. J. Semmens, S. Waage and R. Winthrop (2013). "A comparative assessment of decision-support tools for ecosystem services quantification and valuation." Ecosystem Services **5**: 27-39.

Barrage, L. and W. Nordhaus (2024). "Policies, projections, and the social cost of carbon: Results from the DICE-2023 model." Proceedings of the National Academy of Sciences 121(13): e2312030121.

Barton, D. N. (2023). "Value ‘generalisation’ in ecosystem accounting - using Bayesian networks to infer the asset value of regulating services for urban trees in Oslo." One Ecosystem **8**: e85021.

Barton, D. N., R. Chaplin-Kramer, E. Lazos Chavero, M. Van Noordwijk, S. Engel, A. Girvan, T. Hahn, B. Leimona, S. Lele, R. Muradian, P. Ungar, C. Aydin, P. Iranah, S. Nelson, M. Cantú-Fernández and D. González-Jiménez (2022). Chapter 4: Value expression in decision-making. In: Methodological Assessment Report on the Diverse Values and Valuation of Nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Balvanera, P., Pascual, U., Christie, M., Baptiste, B., and González-Jiménez, D. (eds). Value expression in decision-making. IPBES secretariat, Bonn, Germany.

Barton, D. N. (2022). "Recognising institutional context in simulating and generalising exchange values for monetary ecosystem accounts." One Ecosystem **7**: e85283.

Barton, D. N., Caparrós, A., Conner, N., Edens, B., Piaggio, M. & Turpie, J. (2019). SEEA Experimental Ecosystem Accounting Revision 2020 Working Group #5 – Valuation concepts and accounting treatments. Discussion paper 5.1: Defining exchange and welfare values, articulating institutional arrangements and establishing the valuation context for ecosystem accounting.

Barton, D. (2007). How Much Is Enough? The Value Of Information From Benefit Transfers In A Policy Context. In: Navrud, S., Ready, R. (eds) Environmental Value Transfer: Issues and Methods. The Economics of Non-Market Goods and Resources, vol 9. Springer, Dordrecht.

Bateman, I. J., Mace, G. M., Fezzi, C., Atkinson, G., & Turner, K. (2011). Economic analysis for ecosystem service assessments. *Environmental and Resource Economics, 48*(2), 177-218.

Brander, L. M., R. de Groot, J. P. Schägner, V. Guisado-Goñi, V. van 't Hoff, S. Solomonides, A. McVittie, F. Eppink, M. Sposato, L. Do, A. Ghermandi, M. Sinclair and R. Thomas (2024). "Economic values for ecosystem services: A global synthesis and way forward." Ecosystem Services **66**: 101606.

Brander, L. M., J. P. Schägner and R. de Groot (2022). "On the potential use of the Ecosystem Services Valuation Database for valuation in the System of Environmental Economic Accounting." One Ecosystem **7**: e85085.

Brander, L.M., van Beukering, P., Balzan, M., Broekx, S., Liekens, I., Marta-Pedroso, C., et al. (2018). Report on economic mapping and assessment methods for ecosystem services. Deliverable D3.2 EU Horizon 2020 ESMERALDA Project, Grant agreement No. 642007.

Bronzes, A., L. Hein, R. Groeneveld and A. Pulatov (2025). "A comparison of valuation methods for cultural ecosystem services in support of ecosystem accounting." One Ecosystem **10**: e108556.

Buchholz, M., M. Danne and O. Musshoff (2022). "An experimental analysis of German farmers' decisions to buy or rent farmland." Land Use Policy **120**: 106218.

Buchhorn, M., B. Smets, T. Danckaert, M. van Loo, S. Broekx and W. Peelaerts (2022). "Establishing a reference tool for ecosystem accounting in Europe, based on the INCA methodology." One Ecosystem **7**.

Caparrós, A., J. L. Oviedo, A. Álvarez and P. Campos (2017). "Simulated exchange values and ecosystem accounting: Theory and application to free access recreation." Ecological Economics **139**: 140-149.

Capitals Coalition (2024). Governance for Valuation - A common structure to build confidence in valuation.

Chan, K. M., Satterfield, T., & Goldstein, J. (2012). Rethinking ecosystem services to better address and navigate cultural values. *Ecological Economics, 74*, 8-18.

Clerici, N., Cote-Navarro, F., Escobedo, F. J., Rubiano, K., & Villegas, J. C. (2019). Spatio-temporal and cumulative effects of land use-land cover and climate change on two ecosystem services in the Colombian Andes. Science of the Total Environment, 685, 1181-1192.https://doi.org/10.1016/j.scitotenv.2019.06.275.

Dasgupta, P. (2021). *The Economics of Biodiversity: The Dasgupta Review*. HM Treasury, London.

Dong, F., Y. Gao, Y. Li, J. Zhu, M. Hu and X. Zhang (2022). "Exploring volatility of carbon price in European Union due to COVID-19 pandemic." Environmental Science and Pollution Research **29**(6): 8269-8280.

Ebner, M. (2016). *The economic value of hunting in the EU. Presentation at the "The Economic Value of Hunting in the EU" conference. European Parliament. 27th September 2016*. https://www.face.eu/2016/09/hunting-in-europe-is-worth-16-billion-euros/

Edens, B., & Hein, L. (2013). Towards a consistent approach for ecosystem accounting. *Ecological Economics, 90*, 41-52.

European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, United Nations, & World Bank. (2009). *System of National Accounts 2008*.

European Commission, Joint Research Centre, Grammatikopoulou, I., Chatzimichael, K., Sylla, M., La Notte, A., Zurbaran Nucci, M. and Paracchini, M. (2024), The contribution of ecosystem services in agricultural production: An application of the production function approach, Publications Office of the European Union, Luxembourg. https://data.europa.eu/doi/10.2760/1432445, JRC138967.

Gao, J., Li, F., Gao, H., Zhou, C., & Zhang, X. (2017). The impact of land-use change on water-related ecosystem services: a study of the Guishui River Basin, Beijing, China. Journal of Cleaner Production, 163, S148-S155.https://doi.org/10.1016/j.jclepro.2016.01.049.

Gomes, E., Inacio, M., Miksa, K., Kalinauskas, M., Karnauskaite, D., **Pereira, P.** (2021) Future scenarios impact on land use change and habitat quality in Lithuania. Environmental Research 197, 111101.

Grammatikopoulou, I., Chatzimichael, K., Sylla, M., La Notte, A., Zurbaran Nucci, M. and Paracchini, M., The contribution of ecosystem services in agricultural production: An application of the production function approach, Publications Office of the European Union, Luxembourg, 2024, doi:10.2760/1432445, JRC138967.

Grammatikopoulou, I., T. Badura, R. J. Johnston, D. Barton, S. Ferrini, M. Schaafsma and A. La Notte (2023). "Value transfer in ecosystem accounting applications." Journal of Environmental Management **326**: 116784.

Greaker, M. and L. Lindholt (2021). The resource rent in Norwegian aquaculture 1984-2020 - Calculations applying the National Accounts.

Gren, I.-M., & Kerr, G. (2022). A Meta-Regression Analysis of Hunters' Valuations of Recreational Hunting. *Sustainability*, *15*(1), 27. https://doi.org/10.3390/su15010027

Häggmark-Svensson, T. H., Elofsson, K., Engelmann, M., & Gren, I.-M. (2015). *A review of the literature on benefits, costs, and policies for wildlife management. Swedish University of Agricultural Sciences, Department of Economics. Working Paper Series 2015:01*.

Haines-Young, R. and M. B. Potschin (2017). Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure.

Hein, L., Obst, C., Edens, B., & Remme, R. P. (2020). Progress and challenges in the development of ecosystem accounts. *Ecosystem Services, 42*, 101329.

Huber, C., Meldrum, J., & Richardson, L. (2018). Improving confidence by embracing uncertainty: A meta-analysis of U.S. hunting values for benefit transfer. *Ecosystem Services*, *33*, 225–236. https://doi.org/10.1016/j.ecoser.2018.07.001

IPBES (2022). Summary for policymakers of the methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES secretariat, Bonn, Germany**:** 37.

Johnston, R. J., K. J. Boyle, W. Adamowicz, J. Bennett, R. Brouwer, T. A. Cameron, W. M. Hanemann, N. Hanley, M. Ryan, R. Scarpa, R. Tourangeau and C. A. Vossler (2017). "Contemporary Guidance for Stated Preference Studies." Journal of the Association of Environmental and Resource Economists **4**(2): 319-405.

Laurans, Y., A. Rankovic, R. Bille, R. Pirard and L. Mermet (2013). "Use of ecosystem services economic valuation for decision making: Questioning a literature blindspot." Journal of Environmental Management **119**: 208-219.

Liu, X., M. Wojewodzki, Y. Cai and S. Sharma (2023). "The dynamic relationships between carbon prices and policy uncertainties." Technological Forecasting and Social Change **188**: 122325.

Maes J, Teller A, Erhard M, Conde S, Vallecillo Rodriguez S, Barredo Cano J I, Paracchini M, Abdul Malak D, Trombetti M, Vigiak O, Zulian G, Addamo A, Grizzetti B, Somma F, Hagyo A, Vogt P, Polce C, Jones A, Marin A, Ivits E, Mauri A, Rega C, Czucz B, Ceccherini G, Pisoni E, Ceglar A, De Palma P, Cerrani I, Meroni M, Caudullo G, Lugato E, Vogt J, Spinoni J, Cammalleri C, Bastrup-Birk A, San-Miguel-Ayanz J, San Román S, Kristensen P, Christiansen T, Zal N, De Roo A, De Jesus Cardoso A, Pistocchi A, Del Barrio Alvarellos I, Tsiamis K, Gervasini E, Deriu I, La Notte A, Abad Viñas R, Vizzarri M, Camia A, Robert N, Kakoulaki G, Garcia Bendito E, Panagos P, Ballabio C, Scarpa S, Montanarella L, Orgiazzi A, Fernandez Ugalde O, Santos-Martín F (2020) Mapping and Assessment of Ecosystems and their Services: An EU ecosystem assessment. Publications Office of the European Union, Luxembourg, JRC120383. https://doi.org/10.2760/757183

Marcinkonis, S. (2006). "Nutrient leaching in dominant lithuanian soils." Archives of Agronomy and Soil Science **52**(2): 183-191.

Marques, S. M., Campos, F. S., David, J., & Cabral, P. (2021). Modelling sediment retention services and soil erosion changes in Portugal: a spatio-temporal approach. ISPRS International Journal of Geo-Information, 10(4), 262.https://doi.org/10.3390/ijgi10040262.

Marta-Pedroso, C., T. Domingos, H. Freitas and R. S. de Groot (2007). "Cost–benefit analysis of the Zonal Program of Castro Verde (Portugal): Highlighting the trade-off between biodiversity and soil conservation." Soil and Tillage Research **97**(1): 79-90.

Moeltner, K., K. J. Boyle and R. W. Paterson (2007). "Meta-analysis and benefit transfer for resource valuation-addressing classical challenges with Bayesian modelling." Journal of Environmental Economics and Management **53**(2): 250-269.

Natural Capital Project, 2025. InVEST 3.14.3. Stanford University, University of Minnesota, Chinese Academy of Sciences, The Nature Conservancy, World Wildlife Fund, Stockholm Resilience Centre and the Royal Swedish Academy of Sciences. https://naturalcapitalproject.stanford.edu/software/invest

NCAVES and MAIA (2022). Monetary valuation of ecosystem services and ecosystem assets for ecosystem accounting: Interim Version 1st edition. United Nations Department of Economic and Social Affairs, Statistics Division, New York.

Obst, C., Hein, L., & Edens, B. (2016). National accounting and the valuation of ecosystem assets and their services. *Environmental and Resource Economics, 64*(1), 1-23.

OECD. (2025a). *OECD Data Explorer • Annual Purchasing Power Parities and exchange rates*.

OECD. (2025b). *OECD Data Explorer • Consumer price indices (CPIs, HICPs)*.

Oras, K., Ronk, A., Aun, K., Luukas, G., Ehrlich, Ü., Kosk, A., Adermann, V., & Indres, E. (2021). *Development of the ecosystem accounts. Methodological report. Grant Agreement 881542—2019-EE-ECOSYSTEMS.* (s. 189). Statistics Estonia.

Palacino, B., S. Ascaso, A. Valero and A. Valero (2024). "Regeneration costs of topsoil fertility: An exergy indicator of agricultural impacts." Journal of Environmental Management **369**: 122297.

Panagos, P., G. Standardi, P. Borrelli, E. Lugato, L. Montanarella and F. Bosello (2018). "Cost of agricultural productivity loss due to soil erosion in the European Union: From direct cost evaluation approaches to the use of macroeconomic models." Land Degradation & Development **29**(3): 471-484.

Posner, S. M., E. McKenzie and T. H. Ricketts (2016). "Policy impacts of ecosystem services knowledge." Proceedings of the National Academy of Sciences **113**(7): 1760-1765.

Remme, R. P., Edens, B., Schröter, M., & Hein, L. (2015). Monetary accounting of ecosystem services: A test case for Limburg province, the Netherlands. *Ecological Economics, 112*, 116-128.

Remeur, C. (2020). Carbon emissions pricing - Some points of reference, European Parliamentary Research Service.

Rimal, B., Sharma, R., Kunwar, R., Keshtkar, H., Stork, N. E., Rijal, S., ... &Baral, H. (2019). Effects of land use and land cover change on ecosystem services in the Koshi River Basin, Eastern Nepal. Ecosystem services, 38, 100963.https://doi.org/10.1016/j.ecoser.2019.100963.

Schenau S, van Berkel J, Bogaart P, Blom C, Driessen C, de Jongh L, de Jong R, Horlings E, Mosterd R, Hein L, Lof M (2022) Valuing ecosystem services and ecosystem assets for The Netherlands. One Ecosystem 7: e84624.

Scheufele, G. and S. Pascoe (2023). "Ecosystem accounting: Reconciling consumer surplus and exchange values for free-access recreation." Ecological Economics **212**: 107905.

Schröter, M., Zanden, E. H., van Oudenhoven, A. P., Remme, R. P., Serna-Chavez, H. M., de Groot, R. S., & Opdam, P. (2014). Ecosystem services as a contested concept: A synthesis of critique and counter-arguments. *Conservation Letters, 7*(6), 514-523.

Sharp, E.R.; Chaplin-kramer, R.; Wood, S.; Guerry, A.; Tallis, H.; Ricketts, T.; Authors, C.; Nelson, E.; Ennaanay, D.; Wolny, S.; et al. Sediment Delivery Ratio. In InVEST User's Guide; The Natural Capital Project; Stanford University; University of Minnesota; The Nature Conservancy; World Wildlife Fund: Stanford, CA, USA, 2018; pp. 137–153.

Teng, H., Rossel, R. A. V., Shi, Z., Behrens, T., Chappell, A., & Bui, E. (2016). Assimilating satellite imagery and visible–near infrared spectroscopy to model and map soil loss by water erosion in Australia. Environmental Modelling & Software, 77, 156-167.https://doi.org/10.1016/j.envsoft.2015.11.024.

United Nations (2024). System of Environmental-Economic Accounting Ecosystem Accounting. Statistical Papers Series F. D. o. E. a. S. A.-S. Division. New York, United Nations.

United Nations. (2021). *System of Environmental-Economic Accounting—Ecosystem Accounting (SEEA EA)*. United Nations, New York.

Vigiak, O., Borselli, L., Newham, L. T. H., McInnes, J., & Roberts, A. M. (2012). Comparison of conceptual landscape metrics to define hillslope-scale sediment delivery ratio. Geomorphology, 138(1), 74-88.https://doi.org/10.1016/j.geomorph.2011.08.026.

Walther, F., D. N. Barton, J. Schwaab, J. Kato-Huerta, B. Immerzeel, M. Adamescu, E. Andersen, M. V. Arámbula Coyote, I. Arany, M. Balzan, A. Bruggeman, C. Carvalho-Santos, C. Cazacu, D. Geneletti, R. Giuca, M. Inácio, E. Lagabrielle, S. Lange, S. L. Clec'h, Z. Y. Vanessa Lim, U. Mörtberg, S. Nedkov, A. P. Portela, A. Porucznik, T. Racoviceanu, P. Rendón, D. Ribeiro, J. Seguin, M. Š. Hribar, V. Stoycheva, H. Vejre, C. Zoumides and A. Grêt-Regamey (2025). "Uncertainties in ecosystem services assessments and their implications for decision support – A semi-systematic literature review." Ecosystem Services **73**: 101714.

Zhao, M., He, Z., Du, J., Chen, L., Lin, P., & Fang, S. (2019). Assessing the effects of ecological engineering on carbon storage by linking the CA-Markov and InVEST models. Ecological Indicators, 98, 29-38.https://doi.org/10.1016/j.ecolind.2018.10.052.

Zhu, E., Deng, J., Zhou, M., Gan, M., Jiang, R., Wang, K., &Shahtahmassebi, A. (2019). Carbon emissions induced by land-use and land-cover change from 1970 to 2010 in Zhejiang, China. science of the total environment, 646, 930-939. https://doi.org/10.1016/j.scitotenv.2018.07.317.

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13 Annex

Annex 1 - Overview of primary data collection for valuation in DPs and TSs

13.1.1 Czech Republic

13.1.1.1 Game meat provision

To determine the value of wild game provision, we utilized data on the number of hunted game species, their average cold body weight, and their average purchase price in 2023. Hunting harvest data for 2023 were obtained from the Czech Statistical Office at both the national and regional (14 regions) levels.

A market survey was conducted to establish the average purchase price and weight of game species (excluding heads and extremities). This price encompasses both meat and other valuable parts, such as skins.

The analysis included the following species: red deer, sika deer, fallow deer, mouflon, roe deer, wild boar, European hare, mallard, and pheasant.

Table 27. Description of used data – Wild game provision in the Czech Republic.

Data	**Coverage**	**Source**
Quantity of hunted game species (2022/23)	national level and breakdown for 17 regions	Forest Management Institute and Ministry of Agriculture of the Czech Republic
Average purchase price of game species without VAT (2022/23)	National level	Market survey – meat processing companies
Average purchase weight of the cold body of game species	National level	Survey - meat processing companies

The value of the ecosystem service of wild game provision was calculated as follows:

$$GP = \sum_{i}^{N} a_i * b_i * c_i$$

Where a_i is the average purchase weight of a species I, b_i is the average purchase price of a species I, c_i is the number of hunted species I and N is the number of species analysed, i.e., 9.

The statistics for the hunted game are available for the hunting year (April - March). I labelled the statistics for 2022/23 as the year 2022 in line with the labelling of the years used by the Czech Statistical Office.

In addition, unit value transfer was used to allow for a comparison of methods.

Two meta-regression analyses of hunters' recreational hunting values have been published (Gren and Kerr 2022; Huber et al. 2018). However, these studies express values on a per-person-per-day basis, limiting their direct comparability to the present study, which focuses on annual hunter expenditures.

Consequently, two studies were selected for the benefit transfer analysis: a study conducted in Estonia (Oras et al. 2021) and data on hunters' expenditures presented by Michl Ebner at the "The Economic Value of Hunting in the EU" conference (Ebner 2016).

These studies were chosen due to their methodological similarity to the present study, which also collected data on hunters' expenditures in the Czech Republic.

To estimate the provisioning service value, a benefit transfer method was applied using data from the Estonian study (Oras et al. 2021). The valuation was based on Estonian data for average cold body weight, average meat purchase price (EUR/kg), and red deer skin purchase price (EUR/skin). These values were then multiplied by the Czech Republic's annual hunting harvest for each species.

To ensure comparability, all monetary values were converted to 2022 US dollars using country-specific Consumer Price Indices (CPI) and Purchasing Power Parities (PPP) for household final consumption expenditure (OECD 2025a, 2025b).

13.1.1.2 Recreational value of hunting

Hunting, requiring specialized equipment and licenses, incurs costs for participants. These expenditures can be used to value the ecosystem service of recreational hunting. In our TS, they were valued using two approaches:

1. **Hunting ground rental costs:** In 2022, 87% of hunting grounds were rented, which accounted for 85% of the area of hunting grounds in the Czech Republic (Czech Statistical Office, 2024). Due to limited data on rental prices, the analysis utilized rental prices from the Forests of the Czech Republic, a state-owned enterprise managing a significant portion of hunting grounds.
2. **Hunter expenditures:** Data on the number of hunters (Forest Management Institute, Ministry of Agriculture) and average annual hunting expenditures were obtained from a survey of regional hunting associations. This approach aligns with the methodology used in an Estonian study (Oras et al., 2021).

The types of data and their sources for calculating the value of provisioning and cultural ecosystem services are stated in Table 28.

Table 28. Description of used data - recreational hunting.

Data	Coverage	Source

Hunting grounds information	National level and breakdown for 14 regions	Forest Management Institute and Ministry of Agriculture of the Czech Republic
Rental price of hunting grounds leased by the Forests of the Czech Republic	Hunting grounds leased by the Forests of the Czech Republic (2018 – 2022)	Forests of the Czech Republic
Number of holders of valid hunting licenses permanently practicing game management rights	National level and breakdown for 14 regions	Forest Management Institute and Ministry of Agriculture of the Czech Republic
Cost of hunting - annual average expenditure for hunting	National level	Hunter questionnaire

As with the game meat provisioning service, Gren and Kerr 2022 and Huber et al. 2018 were used for a unit value transfer analysis of the recreational value of hunting.

The direct benefit transfer method was applied to estimate the value of recreational hunting in the Czech Republic. This involved obtaining average yearly hunters' expenditures from the selected studies and multiplying these values by the total number of hunters in the Czech Republic.

13.1.2 Greece

13.1.2.1 Crop provision

To identify the value of Crop provision in the study area, we used the information extracted from the Economic accounts for agriculture by NUTS 2 region (ESTAT, 2025). More precisely, we used the information for the value of the Crop output in the region for (a) basic prices, (b) subsidies and (c) producer prices and, for the regional units of Ileia and Achaia and for the year 2022.

We assume that the typical cultivated area has a uniform average productivity. Then, we calculate the share of the value of production of the test sites based on their share of regional land. Thus, a downscaling of the value of production for the region provided by Eurostat's regional accounts for agriculture is allocated to Achaia and Ilia, respectively, according to their respective shares of land. (Table 29). Thus, the value of production for Achaia is 26.28% of the regional value of production because Achaia possesses 26.28% of the region's Utilised Agricultural Area (UAA).

Table 29. Crop provision values for the Regional Units of the Test Site (Achaia and Ileia).

	Holdings	Area (stremma)	Basic prices (Million Euro)	Subsidies (Million Euro)	Producer prices (Million Euro)
Region of Western Greece (NUTS 2)	80502	2480243	1060,66	8,80	1051,86
Regional Unit of Achaia	22510	651822	278,75	2,31	276,43
Regional Unit of Ileia	28202	846870	362,16	3,00	359,15

13.1.2.2 Global climate regulation

For the valuation of global climate regulation, and for the forests of the study area, we used the estimation of the annual value of carbon sequestration due to the above ground biomass is based on the formula (Albanis et al. 2015):

$$Vsq = Area * (MAI\text{-}Vv) * BEF * CCF * Pc$$

Vsq: the annual value of carbon sequestration in €

Area: the area of the forest in ha

MAI: Mean Annual Increment of the timber stock in m3/ha

Vv: the average annual volume of wood harvested from the forest in m3/ha

BEF: coefficient which converts the volume of timber stock expressed in m3 to aboveground biomass expressed in tones of dry biomass in t/m3

CCF: Carbon conversion factor, the coefficient that converts woody biomass to the corresponding amount of carbon in t, being 0.5

Pc: price of carbon in €/t.

The below ground biomass is a fraction of the aboveground biomass. Thus, total

$$Vsq = [(Area * (MAI\text{-}Vv) * BEF) + (Area * (MAI\text{-}Vv) * BEF)*R] * 0.5 * Pc$$

$$= [(Area * (MAI\text{-}Vv) * BEF) *(1+R)] * 0.5 * Pc$$

The below to above ground biomass ratio R is a coefficient given in Table 4.11 of the IPCC 2003. The selection of the value of R from Table 4.11 of the IPCC, requires the knowledge of the above ground timber biomass in t/ha. For its calculation, however, knowledge of the timber stock in m3/ha is required. Because this parameter is not easily available in Greece, especially for forests that do not have a management plan, the Ministerial Decision suggests that the average timber stock of coniferous and deciduous forests of the country is 59.4 m3/ha and 34.64 m3/ha, respectively. The average aboveground wood stock biomass in t/ha of these, calculated using the BEF coefficients, is less than 50 t/ha for conifers and 75 t/ha for broadleaves. Based on these values, it is suggested to use the value R=0.46 for conifers and R=0.43 for broadleaves.

Using the outcomes of WP5.2 of the ecosystem extent map of Peloponnese, we calculated the carbon sequestration annual value, as presented in Table 30.

Table 30. Annual value of carbon sequestration, for forests and forested areas.

Forest and forested areas ecosystems	Area (ha)	MAI m3/ha	Vv	BEF	CCF	R	Pc: Price of CO2 in €/t	Annual value of carbon sequestration in €
Riparian forest and woodland - L3	41505.06	2.06	0.43	0.80	0.50	0.43	25	967441.44
Temperate, Submediterranean and Mediterranean thermophilous deciduous forest - L3	56422.34	2.62	0.2	0.95	0.50	0.43	25	2318654.90
Coniferous forest - L2 unallocated	141651.8	5.87		0.70	0.50	0.46	25	10622362.24
Pine forest (excluding mires, non-thermophilous) - L3	42758.9	5.87	0.3	0.70	0.50	0.46	25	3042584.36

Mediterranean thermophilous lowland pine forest - L3	59340.46	5.87	0.3	0.70	0.50	0.46	25	4222474.28
Other coniferous forests, excluding plantations - L3	1689.76	5.87	0.3	0.70	0.50	0.46	25	120237.83
Highly modified coniferous forests, in particular plantations - L3	2788.79	5.87	0.3	0.60	0.50	0.46	25	170092.49
Broadleaved evergreen forest - L2 unallocated	101245.62	0.3	0.2	1.00	0.50	0.43	25	180976.55
Mediterranean evergreen *Quercus* forest - L3	122530.44	0.3	0.2	1.00	0.50	0.43	25	219023.16
Olea europaea-Ceratonia siliqua forest - L3	115306.8	0.3	0.2	1.00	0.50	0.43	25	206110.91

13.1.2.3 Biodiversity maintenance

To identify the value of biodiversity maintenance, we used the ecosystem type map produced in WP5.2 for the region and followed the Greek Ministerial Decision FEK 2980, 4.11.2014: "Guide for Implementing a Model for the Assessment of the Value of Forest Land in Greece", that provides a methodology for estimating the Total Economic Value (TEV) of the flow of the different ecosystem services provided by forests and forested areas.

Biodiversity maintenance is estimated as follows (Ministerial Decision FEK 2980, n.d.):

$$Vb = Area * N * Pb$$

Where Vb is the annual value of forest biodiversity in €/year, Area is the extent of the forest resource in ha, N is a coefficient of how natural the ecosystem is, and Pb is the value of annual forest biodiversity per hectare measured in €/ha per year. The coefficient N ranges from 0.1 for fast growing plantations of alien species to 1 for high and non-managed forests. The Ministerial Decision proposes an escalation of the N coefficients as shown in Table 31.

Table 31. Values of the coefficient (N) for different forest types.

Forest ecosystem	N coefficient
Alien species fast grown plantations	0.1
Monoculture of regularly managed deciduous forests	0.2
Mixed forests regularly managed	0.3
Shrubs and Mediterranean maquis	0.4
Complex deciduous forests	0.5
Bush forests under transition to high forests	0.6
High forest monoculture	0.7
Mixed high forests	0.8
Reforestation in the phase of conversion into natural forest	0.8
Multi-layer mixed high forests and riparian forests	0.9
Virgin or primitive forests	1.0

Source: Table 13 of Ministerial Decision 115963/6070.

The value of P_b is approached through two different methods. Using the technique of meta-analysis of the database data, the range, and the average annual value of biodiversity protection in €/ha for each geographical group of countries, for Greece, the value of biodiversity protection is 484.4 €/ha/year according to the Ministerial Decision 115963/6070.

More precisely, for the estimation of the maximum annual value of biodiversity per hectare (€/ha and year), the Greek team that prepared the corresponding legislation considered two alternatives, which were developed during the implementation of the European MASIFF project, implemented in 2009-2010 entitled "Development of a methodology for the analysis of the socio-economic impacts of forest fires and the economic efficiency of fire management" (MASIFF Deliverable 4).

The first alternative is based on the study entitled "Further Developing Assumptions on Monetary Valuation of Biodiversity Cost of Policy Inaction (COPI)" conducted by the Institute for European Environment Policy in collaboration with Alterra, GHK (consultant in UK), Ecologic Institute and other experts (Ten Brink et.al. 2009, MASSIF Deliverable 4, 2010). In the first stage of this study, European countries were grouped according to their latitude, which was considered as a proxy for identifying their sensitivity to climate and the main forest types (Table 32).

Table 32. European countries were grouped according to their latitude, which was considered as a proxy for identifying their sensitivity to climate and the main forest types.

Geographical groups	**Longitude class**	**Countries**

Mediterranean Europe	Latitude N35-45°	Greece, Portugal, Spain, Italy, Albania, Bosnia and Herzegovina, Bulgaria, Serbia and Montenegro, Türkiye, North Macedonia
Central-Northern Europe	Latitude N45-55°	Austria, Belgium, France, Germany, Ireland, Luxembourg, Netherlands, Switzerland, Croatia, Czech Republic, Hungary, Poland, Romania, Slovakia, Slovenia
Northern Europe	Latitude N55-65°	Denmark, United Kingdom, Estonia, Latvia, Lithuania Scandinavian Europe
Scandinavian Europe	Latitude N65-71°	Finland, Iceland, Norway, Sweden

In a second stage, studies carried out in countries around the world on the assessment of the value of biodiversity were collected. Then, a database was created containing the values of biodiversity protection per ha and year of all the above studies. Subsequently, by applying the technique of meta-analysis of the database data, the scope, and the average annual value of biodiversity protection in $/ha for the year 2000 were estimated for each geographical group of countries (Table 33). In row 3 of Table 33, the average values in euros for the year 2000 are given, after their conversion from US dollars to euros (1€= 1.25). Finally, row 4 gives the average values for the year 2009 (conversion factor 1.2771). These values can be used to estimate the value of biodiversity for the countries included in the above table, by applying the benefit transfer method. For Greece, the value of biodiversity protection is 484.4 € /ha/year.

Table 33. Range and average values in euros for the year 2000, and conversion from US dollars to euros (1€= 1.25). Last row presents average values for the year 2009 (conversion factor 1.2771).

	Central Europe	**North Europe**	**Scandinavian Europe**
Range US $ (2000)	356-615	123-182	123-255
Average US $ (2000)	485,5	152,5	189,0
€ (2000)	379,3	119,1	147,7
€ (2009)	484,4	152,1	188,6

The second technique is based on Measure 2.2.4. of the Rural Development Programme (RDP 2007-2013), the purpose of which is to provide annual support per hectare of forest to private forest owners or associations thereof, to compensate for the costs and income foregone due to restrictions on the use of forests and other wooded areas, due to the application of Directives 79/409/EEC and 92/43/EEC in a specific area. The measure was designed to be applied to private forests and woodlands included in the Natura 2000 network, within the framework of the implementation of the above provisions and directives. The beneficiary undertakes certain actions to achieve the objectives of Measure 2.2.4 and for each action is entitled to support for the loss of income suffered. The payment ranges from 40 to 200 €/ha and year, with an average support of 120 €/ha and year and up to seven years.

From the comparison of the two alternatives, the value of biodiversity protection shows a noticeable difference (484 to 120 €/ ha and year). This difference, according to MASSIF Deliverable 4 (2010), is due to the fact that in the first technique the estimate is based on the actual estimate of citizens' willingness to pay for biodiversity protection, which includes, in addition to the use value, the non-use value. In contrast, with the second technique (Measure 2.2.4 of the RDP), the value of biodiversity protection is determined by the additional forest management costs required to enhance biodiversity protection. Thus, the value of biodiversity protection is clearly lower.

To estimate the value of biodiversity protection, the 484.4 €/ha and year estimate was adopted. This is an estimate that encompasses all benefits from biodiversity in contrast to the 120 €/ha and year estimate, which aims to compensate for the costs and income foregone due to restrictions on the use of forests.

Using the outcomes of WP5.2, regarding the ecosystem extent map of Peloponnese, we calculated the biodiversity maintenance value, as presented in Table 34.

Table 34. Value of biodiversity maintenance in forests and forested areas of Peloponnese.

Forest and forested areas ecosystems	**Area (ha)**	**N**	**Pb**	**Vb (Euro)**
Forest and woodland - L2 unallocated	265441	0.9	484.4	115721767.35
Riparian forest and woodland - L3	41505.1	0.9	484.4	18094545.96
Temperate, Submediterranean and Mediterranean thermophilous deciduous forest - L3	56422.3	0.9	484.4	24597883.35
Coniferous forest - L2 unallocated	141652	0.9	484.4	61754518.73
Pine forest (excluding mires, non-thermophilous) - L3	42758.9	0.9	484.4	18641170.04
Mediterranean thermophilous lowland pine forest - L3	59340.5	0.9	484.4	25870066.94

Other coniferous forests, excluding plantations - L3	1689.76	0.9	484.4	736667.77
Highly modified coniferous forests, in particular plantations - L3	2788.79	0.6	484.4	810533.93
Broadleaved evergreen forest - L2 unallocated	101246	0.9	484.4	44139040.50
Mediterranean evergreen *Quercus* forest - L3	122530	0.9	484.4	53418370.62
Olea europaea-Ceratonia siliqua forest - L3	115307	0.9	484.4	50269152.53
Heathland and shrub - L2 unallocated	91769.6	0.4	484.4	17781281.57
Sclerophyllous vegetation - L2	488509	0.4	484.4	94653461.21

13.1.2.4 Recreation

To allow us to evaluate all steps in generating monetary valuation data from survey based valuation methods, we designed and implemented a questionnaire with the aim of generating monetary value estimates from primary data. The main purpose was to evaluate the requirements for using such methods in SEEA EA, and to clarify the challenges and risks to using these.

Sample and Data

The questionnaire was designed to provide data for estimating the value of the public good with two different economic approaches, namely revealed and stated preferences. The method of collection was based on an internet form and respondents were provided guidance to complete. Since the interest of this work is in estimating the value of the resource, and not in ascertaining the population value of trips or visits. The main objective is to collect correct travel cost estimates and bid responses from an as representative, as possible, sample of visitors. Of course, one should take into account that there is no prior knowledge of the typical visitor or at least prior surveys of visitors against which we can just sample representativeness. This is of course a widely acknowledged barrier in any study of this type. The questionnaire was addressed to visitors by invitation. The overall invitation/call was addressed to local and regional citizen organisations, NGOs, etc., but completing the questionnaire was open to anyone who had the link.

1. Revealed preference types of methods to estimate use-values of the resource through a Marshallian type demand function and the relevant welfare measures. The construction of the questionnaire signals the use of the individual vs. the zonal travel cost approach. The questionnaire provides basic information concerning the area's number of visits and days within one year (with or without overnight stay). Thus, the essential "quantity" variable can be calculated as the number of visits or days of stay. The questionnaire also provides information supporting the estimation of travel costs and incorporating or not the time cost. For each respondent, the questionnaire also records the mode of transportation so that average travel costs can be constructed ex-post. In addition, the questionnaire records a range of control variables (demographics and socioeconomic characteristics) to reduce the heterogeneity of the estimating equation and a range of perceptional/attitudinal and behavioural characteristics, which can also be used in the equation. Finally, the questionnaire records the subjective perception of certainty as concerns a range of "critical" responses.

2. Stated preference types of methods to estimate the total value of the proposed Natural Capital management programme (mind you, not of the resource). The construction of the questionnaire signals the use of the take-it-or-leave-it type of contingent valuation (single and double bounded) with specific treatment of protest votes (total rejection of the idea to support a natural resource – public good with private funds or other reasons). Contingent valuation (CVM) estimates the value through Hicksian welfare measures, in this case, willingness to pay (WTP) to support the existence and operation of a public good. The questionnaire records the essential variables, i.e., the bid, the response (yes-no), and a series of other control variables.

The two approaches do not measure the same subject or measure it similarly. TCM measures the use value of recreational (amenities) resources. CVM measures the total value of the programme. Assuming that the programme guarantees the upgrade of the resource's status from B to A, one may infer that this is the total incremental value of the resource when upgraded to status A from status B.

There were three objectives to this exercise:

1. To estimate how to apply the TCM to empirically estimate Willingness-to-Pay (WTP) and consumer surplus. Consider alternative variants (e.g., alternative functional forms) of the TCM, review the emerging econometric and statistical issues and test ways to increase robustness and accuracy.
2. To examine how to apply the CV to estimate Willingness-to-Pay (WTP) and consumer surplus empirically. Consider alternative variants (e.g., single versus double-bounded) of the CV, review the emerging econometric and statistical issues and test ways to increase robustness and accuracy.
3. To compare the results from TCM and CV with those from value transfer methodologies. Examine how and why the results differ and propose a procedure to better align empirical results from locally conducted surveys to value transfers.

<u>TCM</u>

TCM is a widely used approach in environmental economics to estimate the **use value** of recreational and other activities outdoors in nature reserves or other places. TCM is founded on microeconomic consumer theory (Marshallian demand) and welfare economics, particularly concepts from revealed preference theory and demand estimation. In TCM, visitors indicate their willingness to pay (WTP) through travel expenses and frequency of visits. TCM assumes that the travel cost per visit is the "price" for accessing the site, and the number of visits per specific period (usually one year) acts as demand ("quantity"). The demand curve for visits is estimated based on how visit frequency decreases as travel costs increase. In other words, TCM estimates a "demand' function of the following type:

Travel frequency (trips or days) = f (Travel Cost, Income, Time Cost, Demographics)

<u>CVM</u>

CVM is grounded in the Hicksian microeconomic and welfare framework, which defines economic value as the maximum amount an individual is willing to pay (WTP) to obtain a benefit or the minimum amount they require in compensation (WTA) to accept a loss.

Total Economic Value = Use value + Non-Use Value

Non-use values may include option values (future potential use), existence values (values from knowing that the resource exists), and bequest values (values for preserving the resource for future generations as a bequest).

The design proposed by the questionnaire is a dichotomous single (q.21) and double-bounded design (q.22 and q.23 follow up q.21). The weakness in the design is that respondents are offered only one, exactly the same, bid. In theory, with a binary response (Yes/No), we can estimate the probability of accepting the bid using logistic regression (Logit) or probit regression (Probit) under the form:

P(Yes) = f (Bid, Other Variables)

Where P(Yes) is the probability of accepting the bid, Bid is the variable depicting the amount respondents were asked to pay and respond with a “yes” or a “no.” Other variables (optional) could include income, education, or environmental attitudes, as discussed in the TCM above. The functional form may consist of a logit or probit specification. Once the logit or probit coefficient from the model is estimated, we can compute the median (for logit) or the mean (for bounded) WTP.

Data cleaning and pre-estimation procedures

Data cleaning procedures started from 85 collected questionnaires from google forms of which 11 questionnaires failed to pass the screening question “Have you visited the area during the last 12 months?”. This screening question was necessary because all basic estimation variables including the number of visits, travel costs, time spent, etc., were measured on visits undertaken during the last 12 months, i.e., the reference period. Of the remaining 74 questionnaires, 6 had typing errors in crucial variables for the analysis, (e.g., 909 trips to the area) and almost all variables were missing. From the remaining 68 questionnaires there were some obvious outliers or misunderstandings with people having spent more than 8,000 euros in travel costs. This together with pairwise missing values (i.e., missing number of trips or travel cost) raised the number of excluded cases to 27 and the remaining clean questionnaires to 41.

The table below shows descriptive statistics of the number of trips and travel costs variables while the graph below shows the very simple relation between the travel cost per trip and the number of trips in a 12 month period.

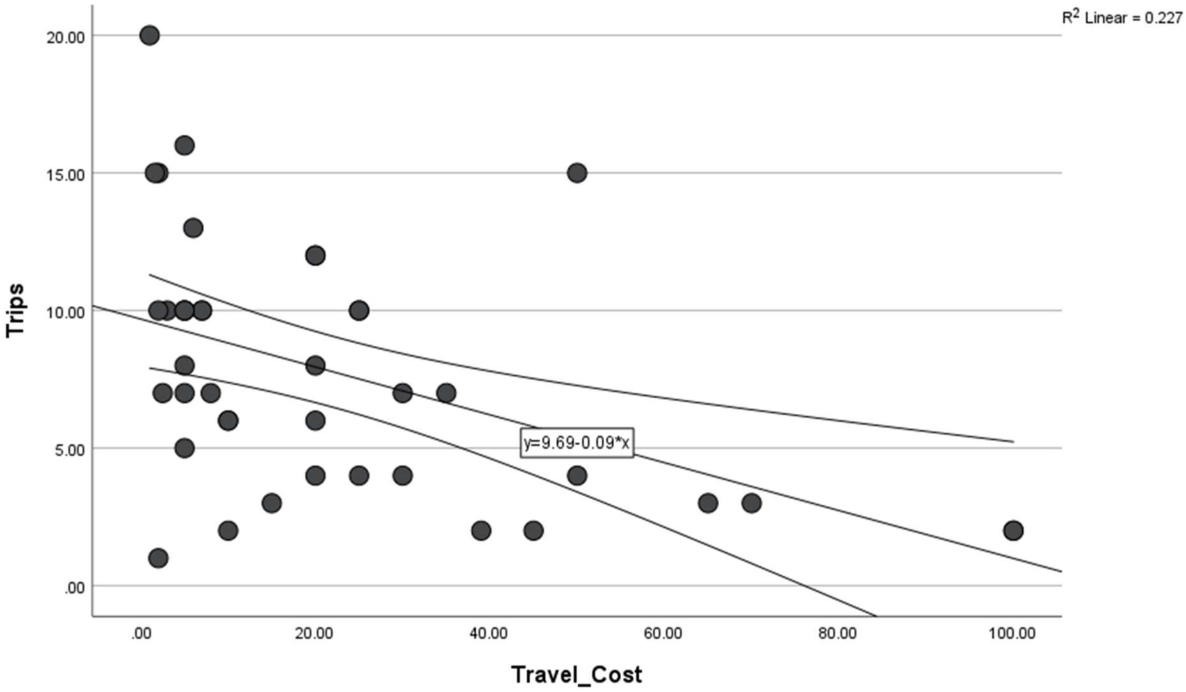

Figure 17. Scatterplot with linear regression function, showing the correlation between travel cost and number of trips.

Table 35. Descriptive Statistics of the travel cost and number of trips variables.

	N	Minimum	Maximum	Mean		Std. Deviation	Variance
	Statistic	Statistic	Statistic	Statistic	Std. Error	Statistic	Statistic
Number of Trips last 12 months	41	1.00	20.00	7.7561	.71549	4.58138	20.989
Travel cost per trip (euros)	41	1.00	100.00	22.2220	3.92408	25.12638	631.335

Fitting the TCM

The Table below shows a series of TCM models fitted using SPSS V.29. The models include three specification forms, of the number of trips-travel cost relationship in linear-linear, log-linear and log-log where the corresponding variables are in different transformations.
The linear model (linear-linear) assumes a constant decrease in visits as travel cost increases but may overestimate demand at high travel costs. The semi-log (log-linear) model assumes a diminishing marginal impact of travel cost and always results in high Consumer Surplus estimates since it suggests visitors are less sensitive to cost increases. Finally, the double-log (log-log) assumes an elastic demand response, meaning visits drop sharply with higher costs,

and provides the lowest CS estimate, implying visitors are highly sensitive to price changes. The third functional form is closer to the behaviour of the Greek consumer after the increasing energy due to the Ukrainian war.
Besides the conventional models we also attempted to incorporate uncertainty estimates by adapting the travel cost and the number of trips to a high, normal and low uncertainty response by respondents. The estimated models include high or low uncertainty in each one of the variables separately and simultaneously for both variables.

The table shows that results, especially the coefficient of the travel cost variable used for calculating the consumer surplus is very robust, i.e., its numerical estimate does not depend on the functional form, the uncertainty or the inclusion of covariates. The estimated coefficient of the log travel cost ranges from a low of -0.223 when the number of trips is highly uncertain (towards underestimation) and an average value of around -0.300.

Table 36. TCM models fitted to the data, showing its constant, TC coefficient and R^2.

Model	Constant	TC coefficient	R^2
Regular models			
Lin-Lin	9.484 (10.820)	-0.086 (-3.254)	0.209
Ln-Lin	2,166 (16,723)	-0,015 (-3,805)	0,271
Ln-Ln	2,535 (11,183)	-0,283 (-3,423)	0,231
Uncertainty – underestimated travel cost			
Lin-Lin	9,516 (10,787)	-0,112 (-3,446)	0,243
Ln-Lin	2,160 (16,286)	-0,018 (-3,675)	0,273
Ln-Ln	2,503 (11,325)	-0,291 (-3,405)	0,244
Uncertainty – overestimated travel cost			
Lin-Lin	9,593 (10,995)	-0,100 (-3,611)	0,261
Ln-Lin	2,162 (16,411)	-0,015 (-3,734)	0,279
Ln-Ln	2,560 (11,008)	-0,297 (-3,464)	0,250
Uncertainty – underestimated trips			
Lin-Lin	8,544 (10,477)	-0,075 (-2,724)	0,171

Ln-Lin	2,064 (15,519)	-0,013 (-2,971)	0,201
Ln-Ln	2,337 (9,566)	-0,223 (-2,425)	0,144
Uncertainty – overestimated trips			
Lin-Lin	9,441 (9,827)	-0,077 (-2,364)	0,134
Ln-Lin	2,158 (15,310)	-0,013 (-2,695)	0,172
Ln-Ln	2,440 (9,531)	-0,222 (-2,308)	0,132
Conf. Trips Low – Conf. Cost Low			
Lin-Lin	8,421 (10,415)	-0,089 (-2,968)	0,206
Ln-Lin	2,044 (15,347)	-0,015 (-3,164)	0,233
Ln-Ln	2,333 (9,795)	-0,245 (-2,646)	0,175
Conf. Trips High Conf. Cost High			
Lin-Lin	9,465 (10,935)	-0,091 (-3,334)	0,246
Ln-Lin	2,154 (15,856)	-0,14 (-3,242)	0,242
Ln-Ln	2,518 (9,869)	-0,266 (-2,816)	0,194

Thus, the estimated consumer surplus for the log-log conservative estimate model is: $CS = \frac{V0}{\beta c}$ where V0 is the lowest travel cost among those visitors taking part in the estimation process and βC the coefficient of the travel cost. In our case ln(V0) = [2.535+[(-0.283) *ln(2euro)]] =2.34 and so the predicted visits at the minimum V0 is 10.38. As such, CS is -10.38/-0.283=36.68 **euros per visit.**
The use value of the resource for recreation is derived by multiplying the total number of visits accepted by the area from all travellers, by 36.68 euros.

Scaling up the consumer surplus estimates to all visitors of the area provides an estimate of the total value of the resource from the recreation service. For this analysis, we make the following assumptions. Visitors to the area are supplied by the population in the regional units of Ilia and Achaia as well as the tourists that stay in local lodges. We assume recreation participation rates among the locals (residency at less than 50-60 km from the resource) aged 18-75 between 20% and 40% and a decay function of 10% for every 10 Km away from the edges of the site. In addition, we consider domestic and foreign tourist in the hotels of the area who undertake the extra cost for visiting the resource from their hotel or other type of accommodation. According to these assumptions and the respective population in Ilia and Achaia, the population contribution to visitors of the resource are the following:

Table 37. Population contribution to visitors of the resource in the Greek test site, their number of visits per year, their consumer surplus per visit and total consumer surplus.

Population source	Population	Participation (rate+decay)	Visits/year	Consumer Surplus per visit	Total Consumer Surplus
Ilia	100,000	40%	7.76	36.68	11,385,472
Achaia	200,000	10%	7.76	36.68	5,692,736
Domestic and Foreign Tourists	15,000	50%	1	36.68	275,100
Total	315,000				17,353,308

One can of course adjust these estimates by applying sensitivity analysis to the participation functions and the visit decay function.

Since the test site covers 96,776 hectares, the value presented in Table 37 can then be converted to a value of **179 euros per hectare per year**.

Fitting the CVM

Having only a single, constant bid, of 5 euros we approach the estimation of CVM through a Turnbull estimator. First, we fit a logit model to estimate which of the socio-demographic or attitudinal factors affect the probability of accepting the bid.

$P(WTP < t) = CDF_t$, i.e., the cumulative distribution function at t, usually noted as $F_{WTP}(t)$. In a binary logistic framework, the cdf is:

$$F_{WTP}(t) = \frac{1}{1 + e^{-(b0+b1X1+b2X2+\ldots)}}$$

Likewise, the survival function (or complementary cumulative distribution function, CCDF) is defined as $S(t) = 1 - CDF_t = P(WTP \geq t)$. In a binary logistic (yes-no) contingent valuation framework with only one bid offered to all respondents, the application of the Turnbull estimator is significantly simplified because there are no bid intervals to estimate. The Turnbull estimator, typically a nonparametric method for estimating the willingness to pay (WTP) distribution across multiple bid levels, reduces to a basic proportion-based calculation when only one bid is used and the WTP is censored into two intervals the $(0, t_1)$ for "no" responses and $[t_1,\infty)$ for "yes" responses.

In our case, the single bid is 5 euros ($t_1 = 5$). We assume a lower bound of $t_0 = 0$ implying that everyone would accept a zero bid and an upper bound of $t_2 = \infty$. We could have used another, user-specified maximum WTP, such as 20 euros, but its responses are almost all "no", i.e., has no variability, and we are not sure if they were understood by respondents. For example, respondents who denied the 5 euro bid accepted the 20 euro bid, etc. Thus, the WTP is divided into two intervals: $(0, t_1)$, implying $(WTP < t)$, i.e., those whose response is "no" to the 5 euros bid, and $[t_1, \infty)$, implying $(WTP \geq t)$, i.e., those whose response is "yes" to the 5 euros bid.

We fit a logit model as shown in Table 38. The data did not provide any relations between the dichotomous bid variable and any economic variable such as income, travel cost, etc. The only two variables with a statistically significant contribution are a dummy capturing the respondents' employment status and a quasi-continuous variable reflecting the respondents' perceptions of the area's attractiveness. From the logit model we predicted the probabilities for each one of the respondents and estimated the CDF and Survival functions. So, there was no need to look for empirical frequencies.

For the first interval, the lower limit, the expected WTP is a conservative estimate calculating the area under the CDF, i.e., the sums of the estimated "no" probabilities by the 5 euro bid. The average predicted "no" probability is 0.292 implying that the average WTP of the lower bound interval is 1.46 derived from:

$$E(WTP \in [0, t_1]) = t_1 * P(WTP < t) = t_1 * F_{WTP}(t) = t_1 * \left(1 - p(\widehat{yes})\right) = 5 * 0.292$$

For the second interval, which is unbounded from the right, the average predicted "yes" probability is 0.707 implying that the average WTP of this bound is 3.535 derived from: $E(WTP \in [t_1, \infty)) = t_1 * P(WTP \geq t) = t_1 * S(t) = t_1 * (1 - CDF_t) = t_1 * p(\widehat{yes}) = 5 * 0.707$

We cannot approximate a mean WTP, i.e., is undefined, because the tail [t1, ∞) contributes an infinite area unless constrained. We could use the 20 euro bid to constraint it, but responses to this question were very problematic.

The estimation of the consumer surplus (CS) in the case of a constant bid contingent valuation is tricky. In our case, the CV exercise is hypothetical and respondents do not actually pay the bid, and so CS can be equated to the total expected WTP for the population. We cannot use the above estimated WTPs because these assume that "yes" respondents have WTP exactly at $t_1 = 5$, and "no" respondents have a WTP of 0. This is highly uncertain because the interval 0 to 5 euros is large to assume that a bid in between, e.g. 2 euros, would not attract some "no" respondents to a "yes" answer, especially if we take into account that there are respondents with travel cost per trip costing less than 2 euros. If the CV assumes respondents who say "yes" would pay the bid, the CS is the difference between their WTP and the bid amount for those willing to pay. However, since WTP is not directly observed, we cannot estimate CS.

Table 38. Results from estimating the logit CVM coefficients.

Variables	B	S.E.	Wald	Sig.	Exp(B)
Dummy variable for employment (1 if fully employed, 0 for any other employment status) (question q32)	2.875	1.035	7.713	0.005	17.732
Site is attractive (1-5) ordinal as quasi-continuous variable (question q8)	1.494	0.674	4.918	0.027	4.455
Constant	-5.818	2.644	4.841	0.028	0.003

$E(WTP)=\sum(Bid\times(1-P(Yes)))$ (the equation is not needed any more, but I leave it here to avoid deleting the comments).

Comparing these CV proxies of WTP to those estimated by TCM, we realise that they are significantly different. This is due to the fact that the CV attempted to measure the value of the incremental change resultant from the execution of a series of infrastructure works for conservation. The TCM measured the actual use value of the resource.

13.1.3 Ireland

13.1.3.1 Recreation

In Ireland, a version of the survey described in section 8.1.2 for the Greek test site was implemented in parallel. Adaptations were made to the site description and bid values, but the design was similar to allow for comparison of implementation across different countries in Europe.

The surveys were carried out in the field using the SurveyMonkey platform on mobile devices to facilitate mobility and reduce paper waste, and for ease of data analysis. A small number of paper copies of the survey were at hand at both sites in case of weak mobile signal coverage; any paper surveys completed were then inputted to the online survey later. To conduct the surveys, surveyors approached people in the main car park of each site, and preferentially selected people who were exiting the woodlands in order to get impressions of each site immediately after a visit. The car parks in both cases were located to the main entrance to the recreational area, so it was deemed unlikely people travelling to the sites by foot or other transport would be missed. A paper flyer explaining the reasons for the survey, with some information about both FOR-ES and SELINA, was offered to all survey respondents.

The survey was divided into three sections; section 1 gathered information on social and cultural values associated with each site, section 2 focused on a willingness-to-pay exercise, and section 3 gathered anonymous demographic data. Surveys were initially planned for the months of October and November 2024, but were postponed due to persistent wet weather. Surveys were rescheduled for December 2024 and January 2025, however two significant storms caused further delays until February. The storms caused considerable damage in both woodland sites, with many trails being closed for safety reasons due to fallen trees. Some respondents at both sites mentioned that these closures negatively affected their enjoyment of the site on the day of the survey. Due to these unforeseen delays, there was no time to estimate monetary values in this test site. However, the survey design and data collection exercise provided valuable input to the evaluation of valuation methods.

13.1.4 La Reunion

13.1.4.1 Recreation

Aim and objective

The primary objective of our study was to conduct a socio-economic assessment of shark mitigation measures in Réunion Island and for or through this assessment, to reveal the value (following different valuation methods) of the recreation ecosystem service of immersed sea-uses in the open sea in Réunion Island.

The initial study was developed in 2022 in partnership with the students of the Master 1 in Economy at the University of La Réunion under the supervision of Dr. Fanny Alivon et Dr. Olivia Ricci, based on a public command formulated by Dr. Erwann Lagabrielle (University of La Réunion, Shark Security Centre).

Table 39: Costs and benefits of shark risk mitigation in La Réunion

Costs	**Benefits**
Impact of fishing on ecosystem and trophic chains	Safety gains (lives saved, accidents avoided)
Possible degradation of the coral reef due to shark net installation	Recreational ecosystem services: recovery and increase of immersed sea uses in the open sea (surfing, diving, sailing…)
Installation and maintenance of equipment (nets, investment)	Recovery and increase in domestic and international tourism
Research and development costs	Creation of direct jobs (lookouts, maintenance) and indirect jobs (restaurants, hotels, etc.)
Human resources (salaries)	

Shark risk public policy context

The beginning of the shark crisis can be traced back to 2011. Since 2014, the French State, through the Centre de Sécurité Requin (CSR, created in 2016) has carried out a shark risk mitigation public policy composed of public policy measures (Table 40). Our analysis is therefore positioned as an intermediate evaluation. This type of evaluation is conducted midway through policy implementation, after a certain period of activity, and prior to the end of the public policy program. Thus, this analysis is made of ex-ante (prior to the implementation of a mitigation measure) and ex-post (after the implementation of the measure) evaluation. Table 40 describes the implementation of the public policy since it started in 2013.

Table 40: Measures of the shark risk mitigation public policy from 2014 to present (2024) in La Réunion.

Public policy measure	**Description**	**Period**
Prefectural decree	A decree prohibiting swimming and certain water sports, such as surfing and bodyboarding, within 300 meters of the coastline of La Réunion, except in designated areas protected by shark nets, within the lagoon or in ZONEX (n=10). ZONEX are zones where experimental risk reduction devices are tested, and wave-based activities are authorized under restrictive conditions.	2013-present
Shark fishing control program	The fishing of potentially dangerous sharks for the purpose of reducing the risk of shark bites can be also referred to as "control fishing". The implementation of this shark fishing program is based on the assumption that removing sharks from a given site reduces the hazard, and therefore the risk of dangerous physical encounters between sharks and humans in the vicinity of that site. Control operations include lethal horizontal longline and SMART drumline devices. Almost all fishing operations are conducted on the West and South coast.	2014-present
Shark exclusion nets	In 2015, two large shark exclusion nets totalling ~1 km length were placed at two locations to allow surfing and bathing activities, but are no longer operational since 2017 due to unsustainable maintenance costs. Yet, 3 smaller nets for bathing remained operational (Boucan-Canot, Saint-Gilles-les-bains, Etang-Salé). A fourth shark exclusion net is operational in the bay of Saint-Paul from 2024. All exclusion nets are within a ZONEX.	2015-present
Water patrol/ Vigie requin	Water patrol (i.e. Vigie Requin) is an in situ human-based (either immersed or jet ski-based) shark detection and alert system, primarily used to secure surfing (i.e. wave-based activities) outside of the lagoon. The system is operational at two moving locations in Saint-Paul and in Saint-Leu.	2014-present

Local study sites

The study was conducted at two local bathing-surfing sites (Boucan-Canot and Trois-Bassins) and at one potential bathing site (Saint-Paul). An objective of this study was to determine whether it was beneficial to implement a shark exclusion net in Saint-Paul.

Table 41: Local study sites in La Réunion.

Site	Description	Illustrative picture
Boucan-Canot	Boucan Canot is a popular beach known for its white coral sand and clear waters. Unlike lagoon beaches, it lacks reef protection, making it suitable for experienced swimmers and surfers but also subject to shark risk. Safety measures like a natural rock pool and shark nets have been installed since 2015, while a lively promenade with cafés and restaurants attracts both locals and tourists. It is a ZONEX.	
Trois-Bassins	The Trois-Bassins surf spot is renowned for its consistent reef break waves that attract surfers. In response to shark risk exposure, the Vigie Requin system has been implemented in the local ZONEX since 2019.	
Bay of Saint-Paul	Beach of Saint-Paul, located in the bay of Saint-Paul on Réunion Island's west coast, is a long black sand beach along the city waterfront. Since 2025 a shark net-protected area allows supervised bathing in a designated ZONEX. Note: There was no shark net and no bathing at the time this study was conducted.	

Questionnaires

We chose to distribute our questionnaires to all individuals over the age of 18 residing on the island of La Réunion, as well as to tourists who might assign significant value to the sites concerned. For the Bay of Saint-Paul, we aimed to reach as many residents of the city of Saint-Paul as possible, as they are considered the most directly affected.

The surveys were conducted using two different approaches: on-line and on-site. For the online surveys, we chose to use social media by posting the questionnaires on the Facebook pages of “VRR Vigies Requins Renforcées,” the Réunion Surf League, and the University of La Réunion, in order to reach the largest possible audience.

Parent population and sampled populations at the three sites

Through our online questionnaires and on-site visits, we were able to collect a sample consisting of 899 respondents during the period of January-February 2021 : Boucan-Canot (n=394, 64% on-line, 36% on-site), Trois-Bassins (n=352, 71% on-line, 29% on-site) and Saint-Paul (n=152, 4% on-site, 96% on-line).

In the context of this economic analysis, understanding the composition of the parent population is essential to ensure the reliability and representativeness of the survey results. The parent population refers to the total group from which the sample was drawn, and its composition includes key demographic and socioeconomic characteristics such as gender, age, place of residence (sub-region on the island : East, West, South, North - or external), education level, and socio-professional category (SCP). Identifying these attributes allows for a better interpretation of the willingness to pay (WTP)/transport cost method (TCP) estimates and helps ensure that the results can be generalized to the broader population. Moreover, knowing the structure of the parent population enables the application of weighting procedures to correct for any sampling biases and improves the accuracy of policy recommendations based on the findings.

In our case, the parent population consists of the residents of La Réunion (n = 861,210), combined with the cohort of tourists who visited the island during the two-month period in which the questionnaire was administered (n = 20,451).

Table 42. Variables describing the parent population and sampled populations of the survey.

Variable	**Parent population proportion**	**Boucan Canot**	**Trois-Bassins**	**Saint-Paul**
n =	881661	394	352	154
Gender (age>18)				
Female	46.73%	66.24%	51.14%	51.95%
Male	53.27%	33.25%	48.86%	47.40%
Other		0.51%		0.65%
Chi² test		15.83	0.781	1.242

Variable	Parent population proportion	Boucan Canot	Trois-Bassins	Saint-Paul
p-value		p = 0.0001	p > 0.05	p > 0.05
Socio-professional category (SPC)				
Craftsmen, shopkeepers, business owners	5.70%	4.06%	7.85%	6.49%
Farmers	1.50%	0.25%	0.00%	0.00%
Executives and professionals	6.80%	6.80%	19.89%	12.34%
Intermediate professions	14.90%	6.60%	7.39%	2.60%
Employees	25.10%	21.32%	23.86%	24.03%
Manual workers	20.80%	0.00%	0.28%	0.65%
Retirees	11.30%	1.27%	0.85%	0.00%
Other individuals without professional activity	13.90%	51.52%	39.77%	53.90%
No answer		0.51%		
Chi² test		163.16	109.65	162.22
p-value		< 0.001	< 0.001	< 0.001
Place of residence				
North	24.03%	32.74%	23.30%	29.22%
West	23.91%	25.38%	34.38%	39.61%

Variable	Parent population proportion	Boucan Canot	Trois-Bassins	Saint-Paul
South	35.32%	28.17%	25.00%	20.13%
East	14.42%	7.36%	4.55%	5.19%
Outside La Réunion (tourists)	2.32%	6.36%	12.78%	5.83%
Chi^2 test		6.35	61.53	29.19
p-value		0.174	< 0.001	< 0.001
Age				
18-24 years	9%	51%	39%	50%
25-31 years	9%	15%	25%	16%
32-38 years	9%	7%	9%	7%
39-45 years	9%	10%	11%	8%
46-52 years	10%	9%	9%	9%
53-59 years	10%	5%	5%	6%
60+ years	19%	3%	2%	3%
Chi^2 test		156.22	103.78	153.25
p-value		< 0.001	< 0.001	< 0.001

The chi-squared test performed on gender distribution across the three sites—Boucan Canot, Trois-Bassins, and Saint-Paul—shows contrasting results in terms of representativeness. At Boucan Canot, the observed gender distribution significantly differs from that of the general population ($Chi^2 = 15.83$, $p = 0.0001$), with a substantial overrepresentation of women (66.24% compared to 46.73% in the parent population). This suggests a possible gender-specific usage pattern of the site or sampling bias in the responses collected. In contrast, for Trois-Bassins and Saint-Paul, the chi-squared values are not statistically significant ($p > 0.05$), indicating that the proportions of male and female respondents are consistent with those of the general population. These findings imply that, while gender distribution is balanced at two of the sites, the results from Boucan Canot should be interpreted with caution, as the overrepresentation of women may influence the perception of risk and willingness to pay associated with shark risk mitigation measures.

The chi-squared analysis conducted on the distribution of socio-professional categories (SPC) across the three study sites—Boucan Canot, Trois-Bassins, and Saint-Paul—reveals statistically significant deviations from the structure of the general population. In all three cases, the p-values are below 0.001, indicating that the observed distributions differ markedly from expected proportions based on regional reference data. At Boucan Canot, individuals without professional activity are heavily overrepresented, accounting for over 51% of respondents compared to just 13.9% in the reference population. Conversely, categories such as manual workers and retirees are significantly underrepresented. A similar pattern is observed at Saint-Paul, where over 53% of respondents fall into the "no professional activity" category. Meanwhile, Trois-Bassins shows a notable overrepresentation of executives and professionals (19.89% vs. 6.8% expected) and an almost complete absence of farmers, manual workers, and retirees.

The chi-squared tests for geographic origin across the three sites—Boucan Canot, Trois-Bassins, and Saint-Paul—show statistically significant differences compared to the reference population ($p < 0.001$ for all sites). This indicates that the spatial distribution of respondents is not representative of the island's overall population. In particular, tourists (non-residents) are notably overrepresented at all sites, especially at Trois-Bassins (12.78% vs. 2.32%). These deviations suggest that visitor origin should be considered when interpreting results related to behaviour, perception, and willingness to pay.

The chi-squared test results for age distribution across Boucan Canot, Trois-Bassins, and Saint-Paul indicate significant deviations from the age structure of the general population of Réunion ($p < 0.001$ for all sites). In each case, the proportion of young adults, particularly those aged 18–24, is highly overrepresented—reaching 51% at Boucan Canot and 50% at Saint-Paul, compared to only 9% in the parent population. Conversely, older age groups, especially individuals aged 60 and above, are heavily underrepresented across all sites. These findings suggest that beachgoers and survey respondents tend to be significantly younger than the general population, which may influence their risk perception, recreational preferences, and willingness to pay for shark risk mitigation measures.

Activities on site

Depending on the area, users can engage in the following recreational activities:

- Swimming, snorkelling, or fins-mask-snorkel activities
- Beach activities : walking, picnicking, racquet games, ball games, beach volleyball, paddle tennis
- Nautical activities : kitesurfing, windsurfing, or any other sail-based activity
- Surfing : including bodyboarding, bodysurfing
- Diving

Due to the location of the beaches on the west coast and the island's tropical climate, recreational activities can be enjoyed year-round, every day. They are especially popular on weekends and during school holidays, both local (Réunion) and mainland (France). Since a respondent can practice none, one or multiple activities. Respondents were asked to identify their main activity on the site.

Purpose for visitation

Trips to the three sites are often made for multiple purposes. For locals, these visits are frequently combined with picnics or time spent with friends enjoying the beach and playing games, for example (first case). Similarly, tourists often visit several places on the island within the same day (second case).

To account for these multi-purpose trips, we asked individuals about their primary and secondary reasons for visiting the sites of Boucan Canot, Trois Bassins, and the Bay of Saint Paul. In the first case, which involves single-purpose visits, our data is based on full days of visitation. In the second case, we focus exclusively on multi-purpose visits.

Hypothesis regarding visits, gender, age and SPC

We subsequently tested the following hypotheses using the Chi-square test of independence. This test allows us to determine whether two categorical variables are potentially related or not. If the critical value (p-value) is greater than 5% (i.e., 0.05), then the hypothesis is considered to be true; otherwise, it is considered to be false.
On site-questionnaire

- H1: The number of visits depended on gender.
- H2: The number of visits depended on age groups.
- H3: The number of visits depended on individuals' socio-professional category (SPC).
- H4: If risk mitigation measures are implemented, the frequency of site visits will depend on gender.
- H5: If risk mitigation measures are implemented, the frequency of site visits will depend on SPC.
- H6: If risk mitigation measures are implemented, the frequency of site visits will depend on the age groups to which individuals belong.
- H7: If risk mitigation measures are implemented, the frequency of visits will be influenced by whether or not individuals feel concerned by the risk mitigation measures.

On-line questionnaire

- H8: In general, site visitation depends on individuals' gender.
- H9: In general, site visitation depends on individuals' SPC.
- H10: In general, site visitation depends on the age groups to which individuals belong.
- H11: In general, whether individuals feel concerned by the risk mitigation measures influences the frequency of site visits.
- H12: There is a link between the importance individuals attach to the risk mitigation measures and the frequency of site visits.

Specific hypotheses for the Saint-Paul site sample on-line only (the risk mitigation measure was hypothetical at the time the study was conducted):

- H13: If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' gender.
- H14: If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' age group.
- H15: If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' income.
- H16: If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' SPC.
- H17: If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' education level.
- H18: There is a link between whether or not individuals feel concerned by the risk mitigation measures and the incentive to visit the site (if the risk mitigation measures are implemented).

Table 43. Hypothesis tested. If the critical value (p-value) is greater than 5% (i.e., 0.05), then the hypothesis was considered to be true; otherwise, it was considered to be false. Note that the Saint-Paul site had specific hypothesis.

Hypothesis formulation		Boucan-Canot	Trois-Bassins	Saint-Paul
On-site questionnaire				
H1	The number of visits depended on gender	TRUE	TRUE	FALSE
H2	The number of visits depended on age groups	FALSE	FALSE	FALSE
H3	The number of visits depended on individuals' socio-professional category (SPC)	TRUE	TRUE	FALSE
H4	If risk mitigation measures are implemented, the frequency of site visits will depend on gender	FALSE	FALSE	-
H5	If risk mitigation measures are implemented, the frequency of site visits will depend on SPC	TRUE	FALSE	-
H6	If risk mitigation measures are implemented, the frequency of site visits will depend on the age groups to which individuals belong	TRUE	FALSE	-
H7	If risk mitigation measures are implemented, the frequency of visits will be influenced by whether or not individuals feel concerned by the risk mitigation measures	TRUE	TRUE	-
On-line questionnaire				
H8	In general, site visitation depends on individuals' gender	FALSE	TRUE	FALSE
H9	In general, site visitation depends on individuals' SPC	FALSE	TRUE	FALSE
H10	In general, site visitation depends on the age groups to which individuals belong	FALSE	TRUE	TRUE
H11	In general, whether individuals feel concerned by the risk mitigation measures influences the frequency of site visits	TRUE	TRUE	TRUE
H12	There is a link between the importance individuals attach to the risk mitigation measures and the frequency of site visits	FALSE	TRUE	FALSE
Specific hypotheses for the Saint-Paul site sample on-line only (the risk mitigation measure was hypothetical at the time the study was conducted):				
H13	If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' gender	-	-	FALSE
H14	If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' age group	-	-	FALSE
H15	If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' income	-	-	FALSE
H16	If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' SPC	-	-	FALSE
H17	If risk mitigation measures are implemented, the incentive to visit the site will depend on individuals' education level	-	-	FALSE
H18	There is a link between whether or not individuals feel concerned by the risk mitigation measures and the incentive to visit the site (if the risk mitigation measures are implemented)	-	-	TRUE

TCM questionnaire

The study sites for the TCM are the 1 km stretch beaches of Boucan-Canot and Trois-Bassins. A questionnaire including TCM-specific questions is deployed for each site. The bay of Saint-Paul is excluded as it was not yet a bathing site at the time of the study (no shark exclusion net)

In questions 4 to 7, the respondent reveals how many daily trips (q7) they have had to the area during the last 12 months.

In question 8 and 9 the respondent indicates its main activity (q8) and secondary (q9) activity on the site. This is used to calculate the cost of practising the activity.

In question 10 to 13, the respondent estimates its expenses for the activity (equipment) and for the stay (restaurant, etc.).

The questionnaire also records if the respondent went on site exclusively to practice the activity (q.14) and if he uses an alternative site (q.15).

In question 16 to 22, the respondents provide information used to calculate the travel cost (way of travel, type of car, etc.). In the case of external tourists, the airfare cost (generally France mainland-Réunion) is integrated in the calculation. Time spent on travel is provided by the respondent in q.21, while the distance is provided in q.22.

Other variables are recorded in the questionnaire and capture awareness of shark mitigation measures (q.23), perceptions of the importance of those measures (q.25).

In question 30 and 31 the respondent is asked if in the region, he visits other sites for the same activities (q.30) and a list of sites is proposed (q.31)

TCM calculation method

- ***One-way transport cost:*** Transport costs (TC) are calculated based on the distance (D) between the visitor's home and the study site, taking into account the vehicle's horsepower and the associated cost per kilometre (CM), as well as the number of people (P) in the visitor group. For pedestrians and cyclists, the one-way transport cost is considered to be 0. The cost per kilometre, varied accordingly with fiscal horsepower from 0.38 € per kilometre (for HP <= 4) to 0.60 € per kilometre (for HP <10).
- ***Equipment cost***: In order to engage in recreational activities, equipment is sometimes required. Users may incur expenses for renting or purchasing equipment. Other users may already own their equipment, in which case this cost is zero.
- ***Accommodation cost***: Accommodation costs apply to tourists whose trip to the island is solely motivated by the desire to engage in their primary recreational activities at sites protected by shark mitigation systems. Per night hotel cost: 120€.
- **Airfare cost (for tourists):** Average cost of one way/return trip Paris-Réunion in February-March 2021 per company. French Bee: 814€, Air France company: 855€

TCM results

Table 44. Willingness to pay per trip per site per category of use along two survey type (on-line and on-site).

Site	Category of activity	Average transport cost per visit (€)	Average number of visits per year	Total respondents	Annual transport cost (€)
Boucan-Canot (on-line)					
	Swimming	27	20	94	51335
	Beach activities	11	17	14	2561
	Nautical activities	270	40	4	42887
	Other activities	16	21	29	9639
	No response			1	
	Total			142	106422
Boucan-Canot (on-site)					
	Swimming	11	8	124	11069
	Beach activities	16	9	24	3449
	Nautical activities	66	12	7	5538
	Other activities	11	8	13	1110
	No response			84	
	Total			252	21166
Trois-Bassins (on-line)					
	Swimming	12	4	45	1993
	Beach activities	7	2	21	340

Nautical activities (surfing excluded)	30	21	8	5188
Surfing	109	55	64	385648
Other activities			1	
No response			111	
Total			142	393169
Trois-Bassins (on-site)				
Swimming	97	14	41	54471
Beach activities	16	3	12	619
Nautical activities (surfing excluded)	12	24	9	2554
Surfing	210	28	33	194590
Other activities	7	32	7	1459
No response			111	
Total			142	253693

Boucan Canot - The recreational value of the Boucan Canot site was estimated using the travel cost method. Users engaging primarily in swimming and nautical activities recorded the highest travel costs, particularly due to the inclusion of travel and accommodation expenses for tourists who visited the site specifically to practice their activities within the secured zone. Surfing was considered marginal at Boucan-Canot, and included with other nautical activities. The online-survey yielded an estimated travel cost per trip of €27 for swimming, €11 for beach activities and €270 for nautical activities. The onsite-survey yielded an estimated travel cost per trip of €11 for swimming, €16 for beach activities and €66 for nautical activities.

Trois-Bassins - Surfing and swimming are the two main activities of the panel who answered. Travel costs per trip were higher for surfers, reflecting their strong preference for this shark-secured site. The online-survey yielded an estimated travel cost per trip of €12 for swimming, €7 for beach activities, €30 for nautical activities (out of surfing) and €109 for surfing. The onsite-survey yielded an estimated travel cost per trip of €97 for swimming, €16 for beach activities, €12 for nautical activities (out of surfing) and €210 for surfing specifically. Highest travel costs can be explained by the inclusion of travel and accommodation expenses for tourists who visited the site specifically to practice their activity within the secured zone.

Saint-Paul - The TCM was not applied to the site of Saint-Paul since the shark net was only a project at the time the survey was conducted.

CVM methods

Our main objective was to provide key insights into individuals' willingness to pay (WTP)—that is, the amount people are ready to contribute financially for the implementation or maintenance of safety measures such as shark lookout systems and protective nets. The underlying principle is that the value an individual assigns to these measures corresponds to the amount they are willing to pay to benefit from them.

To assess WTP, we used different payment scenarios depending on the study site (Boucan-Canot, Trois-Bassins and Saint-Paul). For Trois-Bassins, the payment method was adapted according to the respondents' primary place of residence. For local residents, we proposed a voluntary increase in existing membership fees, with the collected funds being dedicated solely to the improvement, maintenance, and sustainability of the shark lookout system. In contrast, for tourists visiting the island, we suggested a per-session (daily) fee for surfing activities, payable on-site. Differentiating between these two population groups enabled us to examine whether there are significant differences in WTP between local residents and non-residents. For the Boucan Canot and Saint-Paul Bay sites, we proposed two alternative payment mechanisms: an annual membership or a pay-per-visit entrance fee. These options reflect the diversity of user profiles and recreational practices across the sites, and provide a basis for evaluating public support for the implementation of shark risk mitigation strategies.

We applied the contingent valuation method to collect WTP data. This method allows for the direct estimation of individuals' perceived value of environmental goods or services—in this case, the safety measures—based on hypothetical but realistic scenarios. It assumes that respondents provide truthful answers when presented with these scenarios.

WTP values can vary according to multiple factors. These include the specific site concerned, as well as socio-demographic variables such as gender, age, income, education level, socio-professional category, and place of residence (including whether or not the respondent lives in Réunion). Additional factors include the frequency of visits to the site, the type of activities practiced there, on-site consumption behaviours, the level of concern individuals express regarding shark risk, and their perception of the effectiveness of the implemented measures.

To operationalize this method and ensure data reliability, we followed a set of 8 predefined steps, which are detailed in the following section.

Table 45: Methodological steps of the willingness to pay estimation method

Step	Title	Summary
1	Identifying the Change in Ecosystem Service	Define the expected effects of the proposed safety measures (e.g., improved safety, economic boost, increased site attractiveness).
2	Defining the Target Population	Identify the relevant population per site: local residents (Trois-Bassins, Boucan Canot), and both residents and visitors (Saint-Paul Bay).
3	Choosing the Survey Method	Use online and on-site questionnaires (except for Saint-Paul Bay, where only online was used), aligned with the travel cost method.
4	Drafting the Valuation Scenario	Describe the site-specific improvements (e.g., nets or lookout systems) and the proposed payment methods (annual membership, entry fee, or per-session fee).
5	Formulating the Valuation Question	Use an open-ended question to let individuals state their own WTP amount, avoiding biases from predefined options.
6	Designing Auxiliary Questions	Collect data on demographics, visit motivations, environmental sensitivity, and identify “true” vs. “false” zero responses to improve analysis accuracy.
7	Testing the Questionnaire	A pre-test phase was conducted to evaluate the time required to complete the questionnaire, train the survey team, and ensure the online form worked as intended.
8	Data Analysis	Data was analysed by categorizing respondents by residence and perception of the measures. Willingness to pay (WTP) was annualized for comparability, excluding protest (false zero) responses.

Identifying the Environmental Quality Change to Be Valued

The first step of the valuation process involves identifying the changes in environmental quality that are expected to result from the implementation of the proposed safety measures. Three key hypotheses underpin our analysis.

Hypothesis 1 suggests that the introduction of shark risk mitigation systems—such as new protective nets or enhanced lookout (vigie) operations—will increase users' sense of safety. These measures aim to prevent fatal shark encounters, thereby reassuring swimmers and ocean users. By reducing perceived and actual risks, the measures are expected to build greater confidence among beachgoers.

Hypothesis 2 considers the potential of such installations to stimulate economic activity in the areas where they are implemented. These safety systems represent an investment that could yield returns by revitalizing local spaces. In particular, Saint-Paul Bay, which is located near numerous shops and amenities, could benefit from the increased attractiveness of a secured swimming zone. The presence of nets may attract new visitors who are likely to support nearby businesses.

Hypothesis 3 builds on this idea, proposing that improved safety infrastructure could help draw more people to the area, thus boosting its economic potential. New visitors, such as beachgoers or swimmers, may be inclined to stay longer, dine locally, and return more frequently. Over time, this pattern of use could become habitual, further strengthening the link between recreational safety and local economic development.

Defining the Target Population

The target population varies depending on the study site.

For Trois-Bassins and Boucan Canot, both online and on-site surveys were directed at all individuals aged 18 and over who reside on Réunion Island. These respondents are considered the primary stakeholders due to their proximity and frequent use of the coastal areas.

In the case of Saint-Paul Bay, the questionnaires were distributed both online and on-site to reach a broader audience. While residents of the city of Saint-Paul are considered the most directly concerned by the proposed safety measures, non-residents who were present at the site during the survey period were also included. These individuals are viewed as active users of the bay, and their opinions are equally important in understanding the public’s perception and willingness to support the implementation of shark risk mitigation measures.

Determining the Survey Method and Sample Size

The survey methodology adopted mirrors the approach used in the travel cost method, combining on-site and online questionnaires. This dual mode of distribution allowed for broad public engagement, particularly at the Trois-Bassins and Boucan Canot sites. However, this mixed-method approach was not applied to the Saint-Paul Bay site, where only one survey mode (on-line) was implemented. The choice of method aimed to ensure both practical feasibility and representativeness of the respondents.

Designing the Contingent Valuation Scenario

The purpose of the contingent valuation scenario is to simulate a concrete improvement of each site and to assess individuals' willingness to pay (WTP) for the proposed safety measures.

> At Trois-Bassins, the scenario presented to respondents involved the extension and long-term support of the shark lookout (vigie) system. "Extension" refers to longer operating hours of the safety coverage, while "improvement" involves the introduction of theoretical innovative video surveillance technologies to enhance detection capabilities.
>
> At Boucan Canot, the scenario proposed the reinstallation and maintenance of a protective net covering the entire area. This would increase safety for water-based recreational activities, support the resumption of maritime practices, and reduce shark-related risks.
>
> For Saint-Paul Bay, the scenario involved the installation and upkeep of a new net, providing safer conditions for swimming and coastal activities. Once these site-specific scenarios were presented, respondents were asked to indicate the amount they would be willing to invest to support the implementation and operation of these safety devices.

In terms of payment formats, options included an annual membership fee or a site-specific fee, such as a daily entry or activity-based fee, particularly in the case of Trois-Bassins.

Formulating the Valuation Question

The valuation question can be open-ended, closed-ended, or double-bounded. For this study, we chose an open-ended question, which allows respondents to freely state the amount they are willing to pay for the proposed measure.

This approach offers several advantages: It is simpler to administer, as there is no need to predefine realistic price ranges. It encourages a greater diversity of responses, allowing individuals to express their personal valuation without constraint. It captures the "Top of Mind" value, i.e., the first amount that comes to a respondent's mind, reducing potential bias introduced by pre-established options.

However, the open-ended format also presents certain limitations: Respondents may give unrealistic or inconsistent values that do not reflect their actual financial capacity or attitudes toward the proposed measures. The use of mean values in data analysis may obscure important patterns. For example, a very high WTP from one respondent may distort the average if most others gave a value of zero. Full access to individual-level data is required to detect such disparities. The "yea-saying bias" is a potential issue in on-site surveys, where responses may be influenced by external factors such as interviewer sympathy or group dynamics when respondents are not alone. Lastly, respondents are free to answer "0", which is not necessarily a drawback. If an individual sees no value in the proposed changes and decides not to contribute financially, this reveals valuable information about the perceived utility of the measure.

This methodological choice helps to accurately capture the real value individuals attribute to the proposed changes and the level of support for shark risk mitigation strategies.

CVM questionnaire

Based upon the methodological steps, a fictional scenario was provided as follows (here the example of Trois-bassins) to the respondent:

> *"I will now present you with a fictional scenario, for which I would like to know your personal opinion. In order to ensure the long-term operation of the Vigie Requin system at the X site, to expand it (by increasing the hours of surveillance), and to improve it (through the use of innovative video monitoring tools for instance), a membership fee increase could be introduced. This increase would be voluntarily funded by users, and the money collected would be used exclusively for the improvement, maintenance, and operation of the Vigie Requin system. For tourists visiting the island, a per-session (daily) fee could be implemented, payable directly on site."*

In question 26 (local respondent) and 27 (tourist), the respondent estimates the additional financial contribution he/she would be willing to pay **per year (in euros)** to help maintain and improve this safety system?

In question 28, if the respondent answered €0, he/she indicates the main reason to differentiate true from false zeros as indicated in the table below:

Table 46: List of true and false zeros estimates for the WTP

True Zeros	Protest Zeros (False Zeros)
I cannot afford it	It's not up to me to pay
I do not want the system to be improved or maintained	I don't want to pay for others
It is not necessary to add such a system	I think the current membership fee is already high enough
I don't have enough information to decide	Other reasons

CVM results

Boucan-Canot - At Boucan Canot, a total of 394 responses were collected, with over 91% from residents of La Réunion and just over 8% from tourists. Regarding shark risk and related safety measures, no clear dominant opinion emerged. Around 17% of respondents felt completely unconcerned by the measures, while the most common sentiment across both residents and tourists was feeling “somewhat concerned.” Notably, residents tended to be “not very concerned,” while tourists were more likely to report being “very concerned.” In terms of willingness to pay (WTP) for improved shark protection, 50% of residents and 52% of tourists indicated they would be willing to contribute financially. Tourists showed an overall average daily WTP of €6.6 for tourists when non-contributors (true zeros) are included. Residents showed an annual WTP of €26.5 for residents when non-contributors (true zeros) are included, equivalent to €2.4 daily WTP, with a measured frequency of 10.9 sessions per resident per year. Despite having the highest number of respondents, Boucan Canot recorded the lowest total WTP among the sites assessed. This may be explained by a relatively low perceived level of concern among respondents—many only felt “somewhat concerned”—possibly due to the absence of fatal shark attacks at the site since 2011, leading to a perception that the risk has diminished over time.

Trois-Bassins - At the Trois-Bassins site, a total of 352 individuals responded to the survey, with over 87% being residents of La Réunion and just under 13% tourists. Respondents were asked about their level of concern regarding shark risk and the mitigation measures in place. The results show that 59% felt at least “somewhat concerned,” regardless of their place of residence. When asked about their willingness to pay (WTP) to maintain the shark lookout system, 60.6% of residents and 77.8% of tourists expressed a willingness to contribute financially. Those who did not feel concerned, particularly among non-residents, generally did not wish to contribute. Among contributors, the average daily WTP (including true zeros) for tourists was €6.59. For residents, the average annual WTP was €47.10, equivalent to 3.2 euros daily WTP (including true zeros) - with a measured frequency of 14.6 sessions per resident per year. This relatively high figure is driven by strong support from both residents and tourists, particularly those engaged in surfing—an activity highly popular at Trois-Bassins and one that exposes participants to greater shark risk due to its offshore nature. This likely explains why Trois-Bassins recorded the highest annual WTP among the three sites assessed. It should be noted here that the annual fees for surfing at Trois-bassins within the vigie requin was €50 in 2022.

Saint-Paul - The contingent valuation method was applied to assess willingness to pay (WTP) for implementing shark risk mitigation measures at the Saint-Paul coastal site (where no measure had been implemented so far). Based on 153 valid responses, the sample was composed of over 94% residents and fewer than 6% tourists, due in part to the online-only survey method. Among respondents, residents of Saint-Paul were specifically analysed, given their proximity and potential exposure to shark risk. Although nearly one-third of respondents reported feeling “not particularly concerned” about shark risk, Saint-Paul residents displayed a bimodal distribution of concern, with no dominant opinion. This ambivalence influenced WTP outcomes: only 49% of resident respondents expressed willingness to contribute financially—the lowest rate among the sites surveyed. However, when concern was high, the annual WTP rose significantly, reaching up to €90 per year for residents, equivalent to €9 daily

WTP assuming a hypothetical 10 visits per year. On average, Saint-Paul residents showed a daily WTP of €3.1, compared to €2.4 among all Reunion Island residents. Tourists, despite being few in number, were more inclined to contribute, with 66.7% expressing willingness to pay. Yet, due to their low visitation rate (3 visits per year on average), the annual daily WTP of tourists was €2.6. The study suggests that enhancing shark mitigation infrastructure at Saint-Paul could attract more visitors and improve its economic valuation.

13.1.5 Lithuania

13.1.5.1 Crop production

Crop production (CP) in Lithuania has been estimated using data from Agricultural statistics for the year 2022 (https://osp.stat.gov.lt/). The agricultural crop yield data is available in tons per hectare. For the convenience of the calculation, data in ton ha^{-1} has been converted into ton/pixel (10000 m^2). The crop production in Lithuania was calculated using the following yield equation multiplied by area (Rimal et al., 2019).

$$CP_{xy} = \sum_{g=1}^{G} A_{xy}\, Y_g\, C$$

Where CP_{xy} refers to the total crop production of the cell (x and y) in the cropland (C) in a unit of tons, A_{xy} refers to the area of the cell (x and y), which is equal to 1.0 hectares (or 10000 m2) and Y_g is the yield per unit area from crops in ton ha^{-1} y^{-1}.

The crop production was estimated from Corine land cover data i.e. from agricultural land use type[14]. Crops grown on the specific land use were determined based on the Corine LULC types. For example, under the fruits and berry plantation category, the fruits grown (such as apples and strawberries) are mentioned. Finally, these crops' average yield per hectare area were estimated from Lithuania's agricultural statistics.

Monetary values could then be generated with the resource rent method, using statistics on production factors collected by Statistics Lithuania[15].

[14] https://land.copernicus.eu/user-corner/technical-library/corine-land-cover-nomenclature guidelines/html/index.html

[15] Rodiklių duomenų bazė - Oficialiosios statistikos portalas

13.1.6 Portugal

13.1.6.1 Wood provision

Biophysical quantification of timber provision (pre-fire environment) in Northern Portugal was based on the collection and modelling of woody biomass volume data from the Portuguese National Forest Inventory - IFN6 (ICNF, 2015) for maritime pine, eucalyptus and oak forests, combined with spatial data from the Portuguese land use and land cover spatial database (DGT, 2018) (see SELINA D5.1 for a detailed description of the methods). Monetary valuation followed a market price-based valuation method based on public statistics from the Simplified Forest Product Quotation System (SIMeF) in Portugal, which reports sales of standing woody biomass in public areas managed by the Portuguese Institute for Nature Conservation and Forests (ICNF, 2022). Minimum, average, and maximum prices for standing woody biomass (Euro/m^3) of maritime pine, eucalyptus and oak forests in Northern Portugal were collected and adjusted for inflation to 2017 euros. Then, prices were multiplied by the woody biomass volume (m^3) previously estimated for the pre-fire environment.

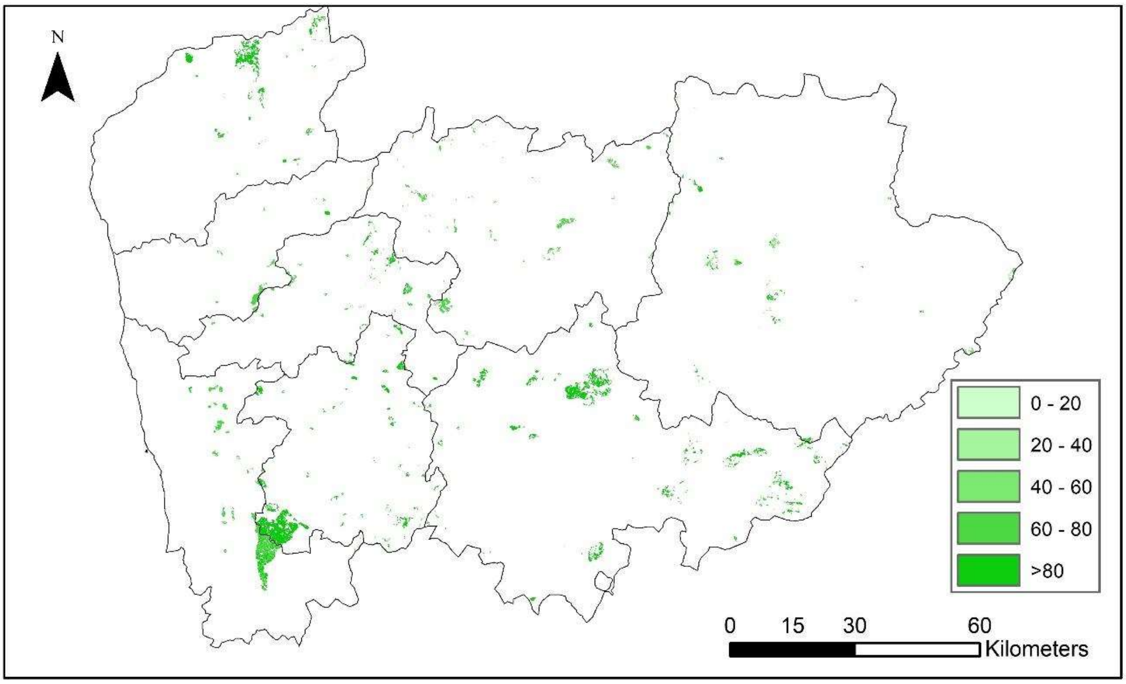

Figure 18. Spatial distribution of timber production (m3/ha) in the pre-fire environment in maritime pine, eucalyptus and oaks forests in Northern Portugal.

Table 47. Biophysical and monetary estimates of timber provision (pre-fire environment) in maritime pine, eucalyptus and oaks forests in the Northern Portugal test site.

Ecosystem type	Area (ha)	Biophysical	Monetary (€/ha)		
		Timber (m3/ha)	Min.	Mean	Max.
Maritime pine forest	13,599.4	116.6	931.5	2,918.2	5,323.7
Eucalyptus forest	10,569.8	74.6	1,296.7	1,756.4	2,409.4
Oaks forest	3,544.0	83.2	1,789.2	2,188.0	2,897.8

13.1.6.2 Cork provision

Biophysical quantification of cork provision (pre-fire environment) in Northern Portugal was based on the collection and modelling of cork oak productivity data from the Portuguese National Forest Inventory - IFN6 (ICNF 2015) in both pure and mixed cork stands, combined with spatial data from the Portuguese land use and land cover spatial database (DGT, 2018) (see SELINA D5.1 for a detailed description of the methods). Monetary valuation followed a market price-based valuation method based on public statistics from the Simplified Forest Product Quotation System (SIMeF) in Portugal, which reports sales of non-woody products by private producers (ICNF, 2024). Minimum (95 Euro/kg), average (366 Euro/Kg) and maximum (583 Euro/Kg) cork prices in Northern Portugal were collected. These prices were adjusted for inflation to 2017 euros. Then, prices were multiplied by cork production (kg) previously estimated in the pre-fire environment.

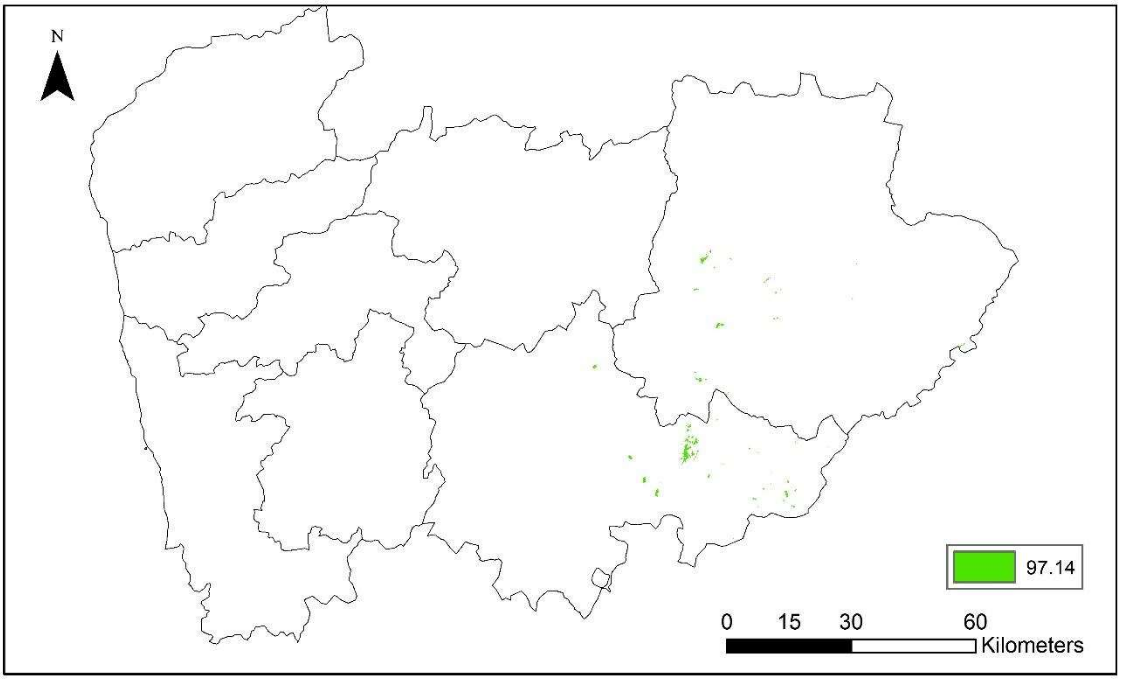

Figure 19. Spatial distribution of cork production (kg/ha) in the pre-fire environment in Northern Portugal.

Table 48. Biophysical and monetary estimates of cork provision (pre-fire environment) in cork oak forests in the Northern Portugal test site.

Ecosystem type	**Area (ha)**	**Biophysical**	**Monetary (€/ha)**		
		Cork (kg/ha)	**Min.**	**Mean**	**Max.**
Cork oak forest	1,735.44	97.1	9,218.5	35,556.9	56,627.6

13.1.6.3 Global climate regulation

Biophysical quantification of carbon storage (pre-fire environment) in Northern Portugal was based on the collection and modelling of carbon stock data from the Portuguese National Forest Inventory - IFN6 (ICNF 2015) in forests and shrublands, combined with spatial data from the Portuguese land use and land cover spatial database (DGT, 2018) (see SELINA D5.1 for a detailed description of the methods). Monetary valuation followed a unit value transfer method based on the social costs of carbon (SCC) and a market price-based valuation method based on carbon credit prices. SCC data were collected from the scientific literature, namely the estimates by Nordhaus (2017) of 16.16 Euro/Mg C (Europe) and 105.31 Euro/Mg C (Global) and estimates by Barrage and Nordhaus (2024) of 213.48 Euro/Mg C (Global). SCC estimates by Nordhaus (2017) and Barrage and Nordhaus (2024) were converted from US dollars to Euro using the average exchange rate of 0.9015 (in 2015) and 0.8931 (in 2019), respectively, and inflation-adjusted to 2017 euros. Carbon credit price data were collected from the Market Climate Trade platform (https://market.climatetrade.com – accessed in October 2024). Minimum (134.40 Euro/Mg C), average (179.38 Euro/Mg C) and maximum (209.97 Euro/Mg C) carbon credit prices were collected from seven carbon offset forest projects in Spain, as currently there is no carbon market in Portugal. These prices were adjusted for inflation to 2017 euros. Then, SCC and carbon credit prices were multiplied by carbon stored (Mg) previously estimated in the pre-fire environment.

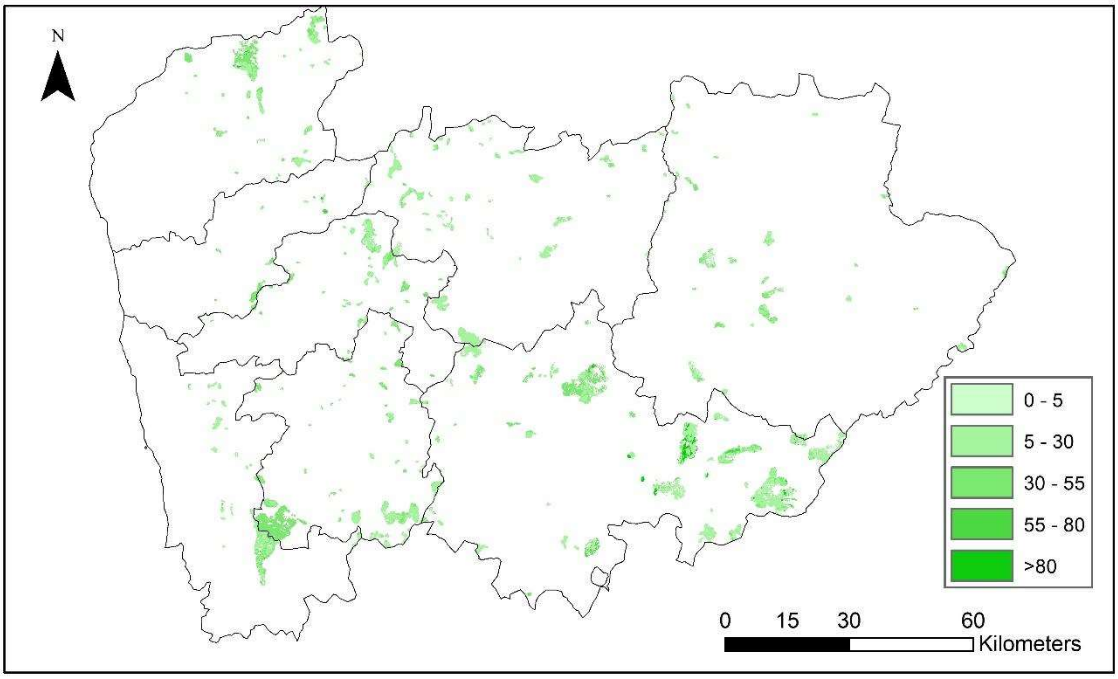

Figure 20. Spatial distribution of carbon stored (Mg/ha) in the pre-fire environment in Northern Portugal.

Table 49. Biophysical and monetary estimates of carbon storage (pre-fire environment) in forests and shrublands in the Northern Portugal test site.

Ecosystem type	Area (ha)	Biophysical	Monetary (€/ha)					
			SCC			Market price		
		Carbon stored (Mg/ha)	Value*	Value**	Value***	Min.	Mean	Max.
Forest	34,247.2	38.6	623.4	4,061.7	8,233.5	5,183.7	6,918.3	8,098.3
Shrublands	39,443.8	13.8	223.5	1,456.5	2,952.4	1,858.8	2,480.8	2,903.9

Source: *Nordhaus (2017) – Europe estimates; **Nordhaus (2017) – Global estimates; ***Barrage & Nordhaus (2023) – Global estimates.

13.1.6.4 Erosion regulation

Biophysical quantification of soil retention (pre-fire environment) in Northern Portugal was based on the application of the Morgan-Morgan-Finney (MMF) erosion model (Morgan 2001) in forests and shrublands (see SELINA D5.1 for a detailed description of the methods). Monetary valuation followed a unit value transfer method based on 1) soil replacement costs and 2) post-fire soil erosion mitigation costs. Data on soil replacement costs were collected from estimates by Marta-Pedroso et al. (2007) for Portugal. The unit value collected was 5.35 Euro/Mg after adjusting for inflation to 2017 euros. This value was then multiplied by the mass of soil retained in the pre-fire environment as estimated from MMF model outputs. Data on post-fire soil erosion mitigation costs were collected from the meta-analysis conducted by Girona-García et al. (2023). The collected unit value was 867.5 €/Mg (after adjusting for inflation to 2017 euros) and corresponds to the median costs of soil erosion mitigation treatments for forest fires that took place in Portugal. Then, this value was multiplied by the mass of soil retained in the pre-fire environment.

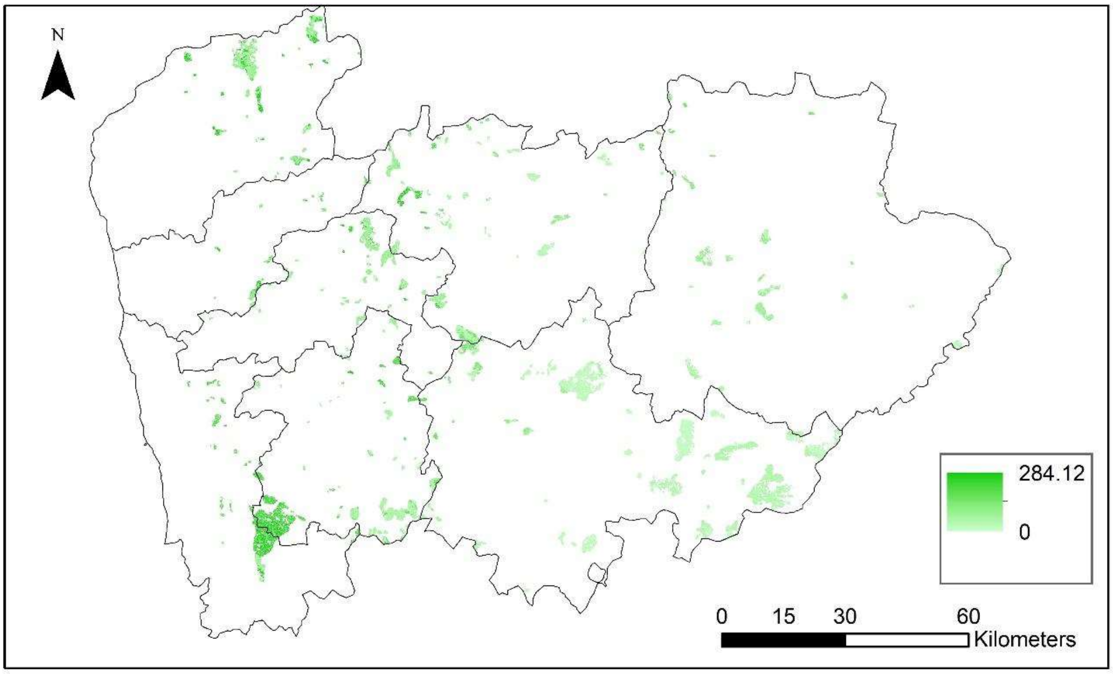

Figure 21. Spatial distribution of soil retention (Mg/ha) in the pre-fire environment in Northern Portugal.

Table 50: Biophysical and monetary estimates of soil retention (pre-fire environment) in forests and shrublands in the Northern Portugal test site.

Ecosystem type	Area (ha)	Biophysical	Monetary (€/ha)			
			Soil replacement costs	Erosion mitigation costs		
		Soil mass retained (Mg/ha)		Min.	Mean	Max.
Forest	34,247.2	74.4	398.2	28,381.4	64,571.3	149,109.4
Shrublands	39,443.8	30.4	162.7	11,596.4	26,383.2	60,924.6

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○ Annex 2 - Table with spatially defined parameters for inclusion in value functions

Table 51. Spatially defined parameters for inclusion in value functions

Indicator	Aggregation method	Dataset unit	Data source	Dates covered by dataset or date of publication of the data	Resolution (m)
Ecosystem Condition	Mean	Index (0-100)	UNEP-WMCM, 2020	Aggregate of datasets with various dates (1970 to present for historical data)	20,000
Protection status	Percentage of site area	Categorical (protected / not protected)	UNEP-WCMC & IUCN, 2023	Protected areas as of February 2023	Not applicable (vector data)
Fragmentation	Mean	km^2	Kennedy *et al.*, 2019	2016	1,000
Biodiversity intactness	Mean	Index (0-1)	Newbold *et al.*, 2016	Modelled image based on data from 2000-2015	927
Pollution (Particulate matter d < 2.5 um)	Mean	kg m^-3	Benedetti *et al.*, 2009; Morcrette *et al.*, 2009	Daily data for 2016-2021, aggregated to Annual data	44,528

Land cover	Percentage of site area for each land cover category	Categorical	ESA, 2017	Annual data for 1992-2019	309
Ecosystem diversity (Global habitat heterogeneity)	Mean	Index (0-1)	Tuanmu & Jetz, 2015	One dataset based on 2001-2005 data	928
Manufactured features/infrastructure	Percentage of area	percentage	Pesaresi *et al.*, 2015	Four datasets covering: years before 1975, 1975-1990, 1990-2000, 2000-2014	38
Human modification index	Mean	index (between 0 and 1)	Theobald *et al.*, 2020	1990, 2000, 2010 and 2015	300
Protected area connectivity	Mean	percentage of connected protected land	Saura *et al.*, 2018	2016	Not applicable (vector data)
Net Primary Productivity	Mean	kg C/m^2	Running & Zhao, 2019	Annual data from 2001 to 2021	500
Human Appropriation of	Mean	kg C/m^2	Imhoff & Bounoua, 2006;	1995	28,000

Net Primary Productivity			Imhoff *et al.*, 2004		
Topographic diversity	Mean	index (between 0 and 1)	Theobald *et al.*, 2015	One dataset aggregating data from 2006 to 2011	270
Non-native species	Mean	number of species per km^2	Pagad *et al.*, 2022	2016	National level
Population density	Mean	persons per square kilometre	Center for International Earth Science Information Network & Columbia University, 2018	2000, 2005, 2010, 2015 and 2020	927.67
Population profile (average age)	Mean	years	WorldPop, 2023	Annual data from 2000 to 2020	92.7
Income per capita	Mean	USD	World Bank, 2021	Annual data. Time coverage varies between countries (oldest year is 1990, nearest year is 2021)	National level

Income inequality (Gini coefficient)	Mean	Index	OECD, 2023	Annual data. Time coverage varies between countries (oldest year is 1976, nearest year is 2021)	National level
Accessibility - time to nearest town/city	Mean	minutes	Howes *et al.*, 2018	2015	1,000
Nighttime light	Mean	nanoWatts/cm^2/sr	Colorado School of Mines, 2023	Monthly dataset from April 2012 to June 2022	463
Accessibility - road density	Mean	m/km^2	Meijer *et al.*, 2018	Aggregation of data sources from 1997 to 2015	8,000