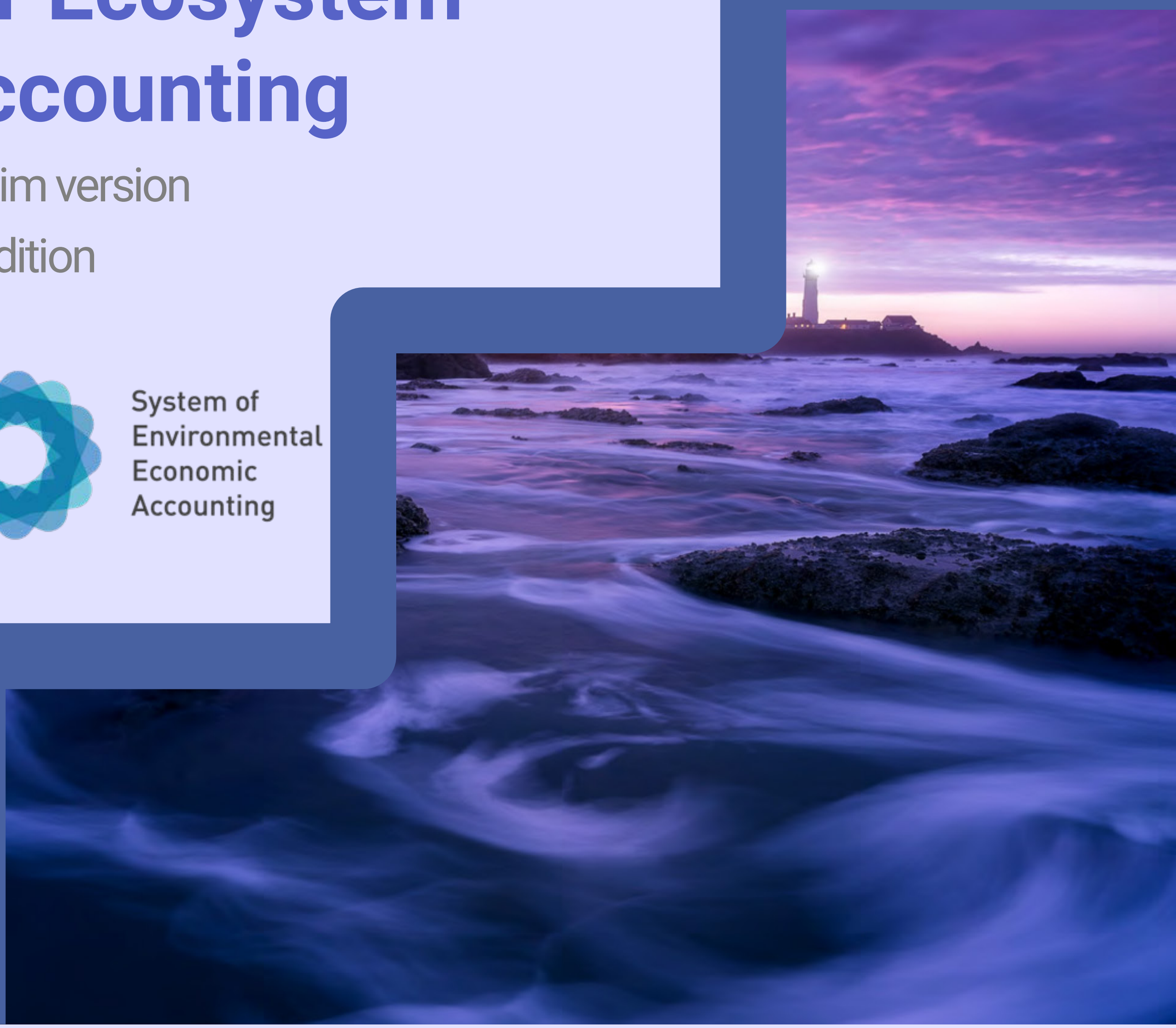
Monetary Valuation of Ecosystem Services and Assets for Ecosystem Accounting

Interim version

1st edition

System of Environmental Economic Accounting



NCAVES
Natural Capital Accounting and Valuation of Ecosystem Services

MAIA
Mapping and Assessment for Integrated ecosystem Accounting

Reference and Licensing

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This reports draws upon the SEEA EA and the issue papers that were developed in support of the SEEA EA revision process: https://seea.un.org/content/seea-experimental-ecosystem-accounting-revision. A study on aggregation of monetary values developed by eftec and commissioned by Eurostat, also provided inputs (2019, unpublished).

The report is intended to provide a useful starting point with suggestions on valuation methods, examples and references for countries that are undertaking valuation as part of the SEEA EA national implementation. It should not be considered as agreed guidelines on how to compile monetary accounts. Some issues related to valuation are being discussed as part of the SEEA EA research agenda and the SNA revision process. The document will remain under review and updated as country experiences are gained in valuing ecosystem services based on SEEA EA and as consensus emerges. This interim status is reflected in the title of the report.

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Abbreviations and Acronyms

ARIES	Artificial Intelligence for Environment and Sustainability
CICES	Common International Classification of Ecosystem Services
CV	Contingent valuation
CW	Comprehensive wealth
DALY	Disability Adjusted Life Years
EEZ	Exclusive economic zone
ES	Ecosystem services
ESVD	Ecosystems service valuation database
ETS	Emissions Trading Scheme
FAO	Food and Agriculture Organization of the United Nations
GDP	Gross domestic product
GE	General equilibrium
GHG	Greenhouse gas
GIS	Geographical Information Systems
IAM	Integrated Assessment Model
InVest	Integrated Valuation of Ecosystem Services and Trade-offs
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IW	Inclusive wealth
MAIA	Mapping and assessment for integrated ecosystem accounting
MC	Marginal cost
MP	Market price
MWTP	Marginal willingness to pay
NCAVES	Natural Capital Accounting and Valuation of Ecosystem Services
NCP	Nature´s Contribution to People
NDP	Net domestic product
NOAA	US National Oceanic and Atmospheric Administration
NPV	Net present value
NSO	National Statistical Office
NWFP	Non-wood forest products

PE	Partial equilibrium
PES	Payments for ecosystem services
PSUT	Physical Supply and Use Tables
QALY	Quality Adjusted Life Year
REDD	Reducing Emissions from Deforestation and Degradation
RR	Resource rent method
RUM	Random utility model
SCC	Social cost of carbon
SDR	Social discount rate
SEEA CF	System of Environmental-Economic Accounting - Central Framework
SEEA EA	System of Environmental-Economic Accounting - Ecosystem Accounting
SEV	Simulated exchange value method
SNA	System of National Accounts
SWAT	Soil Water Assessment Tool
TEV	Total economic value
UNCEEA	United Nations Committee of Experts on Environmental-Economic Accounting
UNEP	United Nations Environment Programme
UNSD	United Nations Statistics Division
USD	United States Dollar
VSL	Value of Statistical Life
VT	Value transfer
WTA	Willingness to accept
WTP	Willingness to pay

1. Introduction

The System of Environmental-Economic Accounting - Ecosystem Accounting (SEEA EA) is a spatially based, integrated statistical framework for organizing biophysical information about ecosystems, measuring ecosystem services, tracking changes in ecosystem extent and condition, valuing ecosystem services and assets and linking this information to measures of economic and human activity. The framework was developed in response to a range of policy demands and challenges with a focus on making visible the contributions of nature to the economy and people (UN et al., 2021).

The United Nations Statistical Commission (UNSC) at its fifty-second session in March 2021 adopted SEEA EA chapters 1-7 describing the accounting framework and the physical accounts as an international statistical standard; recognised that chapters 8-11 of the SEEA EA describe internationally recognised statistical principles and recommendations for the valuation of ecosystem services and assets in a context that is coherent with the concepts of the System of National Accounts for countries that are undertaking valuation of ecosystem services and/or assets; and noted chapters 12-14 as describing the applications and extensions of ecosystem accounting.[1]

The primary purpose of this document is to help statisticians and compilers of ecosystem accounts as well as biophysical modellers of ecosystem services to understand the concepts and methods of valuation of ecosystem services and assets, how the valuations can be carried out, and how they can be of use in the context of the SEEA EA (UN et al., 2021). This document complements the conceptual descriptions of the SEEA EA concerning monetary valuation by describing in more detail the various valuation techniques listed in SEEA EA Chapter 9 and showing how these techniques can be applied for the measurement of many of the ecosystem services in the SEEA EA ecosystem services reference list (SEEA EA Chapter 6). In addition, building on the discussion of net present value techniques in SEEA EA Chapter 10, this report provides additional insight into appropriate measurement approaches. In general, it provides examples of good practice, references where further information can be obtained, and identifies remaining knowledge gaps.

Valuation is understood here as the expression of flows of ecosystem services and stocks of ecosystem assets in monetary units – as described in Chapters 8-11 of the SEEA EA. The SEEA EA applies the accounting principles of the *System of National Accounts 2008* (2008, SNA) (United Nations et al., 2009). Specifically, in the context of monetary valuation, the SEEA EA applies the SNA concept of exchange values in order to support comparisons of ecosystem services and ecosystems assets with the values of products and assets recorded in the national accounts.

Exchange values differ from welfare values which are commonly used in environmental cost-benefit analysis and which include consumer surplus. Thus, while estimates based on exchange values are useful in many contexts, there are also applications in which they are applied together with other economic value concepts, for example in cost benefit analysis. For these types of applications, it will usually be relevant to include the value of the wider social benefits of ecosystems, including many of their non-use values. These differences need to be considered when comparing the two kinds of

[1] https://unstats.un.org/unsd/statcom/52nd-session/documents/decisions/Draft-Decisions-Final-5March2021.pdf

monetary values and when interpreting the ecosystem service values based on exchange values that are used in ecosystem accounts.

More generally, monetary values will not fully reflect the importance of ecosystems for people and the economy. Assessing the importance of ecosystems will therefore require consideration of a wide range of information beyond data on the monetary value of ecosystems and their services. This will include data on the biophysical characteristics of ecosystems, data on the characteristics of the people, businesses and communities that are dependent on them. (UN et al., 2021, p.1)

The SEEA EA is a system conceived as an integrated, internally consistent series of accounts. Its design is such that it can be implemented equally well in parts i.e., the implementation can be flexible and modular. Indeed, the progressive and staged development of the range and detail of the ecosystem accounts is likely an appropriate implementation strategy (UN et al., 2021, p.1)

Generally, the compilation of ecosystem accounts in monetary terms will require the use of data in physical terms since there are a limited number of observable monetary transactions that relate directly to flows of ecosystem services and assets. Further, to support appropriate interpretation and application of the monetary data in policy and decision-making, it is recommended that when monetary accounts are released, the associated data in physical terms (e.g., concerning changes in ecosystem extent and condition and flows of ecosystem services in physical terms) are also released. (UN et al., 2021; para. 1.15). Complementary data in physical terms will enable users to understand the extent to which changes in value are due to changes in price, changes in quality, changes in quantity or some combination of these three factors.

This document is structured as follows. Chapter 2 lays out the conceptual basis for the valuation of ecosystem services (ES). It places in context the concept of exchange values used in ecosystem accounting and other valuation concepts including welfare values, consumer surplus, willingness to pay and willingness to accept. Chapter 2 can be read as a complement to material in SEEA EA Chapter 8, Principles of monetary valuation for ecosystem accounting and Chapter 12, Annex 12.1, Exchanges and welfare values in an accounting context.

Chapter 3 provides a review of the most commonly used valuation methods and the extent to which they are able to approximate exchange values. A key distinction is made between methods that collect and apply primary data and those that use secondary data, collectively referred to here as value transfer methods, which are discussed in Chapter 6. The level of detail provided is intended to be sufficient to understand the methods and their strengths and weaknesses. However, this document does not provide a complete description of how to carry out valuation in practice. Instead, references are provided to relevant implementation materials. Chapter 3 can be read as a complement to SEEA EA Chapter 9.3,Techniques for valuing transactions in ecosystem services.

Chapter 4 describes the application of the methods presented in Chapter 3 to the valuation of selected ES from the reference list of ES in Chapter 6 of the SEEA EA. For each service, the methods applied are discussed in terms of the demand for data and the need for statistical analysis. For countries where these factors are more limiting, less demanding methods are described, with some guidance on their limitations. A tiered approach is taken whereby methods are ranked taking into account their proximity to observed market prices and their expected accuracy and spatial resolution. The chapter also includes examples of valuations and provides information on where the reader can go for further details on specific issues. Chapter 4 can be read as a further elaboration of the short discussion on these issues in SEEA EA chapter 9.4.2,Valuation of different types of services.

Chapter 5 covers the valuation of ecosystem assets. In the SNA, the value of some environmental assets (e.g., timber resources) is recorded in the capital accounts and the balance sheets, but there are many environmental assets, such as wetlands and coastal ecosystems across the terrestrial, freshwater and marine realms[2], whose values are not well reflected in those accounts. This chapter reviews the coverage of ecosystem assets in the SNA capital accounts and balance sheets and provides directions for the valuation of those ecosystem assets that are not included in the SNA. A definition of an asset is given as is an explanation of the basis for measuring the value of an ecosystem asset using the present value of the expected future flow of ES it provides. Issues relating to applying such a valuation method – e.g., discount rates, future values, changes in prices - are discussed. Examples are provided of some national estimates, such as those developed as part of the UK´s natural capital accounts, and from the Inclusive Wealth/Comprehensive Wealth reports. Chapter 5 can be read as a complement to SEEA EA chapter 10, Accounting for ecosystem assets in monetary terms.

Chapter 6 discusses a number of practical aspects to consider when compiling monetary ecosystem accounts: value transfer for ecosystem accounting; data sources and tools to support valuation; fitness for purpose; and aggregation. Finally, recommendations are provided to avoid misinterpretation when communicating the results from monetary ecosystem accounts.

Examples of how the different ES valuation methods have been applied are provided in different sections of the report. Examples include results from pilot studies that have been carried out as part of the NCAVES and MAIA projects.

[3] The 2008 SNA notes a number of cases where market prices do not represent exchange values (e.g., in situations of transfer and concessional pricing (see paragraphs 3.131-3.134).

2. Foundations

2.1 Introduction

This chapter provides the conceptual foundations for valuation as applied in this document. It describes the purpose of valuation in the SEEA EA, lays out the key valuation principles applied in ecosystem accounting and summarizes the economic basis of non-market valuation of ES.

As described in the SEEA EA there is strong evidence of user demand for estimating the monetary value of the environment's contribution to the economy and people. There is also demand for integrated assessments of the connection between the environment and the economy, in particular understanding changes in broad measures of wealth resulting from managed/human and natural causes, for example, from climate change and biodiversity loss. At the same time, monetary valuation will not be appropriate in all decision-making contexts and, in all cases, it will be relevant to use associated biophysical data on stocks and flows. (UN et al, 2021; Section D Overview)

Among statisticians and more broadly, the use of monetary values of environmental stocks and flows in the measurement and assessment of the environment has long been a point of discussion and contention. The existence of multiple perspectives on this issue is well recognised. There are differences of view concerning (i) the underlying framing for valuation of environmental stocks and flows; (ii) the potential of monetary valuation to support decision making; (iii) the ability to produce reliable estimates in monetary terms in practice; and (iv) the role of national statistical offices (NSOs) in producing fit for purpose statistics in this area of measurement. (UN et al., 2021)

While these different perspectives exist, there is support for the exchange value based approach to the monetary valuation of ecosystem services and ecosystem assets described in the SEEA EA (Chapters 8 – 11). Importantly, the valuation approach used in the SEEA EA is based on existing theory and concepts adopted in the SNA, which have been adapted to the environmental context.

More generally, as highlighted in the opening chapters of the SEEA EA, it is emphasized that monetary values from the accounts, and the wider economic values just described, will not fully reflect the importance of ecosystems for people and the economy. Assessing the importance of ecosystems will require consideration of a wide range of information beyond data on the monetary value of ecosystems and their services. This will include data on the biophysical characteristics of ecosystems, for example of extent and condition, and data on the characteristics of the people, businesses and communities that are dependent on them.

2.2 Purpose of valuation in the SEEA EA

A number of motivations exist for the monetary valuation of ecosystem services and ecosystem assets depending on the purpose of analysis and the context for the use of valuations in monetary terms. The different motivations point to different requirements in terms of the concepts, methods and assumptions used for monetary valuation. (UN et al., 2021; para. 8.1) In ecosystem accounting, the primary motivation for monetary valuation using a common monetary unit or numeraire is to be able to make comparisons of different ecosystem services and ecosystem assets that are consistent with standard measures of products and assets as recorded in the national accounts. This requires the use of exchange values. In turn, this facilitates the description of an integrated system of prices and quantities for the economy and the environment that is a core motivation of the SEEA EA. (UN et al., 2021; para. 8.2)

Exchange value based monetary accounts can support: (a) comparing the values of environmental assets (including ecosystems) with other asset types (e.g., produced assets) as part of extended measures of national wealth; (b) highlighting the relevance of non-market ecosystem services (e.g., air filtration); (c) assessing the contribution of ecosystem inputs to production in specific industries and their supply chains; (d) comparing the trade-offs between different ecosystem services through consideration of relative prices; (e) deriving complementary aggregates such as degradation adjusted measures of national income; (f) evaluating trends in measures of income and wealth; (g) improving accountability and transparency around the public expenditures on the environment by recognising expenditure as an investment rather than a cost; (h) providing baseline data to support scenario modelling and broader economic modelling; and (i) 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. (UN et al., 2021; para. 8.3)

In the space of environmentally related monetary valuation more generally, it is common for valuation to focus on measurement of the impacts of changes in ecosystem assets and services on economic and human welfare. For example, valuation may focus on measuring the impacts of improved parks and reduced pollution on human health or the impacts of reduced soil fertility on farm incomes. The valuation of impacts, both positive and negative, is an important requirement in the development of specific policy options and policy settings, project evaluation and incentive design. This may include, for example, detailed cost-benefit analysis and the assessment of compensation and damage claims. Such analysis can be complemented, but not replaced, by data from a set of ecosystem accounts based on exchange values, recognizing that is likely that more detailed and finer scale data and valuations are required for impact analysis. More broadly, SEEA EA accounts provide a coherent framing for the collection and organisation of relevant data and can support an understanding of micro-macro linkages and the assessment of changes over time. (UN et al., 2021; para. 8.4)

As noted in chapter 1, the SEEA EA was developed in response to a range of policy demands and challenges with a focus on making the contributions of nature to the economy and people visible, as well as having a better record of the impacts that economic and other human activity have on the environment. To this end, ecosystem accounting incorporates a wider range of benefits to people than those captured in standard economic accounts, and provides a structured approach to assessing the dependence and impacts of economic and human activity on the environment.
The SEEA EA consists of a system of account covering flow and stocks in physical and monetary terms. The main monetary accounts are described in Box 1.

***Box 1:** Main type of SEEA EA monetary ecosystem accounts and complementary accounts*

The SEEA EA is a system conceived as an integrated, internally consistent series of accounts, which provides a comprehensive and coherent view of ecosystems. The main monetary ecosystem accounts are:

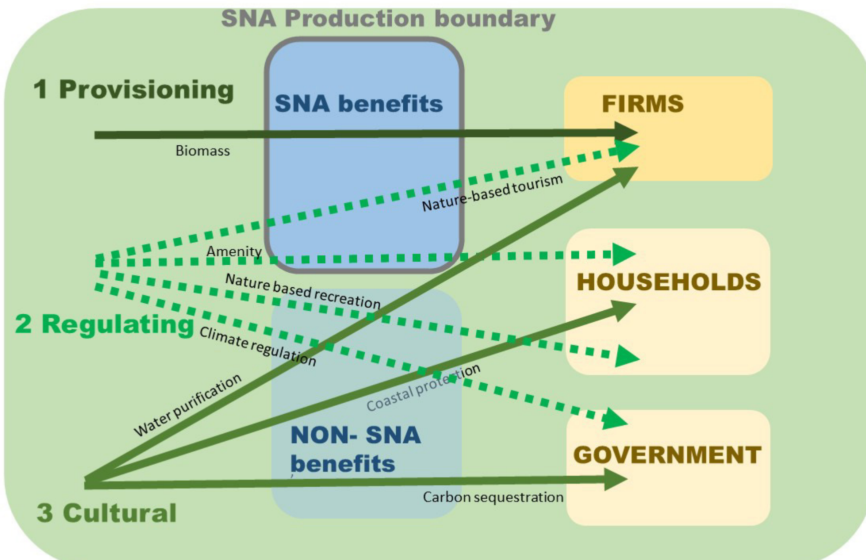
- **Ecosystem services flow account in monetary terms** records the supply of ecosystem services by ecosystem assets and the use of those services by economic units, including households (See SEEA EA Chapter 9 for more information).
- **Ecosystem monetary asset accounts** records information on stocks and changes in stocks (additions and reductions) of ecosystem assets. This includes accounting for ecosystem degradation and enhancement. Asset values can be integrated in extended balance sheets that provide a measure of the wealth of countries (See SEEA EA Chapter 10 for more information).
- **Extended sequence of accounts** integrates the value of ecosystem services and degradation/enhancement in the full sequence of national accounts, allowing derivation of adjusted aggregates for production, income and savings (See SEEA EA Chapter 11 for more information).
- **Complementary accounts** show how the above monetary accounts presented can be related to, and potentially support, other approaches and applications in monetary terms obtained when alternative institutional arrangements are assumed. Examples are: tables that show externalities and ecosystem disservices; alternative measures of income, wealth and degradation; polluter pays recording; restoration cost; Hicksian income (capital gains), and a bridge table between exchange and welfare values (See SEEA EA Chapter 12 for more information).

The measurement scope of goods and services in the SNA is determined by the so-called production boundary. Transactions relating to all goods and services within the production boundary form the basis for the measurement of gross domestic product, industry value added and other aggregates. Flows of ecosystem services are excluded from the production boundary of the SNA since these are not the result of a production process by an economic unit as defined by the SNA. As a result, the role of ecosystem services in supporting economic production is not explicitly recorded in the SNA. In addition, the role of ecosystem services in contributing to benefits received by people and society other than from economic production is also not recorded. As a result, all ecosystem services that contribute to the benefits of people, economy and society are excluded from the SNA.

SEEA EA defines ecosystem services (ES) as the contributions of ecosystems to the benefits that are used in economic and other human activity. By recording flows of ecosystem services, the SEEA EA extends the SNA production boundary. Another example of an extension of the SNA production boundary is known as "the household production satellite account." This satellite account extends the SNA production boundary, recognizing own account production of services such as cooking, cleaning and child-care (Eurostat, 2003).

The SEEA EA makes an important distinction between **SNA benefits** (i.e., goods and services that are included in the production boundary of the SNA) and **non-SNA benefits** (i.e., goods and services that are not included in the production boundary of the SNA). Where ES contribute to SNA benefits, the aim of valuation in ecosystem accounting is to estimate the share of a good or service recorded in the SNA that is derived from an ecosystem service. For example, in the case of crop provisioning services, the ecosystem service would not consist of the full market value of the harvested crops, but measure the ecosystem contribution to crop production calculated for instance by applying a resource rent calculation method that deducts the contributions from labor and produced capital such as tractors. Likewise, in the case of non-SNA benefits, the aim is to estimate the value of the ecosystem contribution to the benefits. The focus of measurement lies on the environment-economy nexus where ecosystem accounts focus on the final ecosystem services that directly contribute to benefits. While the valuation and integration of ES in national accounts will increase output, measures of value added will only increase to the extent of the ES contributing to non-SNA benefits.

***Figure 1**: Framing ES and their values with regard to the SNA production boundary. Source: Adapted from Barton, Caparros et al. (2019)*



The SEEA EA includes a reference list of ES, rather than a full classification system, that provides clear descriptions of the most commonly found ES, in a mutually exclusive way i.e., avoiding double counting of ES. The ES are grouped into 3 main categories (UN et al. 2021; para. 6.51):

- **Provisioning services** are those ecosystem services representing the contributions to benefits that are extracted or harvested from ecosystems.
- **Regulating and maintenance services** are those ecosystem services resulting from the ability of ecosystems to regulate biological processes and to influence climate, hydrological and biochemical cycles, and thereby maintain environmental conditions beneficial to individuals and society.
- **Cultural services** are the experiential and intangible services related to the perceived or actual qualities of ecosystems whose existence and functioning contributes to a range of cultural benefits.

As illustrated in Figure 1 provisioning services will in all situations contribute to SNA benefits, while regulating and cultural services may contribute to both SNA and non-SNA benefits, depending on the ES in question.

2.3 Valuation principles in ecosystem accounting

2.3.1 *Exchange values*

A key characteristic of ecosystem accounts is that they use exchange values, which are defined as: "**the values at which goods, services, labour or assets are in fact exchanged or else could be exchanged for cash**" (2008 SNA, para. 3.118; UN et al. 2009). For the vast majority of entries in the national accounts, exchange values are measured using data from observed transactions involving market prices. ***Market prices are defined as amounts of money that willing buyers pay to acquire something from willing sellers*** (UN et al. 2009; para. 3.119).[3] The use of observed market prices implies that the accounts embody information about the revealed preferences of the economic units involved, recognising that this is not a complete information set about their preferences that would encompass estimation of their willingness to pay (WTP), willingness to accept (WTA) and information on various opportunity costs of revealed transactions.

A key feature of the SNA/SEEA is that it is a transaction-based system, where accounts record the various types of transactions relating to various aspects of economic activity (such as, production, consumption, accumulation) in which institutional units (e.g. businesses, community organizations, households, municipal and national governments) engage. The nature of the accounting system is that each transaction is recorded both as a supply and as a use, so that supply equals use.
Importantly, the recording of transactions is not limited to situations in which goods and services are exchanged for cash or similar financial assets. For example, transactions involving bartering are also recorded provided the good or service being exchanged is within the production boundary. More generally the SNA allows for non-monetary transactions such that flows can be recorded in which the relevant value must be indirectly measured or which are analytically useful to treat as transactions (UN et al. 2009; para. 3.51).

While the majority of transactions recorded in the national accounts are based on observed market prices, there are a number of (often large) types of transactions for which market prices are not observed and therefore need to be estimated. Thus, in the national accounts, where market price-based transactions are not observable, alternative methods are used to estimate them and allow aggregation across market and non-market goods and services in the measurement of production

[3] The 2008 SNA notes a number of cases where market prices do not represent exchange values (e.g., in situations of transfer and concessional pricing (see paragraphs 3.131-3.134).

and consumption.[4] Here, it is useful to distinguish between transactions in goods and services and transaction in assets.

Another effect of using a transaction-based approach is that SNA accounts do not include consumer surplus that may arise with specific transactions since consumer surplus cannot be transferred between transactors. Consumer surplus is the difference between what is paid for a good or service and what a person is willing to pay. For instance, one may buy three litres of milk a week at €3, but be willing to pay more, say €4 giving a consumer surplus of €1.

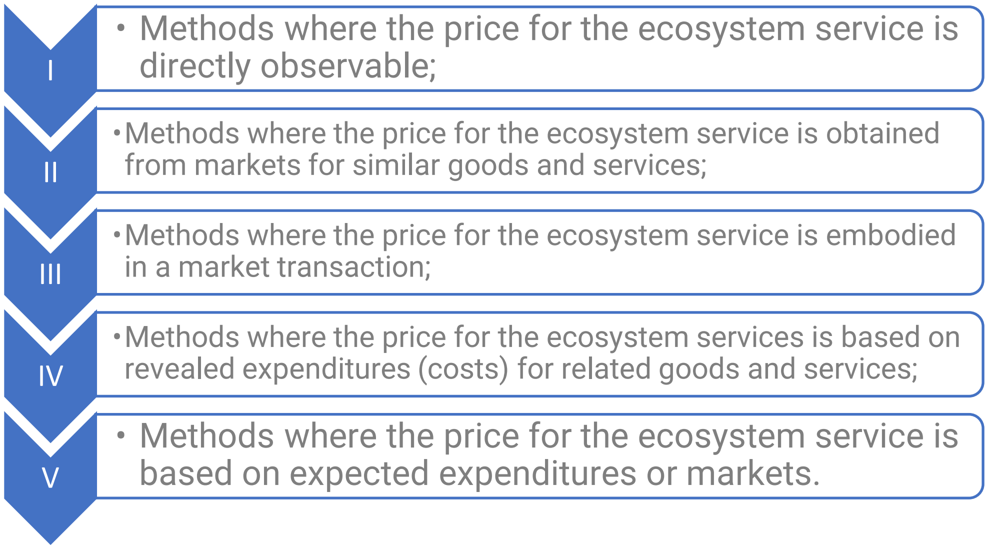
2.3.1.1 Exchange values for ecosystem services

Two primary alternative methods are described in the SNA in relation to transactions in goods and services namely: (a) market prices of similar or analogous items (adjusted for quality and other differences as required) (2008 SNA, para. 3.123); and (b) where no appropriate market exists, prices may be derived by the amount that it would cost to produce them currently (UN et al. 2009; para. 3.135). Cost-based techniques are commonly applied in estimating the value of government supplied services including education, health and defence.

Relevant prices are not often available for ES since many of the services provided by ecosystems are not directly traded in markets. Nonetheless, over the last decades, a number of techniques have been developed for placing a value on non-marketed goods and services including ES. Following a similar framing to the SNA, the SEEA EA recommends that valuation methods for ecosystem services be applied in the following order of preference – see figure 2 below.

[4] Note that the use of these alternative methods to estimate exchange values highlights that the estimation does not require the actual exchange of money (cash or equivalent).

Figure 2: *Preference order for valuing ecosystem services. Source: UN et al., 2021*



2.3.1.2 **Exchange values for ecosystem assets**

Exchange values of ecosystem assets are required to underpin entries in the asset accounts and balance sheets, i.e. exchange values for each asset are required at the opening or closing of the accounting period. The ideal source of exchange values for assets at balance sheet dates are prices observed in markets (e.g., valuing share portfolios using market prices at balance sheet date). Where there are no directly observable prices from markets, the SNA describes two approaches for estimating the exchange value of an asset. The first is the written down replacement cost approach which recognises that the value of an existing asset (most commonly relating to produced assets such as buildings and machinery) at any given point in its life, is equal to *"the current acquisition price of an equivalent new asset less the accumulated depreciation"* (2008 SNA, para. 13.23). The second approach entails using *"the discounted present value of expected future returns"* (2008 SNA, para. 3.137). This second approach is of primary relevance for ecosystem accounting since there are no commonly observable current acquisition prices of ecosystem assets that encompass the range of ecosystem service values supplied by an ecosystem asset.

2.3.2 *Institutional arrangements*

Observed market prices are defined without expectation that the market in which exchanges take place satisfies a specific institutional arrangement or assumption. The 2008 SNA states "a market price should not necessarily be construed as equivalent to a free market price; that is, a market transaction should not be interpreted as occurring exclusively in a purely competitive market situation.

In fact, a market transaction could take place in a monopolistic, monopsonistic,[5] or any other market structure." (UN et al. 2009; para. 3.119). Given this, the general interpretation in accounting is that **market prices should reflect the current institutional context**, i.e., the current market structures and associated legal or regulatory arrangements. Consequently, market prices used in national accounting will likely reflect the presence of various market imperfections from the perspective of economic theory.[6]

Valuation in the SNA is *ex-post*. This means that transactions are described as they have actually occurred, which is distinct from *ex-ante* valuation which describes values under various assumptions, such as that markets are functioning optimally or that externalities would be internalised[7]. In case of ES leading to SNA benefits, the institutional context is evident, as these ES contribute to benefits already exchanged / priced in markets. However, in case of non-SNA benefits, often no market prices are available and a non-market valuation technique should be applied for ES suitable to the institutional context.

Linking back to the transaction-based system of the SNA, another way of interpreting the accounting approach is to recognise that, in principle, each transaction is considered distinct and recorded in relation to the specific economic units undertaking the transaction and the associated context. In this situation, every transaction can be seen to have its own unique context and hence, conceptually, sets of accounts recording every transaction between every transactor might be compiled. In practice, this is not a necessary or appropriate ambition and hence transactors are grouped, e.g. into institutional sectors (households, governments, etc) and industries (agriculture, manufacturing, etc), to show in more aggregate terms the relationships between these different actors in the economic system.

The nature of ecosystems is that they are managed by a variety of institutional units at landscape level. Through its spatially explicit approach, the SEEA EA has the potential to increase the "institutional resolution" for non-market valuation of ecosystem services by integrating information on the spatial context including relevant institutional units. Nonetheless, for non-market valuation, where the values of the transactions are not revealed directly, it may be necessary to develop fin
er level groupings of institutional units than usually done in economic statistics such that the values of ecosystem services and the context surrounding their supply and use is appropriately well-defined. For example, it may be relevant to separate households into different populations or community groups or to group economic units according to their location and proximity to certain ecosystems. Using such finer level groupings it is then possible to assess variations in institutional context that will impact on the estimation of non-market values and thereby recognise that iinstitutional regimes are specific to ecosystems and resource characteristics (Ostrom, 2010).

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[5] Monopolistic and monopsonistic markets are those where there is either a single supplier or single purchaser, respectively.
[6] The existence of market imperfections will also imply that there is a difference between exchange values and associated welfare values.

[7] Prices that would occur when externalities would be internalised are called shadow prices i.e. marginal private cost plus the externality.

Adapting valuation methods to local rights regimes improves representation of household and local community use of ecosystems and will be of policy importance in accounting for themes such as biodiversity and protected areas.

These considerations will be of particular relevance when using value transfer techniques. As noted in the SEEA EA, since market prices are unlikely to be estimated for all transactions in ecosystem services, it will be necessary to apply value transfer techniques that take into consideration variations across location including institutional context and ecosystem type (UN et al. 2021; para. 8.31). Value transfer using meta-analysis may be particularly prone to “institutional mismatch” as estimates often come from many different countries, and are often only adjusted for ecosystem type, but not for variation in institutional regimes (see Grammatikopoulou et al. 2021; Lindhjem and Navrud et al. 2008), notwithstanding the fact that there are likely to be differences in access and property rights (institutional context) in different locations (UN et al. 2021; para. 9.79). Future meta-analysis,purposed specifically for application to ecosystem accounting, may be sensitive to differences in rights regimes across source study sites and with the accounting application location. These and other considerations in the application of value transfer techniques relating to institutional context in the valuation of ecosystem services are described in Chapter 6.

Compilers in different countries must consider their local context and institutional structures. For example, constitutions, legislation or social norms vary across countries in how they grant use and property rights to various private, community or government entities, including households, businesses, and public authorities. This results in variations in the degree to which ecosystem services are marketed or non-marketed. In keeping with ecosystem accounting’s spatially explicit approach, different valuation methods may be needed to reflect different geographically specific institutional settings (e.g. in different jurisdictions). The necessary adaptation of valuation methods to spatially varying institutional conditions can result in a wide range of valuation estimates for the same ecosystem services across types of jurisdictions. Different institutional settings require documentation to underpin cross country comparisons of monetary accounts.

Comparisons of accounts estimates across countries is possible notwithstanding the variation in institutional contexts and methods as the exchange value underpins monetary values recorded in the accounts.

2.3.3 *Consistency between monetary and physical flows*

In the SEEA EA, physical and monetary measures of ES flows are aligned so the monetary accounts reflect physical quantities from the physical accounts multiplied by **unit prices** of the services provided (if such exist). Generally, the compilation of accounts in monetary terms will require the use of data in physical terms. There are however exceptions as it is the case for example with the value of visual amenity services derived from living in an enjoyable landscape. The value of that ecosystem service can be estimated directly in monetary units albeit taking into consideration the number of people benefitting from the services. Where the value is estimated directly, the monetary values can be decomposed into quantities and implied prices to ensure consistency between the monetary and physical ES flow accounts.

The physical quantities of ecosystem services may vary considerably within a country or at a sub-national level. This does not mean, however, that both the price and quantity components of the valuation vary equally. It is important to emphasise that the SEEA EA focuses on valuation of final ES that is on those services that give rise to benefits, that is, if there are users that benefit from the services. For example, air filtration that takes place in remote areas where there are no direct beneficiaries, is not a final ecosystem service and it is not recorded in the supply and use tables. Therefore, physical and monetary ES flows will be dependent upon the geographic location and socioeconomic characteristics of the users. This is why it is important to model and value ES spatially, wherever possible.

SEEA EA recognises that entries in national accounts will usually be an aggregate of multiple transactions in a specific good or service over an accounting period (e.g., all sales of bread in one year) or an aggregate of multiple assets of a specific type at a balance sheet date (e.g., all registered trucks at 31 December). Further, accounts are compiled overtime so as to generate time series. In this way, time series of accounting entries based on exchange values will be compiled for various goods and services and types of assets. All accounting entries are recorded at the respective points in time at their nominal values – i.e., the prices applying at the time of the transaction or balance sheet entry. (UN et al, 2021; para. 8.21).

2.4 Linking exchange values to other measures of economic value

The measurement of economic value in monetary terms can be undertaken using concepts other than exchange values. This section touches upon the discussion about exchange and welfare values. A number of technical aspects of the relationship between exchange values and welfare values that include consumer and producer surplus are described in SEEA EA Annex 12.1. These arguments are not repeated here but interested readers are encouraged to look at the text in the SEEA EA. The focus here is on some specific and common questions that arise in the compilation of the accounts and in their presentation to users.

Price versus value: The price of a good or service is what is paid for a unit of the good. It is not the full economic value of the good to the purchaser because there is normally some consumer surplus derived from the purchase. If there is no rationing involved, people will continue buying goods until their marginal willingness to pay (WTP) equates the market-clearing price at which the goods are offered. That is also called the **marginal value** of the good. Therefore, the economic valuation of a given quantity of a good or service should be distinguished from the estimation of a price or marginal value for a given good or service that is used to derive an exchange value.

Marginal value, total value and consumer surplus: The equilibrium price when demand for a good is equal to the supply is related to the marginal value in the following way: It is equal to the individual's WTP for the last unit and the firm´s WTA to sell the last unit. The total expenditure is the price times the total quantity purchased. The total WTP, however, is greater than that because many individuals had a higher WTP than the price.[8] Consumer surplus is the difference between the total willingness to pay and what is actually paid for the good or service, or would be paid in case of exchange. The accounts record the latter and not the former.

Cost of production versus value: The average cost of producing a good or service is not equal to its value to the consumer, although the more expensive it is to produce the higher its price is likely to be because, for an exchange to occur, the marginal value to the consumer must at least be as high as the cost of supply of the last unit. In the SNA, the exchange value of a number of services is estimated using their cost of production because there is no market for them and hence no price. This is the case with public goods such as defence or public health, which are provided by governments and other authorities.

The use of production cost data in this context, however, does not mean that levels of provision are unrelated to economic values. A link can be made through the political process that determines the level of provision. Thus a given level of spending on health, education, transport etc., reflects society's collective WTP for these services through taxes and user charges.[9]

Replacement vs restoration cost: The SEEA EA makes a distinction between the cost of replacing a specific, individual ecosystem service (e.g., water purification) – replacement costs – and the cost of restoring an ecosystem as a whole – restoration costs. Since the focus of the SEEA EA is on estimating the contribution of ecosystems to a wide range of benefits received by different economic units, all of the ecosystem services that flow from an ecosystem should be valued. The use of replacement cost approaches is consistent with this intention. Moreover, the application of the restoration cost approach would make it problematic to assess the benefits of ecosystem restoration, as these benefit would by definition be equal to the cost.[10]

WTP versus WTA: Many commentators argue that values of ES based on WTP underestimate the worth of the goods and services because they take into account the income limitations of the persons receiving them. The statement that the valuations are constrained by a person´s income is true, but that applies to all valuations of marketed goods and services and is not special to ES. The same commentators sometimes also argue that using WTA to value a loss of ES would get around this limitation. That is a misconception. Both WTP and WTA are related to a person´s demand curve; the difference is the reference level of well-being against which the figures are measured. WTA is not dependent of a person´s income. Reviews of experimental valuation literature have shown the difference in WTP compared to WTA is highest for non-market goods, next highest for ordinary private goods, and lowest for experiments involving forms of money (Horowitz and McConnell 2002).

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[8] A distinction can also be made here concerning the aggregation of private and public goods. Private goods are aggregated horizontally (i.e. summing across quantities used by each individual) while public goods are aggregated vertically (i.e. for a given quantity used by all individuals).
[9] While this argument holds under certain conditions, it cannot be extended to say that marginal WTP is equal to marginal cost of supply and therefore that provision (at whatever level it is) is optimal. Further, there will likely be some consumer surplus that will not be reflected in estimates based on the cost of production.
[10] The SEEA allows to assess costs and benefits separately, thereby allowing to inform return on investment type of policies in the context of ecosystem restoration.

The difference in WTA/WTP ratio measures the consequence of assigning a property right one way or the other (ibid). The initial establishment of property rights, especially for environmental and other public amenities for which property rights are unclear, has large implications for valuation and can have large implications for environmental policy (Knetsch 1990). For example, Horowitz and McConnell (2002) found a WTA/WTP ratio of approximately 7 for preserving land from development in the studies they reviewed. The findings on differences in WTA/WTP is significant to the extent that non-market valuation methods used for accounting purposes assume distributions of rights. This is relevant for instance when applying valuation methods that fall in category iv and v (see Figure 2) and in assessing the source data in meta-analytic functions used for value transfer in accounting.

Ecosystem disservices. As described in more detail in the SEEA EA section 6.3.5, consistent with the accounting treatment of transactions, the recording of ecosystem services includes positive exchanges between ecosystem assets and economic units in a sense of contributing to benefits. However, it is also recognised that there is a range of contexts in which the outcomes of interactions between economic units and ecosystem assets are negative from the perspective of the economic units. Examples include the effects of pests on crop production, increases in disease from environmental vectors, such as mosquitoes or zoonotic episodes, and the presence of flies at a social event. For accounting purposes, although it is possible to record relevant physical flows and quantities, such as the number of pests, these quantifications are not considered to reflect positive exchanges of a good or service and hence are not considered as transactions or flows of ecosystem services (UN et al., 2021).

While ecosystem disservices and other negative externalities are not recorded as transactions in the SNA or the SEEA, their economic significance advocates treating them in other parts of an environmental accounting system. The presentation of information on ecosystem disservices can be undertaken using complementary accounting approaches as described in chapter 12 of the SEEA EA.

3. Valuation methods

As described in SEEA EA Chapter 9, since prices for ES are not generally observed, a range of methods have been developed for estimating them. Chapter 9 introduces a number of valuation methods to estimate prices consistent with the exchange value concept of the SNA. The objective of this chapter is to provide additional detail with respect to those methods and hence support compilers in applying the methods in practice.

The additional detail is provided in two ways. First, Section 3.1 gives a typology of primary valuation methods commonly applied in the environmental economics literature, based on the preference order introduced in chapter 2 and describes how they can be used to estimate exchange values for use in monetary ecosystem accounts. Second, Section 3.2 provides methodological details about each method, examples of their use and a discussion on their utility for accounting.

3.1 Typology of valuation methods

As introduced in Chapter 2, data that reveal relevant prices are not often available for ES since many of them are not directly traded in markets. Nonetheless, over the last decades, a number of techniques have been developed for valuing non-marketed goods and services including ES.

These valuation techniques have been classified in several ways but there exists no formally endorsed typology. Moreover, different names/descriptions of similar or identical methods are often found in the environmental economics literature , which can lead to confusion. Table 1 gives an overview of the most common valuation approaches based on material from the SEEA EA, the ecosystem service valuation database (ESVD) (De Groot et al., 2020) and the ISO standard on monetary valuation (ISO, 2019).

Table 1 lists the methods in the order of preference followed in the SEEA EA, recognizing that all of the methods listed can be used to derive estimates of the target valuation concept of exchange values. In this context, the order reflects the proximity of the methods to the preferred valuation method of observed market prices. There is a strong preference for using methods that translate observable and revealed prices and costs into the values required for accounting purposes.

Table 1: *Typology of valuation methods by SEEA EA preference order*

SEEA EA order	SEEA EA Category of method	Methods	Alternative description / related methods	Conceptual basis
1	**Prices are directly observable**	Market prices	Gross revenue; public pricing; monetary incentives	Market price
2	**Prices from similar markets**	Similar markets		Market price (adjusted)
3	**Prices embodied in market transactions**	Residual value; resource rent	Net factor income	Revealed preference - direct
		Hedonic pricing		Revealed preference - indirect
		Productivity change	Production function method	Revealed preference - direct
4	**Prices from revealed expenditures on related goods and services**	Averting behaviour	Defensive expenditure; averting cost	Revealed preference - direct
		Travel expenditure	As revealed in: consumer expenditure method; zonal based models; random utility model studies	Revealed preference - indirect
5	**Prices from expected or simulated expenditures or expected markets**	Replacement cost	Substitute cost; alternative cost	Revealed preference - direct
		Avoided damage cost	Cost of illness; human capital	Revealed preference - direct
		Simulated exchange value		Modelling

3.2 Valuation methods using primary data

The various methods listed in Table 1 are described below in greater detail. Some methods are more suited to the valuation of certain ecosystem services than others. For example, exchange values for provisioning services are likely to be estimated based on observed market transactions. The matching of methods to different types of ecosystem services is considered further in chapter 4.

3.2.1 *Directly observable values*

Directly observed values or market prices are the most direct method for measuring prices and estimating values for the accounts. For example, if a wetland provides water purification services and the owners of that wetland are able to charge the water company that abstracts the water for municipal uses, there is a transaction in ES provided by the ecosystem that can be recorded. Stumpage values charged to timber logging businesses are also an example of directly observed values. Another example of directly observed values relates to land rental prices in agriculture where markets exist to rent land for crop production or grazing. These rental prices may be used to derive prices for accounting purposes for the relevant biomass provisioning services. In all of these examples there is a direct link to SNA benefits. (UN et al., 2021; para. 9.28)

While the use of directly observed values is the preferred method, the resulting prices may provide accounting entries for the value of ecosystem services that might be considered low (i.e. where the monetary value of the contribution of the ecosystem is negligible). It is fundamental to recognize that this result is most likely a reflection of the existing institutional arrangements and is a result that is well understood in the economics literature. For example, it is well documented that the resource rents for natural resources that are extracted in open-access contexts will tend to zero (Hartwick and Olewiler, 1998). More generally, there is no single, fixed price that will be revealed within a set of institutional arrangements as the balance of factors being considered by the transactors will change over time. (UN et al., 2021; para. 9.29)

Nonetheless, provided the prices are from institutional arrangements that are sufficiently mature and widespread, the resulting prices should still be applied in ecosystem accounting, since the core intent is to show accounting entries that reflect the established market context and hence support analysis of the prices relative to those of other services and assets. Care should be taken to understand the size of markets and their maturity because the use of prices from small or immature markets may not be sufficiently representative for use in ecosystem accounting. To the extent that the recorded values are considered "low", there may be an interest in estimating complementary values on the basis of alternative institutional contexts and market settings. These hypothetical values should not be recorded in ecosystem accounts but may be presented in complementary accounts (see SEEA EA; UN et al., 2021, Chapter 12). (UN et al., 2021; para. 9.30)

Prices may also be observed in relation to non-SNA benefits. For example, in certain circumstances payments for ecosystem services (PES) schemes may provide a direct measure of the value of ES where the payments – for example from a government agency to a land manager – embody an appropriate price for a particular service for accounting purposes. However, most commonly, payments for ES and the associated institutional mechanisms are not designed to reveal prices for specific services. Instead, they are aimed at either supporting land managers in undertaking ecosystem restoration work or similar practices, or implementing broader government social policies – for example concerning income support. The advice is not to use data from payments for ES schemes in the estimation of prices for ES, unless there is clear evidence that the scheme provides a proxy for a market exchange of a specific service. (UN et al., 2021, 9.31)

Increasingly, governments are using environmental markets and PES schemes to establish structures and incentives to encourage behaviours and investments that are "nature positive". As the number and breadth of such policies increases, there will be a need to revisit the advice provided in these paragraphs, and to provide additional guidance on how prices revealed in environmental markets and PES schemes can be used to estimate exchange values for ecosystem accounting. Further investigation in this area will need to align with the treatment of such schemes in the SNA and Government Finance Statistics.

3.2.2 ***Methods where the price for the ecosystem service is obtained from markets for similar goods and services***

When market prices for a specific ES are not observable, valuation according to market price equivalents, or proxy markets, may provide an approximation to market prices. The SNA states the following: *"Generally, market prices should be taken from the markets where the same or similar items are traded currently in sufficient numbers and in similar circumstances. If there is no appropriate market in which a particular good or service is currently traded, the valuation of a transaction involving that good or service may be derived from the market prices of similar goods and services by making adjustments for quality and other differences"* (UN et al. 2009; para. 3.123). (UN et al., 2021; para. 9.34)

The most widespread example of applying this approach in the national accounts is the estimation of the imputed rent for owner-occupied dwellings, where the observed rents paid by tenants are commonly used to apply a "similar markets" method to estimate rents for owner-occupied dwellings (adjusting for variations in rents associated with the location and characteristics of the dwellings.). The technique is also applied in the content of household production of goods, including subsistence agriculture, where market prices for a good are used to estimate the value of production and consumption that is not exchanged on the market.

For example, when non-wood forest products (e.g. mushrooms) from one forest are marketed but those from a similar forest are not, the prices observed in the former can be used to value the non-wood forest products from the latter, adjusting for differences in products and other factors. In applying this method, the price from the similar market will need to be adjusted for any costs incurred to supply the good or service to ensure the price used refers to the ES. Note also that prices from similar markets will reflect prices of the existing institutional context in the same way as the directly observed values method. (UN et al., 2021; para. 9.35)

Implicitly, it is assumed that the flows of (non-marketed) ES (in this example, the harvest of mushrooms) are not significant enough that they would alter the observed price of, and demand for, the good or service from the similar market, i.e. the prices reflect a partial equilibrium.

3.2.3 *Methods where the price for the ecosystem service is embodied in a market transaction*

3.2.3.1 Residual value and resource rent methods

The residual value and resource rent methods estimate the value for an ES by first taking the gross output value of the final marketed good to which the ES provides an input, and then deducting the cost of all other inputs, including labour, produced assets and intermediate inputs (see Box 2).[11] Depending on the scope of the data (e.g. pertaining to a specific location or to the activities of an industry as a whole), the estimated residual value provides a direct value that can be recorded in the accounts or can be used to derive a price that may be applied in other contexts. The relevant considerations in deriving a price are described in the SEEA CF (ibid, Annex 5.1). (UN et al., 2021; para. 9.36)

Box 2: *Resource rent*

Output

- intermediate consumption

- compensation of employees

- other taxes on production

+ other subsidies on production

= gross operating surplus

- consumption of fixed capital (depreciation)

- return on produced assets

- labour of self-employed persons

= resource rent

= depletion + net return on environmental assets

Source: SEEA Central Framework; UN et al. 2012, p.153)

In practice, there can be a number of difficulties in applying these methods. First, the residual may reflect a combination of other non-paid and indirect inputs that could potentially make it difficult to identify the ES contribution . Second, the estimate is subject to uncertainty in calculating the value of all the ´paid´ inputs. For example, a farmer´s own labour or that of his family is usually not paid but has a market value (“mixed-income” in national accounts parlance)., Estimating such a value can be subject to error. Finally, this method is often most readily applied using broad, industry level data and

[11] The resource rent method is commonly applied at the industry (economic activity) level, using data from the national accounts. In case multiple goods are produced by the sector, cost elements need to be pro-rated using suitable assumptions.

the resulting price estimates may lack the granularity required for developing location specific monetary values. At the same time, since this method is applied based on observed data, the values and prices estimated using this technique will reflect the current institutional context and may provide a high-level framing for monetary values. (UN et al., 2021; para. 9.37)

The calculation is subject to variations in prices of outputs and inputs that can be considerable under market conditions, resulting in high annual volatility of estimates (Horlings et al. 2020). For this reason, statistical offices tend to use 3-5 year moving averages when calculating residual value.

Notwithstanding all these challenges, the method is widely applied by NSOs in valuing a number of ES and in preparing asset accounts. Further discussion of the method is given in chapters 4 and 5.

3.2.3.2 Productivity change method

In the productivity change method, sometimes called the production function method, the ES is considered an input into the production function of a marketed good. (UN et al., 2021; para. 9.38)

The productivity change method estimates an exchange value that is consistent with the SNA by estimating a production function directly, based on micro-level data on physical inputs and outputs at the site (e.g. farm) level, such as concerning land area, water use, labour, machinery, fertilizer etc. The econometric estimation of the equation provides a direct estimate of the marginal productivity of the input(s). Multiplying the marginal productivity by the price of the output gives the exchange value of the ecosystem service. (UN et al., 2021; para. 9.39)

Such a method requires the availability of micro data to make the estimation, which can be data intensive. It requires bespoke sampling of a population of production units representing the spatial variation in the use of the ES input. It requires generalizing micro-data assumptions about the cost function (production scale) and size of market (price competition) to the aggregate national level.

The productivity change method has been used to value the services provided by water and other inputs in agriculture, mostly in locations where detailed data to estimate a production function are available. It has also been used recently to value the productivity gains that result from keeping urban areas cool through the planting of vegetation (see the UK example in the context of UK ecosystem accounts in Section 5.11 of this document). Conceptually it is a strong method for the purposes of estimating exchange values and can be applied for many different ecosystem services. However, it relies on being able to define and estimate a production function and may also be most feasible at project or landscape scale rather than at macro-economic scales. (UN et al., 2021; para. 9.39)

3.2.3.3 Hedonic pricing

The hedonic pricing method estimates the differential premium on property value derived from proximity to some environmental attribute (e.g. a local park). In order to obtain a measure of how the specific environmental attribute affects the value of houses or other properties, all other characteristics of the house (e.g. number of rooms, central heating, garage space, etc.) are distinguished. Moreover, any unit of housing needs to be completely described by geographical, neighbourhood and environmental attributes. Once all characteristics and attributes that influence the property value are separated, the differential premium can be estimated assuming additive separability of all characteristics with respect to the total property value. (UN et al., 2021; para. 9.40)

The hedonic pricing method involves collecting large amounts of data on prices and characteristics of properties in a given area, and applying statistical techniques to estimate a "hedonic price function". This function gives a relation between the overall price and each characteristic, so that the slope of the hedonic price function with respect to each characteristic is equal to the implicit price (OECD, 2018). The appropriate functional form for this regression specification is arguable, but many empirical studies have estimated semi-logarithmic regression models of the form:

$$LnHP_{ijt} = \alpha + \beta_{1i}x_{it} + \beta_{2i}n_{it} + \beta_{3i}s_{it} + f_j + \varepsilon_{it}$$

where the dependent variable ($LnHP_{ijt}$) is the natural logarithm of the sale price for each property transaction *i* in property market area *j* in period *t*. The independent variables might include variables such as structural housing characteristics s_{it}, neighbourhood characteristics n_{it}, environmental characteristics x_{it}, or unobserved market characteristics f_j, and other unobserved components ε_{it}. The use of geographical information systems (GIS) and the availability of GIS data on neighbourhood amenities including ecosystem condition have increased the detail, flexibility and accuracy with which these attributes can be linked to residential locations (Kong et al., 2007; Noor et al., 2015; Łaszkiewicz et al. 2021).

Applications of hedonic pricing have grown substantially in recent years. The following are some examples:

- Agricultural activities (Le Goffe, 2000; Samarasinghe and Greenhalgh, 2009),
- Nature views (Paterson and Boyle, 2002; Luttik, 2000; Morancho, 2003; Gibbons et al., 2014),
- Open spaces (Bolitzer and Netusil, 2000; McConnell and Walls, 2005; Panduro et al., 2016; ONS, 2019; Łaszkiewicz et al. 2021).
- Water quality (Walsh et al., 2017)
- Wetlands (Tapsuwan et al., 2010)

See in particular ONS (2019a; b) and Horlings et al. (2020) for applications of hedonic pricing for national level environmental and ecosystem accounts. Although hedonic pricing has been widely used, there are some considerations in applying this method to ecosystem accounting that need to be taken into account:[12]

- Allowing for geographically fragmented and imperfect real estate markets which make it difficult to transfer models and values to different locations
- Spatial multi-collinearity of different variables and omitted spatial variable biases which mean that specification of the hedonic pricing models can be challenging

[12] For a detailed discussion see also Łaszkiewicz et al. 2021

- Non-linear distance decay of implicit prices for ecosystem amenity access which means that using direct distances to ecosystems “as the crow flies” may not be a good indicator of access to an ecosystem from a given dwelling.
- Spatial variation in ecosystem amenity access must be perceived by residents such that the amenity values are reflected in dwelling values.
- Ensuring that asset values are appropriately amortized to annual flows and that spatial aggregation across multiple properties is undertaken to generate ES values linked to specific ecosystem assets
- Checking for double counting with recreation values recognising that some adjustment of the recreation value estimates may be needed to isolate local residents´ use of recreation facilities that is captured in the hedonic estimates. (See also Barton, Obst et al., 2019; Horlings et al., 2020)

As a complementary approach to correct for imperfections Kolbe et al. (2019) suggest to simultaneously use the experienced preference method (see Chapter 3.2.5.3) for estimating marginal values for open green spaces in cities.

3.2.4 *Methods where the price for the ecosystem services is based on revealed expenditures (costs) for related goods and services*

3.2.4.1 Averting behaviour

The averting behaviour method, sometimes called the defensive expenditure method or averting cost method, is based on the assumption that individuals and communities spend money on mitigating or eliminating damages caused by adverse environmental impacts and the revealed expenditure demonstrates the value placed on associated ecosystem services. This is the case, for example, with extra filtration for purifying polluted water, or air conditioning for filtering polluted air. Many other examples exist as reviewed in Dickie (2017), the majority of which are applications of methods used to value reduced mortality and morbidity such that the contribution of ecosystems to those outcomes can be identified. (UN et al., 2021; para. 9.45)

These revealed expenditures are sometimes considered a minimum estimate of the benefits of mitigation. This is only true if it can reasonably be assumed that the benefits derived from avoiding damages are higher than, or at least equal to, the costs incurred for avoiding them. However, that may not be the case if there are spill-over benefits from the defensive expenditures (e.g., double glazing reduces noise damage but also improves thermal comfort / cuts heating bills). (UN et al., 2021; para. 9.46)

In addition, it is possible to include indirect costs related to the actions that individuals undertake to avoid the impacts of a poor environment. For example, the value of time spent indoors, when the preference would have been to go outside. An example is a study by Bresnahan, Dickie and Gerking (1997) who value the impacts of increased levels of ozone on peoples´ behaviour (see Box 3).

An advantage of the averting behaviour method is that it is easier to estimate the expenses incurred than to estimate the avoided environmental damage. A disadvantage is that the expenditures may not be very sensitive to the differences in environmental quality, so they are not spatially sensitive in the way damage functions could be. Also, care is needed to align the expenditure to specific ecosystem services, so to ensure that the expenditures reflect only the cost of avoiding environmental impacts rather than also reflecting matters of taste and consumption preferences.

Data gathered from studies of averting or defensive behaviour track actual transactions and are therefore consistent with exchange values.

Box 3: *Example of Defensive Expenditures*

Acute health damage, particularly in response to peak concentrations of ozone has been documented in a number of epidemiological and medical studies. Moreover, spending less time outdoors on bad air quality days – e.g. days when ozone concentrations exceed recommended standards – can effectively decrease exposure to pollution for certain at-risk groups. The study seeks to evaluate the extent of actual defensive expenditure and averting behaviour among members of these groups living in the Los Angeles area. The study found that two-thirds of the sample reported change in participants' behaviour in some meaningful way on days when air quality is poor. For example, 40 per cent of respondents claimed either to rearrange leisure activities or stay indoors during such days, and 20 per cent of respondents increased their use of home air conditioning units. In summary, bad air quality days appeared in this study to lead to significant changes in behaviour. It is reasonable to speculate that these behavioural changes impose non-trivial economic costs on respondents. For example, these burdens might take the form of the purchase and running of air conditioning with an air purifying unit or the inconvenience imposed by spending time indoors. However, the authors do not attempt to put a monetary value on these actions.

Source: Bresnahan, Dickie and Gerking (1997).

3.2.4.2 **Travel expenditures**

The travel expenditures to reach a site can be used as a basis for calculating a market price for the service provided at the destination. In using data on travel expenditures, it is necessary to make a distinction between two different methods that will be discussed in this section:

- The traditional travel cost method is commonly applied in welfare valuation contexts and hence the results cannot be directly applied for accounting purposes. However, when a demand curve is estimated, this curve can be used to model an exchange value by choosing a suitable point on the curve for example by intersecting it with an estimated supply curve – this is called the simulated exchange value (SEV) method. The SEV method is discussed separately in section 3.2.5.3.

- The consumer expenditures method uses estimated costs directly as a proxy for the value of the service. Travel expenditures (sometimes referred to as “outlays”) that are collected as an input to the traditional travel cost method can be used in this method but alternative sources for expenditure data can also be used. As these estimates are based on actual expenditures on marketed goods and services (e.g. fuel, train tickets etc.) they provide an exchange value.

In this section both methods are discussed, with the traditional travel cost method covered in greater detail as it organizes data that can be used in both the consumer expenditure and simulated exchange value methods. The section also discusses the scope of costs to include in these methods.

Under both the travel cost method and the consumer expenditure method the measurement of costs assumes that the goods and services consumed are a complement to the ecosystem service being valued. Thus, the demand for the ecosystem service can be estimated using the demand for the relevant travel goods and services as a proxy.

3.2.4.2.1 Travel cost method

The traditional travel cost method is based on measuring the costs incurred (and the foregone income) by households or individuals to reach a site and hence receive an ecosystem service from the site, usually in the context of recreation activity. By measuring these costs and the number of trips that take place for groups of visitors, and assuming that people have similar preferences, it is possible to derive a demand curve for the service from a site. The area under the demand curve measures the WTP of visitors for that service.

There is some agreement that the travel cost method is an effective approach to valuing recreation services for welfare analysis purposes (Bockstael and McConnell, 2007; Parsons, 2017). Most of the early research using the travel cost method approach was motivated by estimating the value of visits to recreational sites. In time, the method has also been adapted to be able to value changes in the quality of a site. Indeed, the last 50 years have witnessed a considerable evolution of travel cost method techniques, from simple aggregate demand models to very sophisticated analysis of individual level choices.

OECD (2018) differentiates between individual travel cost models that estimate demand for a single recreational site and models that estimate demand for multiple sites. The latter are particularly relevant for ecosystem accounting which aims to cover all recreation destinations within an accounting area. Each of these categories of models is discussed in turn.

The most basic single site model – discussed for purposes of illustration – requires two pieces of information for a sample of travellers, representing a variety of locations around the site: a) the number of trips that an individual or household takes to a particular recreational area over a period of time (e.g. a year); and b) how much it costs that individual or household to travel to the recreational area.

The information used in the travel cost method is usually collected through surveys carried out at the recreational site or through mail, phone or internet surveys. With these data, assuming people are similar in their preferences, an aggregate demand curve for access to the recreational site can be estimated, which explains the number of visits (i.e. the quantity) as a function of travel costs (i.e. the price) and other relevant explanatory variables. This demand curve is typically downward sloping since the number of trips normally declines as the costs of the trip become higher. Higher costs are associated with people living further away from the site. The points along the demand curve indicate consumer WTP to visit the site. The welfare value associated with the recreation services from the site is estimated as the consumer surplus, i.e. the area under the demand curve (and above the direct costs).

Initial applications of the travel cost method used what is known as the zonal travel cost method (Parsons, 2003). Zonal travel cost methods calculate aggregate visit rates (i.e. number of visits from an area divided by the population of that area) and average cost trips from different pre-defined geographical zones surrounding the recreational site of interest. This enables the estimation of number of visits per capita for each of the zones considered. The approach therefore looks at the average behaviour of groups of visitors rather than at individual choices. The application of this method is illustrated in Box 4.

While both zonal and individual methods provide estimates for a sample of the whole population of visitors, zonal methods have declined in use over time as more data has become available using analytical methods that allow calculations on an individual basis. Of course, where there are data limitations then a zonal travel cost method may be the appropriate approach. The method has been applied to value a wide range of outdoor recreation pursuits such as forest recreation (Christie et al., 2006), lake visits (Corrigan et al., 2007), recreational fishing (Shrestha et al., 2002), National Parks (Heberling and Templeton, 2009), deer hunting (Creel and Loomis, 1990) and many more. For more details on the models used see OECD (2018) and Parsons (2017).

***Box 4:** Illustration of zonal TC approach*

Suppose a national park receives visitors from four zones as shown in the figure below, with direct cost (e.g. fuel, transport) and visitor numbers as given in the table below.

***Figure A:** Location of zones*

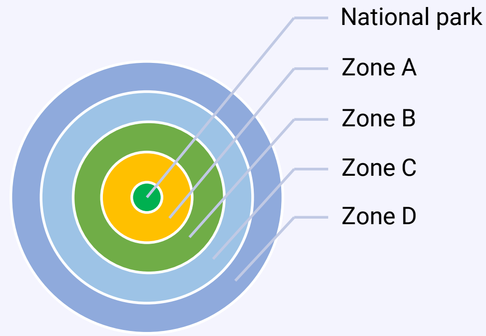
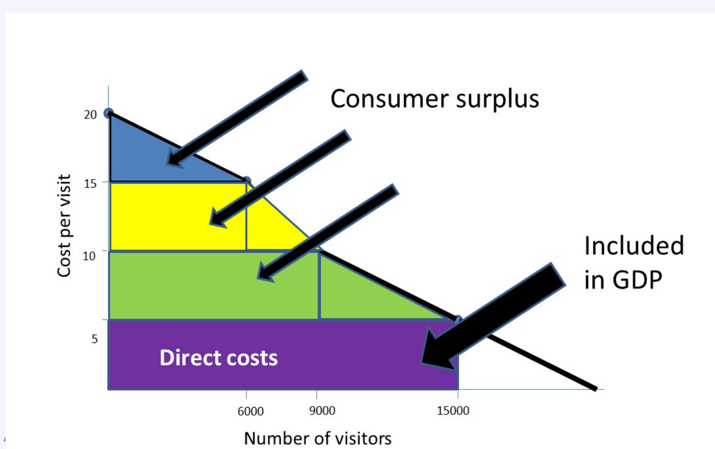


***Table B:** Travel cost, population and visitors from each zone*

Zone	Travel cost	Population	Visitors from Zone
A	5	25,000	15,000
B	10	25,000	9,000
C	15	25,000	6,000
D	20	25,000	0
Total		**100,000**	**30,000**

Based on this information, the demand can be established for each zone, assuming people have similar preferences. For example, while travel costs for people in zone A are 5, we assume that there would be a similar relative percentage of them willing to pay 10 as in zone B. This gives rise to consumer surplus depicted in shaded green in the figure below.

***Figure C:** Estimating consumer surplus based on travel cost*



Similar demand curves can be established for the other zones. The consumer surplus now is the area under the demand curve for each of the sets of visitors. For visitors from Zone C it is the blue area. For visitors from Zone B it is the blue area plus the yellow area and for visitors from zone A it is the blue areas plus the yellow areas plus the green area. The calculations are as follows:

Zone C Consumer Surplus = 0.5*6000*5= 15,000

Zone B Consumer Surplus = 15,000 + 6000*5 + 0.5*3000*5 = 52,500

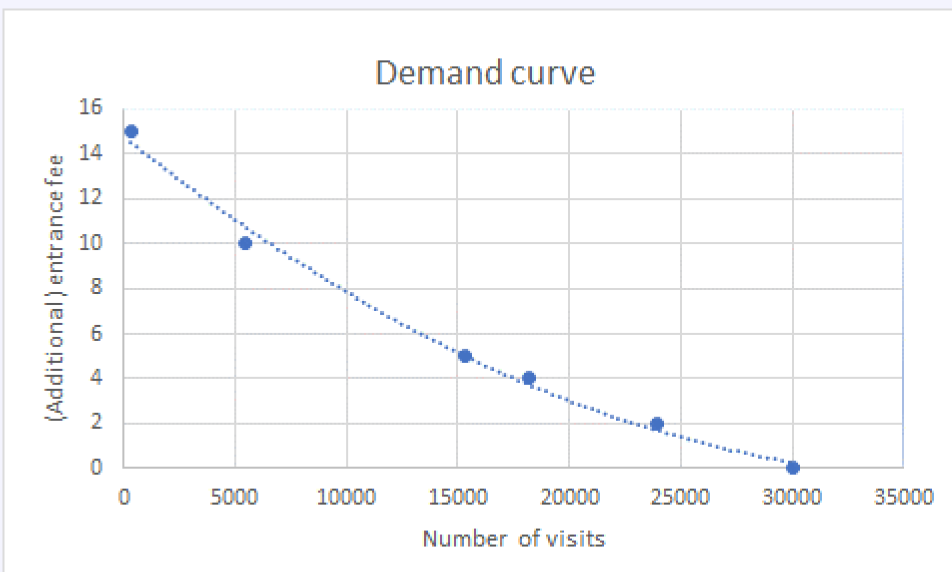
Zone A Consumer Surplus

= 52,000 + 9000*5 + 0.5*6000*5 = 112,500

Giving a total Consumer Surplus of 180,000.

An alternative way of presenting the demand curve is shown in Figure D below. To obtain this curve, first observe that the current situation is an entrance fee of 0 with 30,000 visitors. To obtain the curve, first a regression is carried out using the information provided in Table B of the number of visitors (expressed per 1000 population to allow for different number of inhabitants) depending on cost. In a second step, the actual number of visitors is estimated by applying the regression result, in case a hypothetical (additional) entrance fee would raise costs. For example, in case of an entrance fee of 2, there would be only 12,780 visitors from Zone A, 7980 from Zone B, 3180 from Zone C and 0 from Zone D, in total around 24,000 visitors. If there was an entrance fee of 4, there would then be 17,500 visitors. The choke price is around 15. Integrating under this demand curve also yields a value of 180,000 of consumer surplus.

Figure D: *Integrated demand curve*



A limitation of many single-site individual travel cost models is that they do not accommodate the presence of substitute recreational sites. In many real-world situations individuals have a wide range of substitute recreational sites, e.g. choice of which beach to go to, which river to go fishing in. In such cases an approach capable of modelling the discrete choice that consumers make between sites is required. The model typically used in such cases is the random utility model (RUM) (Bockstael et al., 1987).

The RUM is a discrete choice modelling technique where, in the presence of multiple recreational sites, individuals are assumed to choose which site to visit based on the selected site travel costs and characteristics, in comparison to the costs of travelling to, and characteristics of the different substitute sites. In recent years, the popularity of random utility modelling for recreational choice has boomed, in parallel with a decrease in application of single site or zonal travel cost models. It is now the dominant revealed preference method for recreation demand estimation (Phaneuf and Smith, 2005) and has been applied to a very extensive range of recreational experiences including fishing, swimming, climbing, boating/canoeing/kayaking, hunting, hiking, skiing, and park/forest/river visits, among others.

Moving beyond single site travel cost to a general method for ecosystem accounts, one could implement a bespoke national travel survey with respondents filling in travel diaries that identify individual trips and their purposes. An example of such an approach is the UK MENE survey, with application to national level recreation trip accounts.[13] Data on trip lengths with a purpose to visit natural recreation areas can be used to compute travel expenditures which are generalized from the sample to the population. Trip choice data can be used to estimate demand for each site using a Random Utility Model of travellers' destination choices considering travel costs and entrance price. A simulated exchange value takes a step further assessing what demand would be given that recreation sites charged for entry according to marginal costs of managing the recreation site.

One of the remaining issues is the problem of multiple purpose trips (Parsons, 2017). Many recreational trips are undertaken for more than one purpose. One solution to this problem has been to ask visitors (as part of the on-site survey) to estimate the proportion of the enjoyment they derived from their entire trip that they would assign to visiting the specific recreational area of interest. Total travel costs for the entire trip are multiplied by this percentage and this can be used as the basis for assessing travel costs at the recreational site.

3.2.4.2.2 Consumer expenditure method

The consumer expenditure method involves using estimates of the travel expenditures to visit recreational sites in the form of entrance fees, transport costs and /or accommodation costs to value ecosystem services directly. The rationale for the use of this method is that these expenditures represent the minimum WTP for the ecosystem service. As these costs are based on actual

[13] UK ONS MENE survey and expenditure method: https://www.ons.gov.uk/economy/environmentalaccounts/methodologies/uknaturalcapitalaccountstourismandrecreationmethodchanges

expenditures on marketed goods and services (e.g. fuel, train tickets etc.) they provide an exchange value.

To give an example, in terms of Figure C in Box 4 these expenditures are labelled there as “direct costs”. However, note that these expenditures will not have a clear relationship to the measures of consumer surplus obtained from the traditional travel cost model.[14] For example, using the same zonal travel cost example provided in the Box 4, this would give an estimate of 255,000 as exchange value.

Table 2: *Illustration of consumer expenditure approach*

A	75,000
B	90,000
C	90,000
D	**0**
Total	**255,000**

It is recommended to compare the results of the consumer expenditures approach with results of other methods (like SEV) and select the least cost alternative. Also, the consumer expenditures method should be clearly distinguished from other approaches that estimate the value of economic activity that would be lost (or at risk) in the absence of recreation sites. These estimates are considered complementary valuations to support decision making (see SEEA EA Chapter 12).

The travel expenditures such as entrance fees, transport and fuel, will be already included in the SNA. However, since these expenditures are being used as a proxy to value a complementary ecosystem service that is outside the SNA production boundary, there is no need to reallocate existing expenditures in the SNA sequence of accounts.

3.2.4.2.3 Scope of expenditures to include and other issues

When applying either of these travel cost based methods, the scope of expenditures to include is particularly important. The expenditures may include the following elements: i) direct expenditures in the form of return fares or petrol expenses and entrance fees; ii) wear and tear and depreciation of the vehicle used travelling to the site; and iii) the cost of time spent travelling to the site and iv) the cost of time spent visiting the site.

[14] To see this – the direct cost rectangular shape in Figure C does not determine the shape of the demand curve.

For consistency with accounting purposes the following recommendations are made:

- It is important to identify the proportion of expenditures that are related solely on the service and not for other benefits, such as the pleasure of driving or spending time with family. This can be done, for example, with a multi-criteria regression analysis (at the micro-level) or by using tourism statistics with sufficient granularity.
- The cost of overnight stays for ease of access to a recreation area can be included as long as these are a substitute for travel costs (e.g. choosing to sleep at home and drive, instead of staying at the recreation site). Other expenditures such as food are not recommended to be included.
- The entrance fee should be included in travel expenditures, but where the entrance fee also covers the supply of other services, e.g. a guide, entertainment, the price of these additional services needs to be deducted.
- The cost of time spent travelling to the site should be included. In terms of valuation, an exchange value based on the average wage rate is used as a starting point, noting that empirical work has been undertaken revealing that time spent travelling is valued at somewhere between a third and a half of the wage rate (OECD, 2018). It is common practice in transport research to value leisure time at 1/3 of the wage rate (Cesario 2006).

Box 5: *Example of valuation of recreation time spent in ecosystems*

Recently, mobility data from exercise apps (e.g. STRAVA) calibrated against path counters and mobile phone tracking have made it possible to estimate recreation populations' visitation frequency and time spent at recreation sites (e.g. Venter et al. 2020). The quantification of a physical recreation service metric as time spent on site raises the possibility of valuing recreation time on site directly, instead of indirectly through travel expenditure on complementary goods (food, fuel, lodging). **Value of time** spent in a greenspace is a measurable and intuitive indicator of enjoyment and a straightforward physical indicator of recreation benefit (Barton, Obst et al. 2019). Time spent traveling to a recreation site is usually treated as a cost. The exchange value of recreation time onsite is context specific. In transport literature the Value of Travel Times differentiates between different types of trips of different lengths at different times under different market conditions and different modes of transport. If foregone disposable income is used as a metric for recreation benefit of time onsite, it assumes that the alternative to recreation is work paid by the hour. It assumes that the recreationer has a flexible labour contract and that there is no unemployment. Despite an intuitive understanding of "quality time", measurement of an exchange value is highly subjective and context specific. Still, some kind of standardization of exchange value should be possible in order to take advantage of information provided by 'big data' sources for time on site. Recently, national level data sets from household time use surveys have been used to quantify the trade-offs between indoor leisure and outdoor recreation in forests (Berry et al. 2017), demonstrating the potential for using time on-site as a valuation index for national recreation accounting.

The inclusion of a cost of time has often been considered as inconsistent with SNA exchange values. However, provided the price used is itself an exchange value, (e.g., using observed wage rates) the inclusion of the cost of time is not a priori inconsistent with the SNA. At present, the inclusion of the cost of time is limited to time spent travelling to the site as this can be considered part of the overall WTP for the service. Consideration may be given to also including the time spent visiting the site, which might be particularly relevant in local recreation contexts where travel is not required (see also Box 5). National accounting precedents for the inclusion of time spent can be found in the valuation of own account capital formation. The SNA (UN et al. 2009; para. 6.127) states "As unpaid labour may account for a large part of the inputs, it is important to make some estimate of its value using wage rates paid for similar kinds of work on local labour markets." The cost of time is commonly applied in the measurement of values of unpaid household work. At the same time, a key question in this context is how the cost of time relates to the valuation of ecosystem services as it should not be included if it solely reflects the contribution of human capital.

3.2.5 *Methods where the price for the ecosystem service is based on expected or simulated expenditures for related goods and services.*

3.2.5.1 **Replacement cost**

The replacement cost method estimates the expected cost of replacing a single ES using a process that provides the same benefits but for which there are established costs or prices. It is sometimes called the substitute cost method or alternative cost approach. Replacement cost estimates are consistent with exchange values, as a similar principle is used when estimating cost of consumption of fixed capital.[15] (UN et al, 2021; para. 9.50)

The user of the non-priced service could be households (e.g. those benefitting from water purification services rather than boiling water or using water filters) or businesses (e.g. a farmer benefitting from soil erosion control services of trees rather than building levee banks). In both cases, if the two substitutes provide an identical service, the value of the non-priced good is the cost of the substitute.

The validity of the replacement cost method depends upon three main conditions:

- the substitute can provide exactly the same function of the good or service substituted for;
- the substitute is actually the least-cost alternative; and
- evidence indicates an actual demand for the substitute.

[15] 2008 SNA para. 1.67 states: "Similarly, consumption of fixed capital in the SNA is calculated on the basis of the estimated opportunity costs of using the assets at the time they are used, as distinct from the prices at which the assets were acquired. Even when the fixed assets used up are not actually replaced, the amount of consumption of fixed capital charged as a cost of production should be sufficient to enable the assets to be replaced, if desired."

In some instances, it may be difficult to relate the observed costs of the substitute with the target ES. For instance, mangroves may be planted or restored as a “green infrastructure” alternative to “hard” engineered flood defences. While the alternative cost of the engineering solution may provide a good measure of coastal protection services, it should be recognised that the mangroves are also likely to supply other services, for instance global climate regulation services and nursery services as a spawning ground for fish populations. Values for these other services should be separately estimated. (UN et al., 2021; para. 9.52)

Replacement cost values should be clearly distinguished from restoration costs – i.e. the costs required to restore an ecosystem asset to a previous condition or societally agreed condition. Restoration costs usually cover, implicitly, the supply of multiple services, and hence cannot be used directly to value individual ES. At the same time, restoration costs can provide policy relevant information in certain contexts and are therefore suggested as a possible complementary valuation approach in Chapter 12 of the SEEA EA.

3.2.5.2 **Avoided damage costs**

The avoided damage cost method estimates the value of ecosystem services based on the costs of the damages that would occur due to the loss of these services. Similar to the replacement cost method, the focus is generally on services provided by ecosystem services that are lost if the ecosystem is not present or is in sufficiently poor condition such that the services are not available. The validity of the avoided damage cost method depends also on conditions as listed for the replacement cost method although in this case there is no substitute service. Two conditions are relevant: (i) that the damages avoided can be related to a specific service; and (ii) that people would be willing to pay an amount to actually avoid the damage (i.e. if they are willing to accept the damage then this method is inappropriate). The avoided damage cost method is particularly useful for regulating services such as soil erosion control and flood control, air filtration, and global climate regulation services. (UN et al., 2021; para. 9.52)

The estimation of avoided damage costs will identify certain economic units who would be anticipated to benefit from the avoided damage costs as a result of the supply of ecosystem services. For example, the value of air filtration services may be related to avoided health costs to governments. However, this should not be interpreted as meaning this unit is the user of the service, it is solely a means to estimate the value of the service. In some contexts, prices based on both replacement costs and damage costs may be estimated. If this is possible the lower of the two estimated prices should be used. In most contexts this would be expected to be the prices from the replacement cost method. (UN et al., 2021; para. 9.53)

Sometimes this method is called cost of illness or human capital approach, particularly when applied to estimate the monetary value of a change in an ES such as air filtration through their impact on illness or life . The human capital approach estimates a gain in the quality of the air or water in terms of the increase in the present value of the earnings of the individual, and hence only measures the lost output of the worker due to illness or death. The cost of illness approach is more comprehensive than the human capital approach, as it includes not only medical expenses and lost wages (which the human capital approach would) but also includes a monetary value on suffering and the possibility of reducing the risk of mortality.

To obtain values and prices for accounting purposes, damages should be estimated using prices that are consistent with the exchange value concept. Care should be taken when applying cost of illness or human capital approaches, as gains in life years or values of statistical lives are often based on WTP or WTA studies on avoiding pain and suffering that will therefore include consumer surplus. However, measures based on avoided health care expenditure are in principle consistent with exchange values.

3.2.5.3 Simulated Exchange Value

The simulated exchange value (SEV) method (Caparrós et al., 2017) estimates the price and quantity that would prevail if the ES were to be traded in a hypothetical market. The SEV method is applied by using results from demand functions for the relevant ES (e.g. estimated using the travel cost method, discussed above, or stated preference methods). These are used to calculate the value of the ES that would occur if it was actually being marketed ("could be exchanged for cash" as in the definition of exchange value in Section 2.3.1) (UN et al., 2021; para. 9.55). The economic principles underlying the SEV method are illustrated in Figure 3.

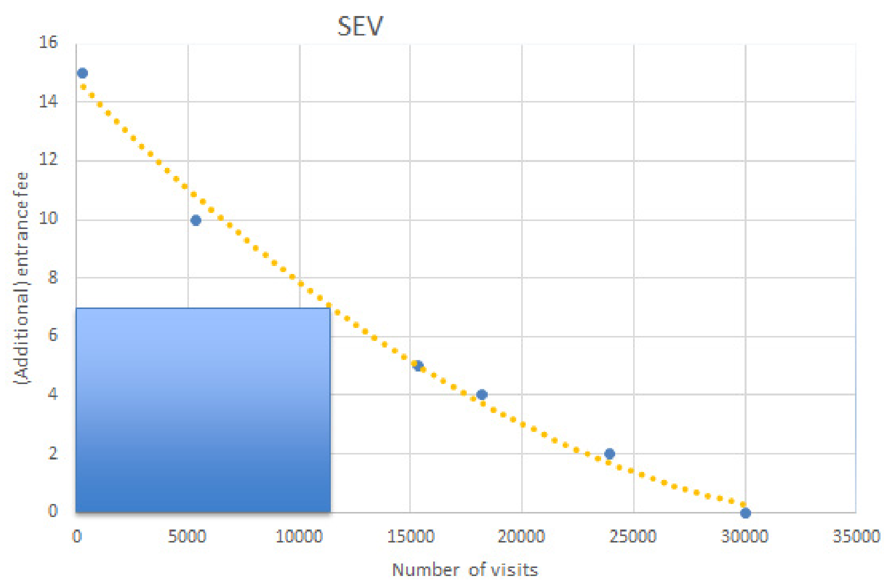
Using an appropriate demand function, standard microeconomic methods are used to yield the simulated price, which can then be used to estimate the value of the ecosystem services. For instance, if we were to use the demand function obtained from a zonal travel cost method discussed previously, an SEV value could be obtained assuming the ecosystem trustee were to maximize societal revenue if monopolistic competition is assumed (see Figure 3). Using the data in Box 4 this would correspond with a price of 7. This approach can be applied at various degrees of complexity and using alternative market structures. For instance, assuming there are only fixed costs (e.g. of maintaining the park) these costs would need to be deducted to obtain the exchange value. In case there are both fixed and variable costs (i.e. depending on visitor numbers), a cost function could be estimated, with the exchange value taken to be the point of intersection of the supply and the demand function. The SEV method has not been as widely applied as have the other methods described above.

The SEV results in an exchange value, specifically the value that would arise in case the ecosystem services in question were to be exchanged for cash (e.g. between the visitors and the ecosystem trustee). Where the simulated quantity (e.g. of visits) differs from the actual observed quantity (e.g. in terms of the number of visits), the price can be adjusted for accounting purposes in a subsequent step so that the simulated exchange value is unchanged, but the ecosystem service flow in physical units (e.g. actual visits) aligns with the physical supply-use table.

When stated preference methods (contingent valuation or choice experiment) are used for obtaining a demand curve they may assume a reallocation of use and property rights relative to the current governance situation in order to establish exclusive use rights, scarcity of access and a potential for a market transaction. This institutional context or market structure is specified in the valuation scenario of the stated preference study and presented to respondents along with a payment vehicle, e.g. a park entrance fee. Protest and zero responses and impacts on WTP can be substantial and have significant effects on the demand curve depending on how they are treated in estimation models (Campos et al., 2007; Oviedo et al., 2016). Given that the transaction and transactors are simulated, practitioners should check stated preference study validity reporting to confirm that protest and zero responses regarding the hypothetical payment scenario are sufficiently acceptable to respondents.

Another method that can be interpreted as an SEV method is the experienced preference valuation (or life satisfaction method) that is mainly used to value environmental conditions near the place of residence, and was also used to value urban green spaces (Krekel et al., 2016). The basis are representative surveys on the relation between well-being measured on a Likert Scale (e.g. 0 to 10) and a set of socio-economic data per person that can be used to explain the individual Likert value by a multi-criteria regression. Including, additionally, data on the provision of open green space at the individual living place into the set of explanatory variables, e.g. by GIS data, gives the opportunity to determine both the contribution of income as well as the contribution of green space to well-being. The first derivative of both relations gives an estimate for the marginal contribution of an additional unit income and green space to well-being and provides an income equivalent of an additional unit of green space. This income equivalent can be interpreted for accounting purposes as the maximum price that would be paid for an additional (marginal) unit urban green if a competitive market for urban green would exist.

Figure 3: *Illustration of SEV method*



Whereas hedonic pricing delivers best estimates when real estate markets are well functioning and everyone is able to choose a place of living that fits best to their different preferences, the experienced preference method provides an estimate under the assumption that preferences for green space play only a marginal role in location selection due to existing market imperfections. Amenity value can be expected to lie somewhere in between. Kolbe et al. (2019) presented a method how to model the influence of market imperfection and to calculate a (second) best estimate, lying between the estimates from hedonic pricing and experienced preference method.

3.2.6 *Other valuation methods*

There is a range of other valuation methods such as opportunity costs of alternative uses and stated preference methods that are found in the environmental economics and ES valuation literature. SEEA EA Chapter 9, section 9.3.7 provides a short introduction. These methods are not preferred methods for ecosystem accounting. If data based on these methods are considered for compilation purposes, then they should be checked for consistency with exchange value principles and adjusted as required before use in the accounts.

A particular note is made in relation to the estimation of non-use values. SEEA EA Chapter 6 – section 6.3.4 explains that non-use values are not within scope of the value of ecosystem services since there are no inherent transactions associated with these values. Where there is interest in estimating these values, a common approach is to use stated preference methods (choice experiment, contingent valuation).

Cost-based approaches have been used to determine the value of ecosystems to provide biodiversity – including as the basis of valuing the non-use values of ecosystems and species appreciation. For example, a cost rate per unit of biodiversity was estimated based on the legal obligation to restore the functions of ecosystems for conserving biodiversity and the cost of planned measures to reach the aims of EU directives (Schweppe-Kraft and Ekinci, 2021). This cost rate was then applied to the complete stock of biodiversity regardless as to whether the ecosystems and their specific contribution to biodiversity were endangered or not. To make sure that costs do not exceed marginal WTP, the cost rates were compared with results from contingent valuation studies on the WTP for additional measures to conserve and enhance the biodiversity stock. The annual flow was calculated backwards from the stock value by multiplying the stock value with a suitable interest rate. Cost-based approaches result in values consistent with exchange values.

4. Valuing ecosystem services

This chapter discusses the valuation of selected ES by applying the methods described in chapter 3. Section 4.1 introduces the reference list of the services as defined in SEEA and discusses the extent to which services are already captured in values recorded in the SNA. Section 4.2 introduces the tiered approach to valuing ES. Section 4.3 describes valuation methods for individual services (with pros and cons) classified into tiers and provides examples of valuation and references for further reading.

4.1 Introduction

4.1.1 *Typology of ecosystem services*

Ecosystem services are defined in the SEEA EA as the contributions of ecosystems to benefits used in economic and other human activity. They are categorized into provisioning, regulating and maintenance and cultural services. The measurement focus of SEEA EA lies on final ES, i.e., flows of ES between ecosystem assets and economic units. The ecosystem accounting framework also supports the recording of flows of intermediate ecosystem services, which are flows of services between ecosystem assets. Examples of intermediate services are nursery services and pollination.

Various classifications and typologies of ES have been put forward including the Millennium Ecosystem Assessment, the TEEB ES typology, the Common International Classification of Ecosystem Services (CICES)[16], the National Ecosystem Service Classification System (NESCS Plus)[17], and more recently Nature's Contributions to People (NCP) – as proposed by the IPBES. Based on these existing schemes, the SEEA EA has developed a reference list of ES (UN et al. 2021, Table 6.1). The list consists of 25 ES that are clearly defined that forms the basis for the ES accounts compilation as shown in Table 3 below. A correspondence/crosswalk between the SEEA ES reference list and the various classifications and typologies mentioned above has also been developed and is available on the UNSD SEEA EA website.[18]

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[16] See: https://cices.eu/resources/
[17] See: https://www.epa.gov/eco-research/national-ecosystem-services-classification-system-nescs-plus
[18] https://seea.un.org/ecosystem-accounting

Table 3: *SEEA EA Reference list of ecosystem services*

PROVISIONING SERVICES	
Biomass provisioning services	Crop provisioning services
	Grazed biomass provisioning services
	Livestock provisioning services
	Aquaculture provisioning services
	Wood provisioning services
	Wild fish and other natural aquatic biomass provisioning services
	Wild animals, plants and other biomass provisioning services
Genetic material services	
Water supply	
Other provisioning services	
REGULATING AND MAINTENANCE SERVICES	
Global climate regulation services	
Rainfall pattern regulation services (at sub-continental scale)	
Local (micro and meso) climate regulation services	
Air filtration services	
Soil quality regulation services	
Soil and sediment retention services	Soil erosion control services
	Landslide mitigation services
Solid waste remediation services	
Water purification services (water quality regulation)	Retention and breakdown of nutrients
	Retention and breakdown of other pollutants
Water flow regulation services	Baseline flow maintenance services
	Peak flow mitigation services
Flood control services	Coastal protection services
	River flood mitigation services
Storm mitigation services	
Noise attenuation services	
Pollination services	
Biological control services	Pest control services
	Disease control services
Nursery population and habitat maintenance services	
Other regulating and maintenance services	
CULTURAL SERVICES	
Recreation-related services	
Visual amenity services	
Education, scientific and research services	
Spiritual, artistic and symbolic services	
Other cultural services	
FLOWS RELATED TO NON-USE VALUES	
Ecosystem and species appreciation	

4.1.2 *Spatial nature of ecosystem accounting*

The SEEA EA takes a spatial approach to accounting, as the benefits a society receives from ecosystems depend on where those assets are in the landscape in relation to the beneficiaries. This spatial focus identifies the location and size of ecosystem assets, the ES provided, and the location of beneficiaries (households, businesses and governments). For example, the beneficiaries of water filtration ES are likely located downstream of the ecosystem asset that provides that benefit.

For accounting purposes, it is assumed it is possible to attribute the supply of ES to individual ecosystem assets (e.g., timber from a forest) or, where the supply of services is more complex, to estimate a contribution from each ecosystem asset to the total provision of services. For each recorded supply of ES, there must be a corresponding use. The attribution of the use of final ES to different economic units is a fundamental element of accounting. Depending on the ES, the user (e.g., a household, business, government or non-resident unit) may receive that service while it is located either in the supplying ecosystem asset (e.g., when it is catching fish from a lake) or elsewhere (e.g., when it is receiving air filtration services from a neighbouring forest). The supply and use of ES in physical terms is captured in Physical Supply and Use Tables (PSUTs).

4.1.3 *Logic chains*

In order to ensure a focus on the value of the contribution of the ecosystem, it is important to distinguish between the final ES and the resulting benefit (SNA or non-SNA). For instance, sales of a harvested crop would be recorded as the SNA benefit, whereas the ES is one factor contributing to the value of the benefit (others include labour and produced capital). Thus, valuing the crop provisioning ES using market prices for crops would overestimate the contribution of the ES.

The construction of logic chains – see Table 4 - is recommended by the SEEA EA to help distinguish final ES from benefits and also establish links with the ecosystem asset supplying the services. Below is an example of the generic logic chain (from SEEA EA). Annex 6.1 of the SEEA EA contains initial logic chains for a range of ES.

Table 4: *Generic logic chain (with example of air filtration services)*

Ecosystem Service	Common ecosystem type/s	Factors determining supply		Factors determining use	Potential physical metric(s) for the eco-system service	Benefits	Main users and beneficiaries
		Ecological	Societal				
Air filtration services	Forest and woodland	Type and condition of vegetation, especially Functional State (e.g. Leaf Area Index) and Chemical State (e.g. ambient pollutant concentration)	Ecosystem management; location type and volume of released air pollutants	Behavioural responses; and location and number of people and buildings affected by pollution	Tons of pollutants absorbed by type of pollutant (e.g., PM10; PM2.5)	Reduced concentrations of air pollutants providing improved health outcomes and reduced damage to buildings (non-SNA benefit)	Households; Businesses (through reduced damage to buildings)

Source: UN et al., 2021

4.1.4 *Coverage of ES in SNA*

The contributions of provisioning services in general are already included in the SNA production boundary to the extent they are inputs to products (e.g. crops; fish; timber), which are exchanged in the economy. The ES flows themselves are not visible in the accounts of the SNA. It is a common misconception that subsistence activities or illegal production (e.g. illegal logging of timber) are excluded from the national accounts, however in principle, such activities should be measured as part of the non-observed economy estimates. In such cases, the household would be treated as a non-incorporated enterprise, and hence the output would be recorded as agricultural output (not as household output). The same treatment applies for kitchen gardens and in theory also non-timber forest resources. It is likely that national accountants will only make imputations in case of significant non-observed activities (e.g. berry picking or hunting for bushmeat etc.). The ecosystem accounts may help improve such estimates.

The provisioning service of water supply is more complex largely due to the need to consider the role of ecosystems in underpinning water supply as discussed in SEEA EA Chapter 6 (section 6.4.2). Where water supply is recorded as a provisioning service, the associated value will be recorded in the SNA as an SNA benefit to the extent it is abstracted by the water supply industry. In case of self-abstraction by industries or agriculture (which is recorded for instance in physical terms in the SEEA CF), a transaction could be recorded in case payments for a permit are made (say to a municipality), and it needs to be assessed whether it can be considered as a proper estimate of the exchange value. In case of self-abstraction by households, this is considered production in the SEEA CF (as is production of energy).

The values of most regulating services are not included in SNA values to the extent they lead to non-SNA benefits. Some regulating services may be indirectly reflected in the accounts, such as air filtration as air quality may affect housing prices, but they will not be identifiable as such. In case of carbon-related services, when a country participates in emission trading schemes, has voluntary offsets or has implemented a carbon tax, related transactions are already recorded in the SNA (e.g. as taxes in the production account, or as financial transactions in the sectoral accounts) but these entries are not within the production boundary and hence are not treated as SNA benefits. As the measurement scope of climate regulation services is broader than the scope of existing carbon markets, there will be limits as to whether this information can be used to estimate exchange values depending on the institutional arrangements involved or the way in which services are quantified within the schemes.

In case of cultural services, recreation related services will be partially included depending on the institutional arrangements. The value of some services (e.g. amenities of urban green parks) will not only be reflected in the current accounts but also in capital accounts (such as housing prices).

Many ES will contribute to a range of physical and mental health outcomes for people and wider society. Such outcomes are beyond the scope of SNA benefits and, as such, the values of these outcomes are not recorded in the SNA. While the full value of health outcomes is not recorded, the contribution of ES to these outcomes should be included in the SEEA EA at their exchange value.

As a general observation, some ES values may be captured in land values, in particular those related to housing values and agricultural production. Therefore, these ES may be implicitly included in the rents charged and hence would be already within the scope of the SNA production boundary. However, regulating and cultural services which contribute to non-SNA benefits will not be incorporated in land values in general.

Where a regulating service, for example pollination, is an input to, for example, agricultural production, it is recorded in the accounts as an intermediate service. In principle the value of this service should also be captured in the production values associated with the agricultural production and hence also with the relevant land rents. Hence a separate valuation of the intermediate service needs to be undertaken.

One exception to the general expectation concerning land rents and non-SNA benefits concerns the valuation of visual amenity services, the values of which will often be associated with the values of property (dwellings and land) in a given location. In these instances, the ES value may be identified using hedonic pricing methods.

4.2 Tiered approach to valuing ES

Chapter 2 describes the overall preference order (5 categories) for valuation methods, following a similar approach as in the SNA, but extended with additional methods required for non-market valuation of ES. Chapter 3 discusses in detail the various methods that are feasible for obtaining exchange values. Not all methods can be applied for all ES, and in some cases, a method can only be applied for a specific ES. Section 4.3 provides recommendations on which of these methods is recommended for what ES. For this purpose, a tiered approach is adopted, whereby methods are ranked (for each ES) taking into account:

- Their proximity with estimating exchange values using the preferred approach of observed market prices (i.e. based on the generic preference order). This means for instance that methods that use prices from proxy markets for similar goods and services are ranked ahead of resource rent methods, which are ranked ahead of cost based methods.

- Whether the ES contributes to SNA benefits or non-SNA benefits. In case of non-SNA benefits, one usually has to apply methods based on revealed or expected expenditures of related goods and services.

- Their expected accuracy and spatial resolution, with Tier 3 providing highest accuracy and resolution. Higher tiers usually require better data availability.

In view of the existence of data limitations and the fact that different methods commonly give different values, it may be desirable to present values as a range, in which the lowest most conservative value is recorded in the accounts and supplementary information is provided to inform users of the range of the alternative estimates and may be complemented by sensitivity analysis for different methods (see for example Horlings et al. 2020). Table 5 summarizes the methods for obtaining monetary ES values in order of preference, that are discussed in greater detail for each ES in the next section.

Table 5: *Tiers of primary valuation methods for ecosystem services*

ECOSYSTEM SERVICE		Tier 3	Tier 2	Tier 1
Provisioning services				
Biomass provisioning services	Crop provisioning services	Land rental values Productivity change method	Residual value (spatial)	Residual value
	Grazed biomass provisioning services	Land rental values	Replacement cost Residual value (spatial)	Residual value
	Livestock provisioning services	Productivity change method	Replacement cost	Residual value
	Aquaculture provisioning services	Productivity change method		Residual value
	Wood provisioning services	Directly observed prices (stumpage values) Land rental values		Residual value
	Wild fish and other natural aquatic biomass provisioning services	Directly observed prices (traded quota prices) Productivity change method		Residual value
	Wild animals, plants and other biomass provisioning services		Similar markets	Residual value
Water supply		Directly observed prices (water rights) Productivity change method	Replacement costs	Residual value
Regulating and maintenance services				
Global climate regulation services	Sequestration component	Directly observed prices (emission trading schemes) – high spatial detail		Directly observed prices (emission trading schemes) – no spatial detail
	Retention component	Social cost of carbon (bespoke model)		Social cost of carbon (literature)
Local (micro and meso) climate regulation services		Productivity change	Averting behaviour	Avoided damages Replacement costs
Air filtration services		Avoided damages		Averting behaviour
Soil and sediment retention services	Soil erosion control services	Productivity change	Replacement cost Avoided damages	
Water purification services (water quality regulation)	Retention and breakdown of nutrients	Directly observed prices	Replacement costs Avoided damages	

ECOSYSTEM SERVICE		Tier 3	Tier 2	Tier 1
Regulating and maintenance services				
Water flow regulation services	Baseline flow maintenance services	Productivity change	Replacement costs Avoided damages	
	Peak flow mitigation services	Averting behavior (e.g. insurance premiums)	Avoided damages Replacement costs	
Flood control services	Coastal protection services	Avoided damages	Replacement costs	
Pollination services		Similar markets	Productivity change	
Nursery population and habitat maintenance services		Productivity change		Residual value
Cultural services				
Recreation-related services	Travel related	SEV+Random utility model	Consumer expenditure (spatial)	Consumer expenditure
	Local	Hedonic pricing		

4.3 Valuation of individual ecosystem services

4.3.1 *Biomass provisioning service*

Biomass provisioning services include: crop, grazed biomass, livestock, aquaculture, wood, wild fish and other natural aquatic biomass, wild animals, plants and other biomass provisioning services. In the majority of cases, the products are well defined and traded in markets. The difficulty is in estimating the contribution of the ES to the product's market value, which is already included in the SNA, and the spatialization of monetary data.

4.3.1.1 **Crop provisioning service**

Crop provisioning services are the ecosystem contributions to the growth of cultivated plants that are harvested by economic units for various uses including food and fibre production, fodder and energy. This is a final ecosystem service (UN et al. 2021).

4.3.1.1.1 Methods

There are three methods that are most commonly applied to value the crop provisioning service. The first approach is to use land rental data, such as payments by farmers to landowners. The reason why land rental data provide a good measure of crop providing services can be seen as follows: The ecosystem contribution for annual and perennial crops is provided by the land, which is combined with other inputs, such as labour, capital, seeds etc., to produce the final product - crops. It is possible to estimate the contribution of each input from a production function in which output (*Y)* is a function of the inputs (Labour, *L*), (Capital, *K*), (Land *W*) and (Other factors *Z)*. The production function is written as:

$$Y = F(L, K, W, Z) \qquad [1]$$

If all factors including land are priced in competitive markets, their prices will be equal to their marginal value products i.e., the increase in physical output of crops as a result of an increase of 1 hectare, multiplied by the price of the output.[19] In the case of land, taking its rental price per hectare as P_W this condition is written mathematically as:

$$P_Y \frac{\partial Y}{\partial W} = P_W \qquad [2]$$

[19] A specific case of the production function is the Cobb Douglas, which has been used frequently in the estimation of (1) for agriculture. For an example, see Grammatikopoulou et al. (2020).

The same applies to all other inputs. Further, if production is taking place in an economy that meets certain conditions for competitive equilibrium, then the production function also meets the following condition.[20]

$$Y = \frac{\partial Y}{\partial L}L + \frac{\partial Y}{\partial K}K + \frac{\partial Y}{\partial W}W + \frac{\partial Y}{\partial Z}Z \qquad [3]$$

Combining (2) and (3) gives:

$$P_Y Y = P_L L + P_K K + P_w W + P_z Z \qquad [4]$$

In other words, the total output (the sales of crops in monetary units) can be decomposed into contributions of labour, machinery, land and other factors. The total land rental value (price P_w times hectares W) provides a proper estimate of the contribution of the ecosystem towards crop production.

In case only part of the land is leased, the prices for non-leased hectares can be imputed based on leased hectares (possibly adjusted for difference in quality, e.g. soil fertility).[21] This is the method used in recent ES accounts for cultivated biomass where land rental data are available, such as the Netherlands and the UK. A key advantage is that rental data usually differ across areas (e.g. more fertile land fetches a higher rental price), so the valuation results are spatially heterogeneous.

The second approach would be to estimate the production function (1) directly, based on micro-level data on physical inputs and outputs at the farm level, such as land area, labour, machinery, fertilizer etc. The econometric estimation of the equation provides a direct estimate of the marginal productivity of the land to output. Multiplying this by the price of the output gives the exchange value of the land as a provisioning service. Such a method requires the availability of micro data to make the estimation, which can be problematic.

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[20] This is the case when the sector exhibits constant returns to scale, something that is assumed to hold in a competitive general equilibrium.
[21] An alternative to rental prices is to use agricultural land values (which often are available), and then calculate the implied user costs. This was done in the Netherlands accounts. They preferred rental prices, but land values provided very similar results.

The third approach is to apply the resource rent approach where the contribution of land may be deduced as a residual from the value of the crops when payments to all other factors have been subtracted (see Box 6 for a comparison of the resource rent and land rental methods). This can provide estimates for the value of land for different crops, sometimes for different administrative regions.

Box 6: *Rental value versus residual value*

The following example considers a case where agricultural output from a piece of land varies up to 60% between the highest and lowest years. Market data on land rentals, however, is relatively stable, increasing in line with inflation at 5%. The residual value calculation takes the gross value of output and nets out the costs of paid and unpaid labour, capital equipment that is either rented (or if owned, the depreciation) and material costs. The first method gives a value of ES that increases with inflation. It may go up more or less than the inflation rate as land prices are often subject to speculation. The residual value, however, is much more volatile and can even be negative. The value of unpaid labour is hard to determine and in many cases is not included in the calculation. Using such an approach it is common to take an average value over a number of years.

Year	-4	-3	-2	-1	0
Land Rent (€)	**100**	**105**	**116**	**127**	**140**
Output (MT)	100	90	120	**75**	110
Price Per MT (€)	5	5.4	**5.5**	**6.1**	**6.1**
Output (€)	500	484	662	455	669
Paid Labour Cost (€)	240	252	265	278	292
Unpaid Labour Cost (€)	80	73.5	100	75.2	116
Capital (€)	75	78.75	83	87	91
Materials (€)	40	42	44	46	49
Residual Value (€)	**65**	**38**	**170**	**-31**	**121**

4.3.1.1.1.1 Data sources and Tiers

In the absence of (detailed) agricultural statistics and assuming a weak estimate of output and value added in the national accounts for the agriculture sector, a Tier 1 approach may consist in applying a basic biophysical modelling technique (as described in UN 2022) to obtain coarse spatially explicit estimates of crop yield for the main crop types harvested in the country, for instance based on crop suitability models. These physical estimates could then be multiplied with a suitable price to obtain the value of the ES (i.e. a bottom-up approach). Such a price may be estimated by applying a resource rent type of calculation based on information in household surveys (if they exist), from an existing scientific study, or by applying a value transfer technique (e.g. using a price from a neighbouring country with similar socio-economic and ecological circumstances, for instance by using FAO databases). As the output of biophysical models will likely be very coarse, and price estimates will be crude, the result of a Tier 1 approach will be subject to high uncertainty.

Assuming availability of agricultural statistics and a good quality estimate of agricultural output in the national accounts, a Tier 2 approach would estimate the resource rent at the macro-level to obtain an estimate of the total ES value. In a second step, agricultural statistics (when available by sub-region) or a suitable biophysical model can be used to spatially distribute the resource rent across the ecosystem accounting area. If the data allows, this top-down approach could be undertaken, disaggregated by (main) crop types, and/or, if regional accounts are available, this approach could be undertaken at the subnational scale. Such a Tier 2 approach would obtain results that are consistent with the national accounts, and in terms of spatial detail align to the extent possible with agricultural statistics.

A Tier 3 approach – assuming availability of micro data - would consist in applying the rental method, as it provides a direct market-based value of the crop provisioning service. Land rental data commonly have also spatial identifiers and other meta-data (e.g. crop type), thereby allowing to also portray the ecosystem service in the form of a map. In some cases where there is speculation in the land market, an allowance for speculation may be necessary. In case rental data are only available for some sites, estimates for other sites can be based on value transfer methods discussed in Section 3.3.

The productivity change method is also considered a Tier 3 method. A strength of this approach is that it reflects spatial heterogeneity very well. However, a disadvantage is that the results may not align well with the macro totals (and concepts) reported in the national accounts.

4.3.1.1.2 Examples

Horlings et al. (2020) compare three approaches for valuing biomass (crops) provisioning service for the Netherlands, namely: resource rent; user cost of agricultural land; and lease/rental payments. The last approach is described here, as this is the preferred approach. In the Netherlands, about 30 per cent of farmers do not own their land, but lease it from landowners. This practice is partly regulated by the government, in the sense that every year maximum prices are specified separately for 14 different agricultural areas, reflecting differences in soil type and average yield. The farmer and landowner can freely negotiate prices as long as the agreed price remains below the government specified cap. The valuation of the ecosystem service itself is done by multiplying the extent of the agricultural land with the applicable rent prices (for each of the 14 areas). By this method, Horlings et al. (2020) obtain a value for this ecosystem service that varies between 1,097 and 1,452 million euro a year (over the study period 2010 - 2017).

4.3.1.1.3 Other considerations

A likely challenge with Tier 1 and Tier 2 approaches exists in estimating the user cost of fixed capital, as this requires information about the capital stock (e.g. of machinery) to estimate cost of depreciation and a return to capital. It should be noted that the use of market prices for crops in combination with crop yield provides an overestimate of the ES, as it includes the return to other types of capital used in production and hence double counts. This approach is therefore not recommended for ecosystem accounting purposes, unless when dealing with subsistence type of activities where other capital inputs are negligible. Another challenge likely consists in estimating the value of mixed-labour (i.e. wages for self-employed farmers). Reasonable assumptions need to be made.

4.3.1.2 **Grazed biomass**

Grazed biomass provisioning services are the ecosystem contributions to the growth of grazed biomass that is an input to the growth of cultivated livestock. This service excludes the ecosystem contributions to the growth of crops used to produce fodder for livestock (e.g. hay, soya meal). These contributions are included under crop provisioning services. This is a final ecosystem service but may be intermediate to livestock provisioning services (UN et al. 2021).

4.3.1.2.1 Method

There are three methods that are most commonly applied to value grazed biomass services. The first method, akin to the previous section on crop provisioning, is to use land rental data. In effect this treats pasture/grazed biomass as a crop.

The second method used to value pasture is the replacement cost method, in which the estimate concerns how much it would cost to feed the livestock if the pasture/grazed biomass was not available. As an example, consider a rangeland area of 2,600 ha currently producing 4,771 tons of forage for sheep and cattle.[22] If grazed biomass were not available then a substitute could be sorghum. From local data it is estimated that one ton of sorghum replaces 1.62 tons' of forage and to buy one ton of sorghum costs USD201/ton. So the value of the grazed biomass = 201*4,771/1.62 = USD592,000 per year.

A third alternative is to value the grazed biomass by applying the residual value method.

An advantage of the replacement cost approach is that it is linked to actual engineering/agricultural estimates of the costs of replacement. Hence, the required economic data for site-specific assessments is almost always accessible. The disadvantage is that there is no guarantee that the farmer would be willing to pay the estimated cost. As a check to the outcome of the replacement cost calculation it would be desirable to make an estimate on the contribution of grazed biomass based on a production function approach. Such a check could be made for some sites to see if there is a significant discrepancy.

4.3.1.2.2 Tiers

The use of land rental values is considered a Tier 3 method due to its proximity to exchange values and spatial resolution. The replacement cost approach is a Tier 2 method and would be well supported by undertaking a check on the estimates using some samples of land using the productivity change method. If there are major differences, an adjustment can be made to the replacement cost estimate. As for crop provisioning services, the residual value method is also considered a Tier 2 method if it is undertaken in a spatially disaggregated manner, of not it is considered a Tier 1 method.

[22] The example is based on data sample data from Africa analysed in Markandya (2002).

4.3.1.3 Livestock provisioning services

Livestock provisioning services are the ecosystem contributions to the growth of cultivated livestock and livestock products (e.g., meat, milk, eggs, wool, leather) that are used by economic units for various uses, primarily food production. This is a final ecosystem service. Note that distinct livestock provisioning services are not recorded if grazed biomass provisioning services are recorded as a final ecosystem service (UN et al. 2021).

4.3.1.3.1 Methods

The methods most commonly used for valuing livestock provisioning services are directly linked to the methods used for the valuation of grazed biomass provisioning services, since the ecosystem contribution to livestock provisioning will be driven, largely, by flows of grazed biomass. Thus, the primary methods include the rental value of the land used to raise livestock and the productivity change method. Residual value methods may also be applied.

When there is no market price information, or when the land is not privately owned, it may be necessary to obtain a value based on the replacement cost method, i.e. the cost of replacing all of the ecosystem inputs to the raising of livestock.

4.3.1.3.2 Tiers

The land rental value method and the productivity change method are considered Tier 3 methods. The replacement cost method is considered a Tier 2 method and the residual value method is a Tier 1 method.

4.3.1.4 Aquaculture

Aquaculture provisioning services are the ecosystem contributions to the growth of animals and plants (e.g. fish, shellfish, seaweed) in aquaculture facilities that are harvested by economic units for various uses. This is a final ecosystem service (UN et al. 2021).

4.3.1.4.1 Methods

Two methods are most commonly used to value aquaculture provisioning services. First, productivity change methods can be applied where the functional form is similar to that shown in equation (1) above. The output in the form of kilograms of fish, shrimp or other species can be seen as a function of the area of the pond, labour and other inputs.

Second, a residual value method can be used, where one can start with the value of the final product and subtract the contribution of each paid input, such as labour, capital etc. The residual can then be considered as the contribution of the ES. The difficulty with this approach (as in all applications of the residual value method) is that there are often other non-paid inputs into the production process, and as a result the residual is not only the return on the pond area. Furthermore, the residual is sensitive to changes in prices of outputs and inputs that vary under market conditions, making the calculation yield unstable estimates.

4.3.1.4.2 Tiers

The productivity change method is a Tier 3 method, while the residual value approach is a Tier 1 method.

4.3.1.4.3 Examples

An example of the productivity change method is from a study in Ghana where output of fish in kg per square metre of pond area was regressed against quantity of feed used per square meter of pond area; quantity of fishmeal fertilizer applied per square meter of pond area; the stocking rate per square meter of pond area and labour per square meter of pond area (Asamoah et al., 2012). The estimated function gives statistically significant estimates of the marginal productivity of each of the inputs listed above and also finds that if all factor inputs are increased by a given percentage (e.g. 1%), output per square meter increases by more than that percentage (i.e. >1%). In this study the increase in output was approximately 1.2 per cent. This additional increase of 0.2 per cent can be interpreted as linked to the return to the services provided by the ecosystem. If the regression had included size of the pond and not normalized by that variable, the coefficient on the size would be related to the returns to scale and would give the marginal product of the service. Such an estimate can be made to determine the contribution of the ecosystem to the flow of production goods from the system.

Greaker and Lindholt (2021) compile the resource rents for Norwegian aquaculture in the period 1984-2020. The starting point for calculating the resource rent is that production of a natural resource can be expressed by a production function, where one or more ecosystem services are included as input factors. Only a certain number of locations worldwide are suitable for aquaculture in terms of climate conditions, sea water quality and protection against the weather. These factors together may provide a resource rent, that is a rent beyond what could be obtained by investing capital and human resources in other activities. Resource rent is also derived from the limited access conferred by the combination of scarcity of these natural conditions - the aquaculture service(s) - and regulation of the number of licences. The study is compatible with the SEEA definition of the components of resource rent (considering intermediate uses, taxes and subsidies on production, input costs of labour, specific subsidies and taxes on extraction, capital depreciation, and return to produced assets). The authors estimate that the annual resource rent from 2000-2020 has on average been 8 billion NOK (roughly 1 billion USD depending on exchange rate). The authors discuss whether they are able to isolate resource rent from regulatory rent (owning a licence), market rent (limited access increasing market prices relative to perfect competition) and intra-marginal rent (due to company-specific knowledge and technology). The authors argue that the value of aquaculture licences (a regulatory rent) should not be considered part of resource rent, because these licences are the source of resource rent. The revenue from a well-designed auction of licences without time limit represents in principle the total Net Present Value of expected resource rents, including expectations about regulatory changes affecting licence access (Greaker and Lindholt 2021)

4.3.1.5 **Wood provisioning service**

Wood provisioning services are the ecosystem contributions to the growth of trees and other woody biomass in both cultivated (plantation) and uncultivated production contexts, which are harvested by economic units for various uses including timber production and energy. This service excludes contributions to non-wood forest products. This is a final ecosystem service (UN et al. 2021).

4.3.1.5.1 Methods

Three methods are most commonly used to value wood provisioning services. To apply these methods it is useful to assess the ownership status and management context (i.e. cultivated and/or natural) of the associated forest land, recognizing that not all forest land is privately owned with some under public ownership and some having unclear, unknown or disputed ownership.

In cultivated contexts, if a private firm pays a price for the right to harvest a given volume of timber or related product from a forest (i.e. the stumpage value), this can be taken as the basis for calculating the contribution of the ecosystem to the final product. The stumpage value represents a directly observed price for the ecosystem service and is the effective price for the right to harvest a tree, which will be less than the market price for the harvested timber after the tree is logged. Payment is made to the government, which implicitly has the right to exploit the land. Since the stumpage value will include costs incurred by the state or other agents in managing such land, those costs need to be netted out from the stumpage value to get the contribution of the ES.

Second, where forest land is privately owned and has a market price, an explicit price from a rental market for forest land may be used (analogous to the stumpage value just described) or an implicit rental value can be calculated from the price of the land as an asset and assuming a market rate of return.[23]

Third, the contribution of the forest ecosystem to the final value can be calculated using a residual value method based on the user cost of the forestland and expected revenue streams from harvesting.

In case of non-cultivated contexts, such as logging without land ownership, it is advisable to apply residual value methods based on estimates of resource rent. Depending on the circumstances, relevant information may be obtained from household surveys. Where the use of fixed assets (e.g. machinery) is negligible, estimation can be simplified. For example, in the case of subsistence logging/harvesting, an assumption can be made to use the market price of timber, as the use of fixed assets may be negligible and the opportunity wage cost may be very low.

4.3.1.5.2 Tiers

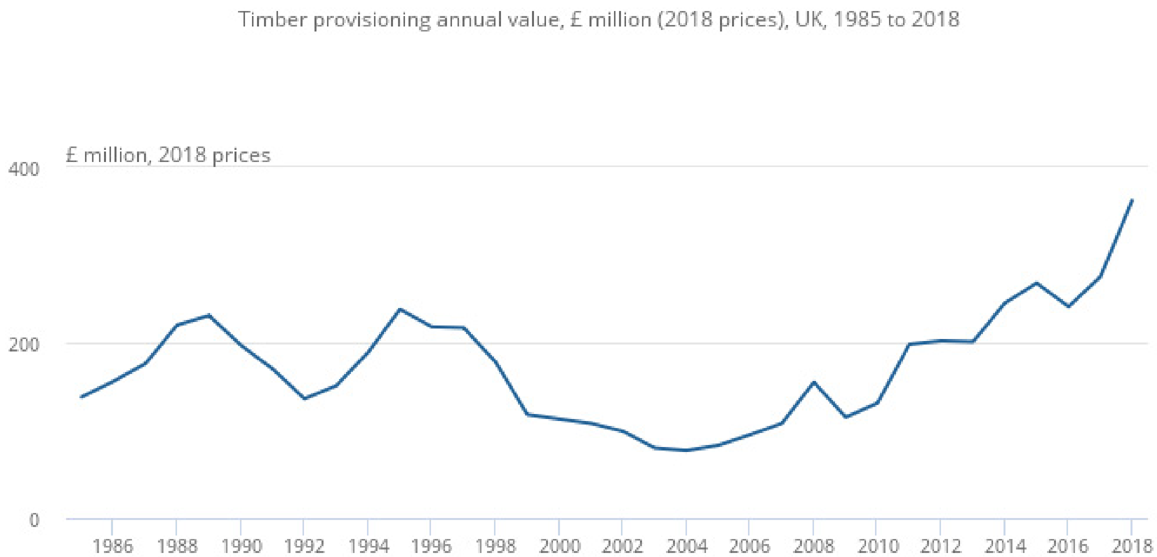
Directly observed prices in the form of stumpage values or land rental prices are considered Tier 3 methods. Residual value methods are considered to be Tier 1 methods.

4.3.1.5.3 Example

The UK (ONS, 2019a) estimates the timber provisioning service using stumpage prices (which are specified per areas) and the actual harvested amounts of timber. The increase in value shown in Figure 4 is to an extent driven by the large increase in stumpage price between 2004 and 2018.

[23] As in the case of agricultural land if the rental market data is incomplete it may be possible to obtain estimates for missing locations based on value transfer methods discussed in Section 6.1.

Figure 4: *Timber provisioning values in the UK.*



Source: Office for National Statistics and Forestry Commission

4.3.1.6 **Wild fish**

Wild fish and other natural aquatic biomass provisioning services are the ecosystem contributions to the growth of fish and other aquatic biomass that are captured in uncultivated production contexts by economic units for various uses, primarily food production. This is a final ecosystem service (UN et al. 2021).

4.3.1.6.1 Methods

There are three methods that are most commonly used to value wild fish provisioning services, noting that outputs included here are wild fish and other uncultivated biomass (e.g. seaweed). ES estimates of fish and other biomass harvested from different bodies of water have been made in several studies using market-based methods.

First, the preferred method is to use e market price data on payments to access wild fish resources if they are available . As the SEEA CF notes, such a price, established by a trading mechanism such as individually transferable quotas, reflects a directly observable estimate . Unfortunately, such mechanisms are rarely applied, with the exception of a few cases where they cover only a few species.

A second method is to apply the productivity change method and estimate production functions for different fisheries, as has been done in a number of studies (e.g. Hannesson, 2011; Armstrong et al., 2016). In these studies, the catch is regressed on inputs to measure the effort made to catch fish (boats, gear, personnel) as well as the stock of fish available. From this, the marginal product of the stock can be estimated in physical units. Multiplying this by the final price of the fish produce gives the contribution of the fish stock to the final value-added based on market prices at point of sale. Complications can arise, however, because the relation between catch and capture fishery depends not only on the stock but also on the quality of the habitat. In conducting the estimation, based on historic and/or cross-sectional data, the results for the contribution of the stock will be biased if measures relating to these factors are not included in the estimation.

A specific application of the productivity change method has been the valuation of seagrass. The valuation of seagrass is quite widespread and a useful review of methods can be found in Dewsbury et al. (2016). Overall, seagrass provides a range of services including the direct use of dead seagrass as insulation and formation of dykes; indirect uses to reduce the impact of wave action, thus reducing erosion and reducing sedimentation; and providing a nursery for juveniles of various fish species. In relation to wild fish biomass provisioning services, productivity change methods have been used based on how seagrasses affect catch-per-unit effort for fish, although there are some criticisms about the way in which this contribution is measured and there is scope for improving the methods.

A third approach is the residual value or resource rent method, where the final value of harvested biomass is estimated and all costs are netted out to obtain a residual that is the contribution of the ecosystem. Countries such as Norway and the UK use a residual value approach to value the contribution of such biomass (see example below for the case in the UK).

4.3.1.6.2 Tiers

The directly observed market price method and the productivity change method are considered Tier 3 methods. It is generally the case that these methods will be difficult to apply since directly observed prices are rarely found and the data required to apply the productivity change method, including estimating a production function that measures of the quality of the ambient environment, may be challenging to source. The residual value approach is considered a Tier 1 method.

4.3.1.6.3 Examples

To calculate marine fish capture in the UK's exclusive economic zone (EEZ), the Marine Management Organisation International Council for the Exploration of the Sea statistical rectangle factors were used. The overall fish capture provisioning service physical flow presented in this article represents landings (tonnage) from UK waters. Physical flow presented in 1997 to 2015 UK Ecosystem Service Accounts was sourced from the Food and Agriculture Organisation and represented the fish capture of UK vessels, not fish capture from UK waters. Valuations are calculated using net profit per ton (landed) estimates by Seafish[24], for different marine species. Net profit per ton is calculated using Seafish economic estimates for fleet segments and 2013 to 2014 Marine Management Organisation data on landings[25] by stocks (landed value and landed weight) and landings by stocks and species (in cases where species are not managed by total allowable catches). Annual net profit per ton (landed weight) is multiplied by tons of fish captured (live weight) for a specific species. This data is aggregated for overall annual valuations of fish provisioning from the UK EEZ.

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[24] A UK public body supporting the seafood industry. See: https://www.seafish.org/

[25] Landed weight is the weight a product at the time of landing, regardless of the state in which it has been landed. Landed fish may be whole, gutted and headed or filleted. Live weight is the weight of a product, when removed from the water.

Net profit per ton is not the same as residual value because the former includes a normal return on capital, which should be deducted before arriving at the residual value. It is thus an overestimate of the residual value.

A key limitation of the fish capture provisioning valuation methodology is that landed weight net profits were multiplied by live weight during fish capture. Based on Marine Management Organisation data on live and landed weights of UK vessel landings into the UK, aggregate landed weight is around 7 per cent less than live weight. At the same time, net profit per ton was not available for all fish species, so not all the physical flow was valued. Based on available net profit per ton annual data, 95 per cent of fish provisioning (live tons) was valued in 2015. In 2016, 92 per cent of fish provisioning was valued. Finally, note that special consideration will be needed to account for the catch of wild fish by non-resident operators in local waters and by resident operators in non-local waters following the guidance of the SEEA Central Framework (UN et al. 2014; sections 3.3.3 and 5.9) and SEEA EA (UN et al. 2021; section 7.2.6) in relation to these imports and exports.

4.3.1.7 **Wild animals**

Wild animals, plants and other biomass provisioning services are the ecosystem contributions to the growth of wild animals, plants and other biomass that are captured and harvested in uncultivated production contexts by economic units for various uses. The scope includes non-wood forest products (NWFP) and services related to hunting, trapping and bio-prospecting activities; but excludes wild fish and other natural aquatic biomass (included in previous section). This is a final ecosystem service (UN et al. 2021).

4.3.1.7.1 Method

Two methods are most commonly used to value wild animal and other biomass provisioning services. First, meat and other products from wild animals are sold and thus have market prices, which can be used to value the quantities that are known to be consumed in the home.[26] These prices may include some processing and marketing costs, which need to be netted out to obtain the residual value of the ES.

At the same time, many ES of this type are likely not marketed, for example the capture of bush meat for own consumption. Such activity should be included in the SNA but in practice it is often not estimated. As the 2008 SNA states: "The SNA includes the production of all goods within the production boundary. The following types of production by households are included whether intended for own final consumption or not: a. the production of agricultural products and their subsequent storage; b. the gathering of berries or other uncultivated crops; c. forestry; d. wood-cutting and the collection of firewood; e. hunting and fishing." (UN et al 2009; para. 6.23)

Where the relevant outputs are not sold on markets, using prices for similar markets is appropriate.

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[26] See http://www.fao.org/3/w7540e/w7540e09.htm for data on prices of bushmeat in some African countries.

4.3.1.7.2 Tiers

The similar markets method is considered a Tier 2 method and the residual value method is considered a Tier 1 method.

4.3.1.7.3 Example

Turpie et al. (2021) contains an estimate of various wild natural resources, which are harvested from ecosystems for subsistence or small-scale production in KZN province, South Africa. They first construct spatially explicit biophysical models for each of the individual resources they distinguish (fuelwood; poles; timber; wild medicines; wild plant foods; thatching grass; reeds and sedges; palm leaves; bushmeat, and fish), estimating both the demand (using information from census) and the supply (estimating available stocks of resources). The ES in physical units is the minimum from the results of the demand and the supply within a specified distance of the source of the demand (e.g. a village). Valuation is calculated by multiplying the physical flows with a market price (per m3 or per kg), using average prices obtained from the literature. The rationale for using a market price and not subtracting opportunity costs of labour and other capital costs is because the harvest practices are mostly undertaken for subsistence purposes, and the rate of unemployment outside urban areas is generally very high in KZN province. Furthermore, capital costs were considered to be negligible. The results are presented both in the form of maps and as supply tables for 2005 and 2011 (see Figure 5 below).

***Figure 5**: Supply table for wild-harvested biological resources in KZN province, SA, 2011, 2010 Rand (millions)*

Biome / Resource	Freshwater ecosystems	Grassland	Indian Ocean Coastal Belt	Savanna	Forests	Estuaries	TOTAL
Fuelwood	3.13	498.66	172.51	590.99	197.15	0.16	1 462.61
Poles	0.12	20.16	6.66	18.28	7.58	0.01	52.81
Timber	0.02	1.85	0.56	3.42	11.44	0.00	17.30
Thatching grass	0.47	491.15	72.01	301.24	0.82	0.04	865.73
Reeds & Sedges	14.95	94.90	29.40	64.46	4.81	0.35	208.88
Palm leaves	0.00	0.00	10.34	0.00	0.00	0.00	10.34
Wild foods & Medicines	2.29	225.39	62.75	177.42	42.23	0.10	510.19
Bushmeat	0.06	17.41	3.30	21.06	2.06	0.00	43.90
Fish	0.32	4.28	0.72	2.98	0.15	0.07	8.51
Total	**21.36**	**1 353.81**	**358.26**	**1 179.86**	**266.25**	**0.72**	**3 180.25**

Source: Turpie et al. (2021)

4.3.2 *Water supply*

Water supply services reflect the combined ecosystem contributions of water flow regulation, water purification, and other ecosystem services to the supply of water of appropriate quality to users for various uses including household consumption. This is a final ecosystem service.

The SEEA EA recommends that water flow regulation and purification are independently measured and recorded as final ecosystem services. In case water flow regulation and/or water purification cannot be separately measured, SEEA EA recommends to use the "volume of water abstracted" as proxy for the ecosystem service – called water supply.

4.3.2.1 Methods

Four methods can be used to value water supply services, recognizing that the aim is to identify the ecosystem contributions to the abstraction of water as discussed in SEEA EA section 6.4.2. As context, water is supplied as input to agriculture, households and industry from a number of sources including freshwater rivers and lakes, seawater (after desalination), water stored in aquifers and rainwater.

First, where payment is made for its delivery, as it is for water for irrigation, household and industrial use, payments cover the cost of treatment and delivery but rarely for the water itself. Nonetheless, if water supply as a proxy ES is valued in accounting terms, it could draw on data for the price of water and net out the costs of treatment and delivery to derive a residual value. Often however, this method can produce unstable results when the intervening stages (between abstraction from the ecosystem and delivery to users) are long and complex.

Second, a productivity change method can be used. Studies have valued the contribution of water to production using partial and general equilibrium models by looking at the impacts of a change (e.g. a reduction) in the supply on the output in different sectors of the economy. Recent work in this area has focussed on the effects of increasing scarcity as demand for water grows while supply is altered negatively, in some cases due to climate change (Roson and Damania, 2016; Calzadilla et al., 2013). As well, there have been studies of the contribution of water to the production of different crops.

However, the estimation of the parameters of the production function at the sector level remains problematic, with the functions used in the computable general equilibrium models drawing on judgments about the right form of the function to use. At the micro level there are more data that can give estimates of the marginal value of the input water, much as was shown for land (see Section 4.3.1) and from that a value of the contribution of water to the value added can be obtained.

A third approach is to apply replacement cost methods, where a source of water is valued based on the cost of obtaining the water from the next lowest cost source (adjusted for water quality). An example would be using the cost of providing water through desalination.

A fourth approach would be to link the value of the ecosystem services to the prices paid for the rights to abstract water, where such rights are separately identified (from land values) and trading in water rights takes place such that a market is established. In theory there should be a close connection between the value of water rights and the marginal value of water obtained from productivity change methods. It should be recognized that the value of water rights can be significantly affected by the availability of water (e.g. as a result of drought).

4.3.2.2 Tiers

Conceptually, productivity change methods and prices based on water rights (similar markets methods) are considered Tier 3 methods but the availability of data is likely a major challenge.

Replacement cost and residual value methods are thus most likely to be applied and are considered Tier 2 and Tier 1 methods respectively.

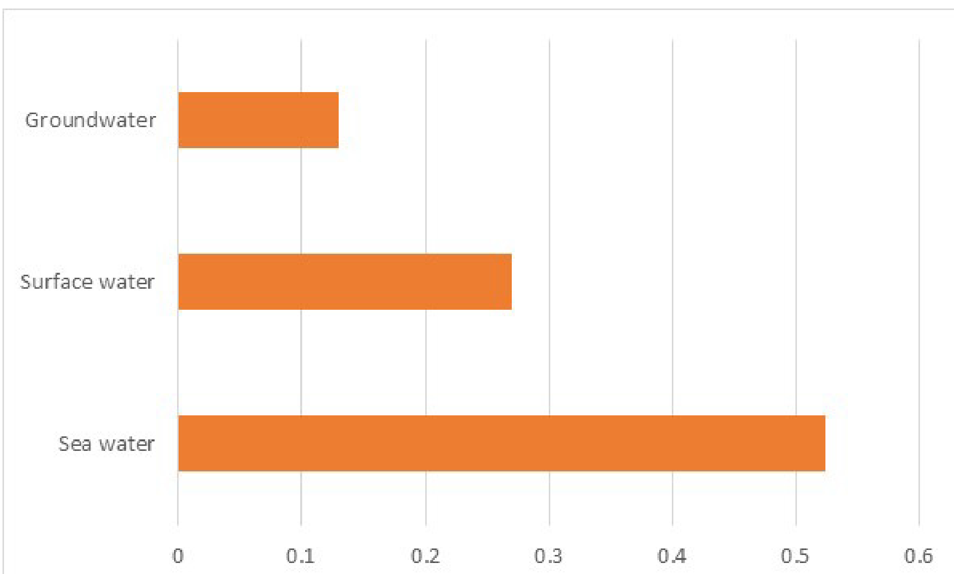
Often the application of methods will vary depending on the context for water supply. Thus, for water distributed by water supply companies (ISIC 37) to households and businesses the use of a replacement cost method is most common while for water used as inputs to agriculture, productivity change methods are most appropriate.

4.3.2.3 **Examples**

The UK (ONS 2019) has estimated the value of water abstraction based on information about economic activities relating to the collection, treatment and supply of water. They followed a residual value/resource rent approach. It was reported that the value of water abstraction in 2017 amounted to £2.54 billion. The results obtained by applying a resource rent approach as in the UK, will reflect the market structure that exists in countries.

For the Netherlands, Edens and Graveland (2013) applied a replacement cost approach to value the various types of water uses by the Dutch economy. They value the water supply service of ground water based on the additional operational cost required to use surface water in case groundwater resources would not be available. The water quality of groundwater is generally better than that of surface water, as evidenced by the cost breakdown of various water supply companies that exist in the Netherlands. Likewise, the least cost alternative in the absence of surface water would be to use desalinated seawater with again higher cost (per m^3) – see Fig. 6.

Figure 6: *Operational costs of drinking water production for various sources, 2010.*



Adapted from Edens and Graveland (2013)

It was found that the least cost alternative of surface water to substitute ground water was 0.13 euro cents per M^3, likewise 0.26 euro cents for seawater as alternative for surface water. After multiplication with the actual amounts of water abstracted, this resulted in a provisioning service of

363 million euros of ground and surface water combined in 2010 as an input for the water supply industry.

4.3.3 *Global climate regulation service*

Global climate regulation services are the ecosystem contributions to the regulation of the chemical composition of the atmosphere and oceans, which affect global climate through the accumulation and retention of carbon and other greenhouse gases (e.g. methane) in ecosystems and the ability of ecosystems to remove carbon from the atmosphere. This is a final ecosystem service (SEEA EA, UN 2021).

The measurement approach recommended in the SEEA EA is to consider global climate regulation services (in case of carbon) as a single service consisting of two components: a carbon retention and a sequestration component, reflecting the importance of ecosystems both in terms of removing carbon from the atmosphere as well as storing carbon over longer periods of time, avoiding its release.

4.3.3.1 **Carbon retention**

The carbon retention component consists of: (i) estimating carbon stocks of relevant carbon pools retained at the beginning of the accounting period; (ii) multiplying this by a suitable carbon price; and (iii) turning this into an annual service flow by multiplying this value by a suitable rate of return (to create an annuity). This framing recognizes that the retained carbon stocks provide a value to society in terms of avoided damages arising from higher levels of carbon in the atmosphere. Thus, in physical terms, the amount stored is a “proxy” for the service flow provided; in monetary units, the service flow is the annual annuity, with higher annuity flows reflecting higher levels of ES provision.

The scope of measurement of carbon retention service is limited to biocarbon in ecosystems (excluding geo-carbon stored in subsoil assets such as oil and gas) and restricted to what the Intergovernmental Panel on Climate Change calls long-lived biomass (e.g., excluding carbon stored in above ground biomass in croplands).

4.3.3.2 **Carbon sequestration**

The carbon sequestration component is measured by the net ecosystem carbon balance, which takes all changes in carbon stocks (e.g. respiration, timber harvest, forest fires) into account. Carbon sequestration has a value to society since it reflects the removal of carbon from the atmosphere thus mitigating the effects of climate change.

4.3.3.3 **Why two components?**

The use of two components recognizes that countries face very different circumstances in relation to carbon stocks and a range of policy instruments exists, some focused on avoiding / reducing emissions from deforestation and degradation (REDD+) and others on stimulating carbon uptake (e.g. flexible Kyoto mechanisms such as the Clean Development Mechanism or Joint Implementation).

In ecosystem contexts where the risk of release of carbon is negligible, the retention service can be valued effectively with price 0. This focus may ensure greater sensitivity of the accounts to assess effectiveness of policy instruments focusing on sequestration. In ecosystem contexts where carbon stocks are declining, for example due to timber harvesting, or land-use changes such as draining of peatlands, recording the retention component ensures that the accounts provide proper signals:

- If an ecosystem loses carbon, lower retention services are recorded.
- Ecosystems with high carbon stocks (e.g. tropical rainforests) obtain high retention values, even though they often have low sequestration as they are in equilibrium, and thus sending the signal that they are worth conserving.
- From an accounting perspective, this framing avoids the need to record ecosystem disservices in case ecosystems would be net emitters of carbon.
- Obtaining estimates of carbon stored (needed input for retention) seems to be easier for most countries and more robust than getting estimates for sequestration.

Finally, double counting strictly speaking is avoided by estimating retention based on opening stocks of carbon (and not average stocks).[27]

4.3.3.4 **Methods**

For the valuation of each component it is necessary to choose an appropriate price for each ton of carbon retained or sequestered. Possible approaches for choosing relevant carbon prices are (Edens et al. 2019): (a) an estimate based on the value of damages avoided; (b) the marginal costs of abatement of carbon, and (c) observed market prices. The first approach is damage based, valuing losses that would result from an increase in carbon using a mixture of market based estimates of the losses. The second approach amounts to placing an estimate on the cost incurred by an economy for avoiding emissions (e.g. better insulation of buildings or more efficient appliances) or for capturing carbon either by technological or biological means (e.g. afforestation). The third approach uses market prices in existing carbon markets such as for emission permits in the European Emission Trading System (EU-ETS). Other approaches can also be found in the literature as well, such as applying the opportunity cost of alternative land uses

4.3.3.4.1 Avoided damage method/social cost of carbon

Given the amount of carbon stored in any one year, its value per ton is a global figure that applies in all situations. Models have been developed to estimate the cost of damages caused by a small increase in the amount of carbon based greenhouse gases in the atmosphere. Because these greenhouse gases are present for a long time, the damages they cause will occur over a long period of time, generally decades.

These damage costs – often called the social cost of carbon – have been reviewed depth in the literature (DEFRA, 2007; Anthoff and Tol, 2013; US Government, 2013). The values are based on the discounted costs arising from adding one ton of CO_2 over the long term and are sensitive to the discount rate adopted. The higher the discount rate, the lower the value attached to future costs and

[27] Carbon retention recognizes carbon sequestration to the extent that it increases carbon stocks over time (and results in higher annuities). In case carbon sequestration would lead to indefinite storage, the amount recorded as carbon retention would be the same as under a standard carbon sequestration service valuation, the difference being this would now be recorded as an ecosystem enhancement (investment) rather than a one-off service flow.

as such, the discounted present value of the costs will be lower. At the same time, the discounted values will increase over time as damages and costs rise with higher levels of GHGs in the atmosphere. The US Government Review of 2013 is probably the most comprehensive recent assessment. Box 7 describes the elements in the calculation of the social costs of carbon from the different models, which have been used and are covered in this review.[28]

Based on a review of the different models, the document gives a range of USD14.9-80.5/ton CO_2 in 2020 rising to USD19.8-94.1/ton CO_2 in 2030 (in USD 2019).[29] The mean values of these ranges are USD47.7 and USD56.9 for 2010 and 2030, respectively. Even this wide range does not encompass all the figures in the literature – a more recent study by Moore and Diaz (2015), for example, suggests much higher values. Other frequently used numbers are Nordhaus (2017) who, based on the DICE model, provided updated estimates of the SCC for a ton of CO_2 emitted in 2015 (USD31.25/ton CO_2 in 2010) and also for CO_2 emissions in a range of future years. These values increased at a real growth rate of 3 per cent per year. Ricke et al. (2018) found much higher values, with a median of USD417/ton CO_2 (in 2020), in part due to the use of empirical climate-driven economic damage estimations. An interesting aspect of this study is that it also provides country specific social cost of carbon (SCC) estimates that distinguish damages as to whether they accrue in the country releasing the carbon or globally.

[28] Studies normally report the social costs of carbon in terms of CO_2 equivalent. The standard factor for converting costs per ton of CO_2 equivalent to cost per ton of carbon is 3.67. See: https://www.epa.gov/energy/greenhouse-gases-equivalencies-calculator-calculations-and-references

[29] These values are averages depending on the discount rate used. The lower bound is the result of a 5 per cent discount rate, while the upper bound is the result of a 2.5 per cent discount rate. There is a much wider range that can be derived, depending on what is assumed about costs, but for this study the above is considered a reasonable representation of the values most researchers would use in sensitivity analysis for the SCC.

Box 7: *Elements in the social cost of carbon*

As explained in the text, the social cost of carbon (SCC) is calculated by running an Integrated Assessment Model (IAM), where the future economic output is estimated under different scenarios for emissions of GHGs. By running the model with a given emissions scenario, calculating the discounted present value of output and then running the model again with a small increase in emissions in the current period, a second discounted present value is obtained. An estimate of the cost caused by that small increase is found by subtracting the discounted value in the second run from the first. Dividing the cost by the change in emissions gives the SCC today. The same calculation can be made starting the model in 2020, 2030 etc. to get the SCC for that year.

The impacts of climate change taken into account vary from one model to another. Three major models are DICE, FUND and PAGE. All include the cost caused by sea level rise, agriculture and energy (higher demand for energy for cooling but less for heating). They also include additional costs of health treatment resulting from higher temperatures and extreme events. Models vary in the cost function they use (i.e. the link between emissions and climate change and between climate change and costs) and there is an element of arbitrariness about the functions. Elements not included in the models are:

1) **Incomplete treatment of non-catastrophic costs:** current IAMs do not assign value to all important physical, ecological and economic impacts of climate change, and it is recognised that even in future applications a number of potentially significant cost categories will remain non-monetised i.e. ocean acidification (not quantified in any of the 3 models), species and wildlife loss.

2) **Incomplete treatment of potential catastrophic costs:** cost functions may not capture the economic effects of all possible adverse consequences of climate change, i.e. **i)** 'tipping point' behaviour in Earth systems; **ii)** inter-sectoral and inter-regional interactions, including global security impacts of high-end warming, and; **iii)** imperfect substitutability between cost to natural systems and increased consumption.

3) **Uncertainty in extrapolation of costs to high temperatures:** estimated costs are far more uncertain under more extreme climate scenarios.

4) **Incomplete treatment of adaptation and technological change:** models do not adequately account for potential adaptation or technological change that might alter the emissions pathway and resulting costs.

Source: US Government (2013)

A recent study in the UK (Bucknall et al. 2021) has developed a carbon pricing model that estimates the degradation caused by the UK's GHG emissions. The model uses a curve that describes the relationship between the loss in global GDP and global temperature increase. It estimates the UK's contribution through the increase in GHG concentration caused by the UK's emissions, the relationship between GHG concentration and temperature increase, and a number of other parameters (such as a discount rate). While the results are used in the study to estimate costs of degradation, they can be used to obtain a measure of the climate regulation service. As the results are driven by the impacts on GDP, the results can be interpreted as exchange values.

4.3.3.4.2 Abatement cost estimates

In this approach it is common to estimate a so-called abatement cost function. Some of these curves have been estimated at the global scale (see below), for specific economies, or for specific sectors (Edens et al. 2019). According to the analysis in Edens et al. (2019), abatement cost curve methods have been criticised for lacking transparency in methods and underlying assumptions. Another key issue is that these methods usually find several negative cost options that any rational actor would choose to directly implement ("no-regrets"). The fact that they are not, indicates that there may be all sorts of barriers (or transaction costs) that prevent their implementation, and hence that abatement cost curves may be underestimates of the actual costs. A specific application of the approach is to calculate the costs to reach a societally agreed target of emission reduction in the near future (for instance as agreed through Nationally Determined Contributions), through estimating a specific time path of required carbon reduction.

As discussed in Edens et al. (2019), while abatement cost approaches may be a feasible approach in concept, in practice their application is likely to be resource intensive (e.g. due to technological development they need to be updated regularly), and therefore generally not recommended for accounting purposes.

4.3.3.4.3 Observed prices in carbon markets

Carbon pricing instruments are becoming increasingly widespread, covering 21.5 % of global GHG emissions in 2021 (World Bank 2021). It is important to distinguish between different types of instruments (Edens et al. 2019). First of all, there are compliance markets (or "cap and trade" systems) where governments set the total amount of allowable emissions, that are subsequently traded (in the form of emission permits) by the participating sectors (such as the EU-ETS scheme). Second, there are voluntary carbon markets where carbon offsets can be purchased (e.g. when buying an airline ticket, people can choose to pay a bit extra to offset their carbon emissions, prompting the airline to purchase certificates). Third, there are carbon markets established in the context of the UNFCCC such as the flexible mechanisms under the Kyoto protocol, REDD(+) schemes etc. Fourth, there are different types of carbon taxes levied by governments.

According to the analysis in Edens et al. (2019), taxes and prices of voluntary emission reduction certificates seem unsuited for the SEEA context.

The main justification for using prices from carbon markets is that such costs are actually being incurred and could be seen as an exchange value of a trade between the government (seller of carbon permit) and the enterprise (buyer of the permit)

Prices from compliance markets will differ across markets, as they will in part reflect the different ways these markets have been set-up. This holds for most transaction in goods and services. But according to some commentators, this would be at odds with the notion that climate regulation services are global and hence should be valued based on a uniform global price.

4.3.3.5 **Tiers**

The two components approach to measuring global climate regulation services allows the use of different prices for the two components.

In case of carbon retention, it is recommended to apply a social cost of carbon, as this aligns with the framing of avoided damages. Different Tiers may be distinguished depending on the sophistication of the model used for deriving the SCC. A Tier 3 approach may consist of using a national model (e.g. Brucknall et al. 2021). A Tier 1 approach may consist in choosing a value from the scientific literature.

When an approach based on long-time forecasts is chosen (e.g. an SCC estimate using an IAM), it is important to be consistent with other valuation methods, in particular when it comes to the choice of discount rate and rates of return. This means that using an SCC estimate is likely to place restrictions on what discount rates and rates of return can be used in other NPV/resource rent calculations (or vice versa). It is also important to choose a best estimate (e.g. median or mean) outcome from an existing IAM model that is aligned with accounting conventions (e.g. excluding human health costs), such as the DICE model, with its focus on lost production.

The SCC estimate should also correspond to the year of the account. Carbon retained in the environment will increase in real value over time. For instance, Nordhaus (2017) estimated that damages grow at a rate of 3 per cent per year.

For the carbon sequestration component, the use of a compliance market price where they are available is recommended. In countries with existing markets, the recommendation is to use these prices as “best available estimates” for those sectors which are not covered by the compliance market.[30] In countries without such markets, the certificate prices of Clean Development Mechanism and/or Joint Implementation projects appear as most compatible approximations. Different Tiers may be distinguished based on the granularity and accuracy of the biophysical model used to estimate the sequestration component (UN 2022)

4.3.3.6 **Examples**

The Indian Ministry of Statistics and Programme Implementation (2021) has estimated carbon retention provided by forests in India for 2015-6 and 2017-18. Hereto they first estimated the total carbon stock consisting of above ground biomass, below ground biomass, dead wood and litter as well as soil organic carbon, using data from the Forest Survey of India. This physical stock estimate was valued using a country specific social cost of carbon estimate. This avoided damage value was turned into an annuity by using 3 per cent rate of return. It was found that the retention service amounted to 2-3 per cent of India’s GDP, and was about twice as large as the gross value added of the Indian forestry sector.

Numerous studies have estimated carbon storage or sequestration. In a study conducted by Turpie et al. (2021), carbon storage was valued in KZN province, South Africa using a SCC approach. The study includes both a valuation based on damages that would impact South Africa only (national SCC), as well as a valuation based on global damages (global SCC). These options resulted in highly different results, by orders of magnitude. i.e. the choice had a huge impact on the value of carbon as an ES, and on the total value of all ES. Using the global SCC meant that the value of carbon ES was much bigger than the value of all other ES.

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[30]The EU ETS covers carbon capture and storage. See: https://ec.europa.eu/clima/sites/clima/files/docs/ets_handbook_en.pdf

Ouyang et al. (2020) valued carbon sequestration using the cost of artificial afforestation, as carbon markets in China were in early stages of development and afforestation is a common policy intervention in China, and default cost estimates (Yuan/tCO_2) exist. It found a value of 4.7 billion Yuan for the Chinese province of Qinghai in 2015, an increase of 67% compared to 2000 (in 2015 constant prices).

4.3.4 *Air filtration*

Air filtration services are the ecosystem contributions to the filtering of airborne pollutants through the deposition, uptake, fixing and storage of pollutants by ecosystem components, particularly plants that mitigate the harmful effects of the pollutants. This is most commonly a final ecosystem service (UN et al. 2021).

4.3.4.1 **Methods**

The most commonly used method used to value air filtration services is the avoided damage method. It focuses on determining the contribution of the ecosystem to improvements in human health and can be applied in different ways. The first type of application involves valuing premature death (mortality) or morbidity using information on the value of a statistical life or disability adjusted life years. As discussed below, the estimated values commonly include a number of components including direct health costs (e.g. hospital costs), loss of earnings and costs of pain and suffering. From an exchange value perspective only the direct health costs should be included. While there is commonly a focus on health outcomes, the avoided damage method can also be applied in the context of building owners who benefit from reduced damage to buildings from air filtration services. If an avoided damage method cannot be estimated, exchange values may also be measured using the averting behaviour method or the replacement cost method. The text below describes the methods in more detail.

The value of the air filtration service is primarily related to the reduced health costs and gain in well-being associated with less exposure to harmful air pollutants. In particular, urban plantations of trees indirectly impact human health by improving air quality through dry deposition of air pollutants on the surface of leaves, twigs, branches and trunks (Nowak et al., 2018; Garcia et al., 2019), thereby reducing concentration of these pollutants in the atmosphere. The beneficiaries are members of the public whose health is improved as a result of the filtration.

Epidemiological studies have established a relationship between the concentrations of a range of such pollutants to which people are exposed and various health impacts, including premature death, hospital admissions, restricted activity days, cases of chronic bronchitis, etc. Such relationships are referred to as dose response functions. The methods used in estimating the additional risks posed by the pollutants and references to the main studies reporting such functions is given in WHO (2013) for Europe and WHO (2018), worldwide.[31]

[31] The references give estimates of functions related to concentrations of PM2.5, PM10, NOx, SO2 and ozone. In the case of heat effects on mortality and morbidity see WHO (2014).

The method applied to estimate the change in health impacts would consist of:

- Calculating the impacts that the vegetation has on concentrations of the pollutants of interest (note that in some cases concentrations may in fact increase). The baseline would be the situation with no vegetation (see also recommendation on biophysical modelling in UN 2022).
- Based on the dose response functions, estimate the changes in number of deaths and cases of morbidity in the exposed population.
- Valuing these changes in health using a range of estimation methods .

For the third step above, estimation methods have been developed and applied to value various reductions in concentrations of pollutants. They can be divided into valuing premature mortality and valuing morbidity.

Reduced premature mortality can be valued in a number of ways. One approach is using a value of statistical life (VSL), where an estimate is made of an individual´s WTP to reduce the risk of death, which is used to value the reductions of risk that the vegetation provides. A second method is to estimate the loss of disability adjusted life years (DALY) that a death or illness would result in and value the DALY based on per capita GDP. The DALY method has been used extensively by the WHO to estimate a range of morbidity impacts and place a value on them.[32] While the VSL approach is based on the basis of individual values, valuation of DALY does not have the same theoretical basis in the theory of value.[33] A third approach is to estimate the number of life years saved and use government guideline values of what it is willing to spend in terms of medical interventions to save a life year.[34] Such values have been used in the UK, the Netherlands and other countries to determine health policies.

Estimation of morbidity is based on a combination of the costs of illness and the costs associated with pain and suffering. For effects such as restricted activity days, impacts include loss of earnings and for hospital admissions or incidence of chronic coughs, cases of bronchitis etc., there are estimates of the direct costs incurred by the individuals concerned as well as estimates of the loss of well-being.

For ecosystem accounting purposes, it is difficult to see full VSL or DALY based values as exchange values. Methods used to determine VSL measure the full WTP in order to reduce the risk of death and therefore include consumer surplus. Similar considerations apply to valuing DALYs. The use of government guideline values is closer to an exchange value, as it is directly related to what the government spends to save a life year. For morbidity, the direct costs of illness (i.e. treatment costs, expenditures on medicines) are within the boundary of the SNA, while others, such as loss of earnings

[32] See: https://www.who.int/healthinfo/global_burden_disease/daly_disability_weight/en/

[33] The literature also uses a related concept of Quality Adjusted Life Year. DALY measures adverse impacts while QALY measures good health. The QALY or DALY measures use the quality of a life year as the basic unit of account, estimated by multiplying the duration of different health states (standardized to one year) and a score (or weight) reflecting the severity of those health states. For QALYs, perfect health is assigned a score equal to one and a score of zero represents death. For DALYs the opposite applies. Thus, numeric values can be assigned to different health states so that morbidity effects can be combined with mortality effects to develop an aggregated measure of health outcomes.

[34] For an application on valuing health costs of air pollution based on both VSL and DALYs. See: World Bank/IHME (2018).

due to illness, are not. Costs of pain and suffering are measured using stated preference methods and include consumer surplus and therefore, would not be admissible for ecosystem accounting. The European Commission reports morbidity costs separately into the direct and indirect components (AEA, 2005), showing the indirect costs to be between 22 per cent and 36 per cent of the total.

In some countries, alternative methods to valuing health impacts have been followed. For example, in China an averting behaviour method has been estimated based on costs of installing air filters in houses to deal with air pollution. Such estimates do not account for the full effects of the presence of air pollution and exposure can only partly be reduced through the measures. There is the further difficulty of estimating the extent to which use of air filtration services reduces the expenditures on such devices.

In summary, it is possible to estimate the value of the services provided by air filtration in terms of reduced costs of lower mortality and morbidity. Some components of the cost reflect exchange values and could be included in the ecosystem accounts.

4.3.4.2 **Tiers**

As detailed in the biophysical modelling guidelines (UN 2022), to measure this service it is essential to start with the physical data on the concentrations of key pollutants and how they are affected by vegetation. This will provide an estimate of the reduction in concentrations, which can then be linked to mortality and morbidity impacts taking proximity to people into account. The impacts can be valued using an avoided damages method related to the value of reduced mortality and the savings in morbidity. As the biophysical model is based on spatial data, the resulting monetary value is also spatially explicit, and Tiering will be related to the level of spatial detail of the model.

If these approaches are not feasible, a partial estimate may be obtained using the averting behaviour method by estimating the savings in defensive expenditures as a result of cleaner air.

4.3.4.3 **Examples**

In the UK accounts (ONS, 2019a), the starting point is the estimated concentration of various pollutants harmful to health in the atmosphere across the UK on a 50x50km grid. Air pollution removal by UK vegetation has been modelled for the years 2007, 2011, 2015 and for 2030 (based on projections of climate and future pollution emissions). Between these years a linear interpolation has been used and adjusted for real pollution levels as an estimation of air pollution removal.

The change in health outcomes is calculated based on the change in pollutant concentration to which people are exposed. Then, avoided damage costs per unit of exposure are applied to the benefiting population at the local authority level for a range of avoided health outcomes. The valuation of the damage is calculated using the value of a QALY estimate.

4.3.5 *Local climate regulation*

Local climate regulation services are the ecosystem contributions to the regulation of ambient atmospheric conditions (including micro and mesoscale climates) through the presence of vegetation that improves the living conditions for people and supports economic production. Examples include the *evaporative cooling provided by urban trees ("green space"), the role of urban water bodies ("blue space") and the contribution of trees in providing shade for humans and livestock. This may be a final or intermediate service (UN et al. 2021).*

4.3.5.1 Methods

The valuation of local climate regulation is particularly important in urban contexts. The two most common methods for valuation of local climate regulation services are avoided damage costs and averting behaviour methods. As for air filtration services, the methods are related to the reduced impacts on human health of extreme weather conditions, especially heat. The same considerations as described in the previous section are relevant in this case. For local climate regulation services, there are also benefits arising to businesses in terms of reduced costs of cooling which could be valued using replacement cost or productivity change methods.

By way of example, an indirect health benefit of trees is their capacity to buffer extreme temperatures. As a result of climate change, the intensity and frequency of heat waves is expected to increase and trees can mitigate such impacts in urban heat islands by reducing exposure to solar radiation. For example, in the summer of 2010 in England, urban trees reduced mean maximum daily soil surface temperatures by 5.7°C, compared to herbaceous vegetation (Edmondson et al., 2016). This lower temperature reduces cooling costs for households and businesses and reduces the risk of illness and premature death, which are common during extreme heat waves. Where the reductions in impacts on human health relate to the avoidance of heat wave conditions, the value can be assessed in terms of lower mortality and morbidity costs. Where the reductions in impacts on human health are unrelated to heat waves, the valuation is based on the reduced costs of cooling as well as higher productivity for outdoor workers.

4.3.5.2 Tiers

The use of a productivity change method would be a Tier 3 approach. The application of averting behaviour would be a Tier 2 approach, while the use of a replacement cost or avoided damages would be a Tier 1 approach.

4.3.5.3 Examples

UK Natural capital accounts (ONS 2021) value urban cooling at £ 453 million/year, defined as the contribution from green (parks) and blue spaces (rivers, lakes, canals) to cooling urban environments on "hot days", valued through avoided loss of labour productivity and reduction in the use of air conditioning. Urban cooling is estimated as the combined city-level effect of aggregate % of green/bluespace extent and a buffer area on temperatures based on empirical values from the literature (ONS 2021). The city level approach does not distinguish temperature reduction of vegetation from shading and evapotranspiration (EFTEC 2018).

Remote sensing data can be used to assess temperature differentials due to different structures of urban landscapes (Hamstead et al. 2016), including vegetation and water. The respective temperature contributions of ecosystem assets such as urban tree canopy can then be computed (Venter et al. 2021)

The pilot accounts for Guizhou province, China (NBS, 2021) adopted a replacement cost method whereby the value of the service is calculated based on the electricity consumption required to obtain an equivalent cooling and humidification effect through air conditioning. This replacement cost considers benefits to indoor activities only.

Horváthová et al. (2021) estimate the exchange value of shading from a generic type tree based on the comparable shading effect and least cost of artificial parasols in Prague, Czech Republic. The authors compared replacement cost intervals of parasols with different life expectancies of urban trees. The method was aimed at providing economic arguments for tree planting programmes. The study demonstrated that urban trees provide higher return on investment for shading specifically relative to artificial shading when tree life expectancy exceeds 40 years (not considering other benefits). The authors demonstrate how an exchange value based approach applied for local policy assessment purposes, is compatible with, and can potentially be scaled up for urban ecosystem accounting.

4.3.5.4 Other considerations

The replacement cost approach for microclimate effects of urban shading does not consider whether the benefits of shading are actually realized. In scaling exchange values of shading from a generic tree to all trees of an urban accounting area, some consideration could be given to differences in shading potential by size (species) and the variation in demand (some tree shading is not accessible to the population). The replacement cost approach of shading does not consider cooling effects of evapotranspiration from vegetation. This illustrates a drawback of the replacement cost approach focusing on individual services that are a composite of more than one ecosystem function.

4.3.6 *Soil erosion control*

Soil erosion control services are the ecosystem contributions, particularly the stabilising effects of vegetation, which reduce the loss of soil (and sediment) and support the use of the environment (e.g., agricultural activity, water supply). This may be recorded as a final or intermediate service (UN et al. 2021).

4.3.6.1 Methods

There are three commonly used methods for valuing soil erosion control services, the productivity change method, the replacement cost method and the avoided damage cost method.

The productivity change method can be used to estimate the value of soil erosion control by assessing the gain in yields from reduced erosion that is attributable to the particular ecosystem and value them using the change in value-added per unit gain in yield. Estimates of loss of yields, when there is a loss of topsoil, are available in the literature. The total loss of cropland due to erosion is put at 10 million hectares a year (Pimentele and Burgess, 2013). To the extent that vegetation in a forest or other non-cropland ecosystems prevents erosion, the part of crop value added that is attributable to that prevention should be attributed to that ecosystem service.

In using the productivity change method, caution should be taken, however, in the use of the proper output price. Consider, for example, a soil protection project where it is expected that the increase in erosion control services will support an increase of the crop production. Two possible scenarios can be considered:

- The first is one where the changes in crop production arising from the increase in erosion control, do not affect the market equilibrium prices of the final good. In this case, only the net operating surplus will be affected and the benefit of the ES can be measured as the additional crop production generated by the reduction of soil erosion by comparing the “with project” scenario and the “without project” scenario. This measure of the producer´s benefit is also referred to as “producer surplus”.

- The second is when prices of agricultural crops decrease because of increases in crop production. In this case, the change in the environmental context due to the soil protection project has an effect on the market equilibrium price.

A second approach is to use the avoided damages method to value the savings from reduced erosion in terms of the additional energy, nutrients and water that would be needed to maintain a given level of production, as well as the costs of siltation and damage caused by soil particles entering streams and rivers and harming habitats. This approach was used in the US by Pimentele et al. (1995), where it was estimated that damages were about USD100 $ha^{-1}yr^{-1}$. However, this method is not as accurate as the first, because it assumes that the potential costs would be paid to replace the services provided by the soil, which may not be the case.

The avoided damage cost method may also be applied considering, for instance the additional costs related to additional dredging of sediment trapped behind dams, or additional costs resulting from reduced hydropower capacity. Such approaches are only possible in case the sediment reaches rivers that have such structures in place.

Note that if the ecosystem that prevents the erosion were not present, there could still be benefits to people downstream (e.g. in terms of topsoil availability in the Nile Delta). These benefits should be accounted for separately recognizing that these downstream gains generally take place over a long period of time and involve a variety of institutional contexts.

Based on the productivity change method, a more sophisticated study was conducted by Ghaley and Porter (2014), who investigated the relationship between soil organic matter in winter wheat production systems in Denmark, ecosystem functions of soil water storage and nitrogen mineralization and the ES of food and fodder production and carbon sequestration. The authors used a soil–plant–atmosphere system dynamic model, which simulates plant growth and soil processes. Based on these findings, the authors estimated the value of changes in soil organic matter. All values are based on market prices of inputs and outputs except carbon, which was taken from the EU-ETS price.

This study demonstrated the importance of accounting for the spatial variation in the role of vegetation as a provider of soil erosion prevention and other services. Accurate valuation relies very much on accurate biophysical modelling, especially for soil erosion prevention.

4.3.6.2 Tiers

The productivity change method is considered a Tier 3 method, where it is assumed to have spatial resolution. The avoided damage cost method are Tier 1 or 2 method. Different Tiers may be distinguished based on the granularity and accuracy of the biophysical model used to estimate the service flow (see UN 2022).

4.3.6.3 Example

Turpie et al. (2021) valued the soil erosion control service in KwaZulu-Natal (KZN) province of South Africa for 2005 and 2011. Applying the InVEST Sediment Delivery Ratio model[35], first for each quaternary catchment the sediment retained and captured by ecosystems was estimated (in physical units), by comparing the baseline of current land cover with a counterfactual situation of barren land. In KZN, sediment gets trapped behind hydropower dams, thereby reducing its capacity. Valuation was therefore based on the cost of lost storage capacity by estimating the avoided damage cost, either of constructing a substitute reservoir or raising the dam wall. The amount of storage required was benchmarked on the baseline amount of sediment reaching downstream locations.

Ouyang et al. (2020) valued the soil retention service for Qinghai province, China. The authors applied the InVEST model to model in physical units the amount of soil retained in the landscape. The valuation was based on the avoided dredging cost in hydropower reservoirs, as well as avoided treatment costs for non-point source pollution in case N and P would exceed carrying capacity of riverine ecosystems for water purification.

4.3.7 *Water purification*

Water purification services are the ecosystem contributions to the restoration and maintenance of the chemical condition of surface water and groundwater bodies through the breakdown or removal of nutrients and other pollutants by ecosystem components that mitigate the harmful effects of the pollutants on human use or health. This may be recorded as a final or intermediate ecosystem service (UN et al. 2021).

4.3.7.1 Methods

The ecosystem acts to remove harmful pollutants from water and thus makes water available to potential users that is cleaner than they what would get if the ecosystem were absent. In this way ecosystems cause saving in the costs of treatment required to make the water fit for a range of uses.

Two methods are most commonly used to value water purification services: (i) the replacement cost method estimating the cost of putting in place structures and equipment to purify water to the same level of quality and (ii) the avoided damage costs estimating the reduction in water purification and treatment costs that arises from having the ecosystem service. Additionally, in some cases, payments may be made by water suppliers to ecosystem managers, reflecting the benefits provided by the

[35] See: http://releases.naturalcapitalproject.org/invest-userguide/latest/sdr.html

ecosystems in which case this payment can be considered a directly observed estimate of the exchange value.

Water purification is similar to the case of avoided soil erosion where an ecosystem, such as a protected forest or wetland, provides a service. Market transactions may be available in some cases indicating a payment to the managers of the ecosystem that provides purification services (see example below). If such data are not available, the service can be valued in terms of the costs avoided in treating the water by other means (e.g. by using chemicals) or the costs of maintaining the purification services of the ecosystem as discussed above.

4.3.7.2 **Tiers**

If data on transactions related to purification services are available to support valuation this is a Tier 3 method. The avoided damage cost method and the replacement cost method are considered Tier 1 or 2. Different Tiers may be distinguished based on the granularity and accuracy of the biophysical model used to estimate the purification service (UN 2022). If it is possible to estimate both avoided damage costs and replacement costs then a comparison of the two should be made and the lowest cost should be used to reflect the exchange value.

4.3.7.3 **Examples**

A well-known example from the USA is New York City, where about 90 per cent of its water never enters a filtration plant, and rather flows from huge reservoirs as far as 125 miles away in the rural Catskill Mountains. New York has spent more than USD1.7 billion to protect this unfiltered water supply since the early 1990s, in return for being granted a succession of federal and state waivers exempting it from costly filtration requirements.[36] The value of the service provided by the mountains could be taken as the cost incurred for the supply of the filtration services by the Catskill Mountains. However, the WTP for the service is much higher than that. According to a New York Times article, in the absence of the Catskills mountains New York City “would have to spend more than USD10 billion to build a massive filtration plant, and at least another USD100 million annually on its operation” and...”Water bills would have to rise significantly to cover the cost.”[37]

In this case, since an exchange of USD1.7 billion is actually recorded, that would be the capitalized exchange value of the service. To get the annual value of the service the cost would have to be annualized. It would further need to be allocated to the ecosystem types in the catchment contributing to the purification service in order to compile the ecosystem supply tables.

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[36] This is the payment the city makes to those responsible for the upstream areas to prevent development and ensure they remain pristine. It is not clear whether this estimate includes the opportunity cost of land under reservoirs and forgone earnings from allowing development. Normally the opportunity cost of land is included in the cost of providing a service if the land is not directly valued and the rental value is not part of the cost.
[37] See: https://www.nytimes.com/2018/01/18/nyregion/new-york-city-water-filtration.html

4.3.8 *Water regulation*

Water regulation services are the ecosystem contributions to the regulation of river flows, groundwater and lake water tables. They consist of baseline flow maintenance and peak flow mitigation (UN et al. 2021).

4.3.8.1 **Baseline flow maintenance services**

Baseline flow maintenance services are derived from the ability of ecosystems to absorb and store water, and gradually release water during dry seasons or periods through evapotranspiration and secure a regular flow of water. This may be recorded as a final or intermediate ecosystem service (UN et al. 2021).

4.3.8.1.1 Methods

Ecosystems can reduce temporal variation in water flows, particularly on an intra-annual basis, relative to the variation in rainfall. Without this service, dry season flows would be lower, increasing the need for storage. Therefore, water supply infrastructure, and reservoir capacity in particular, can be treated as a substitute for the service that is usually provided by ecosystems. The infiltration and temporary storage that is facilitated by ecosystems has the effect of changing the seasonal pattern of surface flows lower in the catchment.

The supply of water flow regulation by ecosystems is determined by the capacity to provide the service and the cost in doing so. In some cases, land may be managed for the purpose of delivering water flow regulation (e.g. maintained dunes, designated areas for flood water retention) and the costs are relatively well understood. In many cases, however, the provision of water flow regulation by ecosystems is an uncompensated public good, in which case the level of supply is determined by other considerations (i.e. private land-use decisions, government regulation, protected area designation etc.), making the costs of delivering the service largely unknown.

Given these contexts for the role of baseline flow maintenance services, there are three relevant valuation methods: the avoided damage costs method; the productivity change method, and the replacement costs method. For the avoided damage costs and productivity change methods, the focus is on understanding the production associated with the maintenance of flows – for example, hydropower generation, agricultural irrigation, river navigation. In the avoided damage costs approach the costs of losing the service are estimated. In the productivity change method a production function that reveals the contribution of the service to the production activity is estimated.

In the replacement cost method, in cases where flow rates serve to ensure a steady supply of water, the value of that flow is based on the costs of constructing storage facilities required in the absence of such services.

4.3.8.1.2 Tiers

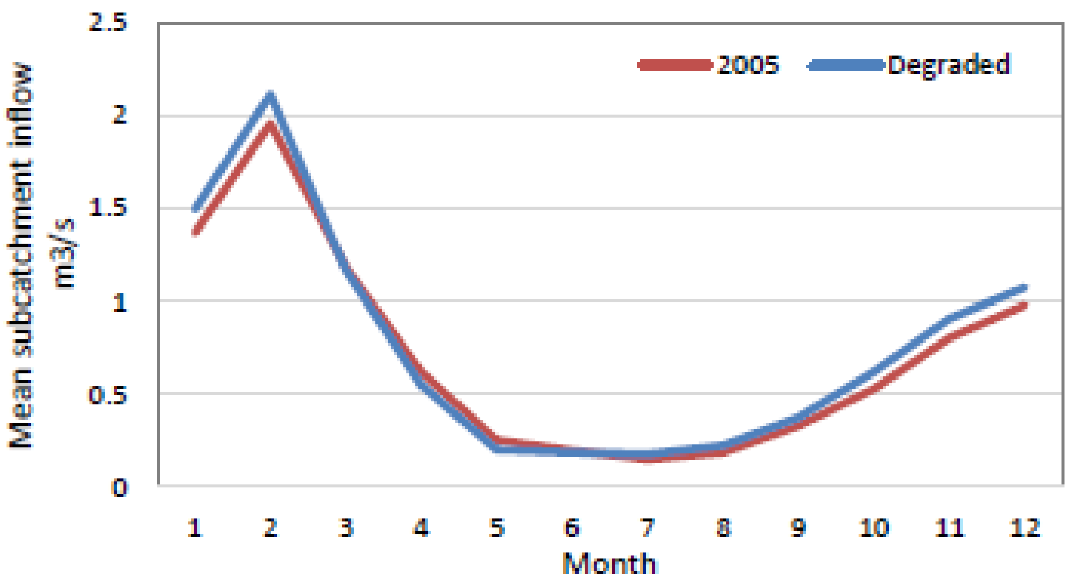
The productivity change method is considered a Tier 3 method. The avoided damage and replacement cost method are Tier 1 or 2 methods.

4.3.8.1.3 Examples

Turpie et al. (2021) measured the water regulation service in South Africa in physical terms as the difference in infiltration relative to a barren scenario, in m^3 per ha. This was obtained from the Soil and Water Assessment Tool (SWAT) model output "Percolation" in mm.

The benefits generated from the ES were considered in terms of the avoided costs of water supply infrastructure for existing supply systems, and in terms of the avoided costs of obtaining water for people that depend on instream flows for domestic water supplies. The estimates did not include run-of-river abstractions for commercial agriculture or other purposes. For each area, the overall storage capacity was calculated, and the modelled runoff for the lowest sub-basin was analysed. The service was quantified based on the ecosystem effects on flow variation, and the influence of this on storage requirements, for the existing yield and reliability requirements.

Figure 7: *Change in mean monthly inflows for 2005 land cover versus a degraded scenario for a sub-catchment*



Source: Turpie et al. (2021)

The unit cost of storage was based on the 2008 inventory of reservoirs for South Africa and inflated to 2010 Rands (R2.81/m3). Converted to 2016 Rands (R2.37/m3), this unit cost was lower than the estimate obtained for Australian reservoirs of R3.96/m3 based on the literature and adjusted using the relative purchasing power parity of the two countries, and may be conservative. Marginal costs are likely to increase as storage level requirements increase, and thus average costs can also underestimate the ES value.

In addition to the benefits of reservoir design, maintenance of base flows benefits people living in the catchment areas who depend on rivers and springs for domestic and agricultural water use. To determine this value, the number of households depending on rivers and springs for water supply was extracted from the 2011 census at the local level, and matched to the sub-basins using spatial data on human settlements. Using the basic human needs allowance of 6000 litres per household per day, the study estimated the monthly water demands of these households within each sub-catchment. These were then compared with the modelled monthly inflows into each sub-basin for the actual land cover in 2005 and 2011 and the corresponding barren (without service) scenarios.

The value of this service in terms of infrastructure cost savings (avoided damage cost method) was estimated to be R1.007 million in 2005 and R982 million in 2011. The biggest change in the estimated average increment in water retention by ecosystems was observed in the grassland and forest biomes.

In addition, the flow regulation service performed by catchment ecosystems contributed an annual cost savings to poor households of an estimated R3 million in 2005, and R2.6 million in 2011, which is significant in terms of the income levels of the beneficiary households. These cost savings were estimated using the replacement cost method based on the cost of purchasing water from local water vendors.

In the pilot for Guanxi province, China (NBS 2021), the water flow regulation (or water conservation value) is calculated using a replacement cost method. The service value provided by forests is estimated based on the cost of construction of a reservoir with similar capacity.

4.3.8.2 **Peak flow mitigation**

Peak flow mitigation services are derived from the ability of ecosystems to absorb and store water, and mitigate the effects of flood and other extreme water-related events. Peak flow mitigation services will be supplied together with river flood mitigation services in providing the benefit of flood protection. This is a final ecosystem service (UN et al. 2021).

4.3.8.2.1 Methods

There are several methods for valuation peak flow mitigation services that all relate to the reduction in flood risk provided by ecosystems. The main methods are the insurance premiums, the avoided damage cost method and the replacement cost method.

Flood mitigation services data on the reduced risks of flooding may be obtainable from the insurance premiums for flood protection. Where rivers and coastal areas are managed to reduce such risks, the premiums should be lower and the difference between the premiums according to risk level would give an estimate of the value of a reduced risk. The problem is that some risks are partly covered by governments and hence premium differences may not fully reflect the risk differences.

Crossman et al. (2019), provide a detailed description of the ES provided and a review of the literature that values such services. The demand for water flow regulation by ecosystems is defined by the benefits of reduced flood risk. The benefits of reduced flood risk are largely determined by the costs of flooding that are avoided, which comprise of two distinct components: damage costs to assets and people in the event of a flood; and mitigation costs including flood protective infrastructure/measures (e.g. levies, dikes, seawalls, beach nourishment), relocation, and averting behaviour (e.g. growing flood resistant crops). It is important to recognize that economic units (households and firms) faced with flood risk, will attempt to minimize the sum of these two cost categories and the mix of cost-minimizing responses to flood risk is highly context specific and requires detailed physical modelling.

A replacement cost approach would consist of estimating the cost of constructing reservoirs that would have similar storage capacity as the storage capacity provided by the ecosystems. This storage capacity can be estimated for instance by comparing the current vegetation cover (e.g. forest) with a counterfactual situation of bare or degraded land.

4.3.8.2.2 Tiers

The use of insurance premiums would be a Tier 3 method while the replacement cost and avoided damage cost method would be a Tier 1 or 2 methods.

4.3.8.2.3 Examples

Broadmeadow et al. (2018) estimate the flood protection service provided by forests in the UK using a replacement cost approach . The physical volume of water (expressed as m3/ha) was estimated using a biophysical model that assessed woodland water use or retention by roughness of the floodplain limited to flood risk catchment areas. The replacement costs were estimated based on the annualized cost of construction and maintaining manmade water reservoirs of similar magnitude.

Turpie et al. (2021) estimated what they call the flood attenuation service provided by urban green space in eThekwini municipality based on an assessment of both the avoided flood damages and the avoided cost of flood protection infrastructure in South Africa. They use the lowest cost estimate of the two analyses. The valuation was based on simulations with a hydrological model.

4.3.9 *Coastal protection*

Coastal protection services are the ecosystem contributions of linear elements in the seascape – coral reefs, sand banks, dunes or mangrove ecosystems along the shore – in protecting the shore and thus mitigating the impacts of tidal surges or storms on local communities. This is a final ecosystem service (UN et al. 2021).

4.3.9.1 **Methods**

World Bank (2016) provides specific guidance on the measurement of the coastal protection by mangroves and coral reefs, with Section 5 of that paper dedicated to valuation.

Based on a detailed review of case studies, the most commonly applied method is the replacement cost approach, such as estimating the cost to construct dams or seawalls that would provide an equivalent amount of protection to that provided by coastal ecosystems (e.g., mangroves) and reefs. However, conceptually and where data are available, World Bank (2016) recommends using the avoided damage cost method applying an expected damage function (EDF), which equates the value of an ecosystem asset that provides coastal protection services with the expected avoided damages.[38] To calculate expected damages, one takes each of the possible avoided damages and

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[38] See also IDEEA Group (2020) for an assessment of valuation methods for disaster risk reduction services. https://www.awe.gov.au/sites/default/files/documents/valuation-disaster-risk-reduction-ecosystem-services-australias-coastal-wetlands-review.pdf

multiplies it by the probability of that specific loss. According to World Bank (2016), the use of EDF functions is grounded on approaches commonly used for estimating risks and benefits in the engineering and insurance industry. It does, however, require probabilities to be defined for the different possible outcomes, which is information not always available.

4.3.9.2 Tiers

The avoided damage cost method using an expected damage function is considered a Tier 3 method and replacement cost methods are considered Tier 2 methods.

4.3.9.3 Examples

World Bank (2016) contains a range of examples. For instance, Barbier (2015) applied the EDF approach for valuing coastal protection in the Gulf of Mexico.

Menendez et al. (2020) computed the benefits of flood risk protection provided by mangroves worldwide. They coupled a 2-D hydrodynamic model of flood risk to economic models of costs of flooding, estimating the global value of flood protection benefits to exceed US$65 billion annually. Monetary valuation was based on the avoided residential and industrial property damage costs based on a global mapping of property types (GAR15 database) and a Global Flood Depth-Damage Function (Huizinga et al. 2017). Economic benefits of flood risk mitigation by mangroves are disaggregated by country.

4.3.10 *Pollination*

Pollination services are the ecosystem contributions by wild pollinators to the fertilization of crops that maintains or increases the abundance and/or diversity of other species that economic units use or enjoy. This may be recorded as a final or intermediate service. (UN et al. 2021).

4.3.10.1 Methods

The main methods used to value pollination services are the productivity change method and the similar markets method.

The estimation of the contribution of pollination is commonly based on how yields of crops have increased as a result of the use of pollination services. Pollination by insects is thought to be the main reproductive mechanism in 78 per cent of temperate flowering plants, and is essential to maintaining plant genetic diversity. Declines in wild bees have been closely associated with similar declines in plants such as bluebells and poppies, which have aesthetic importance to people by improving the overall look of the landscape, gardens and other green spaces from parks to road verges (Breeze et al., 2012). In principle, it is possible to value the landscape benefits of pollination through stated preference studies of the increase in WTP for a landscape with higher presence of such plants. It is also possible through studies of increased visitation to sites with such plants, but none have been undertaken so far in the literature, so only increases in yields have been estimated.

The application of the productivity change method requires an estimate of the amount of pollination a particular crop requires (this varies considerably across crops), and an estimate of the availability of pollinators for the crops. The analysis has to be spatially disaggregated, given the spread of crops and pollinators across the landscape (see UN 2022 for a detailed description of biophysical modelling approaches for pollination). Once the increase in yield for each pollination dependent crop is established, it is multiplied by the market price of the crop in order to obtain a value for the pollination. Note, however, that the pollination value is already part of the value-added from the crop production in the national accounts and the exercise of valuing it helps to attribute part of the value to the pollination.

A second method is to use a similar markets method using the cost of using cultivated bees and other pollinators in place of wild pollinators. In many countries there are well developed markets in the supply of cultivated pollination services.

4.3.10.2 Tiers

The similar markets method is a Tier 3 method and the productivity change is a Tier 1 or2 method, depending on the spatial resolution.

4.3.10.3 Examples

The Dutch ecosystem accounts value pollinators in terms of their contribution to gross revenue of crops, which are allocated to five classes based on their dependence on pollination (Horlings et al., 2020). A measure of pollinator abundance was constructed from habitat quality for the country. Combining the data and applying it to the spatial location of crops that require pollination and the spatial location of ecosystems that are suitable for pollinators, it was possible to estimate the increase in yields of different crops due to pollination. The annual contribution of the ES crop pollination to total crop production was approximately 359 million euros in 2015. The contribution by province is highest in Gelderland (95 million euros), Noord-Brabant (78 million euros) and Limburg (53 million euros) (ibid). River flood basins, which are often situated near fruit orchards, contribute relatively the most to crop pollination services, with an average of 479 euros per hectare. Grasslands, due to their large extent, have a large total contribution to the total crop pollination service.

4.3.11 *Recreation enabling services*

Recreation-related services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems that enable people to use and enjoy the environment through direct, in situ, physical and experiential interactions with the environment. This includes services to both locals and non-locals (i.e. visitors, including tourists). Recreation-related services may also be supplied to those undertaking recreational fishing and hunting. This is a final ecosystem service (UN et al. 2021).

4.3.11.1 Methods

Ecosystems provide a range of recreational and cultural services in urban and non-urban areas. They include activities such as walking and running, hiking, birdwatching, fishing and other water-related pastimes (swimming, boating etc.). Places where the activities take place are usually public areas with access that can be free or based on some payment. A number of countries have a right-to-roam on private land as well.

A distinction is made between recreation that involves a trip to a site some distance from home (e.g. a road trip to a national park) and recreation close to home (e.g. dog walking in the park). In the terminology of tourism statistics, this is the distinction between visitors (who travel outside of their usual environment) and non-visitors. Visitors can further be separated into those undertaking single day trips and those who stay overnight outside of their usual environment.

There are three main approaches to the valuation of recreation-related services where there is travel involved in accessing a site – the travel cost method (including multi-site Random Utility Model - RUM), the consumer expenditure method and the simulated exchange value (SEV) method. As explained in detail in Chapter 3, all of these methods require data on the number of people who visit different sites and information on the costs associated with the recreation activity. The travel cost method cannot be used directly to estimate exchange values, but the data gathered to implement this method can be used to apply the consumer expenditure method and where a demand curve is estimated this can be used as input to the SEV method.

The value of recreation-related ES excludes the value of any health benefits that may also accrue to those travelling to sites for recreational and other purposes. Such health benefits may be measured in terms of increased life expectancy or increased workplace productivity. The exclusion of health benefits is consistent with the valuation scope of the SNA which regards such benefits as an outcome rather than an output of economic production. For example, the SNA records the production and consumption of food, water and medical services, but not the associated health benefits.

4.3.11.2 **Tiers**

A Tier 1 method would consist in applying consumer expenditure based on tourism/visitation statistics cross-classified by purpose. This would be non-spatial. Spatialization would have to occur in a second step, allocating visitation statistics based on a simple metric of relative landscape attractiveness (e.g. footpath density). A Tier 2 approach consist in identifying recreation trip choices by households based on a population representative travel cost surveys / diaries or data from mobile phone tracking applications; then multiplying trip choice by travel expenditures and aggregating to the population. A Tier 3 consist in applying SEV, where the optimal site choice is modelled in a RUM, considering substitute sites. In a base application where only site-specific demand curves are available, one could assume some common revenue maximizing institutional mechanism with homogeneous management costs across sites. In more sophisticated applications, one could model the site specific management costs (supply curves) and estimate point of intersection for each site to obtain the simulated exchange value. This approach is likely to be very data intensive.

In case of local recreation services – especially in urban areas – it is recommended to apply a hedonic pricing method for a Tier 3 approach.

4.3.11.3 **Examples**

The recreation account for the UK compiled by the ONS estimates the total number of visits taken to the outdoors, as well as the total duration and amount spent while visiting the outdoors. For estimates of outdoor recreation in England, Scotland and Wales similar national surveys collect detailed information on people’s use and enjoyment of the natural environment during visits of any duration (e.g. in England the MENE survey or Monitor of Engagement with the Natural Environment). The recreation account estimates the spend incurred to travel to an outdoor location and some expenditure incurred during the visit. This expenditure method considers the market goods consumed as part of making the recreational visit (fuel, public transport costs, admission charges and parking fees). This expenditure is currently assumed as a proxy for a marginal price for accessing the site (ONS 2021).

Simulated exchange value method (see Chapter 3) has been used with contingent valuation techniques to estimate site-specific demands for the main recreational areas in Andalusia (Caparros et al. 2017). The results were based on face-to-face interviews with a sample of 4030 free access visitors to recreational sites in Andalusian forest ecosystems. Campos et al. (2019) present an application where recreational values are integrated with 15 other private and public activities in spatially-explicit accounts for Andalusian forest ecosystems. To obtain an estimation of the recreation service that is considered as "contribution of the ecosystem", only part of the value should be considered. In other words, the value of human inputs to the service needs to be taken into account[39], as shown in Caparrós et al. (2017) and Campos et al. (forthcoming). Only a fraction of the free access recreational value (estimated using simulated exchange value) estimated for Andalusian forests can be considered an ecosystem service in monetary terms.

4.3.11.4 Other considerations

In all cases, visitors can be split into domestic/international and inbound/outbound. In case of international visitors, the recording in the supply-use table will be different as they relate to the import and export of ES.

4.3.12 *Nursery population and habitat maintenance services*

Nursery population and habitat maintenance services are the ecosystem contributions necessary for sustaining populations of species that economic units ultimately use or enjoy, either through the maintenance of habitats (e.g., for nurseries or migration) or the protection of natural gene pools. This service is an intermediate service and may input to a number of different final ecosystem services including biomass provision and recreation-related services (UN et al. 2021).

The focus here is on valuation of nursery services that supporting the supply of biomass provisioning services. Note that the role of habitat and biodiversity in supporting the conservation of significant species is not considered an ecosystem service and the estimation of the associated non-use values are not discussed here.

4.3.12.1 Methods

Nursery services support various provisioning services and can be valued in terms of the contribution to the market value of the latter. Examples of provisioning services are wood and non-wood biomass provisioning services for nursery services provided by land-based ecosystems, and wild fish provisioning services for nursery services provided by seagrasses, mangroves and other marine/coastal ecosystems. In the case of uncultivated biomass, the value of the provisioning service is embedded in the value of the final product, which has a market price. This value links to the value

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[39] Caparros et al. 2017 (op.cit) and

of the associated nursery services. Changes in those nursery services impact on the provisioning services and the link between the two can be estimated using the residual value method or productivity change method (Forestry Commission, 2016). The same applies for marine biomass, where productivity change methods described for estimating the contribution of the stock to catch per unit effort can also include variables measuring the quality and role of the nursery services provided by marine ecosystems.

Note that both productivity change and residual value methods can also be applied to value habitat provision by establishing the links and contributions to other ecosystem services. For example, an important aspect in the supply of pollination services will be the availability of suitable habitat for pollinators.

4.3.12.2 **Tiers**

The productivity change method is considered a Tier 3 method and the residual value method is considered a Tier 1 method.

4.3.12.3 **Examples**

Anneboina and Kumar (2017) use the productivity change method to value the nursery function of mangroves for commercial marine fisheries in India. Most studies to date had estimated mangrove contribution to total fisheries productivity (10-32%), but not specifically for commercial fisheries. This study determines that the marginal effect of mangroves on technical efficiency of the commercial marine fishery is 1.86t/ha fringe mangrove per year. Marginal mangrove productivity is found by (1) estimating the technical efficiency of the fishery using a stochastic production frontier model, and then (2) regressing the technical efficiency on mangrove shoreline area per state. The authors use landing prices of fish species to estimate a gross value of output, rather than gross value added; marketing and input costs are not excluded. Since the marginal productivity increase already controls for technical efficiency of other inputs to the commercial fishing fleet, using a market price/gross value approach for the (costless) mangrove nursery provision is accounting compatible. Gross value of output is multiplied by average marginal contribution per hectare mangrove to obtain an estimate of 146,000 Rs/ha yr (approximately US$1,900/ha yr).

5. Asset valuation

This chapter describes the methodology for compiling monetary ecosystem asset accounts. The sections discuss issues around discount rates and projecting future service flows. Applications of the valuation of ecosystem assets in countries and as part of the World Bank Comprehensive Wealth Accounts (Lange et al., 2018) and the UN Environment Programme's (UNEP) Inclusive Wealth Approach (Managi and Kumar, 2018) are also presented and reviewed.

5.1 Introduction and approach

Assets as defined in the SNA, are entities that must be owned by some economic unit, or units, and from which economic benefits are derived by their owner(s) by holding or using them over a period of time. Consumer durables and human capital, as well as natural resources that are not owned, are excluded. Land is included as an asset but the atmosphere and the oceans are not. Also, the value of land as an asset does not cover the value of services its owner cannot capture, such as sequestration of carbon. The SEEA CF extends the SNA asset boundary by having a broader understanding of benefits, and as a result the whole physical environment becomes in scope of measurement. The SEEA EA in addition extends the SNA production boundary to include non-marketed ecosystem services derived from ecosystems.

The valuation of assets in the SNA is based on economic principles, in the absence of market prices for these assets, to estimate their value based on the net present value (NPV) of the flow of benefits (income flows) that the assets are expected to provide. In mathematical terms, the value of an asset *V* is written as:

$$V = \sum_{t=0}^{t=N} \frac{R_t}{(1+r)^t} \qquad [1]$$

Where R_t is the expected net income from the asset in period *t*, *r* is the discount rate, and *N* is the lifetime of the asset, which may be infinite for some ecosystem assets if used sustainably.

In the context of ecosystem accounting, we apply the same approach, by valuing ecosystem assets based upon the set of ecosystem services they provide, using the information from the monetary ecosystem services accounts, and additional assumptions. In mathematical terms, and following SEEA EA Chapter 10, the value (V) of a single ecosystem asset is written as:

$$V_t(\boldsymbol{EA}) = \sum_{i=1}^{i=S} \sum_{j=t+1}^{j=t+N} \frac{ES_t^{ij}(\boldsymbol{EA}_t)}{(1+r_j)^{(j-t)}} \qquad [2]$$

where ES_t^{ij} is the value of ecosystem service *i* in year *j* as expected in base year t (e.g. 2020) generated by a specific ecosystem asset $\boldsymbol{EA}_t$; S is the total number of ecosystem services; r_j is the discount rate (in year *j*), and *N* is the lifetime of the asset, which may be infinite for some ecosystem assets if used sustainably.[40]

Since each flow is measured using an exchange value concept, the resulting asset value is also an exchange value. The corresponding welfare value of ecosystem assets would use the expected flow of welfare values over time.

Equation 2 assumes that ES are separable. Two key parameters in determining the value of an ecosystem asset are the discount rate and estimating the future flow of ES. Each of these is considered in more detail below.

The other important parameter is the expected asset life. SEEA EA (UN et al., 2021; section 10.3.5) notes that estimates of the asset life should be based on consideration of the condition of the ecosystem asset and its capacity to supply the set of ecosystem services being considered in the valuation of the ecosystem asset. It is possible to assume an infinite asset life when it is expected that the ecosystem asset will be used long into the future. In practice, rather than assuming an infinite asset life a maximum asset life of 100 years is applied which is computationally as easy to derive but which may be more straightforward to explain to users. Further, where discount rates are relatively low (e.g. less than 3%) NPV estimates will be relatively unaffected by using 100 years as a maximum asset life rather than an infinite asset life.

In selecting an asset life however, it remains fundamental to understand the condition of the ecosystem and the likely pattern of use. Thus, as noted in the SEEA EA, it is recommended that estimates of asset life be based on patterns of ecosystem use that have occurred in the recent past rather than on the utilization of general assumptions regarding future sustainability or intended or optimal management practices (UN et al., 2021; para. 10.72)

5.2 The Discount Rate

5.2.1 *Definition*

In order to compare income and costs at different points in time, it is standard practice to apply a discount rate to future values. The value attached today to receiving one euro a year from now is expressed as: $\frac{1}{(1+r)}$, where r is the discount rate. If *r* were 0.05 (5 per cent), the present value of a euro in one year's time would be 0.95 euro. If the discount rate is constant, and one wants to know the present value of receiving one euro two years from now, one needs to discount twice and obtains

[40] The assumption made here is that the returns accrue at the end of the accounting period and hence the first future period's flows (e.g., 2021) are discounted. This assumption is used to simplify the explanation and the associated notation but has no impact on the underlying relationships described.

0.91 euro. The mathematical expression for that could be written as: $\frac{1}{(1+0.05)^2}$. Extending this over a number of years would result in a value that declines geometrically.

5.2.2 *What discount rate should be used to value ecosystem assets?*

Arrow et al. (1996) distinguish between two fundamentally different approaches to choose the discount rate in order to compare consumption in different time periods: the *prescriptive* approach and the *descriptive* approach. The prescriptive approach begins by asking how trade-offs between present and future generations *should* be made. The descriptive approach, by contrast, begins by asking what choices involving trade-offs across time do people *actually* make? (Arrow et al., 1996; pp. 129).

5.2.2.1 **The social discount rate**

The distinction between these two approaches can be better understood by considering the following equation, which seeks to determine a social discount rate (SDR) as follows:

$$SDR = \rho + \theta g \qquad [3]$$

Where *SDR* is the social discount rate, ρ is the pure rate of time preference or utility discount rate, θ is a measure of the rate in which consumption increases welfare (technically referred to as the elasticity of the marginal utility of consumption), and *g* is the growth rate of per capita consumption.[41] The equation states that the discount rate is made up of two terms:

- The pure rate of time preference (PRTP), (ρ) as a measure of individual or societal "impatience". If $\rho > 0$, current consumers care less about future utility (or welfare) than about today's welfare.

- The growth in *per capita* consumption over time, expressed in units of utility (θg).

Among economists and philosophers there is less controversy about the second term. If future generations are expected to become richer (a common assumption of most socioeconomic scenarios underlying impact assessments of investment in natural capital), a future consumption bundle will be worth less than an equivalent consumption bundle today. How much less depends on the growth of per capita consumption, *g*, and the elasticity of utility, θ (i.e. the marginal contribution of one extra unit of consumption to utility). Disagreements arise, however, over what value of *g* can be assumed over long periods of time. Those favouring a lower discount rate argue that society cannot sustain positive

[41] θ can also be interpreted as society's aversion to inequality. The higher the value the greater the penalty attached to future generations if they are richer than present ones.

rates of growth of per capita consumption, and suggest that over the long term *g* should take a value close to zero.

Greater controversy surrounds *ρ*, the PRTP. Advocates of the prescriptive approach to discounting argue that the PRTP should be zero or close to zero, because a higher rate could leave future generations short-changed, without the possibility of redress through intergenerational transfers. Advocates of the descriptive approach, however, argue that the PRTP should be inferred from actual (savings) decisions people make (revealed preference), and that a PRTP of zero, for example, is incompatible with current rates of savings in developed and developing countries. The problem with that approach, however, is that is also involves assumptions about the parameter θ.

The debate regarding the value of *ρ* in selecting the SDR is not new. Frank Ramsey, arguably the "father" of the debate on discounting, stated that taking a value greater than zero was ethically indefensible (Ramsey, 1928). Of course, individuals may apply a higher pure rate of time preferences in making their decisions, as they face greater risks as individuals, including their possible death during the decision period. Society, however, can be seen as "infinitely lived", or close to it. More recently, Stern (2006) has argued that the only reason for taking a positive value for the social PRTP is the "exogenous possibility of [human] extinction" (Stern, 2006; p. 60). On this basis he proposes a value of 0.01. Taking the viewpoint that even the representative agent must die and using demographics to inform long run discount rates, Addicot et al. (2019) conclude that the value of *ρ,* based on national mortality rates and life expectancy, lies in the range of 1-3 per cent across countries.

The prescriptive approach to discounting, advocates for a lower rate of discount than the descriptive approach. The growth rate of per capita consumption, *g*, depends on the underlying socioeconomic scenario, and may differ between countries and time periods. With a high *g*, discounting the future is justified by the assumption that those living in the future will be better off than those living today. World median household income today is about USD10,000 (Phelps and Crabtree, 2013). At a 2 per cent growth rate, real household median income in the year 2100 will be about USD54,000, or 5½ times higher than today. At the 1.3 per cent income growth rate used by Stern (2016), it would be USD35,000, or 3½ times higher. One argument against taking such a growth rate for a long time into the future is that the past experience of growth may not continue. Some take the view that there are good reasons to believe it will slow down (Piketty, 2014).

Estimates of the elasticity of marginal utility, θ, usually range between 1 and 2. This implies that the marginal utility of consumption drops by 1 per cent or 2 per cent when the level of consumption increases by 1 per cent. The higher the value of θ, the greater is the weight given to a poor person's income relative to a rich person. Thus, if future generations are going to be richer than the present generation on account of economic growth, their utility is discounted more highly. Of course, if future generations were to be poorer, the same value of θ would imply their utility holds a greater weight. Some economists have attempted to infer a value of θ from government decisions relations to intra- and inter-generational transfers, tax principles and risk aversion in insurance markets. They come up with values in the range of 1–2 (Stern, 1977) and 1.5 (Groom and Maddison, 2019).

What then should the SDR be? A recent survey of over 200 well-known economists/philosophers on the long-term SDR found a mean (median) recommended long-term SDR of 2.25 per cent. While there is considerable disagreement on precise SDRs, 92 per cent of experts were comfortable with SDRs somewhere in the interval of 1 per cent to 3 per cent (Drupp et al., 2015).

5.2.2.2 Declining discount rates

The above discussion of the discount rate does not account for how rates change over time. To date, most governments use a constant rate, but economic theory does not make a case for that. In particular, there are a number of scholars who have presented reasons why the discount rate should decline with time. What follows is a short summary of these arguments.

There is evidence to suggest that individuals and societies do not discount the future at a constant rate but rather that they adopt a "hyperbolic" path (Kim and Zauberman, 2009; Settle and Shogren, 2004). To illustrate this, consider the following (based on consumption discount rates), an individual is faced with two choices: (1) postponing consumption for one year from now; or (2) deferring an equal amount of consumption for one year but only 50 years from now (year 50 to year 51). Most individuals are likely to respond differently to these two choices. While postponing the consumption right now might mean a lot, postponing it for an equal amount of time in 50 years from now might not. In other words, the weight placed on an extra year in the future is declining with time. However, the standard formula for constant discounting gives the same value to both types of postponement. If one accepts this line of reasoning the discount rate should decline over time.

Another argument for declining rates is that if the 'right' discount rate is not known with certainty, society should take a weighted discount rate based on ignorance. Newall and Pizer (2003), have shown that future rates decline because of dynamic uncertainty about future events.

To summarize the discussion on the SDR, there is general agreement that a value in the range of 1-3 per cent is appropriate. There is also a strong case for applying rates that decline over time and that is indeed what some governments are currently doing. The UK Treasury in its Green Book (HM Treasury, 2003), proposed the rates given in Table 6.[42]

Table 6: *UK Treasury recommended discount rates*

Period of Years	0-30	31-75	76-125	126-200	201-300	300+
Discount Rate	3.5	3.0	2.5	2.0	1.5	1.0

Source: HM Treasury (2003)

In a similar fashion, France decided in 2004 to replace its constant discount rate of 8 per cent to a 4 per cent discount rate for maturities below 30 years, and a discount rate that decreases to 2 per cent for larger maturities (Ni, 2017).[43] Finally, the Office of Management and Budget of the US Government, recognize the possibility of declining rates (see appendix D of US Government, 2013).

The above discussion raises the question of whether the same discount rate should be used to value: (a) all assets, or (b) all environmental assets? If one adopts declining rates (as some governments

[42] It is worth noting that with such rates, about two-thirds of the value is discounted away in the first step, which remains the dominant one.

[43] This should be interpreted as using a discount factor equalling $(1.04)^{-t}$ if the time horizon t is less than 30 years, and a discount rate equalling $(1.04)^{-30}(1.02)^{-(t-30)}$ if t is larger than 30.

have done) the average rate for different assets will not be the same; those with a flow of services over a longer period of time will have a lower average.[44] If, however, a constant rate is used, for long-lived environmental assets such as forests, the chosen rate was often lower than that for physical assets even before declining discount rates were thought of. So in some countries, a different rate has already been applied for some environmental assets compared to non-environmental assets. There are, however, differences of opinion with using different rates. They are discussed in the next sub-section.

5.2.3 *The market discount rate and the recommendations in the SEEA*

Individuals and firms do not apply the SDR when deciding on their investments or savings. In the case of individuals, they often borrow at very high rates of interest, implying a correspondingly high discount of the future. For firms, the key factor is the return on capital they could receive if they invested the same amount of money in some other asset or project, with the same degree of risk as with the asset being valued. Such rates (adjusted for inflation) can be very high, especially in developing countries. Typical rates are in the range of 10 per cent and above, depending on the risks involved.

The SEEA CF (UN et al., 2014) discusses the use of discount rates as a key component of the *Net Present Value* approach. While there is some discussion about the SDR and how it may be determined, the SEEA CF does not recommend the use of a social rate. The final statement declares: "It is recommended that a discount rate be determined that is consistent with the general approach to valuation in the SEEA and the SNA, i.e., consistent with valuation at market prices. This suggests the choice of an individual discount rate that reflects the return needed by those undertaking an activity to justify investment in that activity. Consequently, the relevant rate should be descriptive and, ideally, should include any activity-specific risks" (UN et al., 2014; p.231). It further explicitly states: "Because judgements are required regarding societal preferences, it is not recommended that prescriptive approaches to the determination of discount rates be used for the purposes of official statistics" (UN et al., 2014.; p.232).

This view has been modified in the revised SEEA EA (UN et al., 2021; para. 10.77), which states that the following should be applied in selecting a discount rate:

- Individual, market-based discount rates should be applied in the valuation of ecosystem services whose users are private economic units; and

- Social discount rates should be applied in the valuation of ecosystem services that contribute to collective benefits, that is, benefits received by groups of people or society in general.

Many of the ecosystem assets generate returns over very long periods and, as was shown in the first section, a high discount rate places very little value to returns after about 50 years. When the valuation involves a *change* in the flow of services, such as carbon sequestration, the use of market-based rates

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[44] The average can be computed as the constant rate that gives the same NPV and that obtained from the declining rates.

would result in very low values that are not socially defensible. Thus, there is a case for using social discount rates to value such assets.

Specifically, SEEA EA recommends (in section 10.3.7) that the selection of a SDR for SEEA EA purposes should be based on rates as specified in relevant government guidelines. Examples of such prescriptive rates can be found in the United Kingdom, France and the United States. In case such rates are not available, compilers may consider using long-term government bond rates. In applying discount rates, SEEA EA recommends that compilers use a constant rate over the asset life.

Care should be taken to ensure that the discount rate applied is consistent with the assumptions made in projecting future returns of ecosystem services. Specifically, if future returns are estimated in nominal prices then the discount rate should include an allowance for expected inflation. Most commonly, future returns will be estimated in real terms and thus the discount rate applied should also be in real terms. Since the essential function of a discount rate is to reflect the time value of money, the appropriate measure of expected inflation is likely to be one that is economy-wide in scope, for example, the GDP deflator.

Finally, compilers are encouraged to undertake an assessment of the sensitivity of monetary valuations to different assumptions, in particular through the application of alternative discount rates. Such assessments can be published as part of the general documentation of the accounts.

5.2.4 *Projecting future ES flows*

One of the most difficult steps in valuing assets is to determine the value of future ecosystem services flows. In making an estimate, account needs to be taken of both the evolution of demand for the services of the ecosystem and the evolution of supply.

The principal factors that influence demand include population growth, real incomes and changes in preferences. Demand can normally be expected to increase proportionally with the population that uses the ecosystem service (demand for carbon emissions is an exception). The relationship between growth in *per capita* real income and individual demand is measured through two parameters: how the WTP for a given quantity of the ES varies with real income (the WTP elasticity of demand) as well as how the quantity of the ES demanded varies with income (the income elasticity of demand) (Flores and Carson, 1997). Estimates of the WTP elasticity of demand are found to be greater than zero, but less than one (Hökby and Söderqvist, 2003), while studies of the income elasticity of demand indicate values greater than one (at least relating to recreational services) (Ghalwash, 2008).[45] In projecting demand for a given ES, both elasticities will be combined to determine the change in the WTP. With regard to recreational services, it will be a combination of the change in the WTP for a given visit multiplied by the change in the number of visits. Lastly, there may also be a change in preferences, driven possibly by climate. For example, studies have shown that a rise in temperature for a specific site has a non-linear impact on the number of visits to a site, with a temperature of around 21.6ºC being optimal (Lise and Tol, 2002). With input services, such as

[45] For the UK, however, an income elasticity of 0.5 was used.

water in agriculture, demand can be expected to increase with global warming.[46] All these factors have to be taken into account in making projections of future demand for specific ES.

The supply side of ES is impacted by the current and past use rate of the services, as well as factors affecting the ecological condition of the ecosystem asset, such as climate change and land management practices. If an ecosystem is being overused (such as grazing land), its future productivity (what has been called the capacity of the ecosystem asset) will be reduced and the extent of such an effect should be taken into account. For air filtration and water purification, a key factor will be expected changes in the concentrations of pollutants, which could be mediated by the ecosystem. For flood protection, local climate regulation and storm protection services, climate change impacts will be most relevant. In all cases, the loss of habitat and associated biodiversity losses are key factors that drive the change in supply of the ES. In terms of exogenous impacts, climate change can also alter the productivity of agricultural land, forests and other ecosystems (Field et al., 2014).

Both demand and supply of each ES need to be analysed to determine which will be the limiting factor in determining the quantity that is utilized and, then, to determine the value of that ES for each year in the future. Examples of how projections have been made in actual estimates of the value of different ecosystems types are given in the next section.

5.3 Examples of asset valuation

This section discusses several examples of studies that have valued ecosystem assets as part of a natural capital accounting exercise, namely: in the UK (ONS, 2019), in KwaZulu-Natal in South Africa (Turpie et al., 2021), and by the World Bank (Lange et al. 2018) and UNEP (Managi and Kumar, 2018) for a large set of countries. Here only the valuation of ecosystem assets (through the valuation of ES) is considered; abiotic assets such as minerals are outside the scope of the SEEA EA. In the following summaries, the emphasis is on how information about the current level of ES values is converted into an asset value.

In the UNEP and World Bank examples, no attempt has been made to value individual ecosystem assets at a detailed spatial level. This is partly because the projections of future service flows can incorporate assumptions about future population growth, income growth, etc. at a national level, but these projections could not be applied at the level of individual ecosystem assets in a global exercise. In the case of KwaZulu-Natal, projections were made at a more detailed level, but only the future sustainability of wild resource extraction was taken into account. The UK national accounts have a disaggregation of assets by habitat type.

The Comprehensive Wealth (CW) Approach of the World Bank is based on the NPV method, with rental rates for different services, which seek to measure net income attributable to the ES, applied to estimates of the physical quantity of service provided. Services valued are: timber, a range of non-timber forest services (non-timber products; hunting, fishing and recreation; and watershed protection), agricultural biomass and services from protected areas. Further information is available

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[46] For detailed projections of crop water demand under climate change in Europe, see: https://www.eea.europa.eu/data-and-maps/indicators/water-requirement-2/assessment

in Lange et al. (2018).[47] A discount rate of 4 per cent is applied to all future flows, but the asset life over which the service flows are projected varies for different services.

The Inclusive Wealth (IW) approach of UNEP is similar in many respects to the CW approach, based on the NPV method. Similarly, ES taken into account in the valuation of ecosystem assets are agricultural biomass, timber, non-timber forest services, and aquatic biomass. Details are available in Managi and Kumar (eds) (2018). A discount rate of 5 per cent is applied to all future flows, with the assumption of infinite life.

Compared to the IW and CW studies, the UK asset valuation assessment (ONS, 2019) is more comprehensive, covering a wider range of services and a finer breakdown by ecosystem types. Services covered are fish capture, agricultural biomass, timber, water abstraction, carbon sequestration, air filtration, noise mitigation, urban cooling and recreation and amenity services.

The discount rate used is a SDR of 3.5 per cent that declines over time, reducing to 3.0 per cent after 30 years and to 2.5 per cent after 70 years. This is based on the UK Government's recommendations (HM Treasury, 2003). Flows are discounted over a 100-year asset life. The results for 2016, as released in 2019 are summarized in Table 7.[48] The asset values are based on exchange valuations: the only values that could pick up non-exchange values are those related to air pollution removal. The largest component is recreation, which accounts for nearly half the total, followed by agricultural biomass, which accounts for about 12 per cent.

[47] The resulting wealth estimates (per country) are available at: https://datacatalog.worldbank.org/dataset/wealth-accounting.
[48] Noise mitigation was considered too experimental to be included.

__Table 7:__ UK Net Present Values of Ecosystem services: Results for 2016

Service	Value £Mn.	Method Used
Provisioning		
Agricultural Biomass	118,426	Residual value resource rent after allowing for all inputs
Fish Capture	7,584	Marine fish capture in the UK EEZ valued at net profit per tonne
Fossil Fuels	95,285	Rents per tonne removed based on residual value
Mineral	5,483	Rents per tonne removed based on residual value
Timber	8,517	Stumpage price times the physical amount of timber removed
Water Abstraction	76,370	Resource rents from abstraction after deducting costs
Renewables Generation	7,887	Residual value resource rent after allowing for all inputs
Regulating		
Carbon Sequestration	103,947	Valued at the projected price of non-traded carbon, based on MAC to meet a given target
Air Pollution Removal	43,907	Valuation of health benefits based on WTP to avoid illness and premature death
Urban Cooling	11,398	Estimated cost savings from air conditioning and benefit from improved labour productivity
Noise Mitigation	-	
Cultural		
Recreation	393,707	Expenditure incurred to travel to the natural environment and incurred during visit
Aesthetic (House Prices)	9,428	Hedonic price study
Recreation (House Prices)	68,552	Hedonic price study
Total	**951,323**	

Source: ONS (2019)

The study in KwaZulu-Natal in South Africa by Turpie et al. (2021), covered a wide range of ES: wild resources, animal production and cultivated biomass; pollination, carbon storage, water flow regulation, water filtration, flood regulation, sediment control; nature-based tourism and amenity values. Future flows of all but wild resources were assumed to be the same as the current level, projected over a 25-year period, and discounted using a social discount rate of 3.66 per cent.

5.3.1 *Wood provisioning services*

In the CW approach, timber rents per cubic metre are taken as a five-year moving average and quantities are based on recent rates of extraction. The lifetime of timber resources is determined by the rate of timber extraction (Q) relative to the rate of natural growth (N). If Q > N, then current rates of extraction are unsustainable, and the lifetime of the resource is limited by the values of Q and N. If Q ≤ N, then extraction is assumed to be sustainable, and the lifetime of the resource is taken as infinite. Data on prices, output and extraction rates relative to natural growth are at the country level. Data for the rental rates, however, are estimated regionally.

In the IW approach, the future rental rates, areas and prices are assumed to be constant. The price used is the average of the last 25 years. Unlike the CW approach, no adjustment for the sustainability of extraction is made.

In the UK accounts, projected flows of timber provisioning services were based on Forestry Commission forecasts of timber availability to estimate the pattern of expected future flows of the service over the ecosystem asset lifetime of 100 years. Rental prices were based on a moving average of stumpage prices (see Chapter 4 for a discussion on this approach).

5.3.2 *Wild animals, plants and other biomass provisioning services*

In the CW approach, non-timber forest services are based on estimates of per hectare values for three benefit categories: non-wood forest products; recreation, hunting, and fishing; and watershed protection. The capitalized value of non-timber services is equal to the present value of annual services, discounted into the indefinite future. Similarly, the IW approach combines estimates of a range of services. In this case, estimates are made for three different types of forest. These ES values are then assumed to continue, discounted over an infinite time horizon.

In the UK accounts, recreation and amenity service flows are projected separately (see below); non-timber forest products, hunting and watershed protection are not covered within the accounts.

In the KwaZulu-Natal accounts, the future use flows of wild resources (wood biomass, grasses and reeds, palm leaves and wild animals) were projected by comparing current extraction rates with estimates of sustainable yield. Where current extraction exceeded sustainable yield, it was assumed extraction would continue at that level until the resource was depleted; where current extraction was at or below the sustainable level, it was assumed that future extraction would continue at the current level.

5.3.3 *Fish provisioning services*

The CW approach does not cover fish provisioning services.

The IW methodology annex presents a modelling approach to estimate the stock of different fish species based on estimating a harvest function or on tracking total resource dynamics. In most cases, both methods make data demands that are not possible to meet. Therefore, a simpler rule that is applied estimates the stock according to the following:

- If the year being studied follows the year of the maximum catch, then the biomass stock is estimated as twice the catch; and
- Otherwise, the biomass stock is estimated as twice the maximum catch, net of the catch (i.e. 2 x Maximum Catch – Catch).[49]

[49] The approach has significant limitations, since if the stock has been overfished at some point in the past, it will assume a higher catch is sustainable than is actually the case. In addition, the rule of thumb of biomass being twice the maximum sustainable catch also needs to be checked for the species to which it is applied.

The time series data of the catch (tonnage and value) of each country's EEZ, either by domestic or foreign fleets, is for the period of 1950-2010. The catch from the stock is valued using a period average, i.e. species average market price multiplied by the rental rate. Future catch levels are estimated from a model, which adjusts the stock if the catch exceeds, or is less than, the natural increase. The demand for the catch includes population and GDP growth and changes in fishery management systems in some of the major fishing countries.

In the UK accounts, valuations are calculated using net profit per ton (landed) estimates for different marine species. Annual net profit per ton (landed weight) is multiplied by ton of fish captured (live weight) for a specific species. This data is aggregated for overall annual valuations of fish provisioning from the UK EEZ. This was assumed to remain constant over the future, which is a questionable assumption in light of changes to stocks.

5.3.4 *Crop and grazed biomass provisioning services*

In the CW approach, rents for crop and grazing biomass assume a growth rate, *g,* in agricultural productivity. For crops, a rate of 1.94 per cent is assumed for g for all low- and middle- income countries, and a rate of 0.97 per cent is assumed for g for all high-income countries. For grazed biomass, proxied by livestock products, 2.95 per cent is assumed for low- and middle-income countries and 0.89 per cent for high-income countries (Rosengrant, Agcaoili-Sombilla, and Perez, 1995).

In the IW approach, crop biomass service flows are based on estimated rent for different parcels of land multiplied by the quantity of different crops produced. Future rents are assumed to be the same as present rents and based on that a simple net present value is computed. To avoid fluctuations in the adjusted price an average of the last 25 years is used.

For grazing biomass, it was not possible to calculate the NPV from available data, so the rent was assumed to be the same as the shadow price for crop biomass.

In the UK accounts, residual values based on a five-year moving average for agricultural production, were assumed to be constant in future years.

5.3.5 *Water supply*

In the UK accounts, residual values for public water supply (including water treatment) were estimated based on a five-year moving average. They were assumed to be constant in future years.

5.3.6 *Global climate regulation*

In the UK accounts, the annual value of the service is based on the estimates of the carbon captured by ecosystems multiplied by the carbon price.

The projections of physical flows are based on forecasts of future sequestration under a central reference scenario, which incorporates funded commitments such as additional tree planting.

The carbon price used in calculations is based on the projected non-traded price of carbon schedule. This is contained within the Data Table 3 of the Green Book supplementary guidance. Carbon prices are available from 2010 to 2100. Prices beyond 2100 are assumed to be constant at 2100 levels. The non-traded carbon prices are used in appraising policies influencing emissions in sectors not covered by the EU Emissions Trading System (ETS) (the non-traded sector). This is based on estimates of the marginal abatement cost (MAC) required to meet a specific emission reduction target. Beyond 2030, with the (expected) development of a more comprehensive global carbon market, the traded and non-traded prices of carbon are assumed to converge into a single traded price of carbon. Further discussion can be found in Section 4.3 of this document.

5.3.7 *Air filtration services*

In the UK accounts, the starting point is the estimated concentration of various pollutants harmful to health in the atmosphere across the UK on a 50x50km grid. Air pollution removal by UK vegetation has been modelled for the years 2007, 2011, 2015 and also for 2030 based on projections of climate and future pollution emissions (noting these are expected to fall and hence the ES flow is declining over time). Between these years, a linear interpolation has been used and adjusted for real pollution levels as an estimation of air pollution removal.

The health benefits were calculated from the change in pollutant concentration to which people are exposed. Damage costs per unit exposure were then applied to the benefiting population at the local authority level for a range of avoided health outcomes. For methods of valuing the health impacts and issues related to them see the discussion on air filtration in chapter 4.

Future physical flow (concentration reductions) beyond 2030, were assumed to be constant. Future demand for the service took into account an average population growth rate and an assumed 2 per cent increase in income per year (declining to 1.5 per cent increase after 30 years and 1 per cent after 75 years). Income elasticity was assumed to be one.

5.3.8 *Local climate regulation – urban cooling*

In the UK accounts, the physical benefit of urban cooling is based on an estimate of the proportional reduction in city-level temperatures caused by the urban cooling effect of blue and green space features and their buffers. This service is monetised through the estimated cost savings from air conditioning and the benefit from improved labour productivity. The benefit from improved labour productivity makes up most of the value, where avoided air-conditioning energy costs only account for a small fraction.

The monetary account of the future provision of the ES, or future benefit stream, incorporates a projection for an annual increase in working day productivity losses due to climate change, which increases the value of urban cooling over time. The assessment of future climate impact relies on the broad estimation of the number and degree of hot days in the future across the UK. As well as including climate change impacts, an annual uplift is applied to the monetary values to account for year-on-year increases in GDP over the 100-year assessment period. For the first 30 years this uplift is 2 per cent annually, decreasing to 1.5 per cent for years 31 to 75, and 1 per cent for years 76 to 100.

5.3.9 *Recreation and amenity services*

The UK accounts take two approaches for the measurement of the asset valuation of outdoor recreation. The first approach is for visits which entail some expenditure, and is referred to as the consumption expenditure approach. Future flows were projected using population growth forecasts from population statistics by the Office of National Statistics and an income uplift assumption. The income uplift assumptions are 1 per cent, declining to 0.75 per cent after 30 years and 0.5 per cent after a further 45 years. These assumptions project the annual value to increase over the 100 years.

The second approach values the service enjoyed by residents by simply living close to open spaces and enjoying the visual benefits they provide and being able to visit them without incurring any travel costs. These benefits are measured through the hedonic approach (see Section 3.2), where house prices are regressed on a wide set of variables, including the proximity to open spaces. The resulting estimate is a capital value that does not then require the NPV approach to arrive at the asset value of the service.

5.3.10 *Other values*

The last category in the CW accounts relates to protected areas. It is argued that such areas provide a range of services to the country. For example, "wildlife reserves can generate significant revenues for developing countries, in particular from international tourism activities. And about one-third of the world's big cities get their drinking water from sources in or downstream of protected areas, saving billions of dollars in supply and treatment costs thanks to forests and wetlands that regulate the flow of water and remove contaminants" (Dudley et al., 2010). Valuing these individual ecosystem services on a national basis, however, is difficult. For this reason, protected areas are valued in the World Bank's wealth accounts using a simplified approach. Under this approach, the quasi-opportunity cost of protection per unit area of land contained in terrestrial protected areas is estimated as the lower of returns to cropland and pastureland. The values of the latter are estimated as part of the national agricultural land component of wealth. This is likely to be a lower bound on the true value of protected areas. Future values are assumed to be constant as the returns on crop/pastureland are constant.

5.4 **Conclusions**

In practice, international attempts to obtain an estimate of the value of natural capital contain a mixture of the exchange value and the welfare value approaches. The Inclusive Wealth approach by UNEP (Managi and Kumar, 2018) estimates values of agricultural land and forest timber services derived from an exchange value. The estimates are, however, consistent with a welfare value. Non-timber forest services and value of carbon sequestration are based on welfare values, so the whole estimation of natural capital is consistent with welfare value.

The World Bank's Comprehensive Wealth Approach (Lange et al., 2018), seeks to be consistent with exchange values and its estimates for cropland, pasture, timber and protected areas are based on exchange values. Estimates for recreation, hunting, and fishing and non-wood forest products on the other hand seem to be based on welfare values.

At the same time, the CW and IW work demonstrate that the calculation of NPV is possible and, through the use of alternative, exchange value consistent methods for all ecosystem services (as described in chapter 4), exchange value based NPV for ecosystem assets could be obtained. This potential is also evidenced by the results that have been published on a regular basis by the UK ONS (as described in this Section), the work of Turpie et al. (2021) and the work of Statistics Netherlands.

Given the potential that has been demonstrated in this area, further work to better align methods and assumptions (e.g. concerning discount rates and asset lives) will contribute greatly to the development of comparable estimates.

6. Other considerations in compiling monetary ecosystem accounts

This chapter describes a number of relevant topics when compiling monetary accounts. These topics are: (i) the use of value transfer approaches in ecosystem accounting; (ii) the availability of platforms and tools to support valuation work; (iii) assessing the accuracy and reliability of valuation, including its fitness for purpose; (iv) considerations in aggregation of values; and (v) approaches to communicating monetary values.

6.1 Value transfer for ecosystem accounting

Studies of the values of ES are location and time specific and usually do not cover all locations within a country. In cases where there is no time or resources to conduct primary valuation, it is common to use **value transfer** (VT) methods, i.e. use research results from pre-existing studies at one or more sites and for a range of time periods to predict value estimates for other sites and time periods.[50] Transfer of values refers to both physical as well as monetary metrics, although the method is most commonly applied for monetary estimates. Since the intent in ecosystem accounting is to provide a comprehensive geographical coverage of all flows of ES within an accounting area, some form of spatial generalization (extrapolation or interpolation) will always be required for physical and monetary methods that quantify ecosystem services. Below we discuss value transfer applied to monetary valuation of ecosystem services.

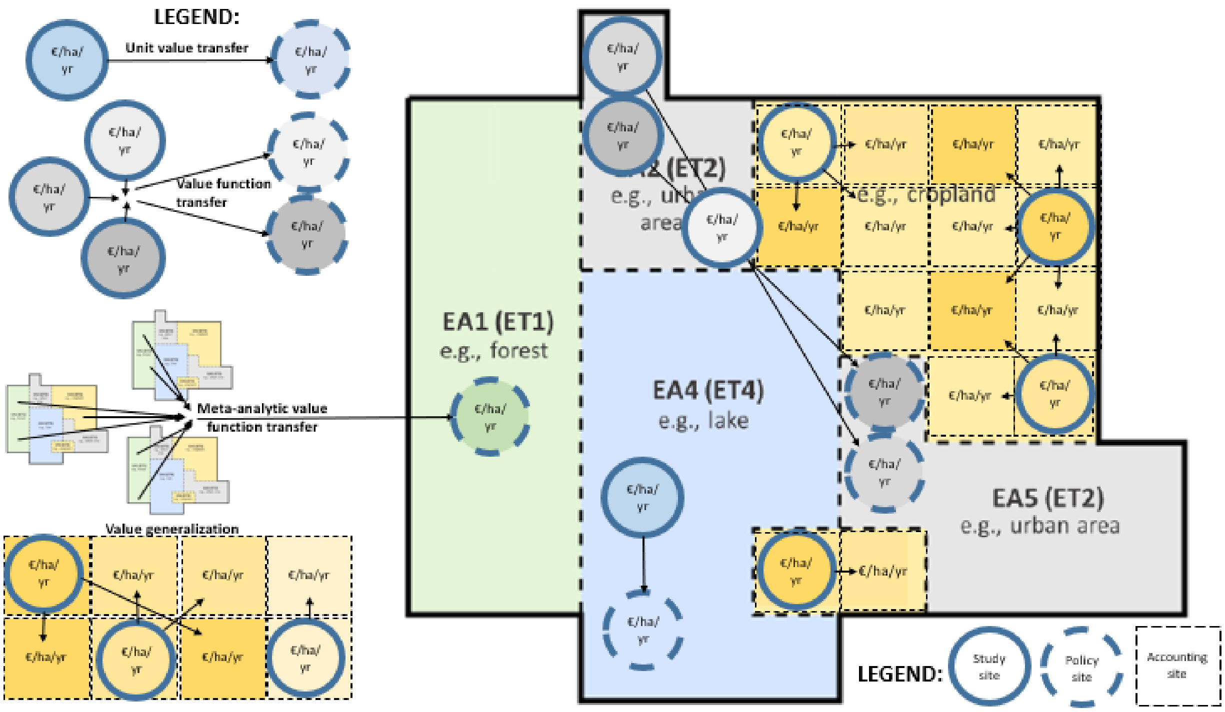
Value transfer (see Figure 8) should be distinguished from **value generalization,** which involves applying data from one or more study sites to describe ecosystem service values of all ecosystem locations within an accounting area (see Figure 8). An example of the latter could be measuring the biomass at a few sampled points within an ecosystem asset and extrapolating those numbers spatially to all grid cells of that same ecosystem asset. Value generalization occurs frequently (also in economic statistics when collecting data based on a sample) and is not the topic of discussion in this report (see also the Guidelines on Biophysical modelling (UN, 2022) for a discussion on spatialization).

Value transfer is now the subject of a large body of literature to ensure that the procedures used in such transfer are valid and reliable (Johnston et al. 2021). All value transfer methods can be used for value generalization in accounting; the difference lies in extrapolating to a few sites for location specific policy/project analysis purposes common in the environmental economics literature, and the full geographical coverage required for ecosystem accounting purposes. Value transfer methods can be considered variations of the same approach along a gradient of increasing information and

[50] In environmental economics, these approaches are also commonly referred to as benefit transfers or secondary valuations. Through the SEEA EA revision process, discussion revealed no definitive distinction between these terms with different experts holding different explanations for the use of either benefit or value transfers as the appropriate term. Ultimately the term value transfer was chosen, in part because of the specific use of the term benefit in the SEEA EA framework, and in part because the term value can be readily interpreted with respect to monetary and non-monetary estimates and with respect to incomes and costs. The choice of term has no impact on the description of the methods themselves.

statistical power, and there are many variations in the literature. Figure 8 aims to illustrate some basic conceptual differences among the approaches. Individual VT methods are described in greater detail below.

***Figure 8**: Conceptual differences in value generalization and value transfer methods applied to ecosystem types (ETs) and ecosystem assets (EAs) within accounting area*



Unit value transfer often uses a single available valuation study, applying the unit value from the study site to a policy site (see 6.1.1.1). Simple adjustments are sometimes made, e.g. by scaling WTP estimates by differences in respondent income levels between the study and policy sites (see 6.1.1.2 on adjusted unit value transfer). **Value function transfer** (6.1.2) uses variation of respondent characteristics in a sample to predict values out of sample in a different population. A "site" may encompass a study sampling households across spatial variation in ecosystem condition or types in which case the value transfer function can be used for generalization across whatever gradient it has sampled from (this is illustrated in Fig.8, with a value function transfer sampling from one urban ecosystem to predict urban ES at multiple locations in another city). **Meta-analytic value function transfer** (6.1.33) is used when data from values studies within a country (domestic values) is unavailable. Value estimates are used from a large enough number of studies in other countries/locations to provide statistical power to adjust for foreign study and location characteristics to domestic policy sites. The different value transfer approaches aim to adjust for differences between values at study sites and *a selection of policy sites*. **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 may use different value transfer approaches depending on the data available.

A meta-analytic value function (with a large number of studies identifying household and ecosystem condition variables) is potentially more generalizable for accounting purposes than a single site value function – this depends on how much each data sample identifies socio-economic and ecological gradients that explain ES values in the accounting area. Different value transfer methods have different accuracy, which should be considered when choosing to apply a method to value generalization for ecosystem accounting (see section 6.3.2). The following sections discuss in more detail the three value transfer approaches in which data from a site where the valuation has been made can be transferred to other sites (OECD, 2018).

6.1.1 *Unit value transfer*

6.1.1.1 **Unadjusted unit value transfer**

The procedure here is to "borrow" an estimate of the ES value from context *S* (the study site) and apply it to context *P* (the alternative site also referred to as the policy site and in the SEEA as the accounting area) (see Fig. 8). The unit that is usually transferred is value per spatial unit, such as euros per hectare; or per physical unit of a service, for example euros per cubic metre of water. The unit value may also be adjusted for the number of beneficiaries, transferring, for example, euros per hectare per person for an amenity.

In the discussion below we refer to willingness to pay (WTP) because it is the most common in the value transfer literature, but for the purposes of accounting this refers to an estimate from any accounting compatible valuation method that is generalized.

The virtue of this approach is clearly its simplicity and the ease with which it can be applied once suitable original studies have been identified. Of course, the flipside of this is that it fails to capture any differences that exist between the characteristics of an original study site(s) and a new site. If these differences are significant determinants of WTP, then this transfer approach – which is sometimes more prescriptively known as a naïve transfer – will fail to reflect likely divergences in WTP at the study and policy sites. Determinants of WTP that might differ between study and policy sites include:

- The socioeconomic and demographic characteristics of the relevant populations. This might include income, educational attainment and age.
- The physical characteristics of the two sites. This might include the environmental services that the good provides such as, in the case of a river, opportunities for recreation in general and angling in particular.
- Differences in the "market" conditions applying to the sites, including the location of populations relative to the ecosystems. For example, variation in the availability of substitutes in the case of recreational resources such as rivers. Two otherwise identical rivers might be characterized by different levels of alternative recreational opportunities. Other things being equal (by assumption in this case), mean WTP to prevent a lowering of water quality at a river where there are few substitutes should be greater than WTP for avoiding the same quality loss at a river where there is an abundance of substitutes. The reason for this is that the former is a scarcer recreational resource than the latter.
- Temporal changes. There may be changes in valuations over time, perhaps because of increasing incomes and/or decreasing availability of clean rivers.

As a general rule, there is little evidence that the conditions for accepting unadjusted VT apply in practice. Effectively, those conditions amount to saying that the various conditions listed above do not hold, i.e. “sites” are effectively “identical” in all these characteristics (or that characteristics are not significant determinants of WTP, a conclusion which sits at odds with economic theory). Unit VT has been applied in well-known studies with global valuations of ecosystem services (Costanza et al., 1997; Costanza et al., 2014).

6.1.1.2 **Adjusted unit value transfer**

Due to the issues mentioned above, when applying unit value transfer, estimates from the original site are often corrected for purchasing power parity adjusted income. A widely used formula for adjusted unit value transfer is:

$$V_P = V_S\,(Y_P/Y_S)^e$$

where Y is purchasing power adjusted income per capita, V is the value estimate for a given service and *e* is income elasticity, P is the policy site for which an estimate is required and S the study site for which a value already exists.

The service may be measured in units such as euros per person for a reduction in risk or mortality or morbidity. Other units value examples given above may also form the source of the transfer. The elasticity is an estimate of how the WTP for the (non-market) good in question varies with changes in income. According to this expression, if *e* is assumed to be equal to one, then the ratio of WTP at sites *S* and *P* is equivalent to the ratio of per capita incomes at the two sites (i.e. $WTP_P/WTP_S = Y_P/Y_S$). In this example, values are simply adjusted upwards for projects affecting people with higher-than-average incomes and downwards for projects that affect people with lower than average incomes.

As an example, Hamilton et al. (2017), based in turn on OECD (2014), transfer WTP for various health states (particularly mortality risks) using the ratio of incomes between two areas (and various assumptions about the income elasticity of WTP) in order to estimate the health burden of $PM_{2.5}$ which is co-produced by industrial processes along with carbon dioxide.

In the most commonly used adjustment, the only feature that is changed between the two sites is income per capita. The rationale for this is that income per capita is the most important factor determining changes in WTP, as meta-studies, such as OECD (2014), appear to find. Of course, to the extent that, say income is not the sole determinant of WTP, then even this improvement will fall short of approximating actual WTP at the study site. However, it is also possible to make a similar adjustment for, say, changes in age structure between the two sites, changes in population density, and so on. Making multiple changes of these kind amounts to transferring value functions and this transfer approach is considered below.

6.1.2 ***Value function transfer***

A more sophisticated approach is to transfer the *benefit* or *value function* from *S* and apply it to *P*. Thus, if it is known that WTP at the study site is a function of a range of physical features of the site and its use as well as the socioeconomic (and demographic) characteristics of the population at the site, then this information itself can be used as part of the transfer.[51] For example, if $V_S = f_S(A_s, B_s, C_s, Y_s)$ where A_s, B_s, C_s are additional and significant factors affecting WTP (in addition to Y) at site *S*, then WTP_P can be estimated using the coefficients from this equation, but using the values of A_p, B_p, C_p, Y_p at site *P*: i.e.

$$V_P = f_S\,(A_p, B_p, C_p, Y_p)$$

where the term a_i refers to the coefficients which quantify the change in WTP as a result of a (marginal) change in that variable. For example, assume that WTP for a reduction in the risk of death depends on the income, age and educational attainment of the population at the study site, and that the analysts undertaking that study estimated the following relationship between WTP and these (explanatory) variables:

$$\log WTP_S = 3 + 0.5\log Y_S - 0.3\ AGE_S + 2.2\ EDUC_S$$

That is, the log of WTP*S* increases with the log of income and educational attainment (EDUC), but decreases with age (AGE) as described above. In this transfer approach, the entire benefit function would be transferred as follows:

$$\log WTP_P = 3 + 0.5\log Y_P - 0.3\ AGE_P + 2.2\ EDUC_P$$

The use of value function transfers does not always result in more accurate estimates relative to adjusted unit value transfer. Johnston et al. (2021) argue there is insufficient weight of evidence to identify the specific variables with which value function transfer enhances validity and reliability.

6.1.3 *Meta-analytic function transfer*

A still more ambitious approach is that of using meta-analysis to define the value transfer function (e.g. Bateman et al., 2000). This is a statistical analysis of summary results of a (typically) large group of studies. A meta-analysis seeks to explain why different studies result in different mean (or median) estimates of WTP. At its simplest, a meta-analysis might take an average of existing estimates of WTP, provided the dispersion about the average is not found to be substantial, and use that average in policy site studies (i.e. make a unit value transfer). Alternatively, average values might be weighted by the dispersion about the mean; the wider the dispersion, the lower the weight that an estimate would receive. Most meta-regressions use data from more than one country. This is acceptable as long as the meta-regression accounts for inter-country differences through the selected set of explanatory variables.

The results from past studies can also be analysed in such a way that variations in WTP found can be explained. This should enable better transfer of values since the analyst can learn about what

[51] The example given is for a WTP study but of course the method can be applied in other contexts as well. See for example, Lara-Pulido et al. (2018) for an application to ecosystem services more widely and Pettinotti et al. (2018) for benefits from water related ecosystems and climate change.

WTP systematically depends on. In the meta-analysis case, whole functions are transferred rather than average values, where the functions do not come from a single study, but from collections of studies. As an illustration, assume that the following function is estimated using past valuation studies of wetland provision in a particular country:

$$WTP = a_1 + a_2 \text{ TYPE OF SITE} + a_3 \text{ SIZE OF CHANGE} + a_4 \text{ VISITORS} + a_5 \text{ NON-USERS} + a_6 \text{ INCOME} + a_7 \text{ ELICITATION FORMAT} + a_8 \text{ YEAR}$$

This illustrative meta-analysis attempts to explain WTP with reference not only to the features of the wetland study sites (type, size of change in provision in the wetland as well as distinguishing between visitors and non-users) and socioeconomic characteristics (income), but also process variables relating to the methods used in original studies (elicitation format in stated preference studies and so on) and the year in which the study was undertaken. Application of meta-analysis to the field of non-market valuation has expanded rapidly in recent years. Studies have taken place with respect to urban pollution, recreation, the ecological functions of wetlands, values of statistical life, noise and congestion.

Meta-analysis VT has been applied for thematic assessments of ecosystem services: wetlands (Ghermandi et al., 2010); forests (Chiabai et al., 2011; Grammatikopoulou and Vačkářová, 2021); mangroves (Brander et al., 2012); and lakes (Reynaud and Lanzanova, 2017).

6.1.3.1 **Structural preference-calibration transfers, value scaling and generalization**

Spatial "scaling up" or "scaling down" is the equivalent term used in the benefit transfer literature to "value generalization" as applied to accounting in this report. Value transfer scaling at its extreme is provided in the example of the Costanza et al. (2014) study using unit value transfer to value changes in global ecosystem services due to land use change between 2007-2011, estimated at $4.3–20.2 trillion/yr (global GDP in 2011 was $73.1 trillion/yr).

Structural preference-calibration transfers are a more advanced form of value function transfer that allow for variations in marginal values as a function of the quantity of change or affected areas, and hence enable spatial scaling of transferred values to be applied in a way that is consistent with theoretical expectations (Johnston et al. 2021). In the case of valuation of prospective policies, the benefit transfer literature cautions that naïve "scaling up" of stated preference estimates from a small study area to a large policy area can lack credibility (ibid).

The value transfer literature has not explored this issue in the case of retrospective valuation of historical changes in ecosystem services recorded in ecosystem accounts. If the change in ecosystem extent/condition is similar in the accounting period to the scale assumed in the valuation studies used for value generalization, there should not be a scaling issue. Where ecosystem accounts record significantly larger (or smaller) losses than assumed in source studies documentation, caution and some form of correction is warranted. Greater caution regarding scaling/generalization should be taken for local than for global ecosystem services; and consideration should be given to whether the size of the market relative to the ecosystem change could lead to change in scarcity and marginal values.

6.1.4 *Guidance for conducting value transfer*

Certain implementation steps for conducting transfers have been suggested in the literature (Johnston et al., 2015; Boyle and Parmeter, 2017) – see Figure 9.

Figure 9*: Implementation steps for value transfer*

Preparation: Steps 1 to 4

Define the valuation policy context

Establish the need for a value transfer

Define the *good* to be valued and the affected population

Specify the baseline and current conditions of the good to be valued

Implementation: Steps 5 to 9

Gather and evaluate valuation data/evidence

Select the value transfer approach

Implement the transfer

Aggregate values over population, areas and time periods

Conduct sensitivity analysis and test reliability

Reporting

Report results

A range of challenges exist when conducting a VT. Of key importance is ensuring that during preparation, only those studies compatible with exchange values are selected. A systematic review is a step-wise methodology that aims to collect, assess and synthesize existing research data. These steps consist of: review scoping (keyword selection); abstract and title screening; full text screening (inclusion criteria); data extraction (applying a specific template); and reliability assessment (quality criteria). Systematic reviews have been used to compile valuation databases that can be used for different value transfer approaches. For example, Jiang et al. (2021) review the last 20 years of ecosystem services valuation in China with the aim of developing an ecosystem service valuation database. Starting from a systematic review, a meta-analysis goes on to do statistical tests for patterns in methodology and location specific features across valuation studies (e.g. Reynaud and Lanzanova 2017). Meta-analysis adjusting only for location specific features determining ES values, holding methodological features constant, is referred to above as meta-analytic transfer (e.g. Brander et al. 2012).

There are open access datasets that report the economic value of ES for various ecosystems and which can provide data for VT applications. These include the Ecosystem Service Valuation Database (ESVD) (de Groot et al., 2020); the Environmental Valuation Reference Inventory (EVRI)[52], COPI (Braat and ten Brink, 2008), ENValue (2004), EcoValue (Wilson et al., 2004), Consvalmap (Conservation

[52] See: https://www.evri.ca/Global/Splash.aspx

International, 2006), CaseBase (FSD, 2007), ValueBaseSWE (Sundberg and Söderqvist, 2004) and FEEM (Ojea et al., 2009).[53]

Much of this work is summarized in The Economics of Ecosystems and Biodiversity (TEEB) study (TEEB Synthesis report, 2010), which was launched by the G8+5 Ministers of the Environment in 2007 to draw attention to the global economic benefits of biodiversity and the costs of biodiversity and ecosystem loss. A more comprehensive set of background papers and sectoral and country studies have been undertaken since (Russi et al., 2013; McVittie and Hussain, 2013).[54] A good reference to the economic valuation of ecosystems undertaken as part of TEEB is Ten Brink (2011). The values in these studies, however, do not always provide unit values for services that are compatible with exchange values as required from incorporation of ecosystem values in national accounts. Thus extracting the exchange value component can remain an issue.

To demonstrate the potential use of these databases and relevant considerations in the application of the data the following section describes the ESVD (de Groot et al., 2020).

6.1.5 *Demonstrating the use of data from the ecosystem services literature: the Ecosystem Services Valuation Database example*

The ES valuation literature has resulted in thousands of studies over the past decades. The Ecosystem Services Partnership[55] has assessed, categorized and summarized the literature (de Groot et al., 2020). They identified 693 studies over the period 1960 to 2020 and extracted 2,917 data points that could be used to calculate the flow of services in terms of international dollars per hectare per year.[56] Studies in different currencies were converted into US dollars using purchasing power parity exchange rates and the assessment took into account inflation between the year of study and the standardized year.

Table 8 provides some examples of results that emerged from this literature review. They do not indicate what values would apply in a particular location, nor do the studies from which they are derived adhere to the concept of exchange value. Nevertheless, they are useful as a benchmark of the ranges and orders of magnitude found in studies predominantly focusing on the welfare values of ES.

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[53] Access to most of these databases can be obtained at www.es-partnership.org
[54] Further information on the TEEB publications is available at www.teebweb.org/our-publications/all-publications/
[55] See: https://www.es-partnership.org/
56 International or Geary-Khamis dollar is a unit of currency constructed to standardize money values by correcting money values across countries to the same purchasing power that the US dollar has at a point in time. This involved using purchasing power parity exchange rates, as has been done in the table cited.

Table 8: *Summary of monetary values for each service by biome(International dollars per hectare per year, 2020 price level)*

		Open Sea/ Ocean	Coral reefs	Coastal systems	Mangrove	Inland wetlands	Rivers & lakes	Tropical forests	Temperate forests	Wood lands	Grass lands	High Mountain Polar	Cultivated areas	Urban green-blue
1	Food	43	6231	9892	6717	6030	2288	602	4	8		2488	12	520
2	Water			5172	10,496	1934	9198	47,869			313	58	604	
3	Raw materials	9		44	4,454	1682	92	11,739	33	1	637	377	6	
4	Genetic resources			11		60		16						
5	Medicinal resources							3		1				
6	Ornamental resources											5		
7	Air quality regulation			15	1323	34		309	1593	7	8		10	
8	Climate regulation	69		262	1698	150	251	658	481	89	73	190	10	1722
9	Disturbance moderation		15,312	12,730	16,960	13,320	18	108	6			419	993	
10	Water flow regulation			104	2285	3638	4221	442	68	71	43		17	620
11	Waste treatment	28,910	61,013	36,556	4079	2043	50,760	12					40	
12	Erosion prevention		22,158	55	3998			604	6				173	
13	Soil fertility maintenance			4019	5576		6189	42	117			160	34	
14	Pollination							877					1498	
15	Biological control						142	14					621	

		Open Sea/ Ocean	Coral reefs	Coastal systems	Mangrove	Inland wetlands	Rivers & lakes	Tropical forests	Temperate forests	Wood lands	Grass lands	High Mountain Polar	Cultivated areas	Urban green-blue
16	Maintenance of migratory species life cycles			375	1658	1886	803	19						
17	Genetic diversity			165	6645	3427	17,987	7						
18	Aesthetic information		1200	268	334	49	2276		35	38			395	
19	Recreation/Tourism	2473	14,057	7694	4366	2660	13,633	52,789	281	124	92	167	3101	
20	Inspiration for culture		244	145	3890	114	310	5	196	214	284		16	
21	Spiritual experience					1	76							
22	Cognitive development		90	5863	1429	120	116		147	214	147			
23	Existence & bequest values	2	38,255	972	2146	11,498		2960	2416	2				
Total economic value		**30,794**	**158,560**	**84,163**	**78,052**	**48,647**	**108,361**	**119,076**	**5383**	**769**	**1597**	**3822**	**8026**	**11,759**

Source: Adapted from: Rudolf de Groot, Luke Brander, Stefanos Solomonides, 2020. Note: Coastal systems include estuaries, continent shelf areas and seagrasses, but exclude wetlands like tidal marshes, mangroves and salt-water wetlands. The numbers in the cells are the simple mid-points of strides with very wide ranges.

These results show significant benefits from the different ecosystem types, ranging from a high of USD158,560 per ha. per year for coral reefs to a low of USD 769 per ha. per year for grasslands. In terms of services, items 1-6 are provisioning services, 7-17 are regulating and maintenance services, and 18-23 are cultural services. Within the provisioning category, food is the most important service. In the regulating services, ecosystems provide an important source of waste treatment, moderation of disturbance and erosion prevention. In the cultural service group, recreation and tourism is a major category, along with existence and bequest values.

While the work summarized in Table 8 is impressive, there are a number of aspects that need further consideration. First, it is not appropriate to take the average values given in the table and use them as single figures that apply to all services provided by a given biome. These values per hectare will vary significantly across countries, regions etc. Thus, they cannot be applied to other areas or regions without some consideration of local factors, such as: population density (an ES may be more valuable when more people are living nearby), substitute and complementary services, local property rights, the degree of development of the region etc., because these factors can lead to major differences in values.

There is a concern that one cannot add the service categories to obtain a total economic value (as noted above). As Chevassus-au-Louis et al. (2009) and others have observed, the service categories are the source of the other values and including both would amount to double counting. While this is true to some extent, it is also the case that in Table 8, the regulating/service functions are combined and many of these are not captured in the provisioning or cultural categories. Climate regulation is clearly a case in point – it has benefits but the standard valuation of food, water and raw material provision does not include these benefits, so no double counting is taking place. Likewise, the benefits of water flow regulation and waste treatment are fully captured in the food and water provision estimates. Thus, while there is an element of double counting that needs to be avoided (e.g. pollination benefits are picked up in food provision through agroecosystems, represented here by grasslands), not all categories of regulating benefits constitute double counting.

There is variability owing to changes in ecosystem condition. Ecosystem condition is measured in SEEA EA, but there may not be a direct correlation between areas that are considered to be the highest scoring (or meeting the "reference" condition) in SEEA EA and what humans value in a landscape. For instance, the insertion of hedgerows into a landscape takes such a landscape further away from reference conditions, but evidence from the economic valuation literature suggests a higher WTP for landscapes with hedgerows. But this issue can be set aside, since there is obviously variability across (say) all forests vis-à-vis their ecosystem condition, which will affect their value. These issues are discussed further in section 6.1.2 under the sub-section on VT.

There is also variability owing to socioeconomic conditions and institutional conditions. For example, take two 10-hectare plots of forest that in ecological terms might be considered identical—in terms of species richness, the provision of freshwater provisioning services etc. Even though they are identical in ecological terms, their value will depend on local socioeconomic conditions. For instance, WTP estimates will likely be higher in higher-income countries, all else being equal. Further, if the 10-hectare plot is close to an urban centre and/or is easily accessible via transport infrastructure, then it is likely to provide more ES and thus be more valuable. Standard economic theory assumes that "diminishing marginal utility" applies, i.e. the more you already consume of something the lower is your WTP for one more unit. In the case of the forest, value thus depends on how many hectares of forest are available/accessible in the same region as the hectare being valued and what complementary services are available.

One can see the extent of variation around the mean in the studies summarized in Table 9[57] which shows that the mean and median values are very different, where the mean is much higher than the median in all cases. This indicates a strong positive skewness in the distribution, indicating a few locations where the values are much higher than the average. The underlying dataset has many studies with values much lower than the mean (in some cases by an order of magnitude) and a smaller number with values well above the mean. This does not indicate any valuation problem, but rather that local factors will matter greatly when transferring ecosystem values from other contexts into any accounting application.

Second, the coverage of ecosystem services in these studies is far from complete. Although Table 8 provides numbers under most categories, the items included do not pick up all the linkages between the service and the state of the biome. For example, the role of oceans in climate regulation is still being investigated, and the studies from which current values have been derived are based only on a partial understanding of the underlying physical phenomena. The same applies to the value of genetic resources and genetic diversity in different biomes and to a number of other categories of services.

Third, for some categories of services, the studies from which the average values have been derived are disproportionately from developed countries. While this is not true for provisioning services or for services from biomes such as ocean systems, coral reefs and coastal systems, it is true for recreational and other cultural services. The consequence is that for these categories of services, the transferability of the numbers to developing countries with different socioeconomic and institutional contexts may be problematic.

Table 9: *Range of values in studies of selected ecosystem services (International dollars per hectare per year, 202020 price level)*

Ecosystem Service	Mean	Median
Food	3953	226
Water	3865	360
Raw Materials	2366	27
Genetic Resources	344	56
Medicinal Resources	4	1
Ornamental Resources	5	1
Air Quality Regulation	4226	912
Climate Regulation	1196	172
Disturbance Moderation	4095	262
Regulation of Water Flow	1785	73
Waste Treatment	6552	250

Source: Adapted from De Groot et al. (2020).

[57] Note that these are figures from studies across the world and the base is evolving. Some categories are underrepresented in the table, such as recreation in urban green-blue.

6.1.6 *Conclusions on value transfer*

VT can be a cost-effective method and could allow for periodic and consistent updates of ecosystem accounts. When dealing with relatively standard cultural and provisioning services, involving recreation, nutrition and biomass, VT that takes account of geographical proximity can provide reasonably accurate national estimates of the relevant services, with a geographical disaggregation that is of considerable benefit to policymakers. In making such transfers, the more information that can be used about how values vary with site characteristics and with local populations, the better. It is important to note that the estimates derived from the meta-analytic transfers will have the value characteristics of the original studies; if the latter include consumer surplus, so will the transferred value. That will have a bearing on the way in which the estimate is used in the context of the SEEA EA. There will be some ES, however, where values are so specific to each site that VT is not possible, and some primary study needs to be conducted.

6.2 Platforms and tools to support valuation of ecosystem services

There exist a range of tools and platforms that can be used to support valuing ES (see Table 10 for a non-exhaustive list of examples). These platforms include:

ARIES (Artificial Intelligence for Environment and Sustainability)[58], is an integrated, open-source modelling platform for environmental sustainability, where researchers from across the globe can add their own data and models to web-based repositories. It uses machine-reasoning principles to generate solutions for user-specified contexts (ecosystem accounting area and time period). The ARIES Explorer[59] contains a number of models for individual ES. Within the ARIES platform, the ARIES for SEEA Explorer[60] has been developed specifically for the purpose of jumpstarting ecosystem accounts (extent; condition; supply and use tables of ES). As of 2022 the tool allows compilation of the following ES in monetary units: crop provisioning (ecosystem contribution); crop pollination (insect pollinator contribution); global climate regulation services (carbon storage).

Co$ting Nature[61] covers thirteen ecosystem services, including the (total) economic valuation of these services (i.e. welfare based). It is a web-based tool and is free of use for non-commercial users.

InVest[62] (Integrated Valuation of Ecosystem Services and Trade-offs) is a suite of open-access models that can be used to map and value a range of ES. While InVEST covers a wide range of ES, not all ES can be assessed in monetary units (however, the biophysical results can be used as input in subsequent valuations).

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58 See: https://aries.integratedmodelling.org/
59 https://integratedmodelling.org/
60 https://seea.un.org/content/aries-for-seea
61 See: http://www.policysupport.org/costingnature
62 See: https://naturalcapitalproject.stanford.edu/software/invest

Table 10: *Examples of platforms for spatially differentiated estimation of ecosystem services that include monetary outputs*

Name	Input type	Regulating	Provisioning	Cultural	Focus Area*s
ARIES	Spatial data; selection from interface	Yes	Yes	Yes	Crop provisioning; crop pollination; global climate regulation services (carbon storage); nature-based tourism (non-domestic); soil erosion control services; water supply and water flow regulation
Co$ting Nature	Spatial data	Yes	Yes	Yes	Timber (softwood, hardwood), Fuelwood (softwood, hardwood), Grazing/fodder, Non-wood forest products, Water provisioning (quantity, quality), Fish catch, Carbon, Natural hazard mitigation (flood, drought, landslide, coastal inundation), Culture-based tourism, Nature-based tourism services, Environmental and aesthetic quality services, Wildlife services (pollination, pest control), Wildlife dis-services (crop raiding, pests), Biodiversity, Pressure and threat
Coastal Resilience Decision Support Tools	Selection from interface	Yes	No	No	Coastal protection
Ecosystem Valuation Toolkit	Unknown	Yes	Yes	Yes	Varies with dataset
ENVISION	Spatial data	Yes	Yes	Yes	N/A
InVEST	Spatial data	Yes	Yes	Yes	Carbon, Coastal blue carbon, Coastal vulnerability, Crop production, Crop pollination, Fisheries, Habitat quality, Habitat risk assessment, Marine fish aquaculture, Offshore wind energy, Recreation, Reservoir hydropower production (water yield), Scenic quality, Sediment retention, Urban Cooling, Urban Flood risk mitigation, Water purification, Wave energy
Resource Watch	Selection from interface	Yes	Yes	Yes	Varies with dataset
UN Biodiversity Lab	Selection from interface	Yes	Yes	Yes	Varies with dataset

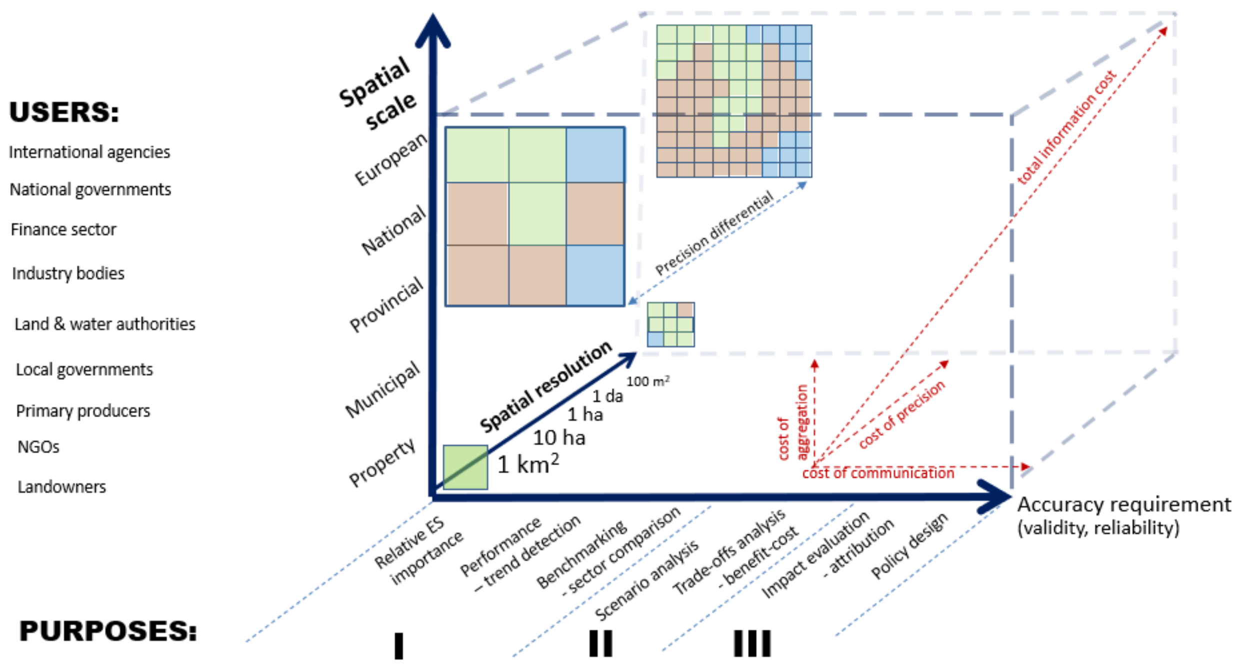
Source: Adapted from GGKP (2020). *This column lists all ES covered in the platforms, some of them are only assessed in biophysical units.

6.3 Accuracy and reliability in ecosystem services valuation

6.3.1 *Fitness for purpose*

The accuracy requirements of ecosystem accounting methods depend on the needs of the users of the information at different political, jurisdictional and land-use management levels. The information costs of ecosystem accounting (red arrows in Figure 10) are driven by: (i) the costs of communicating the validity and reliability of the information for the specific purpose of the end users, (ii) the costs of aggregation defined by the geographical scale at which users must make decisions, and (iii) the costs of precision defined by the spatial and temporal resolution needed to discriminate between decision alternatives. Figure 10 visualizes these joint considerations, as a framing of guidance on a selection of methods for the purposes of ecosystem accounting and beyond.

Figure 10: *Contexts and purposes for ecosystem accounting and valuation*



Source: adapted from Zulian, G. et al. (2017)

These costs are not independent –supporting requires identifying the accounting scale and resolution required. Lower-scale problems require higher spatial resolution of ecosystem service mapping and values.

In a situation with costly and limited information, valuation methods should be selected stepwise based on the value of the information for supporting decisions. This depends on the cost of information, the accuracy of the method and the minimum accuracy threshold of the accounting purpose, and the ecosystem values at stake (Barton, 2007). Brander et al. (2018) provide a list of possible uses of ES values, divided into primary, secondary and tertiary.

Primary uses of ES values are to answer questions such as:

- **Relative importance of ecosystem service contribution to the economy**. If the ecosystem service has never been valued economically, accounting answers the simple awareness raising question – is the value a "big number"? More specifically, ecosystem accounts answer: "how large is the annual monetary value of ecosystem service contribution compared to GDP?"

- **Trend.** Is there a trend in the physical flow of ecosystem services or in the monetary value of supply and use over accounting periods?

- **Benchmarking - sector comparison**. Are there differences in the annual contribution of ecosystem services to the economic product of different economic sectors, jurisdictions, or administrative or management areas?

Secondary uses of ES values address the following questions:

- **Scenario analysis**. How will exchange values of ecosystem services change in different alternative futures due to global change drivers such as climate change, species loss and population growth?

- **Trade-offs**. Use of ecosystem accounting data as input to financial and social cost-benefit and multi-criteria decision analysis. What is the exchange value of ecosystem service supply and use, relative to the exchange value of alternative land uses? How important are exchange values of ecosystem services relative to economic welfare values and other non-monetary values from alternative land uses?

Finally, tertiary uses of ES values inform decisions relating to:

- **Impact evaluation – attribution**. What was the impact on the exchange value of ecosystem services during a specific time period and in a specific area and population of a policy instrument or management measure? Attribution requires the use of before-after-control-impact approaches.

- **Policy design**. What regulatory standards and levels of economic incentives will attain policy objectives for the exchange value of ecosystem services?

This classification should help decision-makers seek data that is relevant for the kinds of uses they have in mind. By applying it, it is important to take into consideration physical and monetary information together. This is especially true for trends and comparisons over time. An increase in total exchange value, for example, could mean an increase in the physical quantity of the service, or, in the case of inelastic demand, it could mean a decrease in the physical quantity of the service with an increase in the unit value. Hence, it may or may not be a sustainable use of the asset, and any such determination would necessitate looking at the physical ecosystem accounts (ecosystem service and/or condition accounts).

The information in Figure 10 above indicates that ecosystem service mapping of physical flows and exchange values produced for the primary ecosystem accounting purposes will not necessarily be reliable enough or valid for the secondary and tertiary purposes. Regarding validity, the primary purpose of ecosystem accounting is to target monetary exchange values of ecosystem services, while the secondary purposes may require other measures of value (e.g. economic welfare values for social benefit-cost analysis). Regarding reliability, users would ideally go back to the highest resolution available in accounting databases to recompile information, with a purpose specific scale and resolution and an understanding of accuracy fit for purpose. Due diligence, in the sense of checking the origins of information before applying it to secondary and tertiary purposes, is therefore needed.

National accounts data are considered informative in providing a broadly reliable estimate of the values of different goods and services, of flows between suppliers and users and of changes in the flow of these services, over time. This is not done at a micro level but at a considerable level of aggregation. In this regard, the assembly of ES accounts should seek to meet similar standards, given the limitations of the physical data available. It should meet the needs for primary uses as defined above but will probably need to be supplemented for secondary and tertiary purposes.

6.3.2 *Accuracy and reliability in value transfer*

The accuracy of VT has been investigated by comparing the estimated value of an ES for a site based on applying one of the above VT methods with the value estimated by a primary valuation study conducted at the same site. It is important to note that the "true" ES value is unobserved and that the accuracy assessed refers to convergent validity. In the following, we provide some examples of transfer error ranges for different methods observed in the benefit transfer literature.

Kaul et al. (2013) provides a test of transfer errors using a relatively comprehensive meta-analysis study of more than 30 past studies, comprising in total more than 1000 estimates of transfer error (although mostly drawn from the United States and Europe). A number of findings emerge, including that the possible ranges of error are extremely large. For a typical study, the error can vary from just a few per cent to an order of magnitude of that amount (and sometimes even more). Controlling for extreme outliers, the average transfer error is about 40 per cent. In the Kaul et al. study, more sophisticated approaches (based on benefit function transfers) outperform simpler approaches (based on largely unadjusted VTs) in terms of reducing the likely error range, although pooling estimates (i.e. combining estimates from several studies to obtain an average) also helps reduce error. Geographical proximity between policy and study sites reduces transfer error. In addition, transfer errors are smaller for policies involving changes in environmental quantities than for those involving changes in environmental quality.

In another study of meta-analytic transfer error Johnston et al.(2019) reported a mean average transfer error of 68-78% for transfer of WTP for quality improvements between US waterbodies. By comparison, for unit value transfers regarding non-timber forest benefits in Nordic countries, Lindhjem and Navrud (2008) find a mean average transfer error of 86% (mean of domestic studies, similar site characteristics to the policy site) and 62% (best study estimate chosen from a domestic study). Using benefit function transfer for similar sites within Costa Rica, Barton (2002) found mean average transfer errors of 11-25% of WTP for waste water treatment between towns in the same country.

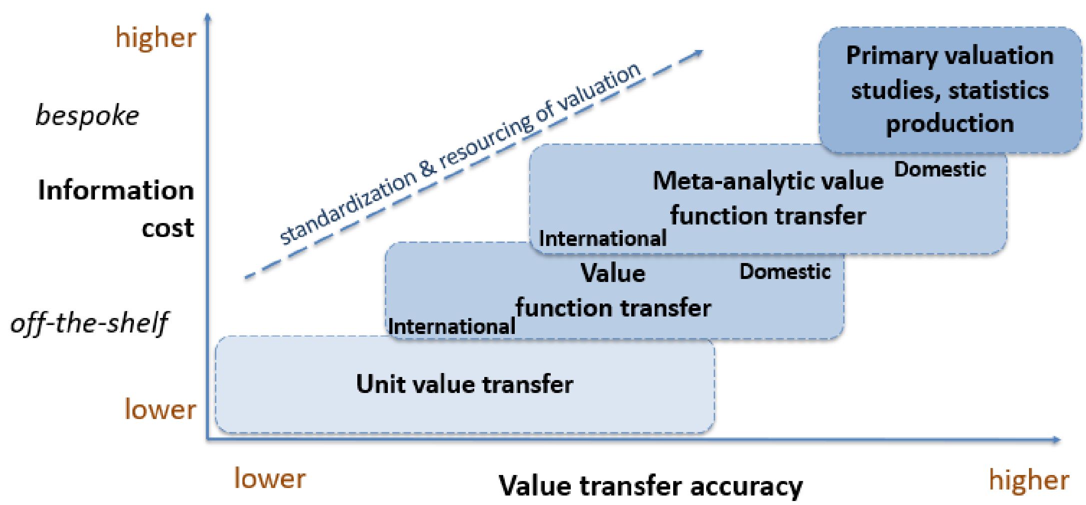
In view of these findings, an important question is when VT is a valid method. Johnston et al. 2021 recommend "proceeding with caution and justification of how the study site value estimates match the policy-site value question either directly or with simple adjustments". The benefit transfer literature highlights that economic values of ecosystem services are policy (institution) context specific.

Each valuation generalization method is associated with errors/uncertainty. The expected errors should be considered against the costs of obtaining better information with primary valuation and with the purposes of monetary ecosystem accounts (see previous section).

No single value transfer method can be considered universally superior to others across all possible circumstances (Johnston et al. 2021). The choice of method is intertwined with available study-site data and the credibility of study-site value estimates (ibid p.23). Some insight into accuracy of value transfer is available from convergent validity tests; unadjusted value transfers are the least accurate and not preferred when value function transfer is available. In turn, primary valuation using a sample from 'most similar' sites to the accounting area is preferred over function transfer (ibid).

Figure 11 illustrates this broad tiering of value transfer versus primary valuation studies in terms of their information cost and expected errors. Value transfer experiments tend to show that transfers in geographical similarity tend to have lower transfer errors (relative to a primary study at the policy site) (Johnston et al. 2021). Also, domestic studies with 'most similar' sites tend to have lower transfer errors than international transfers (noting that large countries may exhibit similar variation to international transfers). However, this ranking is not universal. Unit value transfers of a single study or a small pool of domestic studies from a local accounting area with a similar population and policy could still be more accurate than a value function transfer from other jurisdictions/countries (Lindhjem and Navrud, 2008), and a domestic meta-analytic value function (based on a larger number of domestic valuation data points) could outperform a value function transfer using a study from another country.

Figure 11: *Tiered approach to value generalization in accounting*



As long as compilers rely on using off-the shelf valuation studies for value generalization, the relative accuracy of methods is an empirical, rather than a conceptual, question. Increased resourcing and standardization of valuation methods will enable a transition from the current use of "off-the shelf" valuation, towards standardized fit-for purpose statistics produced for monetary valuation in ecosystem accounting. This will reduce value generalization errors, thereby increasing the accuracy of monetary accounts for answering policy relevant questions.

6.4 Aggregation of ecosystem service values across individual services, regions and over time

The issues relating to aggregation of ES values to input into ecosystem accounts has been investigated in some depth by eftec (2019, unpublished). This section summarizes their main findings, with some additional comments.

Aggregation of ES values is considered across space, over time and across services. It is primarily focused on economic valuation, not biophysical measurement/aggregation, which may require a slightly different focus. Implicit assumption is that a biophysical approach is consistent or otherwise sits outside of aggregation for valuation. In each case, the key factor is to ensure consistency in the way that the values are calculated before adding them up to get a total that covers a number of regions, or a number of years, or a number of services. Consistency here means the use of the same or comparable methods to estimate the values and the use of the same or comparable rules to aggregate them across the different dimensions.

Eftec proposes the classification of types of consistency for aggregation as laid out in Table 11 below. The terminology they use for the type of consistency is as follows:

- “Consistency required” means that the component must be consistent for that level of aggregation;
- “Consistency preferable” means that consistency is not strictly required, but is important to the interpretation of results;
- “Consistent approach” means that although the specific values or assumptions applied need not be the same, the general approach should be applied in a consistent manner. In practice with current accounting activities, this is the most likely achievable level;
- “Internal consistency” means that the component must be approached with consistent internal logic for aggregation, but not necessarily that the same values must be applied throughout. For aggregation over time, a consistent approach to profiling and discounting must be applied, but it is not necessary to use the same discount rate or profiling calculation across all years to be aggregated;
- “Not applicable” means that consistency is not necessary for that component.

Key considerations for each component are discussed further in the remainder of the section.

Table 11: Summary of required consistency for each aggregation component

Component for aggregation	Over time	Between regions	Between services
Physical boundaries	Consistency required	Consistency required	Consistency required
Ecosystem classification	Consistency required	Consistency preferable	Consistency preferable
Ecosystem service classification	Internal consistency	Consistency required	Internal consistency
Beneficiaries	Internal consistency	Consistency required	Consistency required
Valuation	Not applicable	Consistent approach	Consistent approach
Time horizons	Not applicable	Consistency required	Consistency required
Profiling	Internal consistency	Consistent approach	Consistent approach
Discount rate	Internal consistency	Consistency required	Consistency required

Source: Modified from eftec (2019, unpublished)

Physical boundaries should align for all items being aggregated as far as possible. For aggregation over time, this simply means that the boundary should be consistent over the time horizon to be assessed. For aggregation between regions, boundaries should be mutually exclusive, and as far as possible, exhaustive over the study area. There is some complication where boundaries follow geopolitical delineations that are not aligned with ecosystem delineations. In such cases, some ecosystem services may be provided to beneficiaries outside of the boundary of where the ecosystem is located, and this can create challenges both for capturing the total value from ecosystems within a given boundary, and for double counting of ecosystem services which cross boundaries.

Ecosystem classification and categorization needs to be consistent in order to provide an accurate assessment of value for an ecosystem type. For aggregation over time this is not a specific issue; however, for aggregation between regions and ecosystem services, consistent ecosystem type classification allows for like-for-like comparison and compilation, though is not strictly necessary if the purpose of aggregation is restricted to the overall monetary values of benefits provided.

Ecosystem service classifications need to be consistent between regions if values are being aggregated across them. The same applies over time. The ES classifications should be applied consistently across ecosystem types to avoid double counting or missing ecosystem services.

In the case of beneficiaries, the groups considered for the determination of the physical and monetary flow of supply and use should be the same across ecosystems and over time (insofar possible), so that the base of beneficiaries is equivalent. For aggregation between regions, the beneficiary groups should be determined in the same way (e.g. all farmers, all individuals over 60).

Valuation principles need to be applied such that the same aspects of the ecosystem service are being measured and valued in monetary terms in each region. If VT is used, secondary estimates need to be obtained (transferred) preferably from domestic data sources. If no domestic data are available, value estimates may be obtained from similar services in countries with similar institutional regimes/market conditions (e.g. through meta-analysis).

These monetary values need to be converted into the same price, year and currency, adjusting for the purchasing power of the country that each ecosystem services relates to. The need for consistent valuation does not imply that the same estimation method must be applied in all circumstances and for all ES. Indeed, a variety of different techniques are likely to be required to cover the range of situations and the different types of ecosystem services. For a given ES, however, it is desirable for the method of valuation to be the same across all regions that are being aggregated.

In the case of the time horizon, the issue is how consistent it should be. For assets in a given category (e.g. metals, forests) the time period over which the NPV is estimated should be the same across regions, unless there are strong grounds for varying it (e.g. a particular region has highly unsustainable use of a renewable resource). The time horizon typically will vary across asset classes but that is not a problem. It is important to note, however, that the time horizon applied may have a large impact on the asset value, especially for shorter time periods and so the need for consistency across regions for a given asset is important.

In Table 11 profiling refers to the expected pattern of all future flows of the ecosystem services that an ecosystem provides. These are not observed however, and so assumptions concerning the flows over time must be made; firstly, regarding the physical amount of the benefits provided and, secondly, the economic value of these benefits. Regarding the physical flow of ecosystem services, a simple assumption would be to assume a constant flow or linear trend (i.e. based on what was observed over the assessment period). However, the SEEA EA notes that depending on the specific asset and service, it may be more realistic to model an increasing or declining flow over time to take into account any predicted changes to the condition of an ecosystem asset or its use. Examples of such changes to the physical flows would include increased carbon sequestration over time as a woodland matures, increased recreation in an area with a projected population rise, or an increase in flooding incidents, as habitats provide coastal protection are degraded.

In addition, the monetary value of each unit of service might also be expected to change over time. For instance, in the case in the UK, the value of greenhouse gas emission reductions used in economic appraisals rises over time to reflect the increasing marginal abatement costs required to meet the UK’s emission reduction target. In such cases, assuming a constant flow of benefits may not be appropriate. Trend profiling is important to consider when aggregating values. It is an internal aspect of aggregating over time rather than a component that needs to be consistent for aggregation; therefore, as long as the profiling logic is internally appropriate, it is reasonable to aggregate over time following an NPV approach. To aggregate between regions, profiling should be based on a consistent methodology but may not be applied identically, as different trends may be prevalent in different regions. Likewise, and in order to aggregate between services, different trends may be prevalent for different services, but the overall approach to profiling should be consistent.

Discount rates have been discussed in some detail in section 5.2. Applying a high discount rate reduces the stream of values more sharply over the selected time horizon, whereas a lower rate maintains a moderate decline in value over time. This results in a lower NPV than the equivalent analysis with a lower discount rate. However, discount rates can also decline over time, as is the case in project appraisal and asset valuation in the UK.

Discounting is an internal aspect of aggregation over time, rather than a component that needs to be considered for consistency. For aggregating between regions, the discount rate applied should generally be consistent, although it is possible that different countries, especially in different stages of economic development, apply different discounts rates and this does not necessarily impede aggregation.

The approach to consistency laid out in this section has been tested. Findings indicate that in practice these requirements are often not achievable and may need to be relaxed. They could therefore be seen as an "aspirational" level of consistency for robust estimation.

6.5 Communicating monetary values for ecosystem services and assets

Monetary valuation of ES and ecosystem assets can present issues in interpretation and communication. It is therefore important to communicate the monetary ecosystem accounts in a proper manner. The following recommendations are made to support appropriate communication of results:

- Reinforce that monetary valuation in SEEA EA does not aspire to generate a full value or true value of nature, or put a price on nature. Rather its purpose is to make contributions of ecosystems to the economy/society visible and to make comparisons of different ecosystem services and ecosystem assets in a manner consistent with standard measures of products and assets as recorded in the national accounts. Valuation is undertaken in order to improve decision-making, which all too often takes nature for granted (for instance in national accounts treats nature as an abundant/non-degradable resource).

- Place emphasis on the fact that a range of both monetary and non-monetary metrics are needed to assess the importance of ecosystems, and that such assessments may not require compilation of ecosystem accounts in monetary terms. To this end, and to support interpretation of valuation outcomes, it is recommended that when monetary accounts are released, the associated data in physical terms (e.g. concerning changes in ecosystem extent and condition and flows of ecosystem services in physical terms) are also released. This will aid appropriate interpretation and application of the monetary data in policy and decision-making. Interpretation and analysis of ecosystem accounting data will also be supported through the use of other data such data that concerns environmental protection expenditure, industry value added, employment and population.

- Recognize that exchange values have particular applications that are different from applications that require the use of wider economic values or which use alternative assumptions. Thus, for example, it will be relevant to recognize that:

    - Exchange values will not reflect the full importance of ecosystems for people and the economy. There are ethical and other considerations concerning human beings' relationship to nature that lie beyond the realm of economic analysis.

    - Values in monetary ecosystem accounts are generally limited in their scope to use/instrumental values.

    - Monetary values may not reflect people's direct dependence on a natural resource or ecosystem if the price of the service is low, which may be the case if the users are very poor.

- Exchange values do not capture the potential or capacity of ecosystems to generate or sustain values under alternative management arrangements.

- Explain clearly the difference between valuation approaches that include consumer surplus and non-use values and the SEEA EA exchange values which exclude these elements of economic value, and complementary valuation approaches that for instance estimate the economic value dependent on nature, as the results can be very different.

    - For instance, in the Netherlands Horlings et al. (2020) estimate aggregate ecosystem service values for 2015 of 13,0 billion Euro/yr. (1.9% of GDP) using a broad valuation scope, or 6.3 billion Euro/yr. using a limited scope (0.9% of GDP). By comparison, the most comprehensive global estimate due to Costanza et al. (2014) - based on value transfer including welfare value estimates - suggests that ecosystem services provide benefits of USD 125-140 trillion per year - more than double global GDP. Reporting conservative ES estimates in ecosystem accounts may be a challenge with regard to an “awareness raising” purpose of ecosystem accounting.

    - e World Bank (Johnson et al 2021), using a GVA approach, estimates that partial ecosystem collapse relative to a ‘no tipping point’ scenario would result in a global change of -2.3 % of real global GDP in 2030 (0.7% in high income countries; 10% in low income countries). In another example, the World Economic Forum (2020), using a Gross Value Added at risk approach reports that half of the world’s GDP is moderately or highly dependent on nature. Industries highly dependent on nature generate 15% of global GDP ($13 trillion), while moderately dependent industries generate 37% ($31 trillion).

- In view of data limitations and the fact that different methods commonly give different values, it may be desirable to present values as a range, in which the lowest most conservative value is recorded in the accounts and supplementary information is provided to inform users of the range of the alternative estimates and may be complemented by sensitivity analysis for different methods.

Despite its accepted limitations, the use of monetary valuation can, in certain contexts, help mainstreaming nature within discussions on economic development, thereby helping decision makers more clearly understand trade-offs of economic activity and nature and as result be more able to make more informed policy decisions.

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