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Monetary Valuation in Ecosystem Accounts:

First Draft for Discussion

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Issues Paper

Monetary Valuation in Ecosystem Accounts: Key Challenges and Potential Responses

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Prepared in Support of the Experimental Ecosystem Accounting Framework Development Process

First Draft for Discussion

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Contents

1. Introduction

This Issues Paper is prepared in the context of the UNSD coordinated process to design a framework for Experimental Ecosystem Accounting. The EEA is part of the SEEA 2013 Revision, which consists of 3 Volumes. Volume 1 is the SEEA Central framework, and it has recently been adopted as global standard for environmental accounting. Volumes 2 and 3 are yet to be prepared, Volume 2 is the SEEA Experimental Ecosystem Accounts (EEA) and Volume 3 the SEEA Extensions and Applications.

The EEA will describe the measurement of the flow of benefits to humanity provided by ecosystems, and the measurement of environmental conditions in terms of the capacity of ecosystems to provide benefits. The SEEA Experimental Ecosystem Accounts will not be a statistical standard but should provide a consistent and coherent summary of the state of the art of using a systems approach to the measurement of ecosystems in the context of SEEA. It is meant to provide the basis for countries to further advance the implementation of ecosystem accounts using terms and concepts which facilitate the comparison of statistics and the exchange of experiences.

This Issues Paper on monetary valuation in ecosystem accounts has been prepared in support of the development of the EEA. The objective of the paper is to provide a comprehensive overview of the key challenges related to expressing ecosystem services in monetary terms , in an accounting framework, and to identify potential ways to address these challenges. The paper is not meant to provide any definite guidance but instead meant to serve as a basis for discussion.

The paper builds upon Issues Papers prepared by M. Eigenraam et al., M. Pittini and D. Simpson in the context of the Expert Group on Ecosystem Accounting and that were presented during the December 2011 London Meeting of this Group. In addition, some parts of the concept note of Carl (Obst, 2011) have been inserted where relevant to the material presented in this paper.

Chapter 2 presents a list of 4 key themes in which the various challenges can be grouped, at least for the purpose of this report in support of further discussion. Chapters 3 to 6 present some preliminary thoughts on the various ways in which these challenges can be addressed in the EEA.

The focus of the paper is on the ecological and (micro-)economic considerations relevant for the development of the EEA rather than on the accounting principles per sé.

2. List of Key Challenges

A list of key challengesto be addressed in the Monetary Valuation section of the EEA was prepared on the basis of (i) a review of the papers prepared for the London Expert Group on Ecosystem Accounting Meeting in December 2011 (i.e. Eigenraam et al., 2011; Pittini, 2011; and Simpson, 2011); (ii) a review of relevant literature including peer reviewed literature and documents available in the context of the 2013 SEEA Revision process; and (iii) discussions with UNSD and the EEA Editor. The challenges are listed below, groupedfour key themes.

1. **How to include ecosystem services in the EEA**. There is still no consensus among experts in the field whether ecosystems should be seen as assets or units. Nevertheless, the EEA needs to be consistent and, presumably, prescriptive in this respect. A question therefore is how to define the object of valuation in relation to ecosystem accounting (page 9, Pittini). Where is the 'boundary of the ecosystem' ? For instance, on agricultural land, is the ecosystem (i) the soil providing the farmer with a substrate for farming, or (ii) is the ecosystem the farmland where labour (from the farmer), machinery and soils are combined to produce outputs such as crops (and perhaps other ecosystem services such as carbon sequestration). From this distinction follows the approach that can be taken to distinguish services from benefits. Related to this:

- Is it appropriate to attribute the value of an ecosystem to the services it provides rather than the physical characteristics of the ecosystem itself ?
- There needs to be clarity regarding the issue of 'final' versus 'intermediate' ecosystem services.
- Do the intermediate services (e.g. pollination) need to be included in the EEA ? If so how can this be achieved ?
- And finally: how degradation can be recorded in the EEA, and can it be captured in meaningful and measurable indicators ?

2. **Biophysical quantification and aggregation of services**. Monetary valuation of ecosystem services follows the biophysical quantification of ecosystem services.Relevant questions for the EEA are therefore:

- What are the uncertainties and what is feasible in terms of analysing ecosystem services in biophysical terms ?
- At what scale should the service be measured prior to aggregation to the national level required for the EEA ?
- How should aggregation take place in an manner that is consistent and avoids double-counting, given that ecosystems seldom have distinct boundaries ?
- Should only non-market or all ecosystem services be included ?

Related, the EEA may attempt to clarify which ecosystem services appear most feasible for inclusion in SEEA (– noting that this may differ between countries). The aspect of biophysical analysis including the design of (composite)indicators to indicate ecosystem stock or capital is critical to EEA, but considered outside of the scope of this Paper and not further discussed in this paper.

3. **Monetary valuation methods for ecosystem services**. In principle, the EEA should present a clear valuation approach plus recommendations for specific valuation methods as they may be applied to specific ecosystem services. Relevant questions are therefore:

- To what degree existing methods for ecosystem services valuation are aligned with the requirements for SEEA ?
- And how can they be applied at a national scale, given the uncertainties, scales and aggregation issuesinvolved ?
- To what degree can benefit transfer and interpolation of value estimates be relied upon to provide insights in the values of ecosystem services at aggregated scales ?

4. Sustainability. There are different interpretations of sustainability, and the EEA needs to make clear how ecosystem accounting relates to these different interpretations. This is of course related to the degradationis included in the EEA.A key issue in relation to sustainability is always the reference situation with which changes in ecosystems are compared – and which is often very difficult to define because human influence has modified ecosystems in most parts of the world since many centuries, and because of naturally occurring variations in ecosystems. In addition, there is a need to examine if there are particular aspects related to ecosystems that are not captured in the EEA and that are still relevant for sustainability (perhaps ecosystem services that are currently not scarce and therefore have no economic value, or the resilience of ecosystems).

2. Defining ecosystems and ecosystem services for the purpose of the EEA

2.1 Basic concepts

Ecosystems. The UN Convention on Biological Diversity has provided the following definition of an ecosystem: 'A dynamic complex of plant, animal and micro-organism communities and non-living environment interacting as a functional unit'. For ecosystem accounting, it is required that the elements to be included in the accounting system are clearly defined. In the case of ecosystems, based on their components and structure, ecosystems can generally be linked to a specific spatially defined area. There are different interpretations regarding the degree of naturalness associated with ecosystems. Traditionally ecosystems were often associated with more or the less 'natural' systems, i.e. systems with only a limited degree of human interference. However with the publication of the Millennium Ecosystem Assessment (MA), a wider interpretation has become common and agricultural land is now also often interpreted as being an ecosystem that provides services (in addition to crop production, for instance: carbon sequestration, nature conservation, or supporting tourism and recreation). However, it needs to be examined if and how the MA interpretation is the most suitable for the EEA.

Ecosystem services. Three different categories of ecosystem services can be distinguished: (i) provisioning services; (ii) regulating services; and (iii) cultural services. The Millennium Ecosystem Assessment, but not later assessments such as TEEB and CICESalso included the category of (iv) supporting services. Supporting services represent the ecological processes that underlie the functioning of the ecosystem. However, there are a very large number of ecological processes that support the functioning of the ecosystem, and the their inclusion in valuation may lead to double counting as their value is reflected in the other three types of services. Therefore this service is of less relevance in the development of Ecosystem Accounts. The other three types of services are briefly described below.

(i) Provisioning services reflect goods and services produced by or in the ecosystem, for example a piece of fruit or a plant with pharmaceutical properties. The goods and services may be provided by natural, semi-natural and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.

(ii) Regulating services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect. For instance, the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest. The nursery service can also be classified as a regulation service. It reflects that some ecosystems provide a particularly suitable location for reproduction and involves a regulating impact of an ecosystem on the populations of other ecosystems.

(iii) Cultural services relate to the non-material benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies), or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved. The latter may occur without having the intention of ever visiting the area. The category cultural services also includes the biodiversity conservation, or habitat service, that represents the benefits that people obtain from the existence of biodiversity and nature (not because biodiversity provides a number of services, but because people believe it's conservation is important in itself).

There are different list of ecosystem services within these categories. A total of 59 different types of ecosystem services were distinguished in CICES, which compares to 22 ecosystem services in TEEB, 18 services in MA (2003) – excluding supporting services, and 17 in Costanza et al. (1997). The substantially larger number in CICES is because some services have been split, e.g. commercial and subsistence cropping are now distinguished, and several new services have been added, such as seed dispersal and solar energy provision.

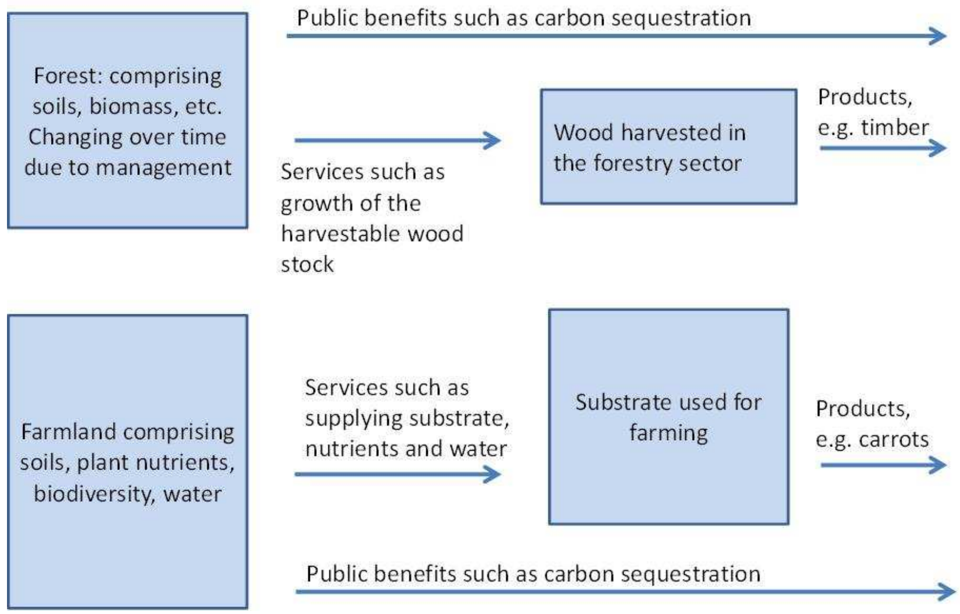
2.2 Options for incorporating ecosystems and ecosystem services in the EEA

Fundamental to the concept of ecosystems, and to analysing the functioning of ecosystems, is that ecosystems depend upon the connectedness of it's different components. Ecosystems change as a function of (i) natural processes, that may take place without any human intervention; (ii) external pressures (often due to human actions, e.g. climate change) and (iii) human interventions (e.g. cutting a tree). Carl conceptualised this as follows: 'ecosystems are analogous to a produced asset (e.g. a computer, or a building) that itself has many components (e.g. the foundation, the roof, the walls, the windows) with each component having a different purpose/role but noting that without each component the asset would not operate effectively. Some components may seem more critical than others, but the ecosystem does operate as a whole'.

Although the 'asset like nature' of ecosystems is broadly recognised, and aligned with the dominant thinking in environmental economics (as expressed in e.g. Barbier2012), there are nevertheless different options for subsequently incorporating ecosystems in EEA. These three options are briefly summarised below. Each model has advantages and disadvantages, and the main purpose of the text below is to point out these advantages and disadvantages as a basis for further discussion.

Model 1 (partly based on comments Mark De Haan).An ecosystem is a composite asset (reflecting that it generates several services at the same time) but functions asa separate entity or sector. This entity produces services that people use as input into the production process.The ecosystem is 'owned' by the sector 'Nature'. Fundamental in this reasoning is that ecosystem services accrue to a diverse set of beneficiaries, and many of these benefits do not show up in the decision criteria of the person managing or benefiting from the ecosystem. In other words, a lack of ownership rights is a fundamental element of ecosystems that needs to be reflected in the EEA. Assigning ecosystems and their output to private or public owners is not in line with the *conflicting* interests of different stakeholders that need to be considered.

The consequence of this approach is that ecosystems are essentially seen as a system that functions at a different level than people benefitting from the ecosystem. The ecosystem provides a range of outputs that are subsequently used by people, see Figure 1.



The sector 'Nature' provides services to other sectors (e.g. forestry sector, agriculture)

Figure 1: Relation between ecosystems, ecosystem services and benefits in Model 1.

From an ecological perspective, there are two concerns with Model 1 that need further consideration. First, it assumes that nature operates rather independently from people which is not in line with observations across ecosystems on the planet. Second, it is difficult to record degradation, since degradation is recorded as part of the sector 'nature/ecosystems' rather than recorded as a decrease in the production capacity of the farmer or forest owner.

It is assumed in Model 1 that the value of ecosystem services accruing to the private land owner, e.g. the farmer or the forester can be revealed on the basis of the purchase price of the land (assuming that the farmer only pays for the capacity of the land to produce crops rather than the land's capacity to support other ecosystem services such as carbon sequestration - i.e. reflecting the resource rent of the land). This reflects that analysing the relation between ecosystem characteristics (soil properties, etc.) and the productive capacity is complex and data intensive. For example, the cultivation of crops and the yields obtained depends upon the combined input of substrate, nutrients, water, labour from the farmer, equipment (tractor, irrigation equipment, type of irrigation equipment). There is a degree of substitutability - e.g. in case of water scarcity more efficient irrigation equipment can be applied; and with a lack of phosphorous (P) and ample nitrogen (N), plants can adapt and be less efficient with N and more efficient with P. Hence, in practise, the individual contributions of these 'ecosystem services' are very hard to separate. It is typically the most constraining factor (e.g. nutrients, or water) that has the largest impact on the final yields. Moreover, because there is a degree of substitutability between ecosystem services as measured in this way and labour and equipment, it is also not always straightforward to analyse the benefits provided by ecosystem services separately from the benefits resulting from the use of labour and equipment. It needs to be further discussed how these complex realities facing the farmer and the forest land owner can be reflected in Model 1.

Model 2. Ecosystems are assets not units, and produce services that are combined with other capital goods to produce benefits (Model Carl).This model sees ecosystems as assets, that change over time as a function of human action and natural processes. This model is expressed in Carl's Figure, see Figure 2 below.

Model 2 is partly consistent with TEEB in that TEEB also distinguishes ecosystem services from benefits. In the thinking of TEEB, for instance, an example of an ecosystem function is the deposition of particulate matter in forests, an ecosystem service is the resulting improvement in air quality, and the benefit is the subsequent improvement in health that people enjoy (and that may be valued in monetary terms through CVM and benefit assessments of reduced hospital treatments). Or, to give another example, the ecosystem function is the growth of trees in a forest, the ecosystem service is the wood harvested from the forest, and the benefits are the benefits enjoyed from the table made from that wood or from the heat generated in the fireplace. Note that there is still discussion among ecological economists if the distinction between services and benefits proposed in TEEB is an improvement vis-à-vis the Millennium Ecosystem Assessment which did not make this distinction. Note also that, in the thinking of TEEB, there is a still the need for the deployment of labour and equipment (e.g. a saw) to generate the ecosystem service and that this aspect is therefore not aligned with Model 2.

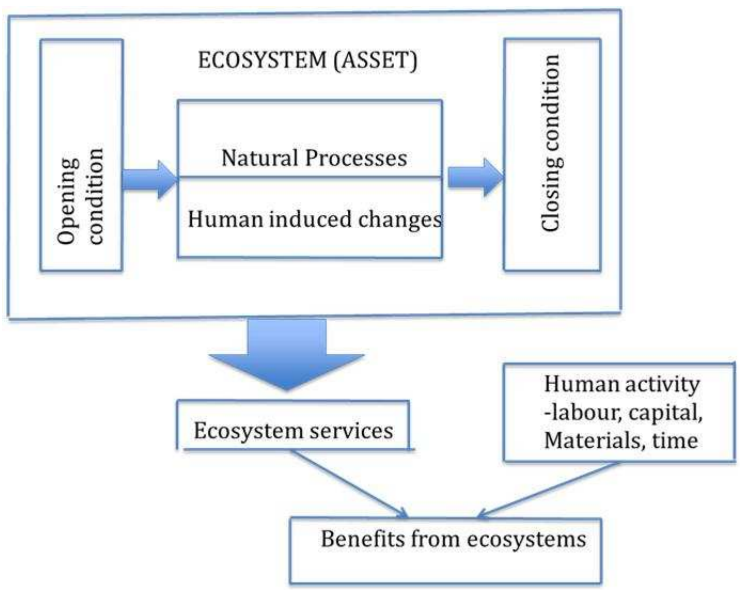


Figure 2. Model 2 (source: Concept Note Carl).

A crucial difference with Model 1 is that ecosystems are not an entity in it’s own right, but that ecosystems are owned by specific sectors, be it the farmer, the forest owner or, for public land, the government. The land owner derives private benefits from the system and is likely to manage the landaccordingly. Public benefits, such as carbon sequestration, are externalities. Note that both positive (C sequestration) and negative (e.g. ammonia emission from cropland) externalities may occur.

There are two potential concerns with Model 2. First, as in Model 1, some of the costs of degradation need to be assigned to the land owner (reflecting the loss in capacity to produce private benefits), whereas other costs of ecosystem degradation need to be attributed to society as a whole, potentially the government.Second, human activity is recorded twice in Figure 2, first as a driver of ecosystem change, second as part of the process to turn ecosystem services into benefits. In reality these two types of human action may be difficult to separate.

Model 3. Model Millennium ecosystem Assessment, adopted in the paper byEdens&Hein. In this model, ecosystems are assets, and the supply of ecosystem services is a function of the joint application of natural processes and human interventions. Crucially, in this model, ecosystems are not seen as separated from human activity, in recognition of the situation in most parts of the planet that ecosystems are heavily modified by people, in terms of components and processes. As in model 2, the ecosystems are owned by the land owner holding title to the physical location of the ecosystem, but it needs to be recognised that some of the benefits accrue to the owner whereas other benefits accrue to other stakeholders or perhaps society as a whole. In an accounting system, in this model as well as in Model 2, the supply of ecosystem services is attributed to the standard institutional sectors and activities. Agricultural land, in this model, is also seen as an ecosystem. Model 3 is presented in Figure 3, for both a forest ecosystem and for farmland.

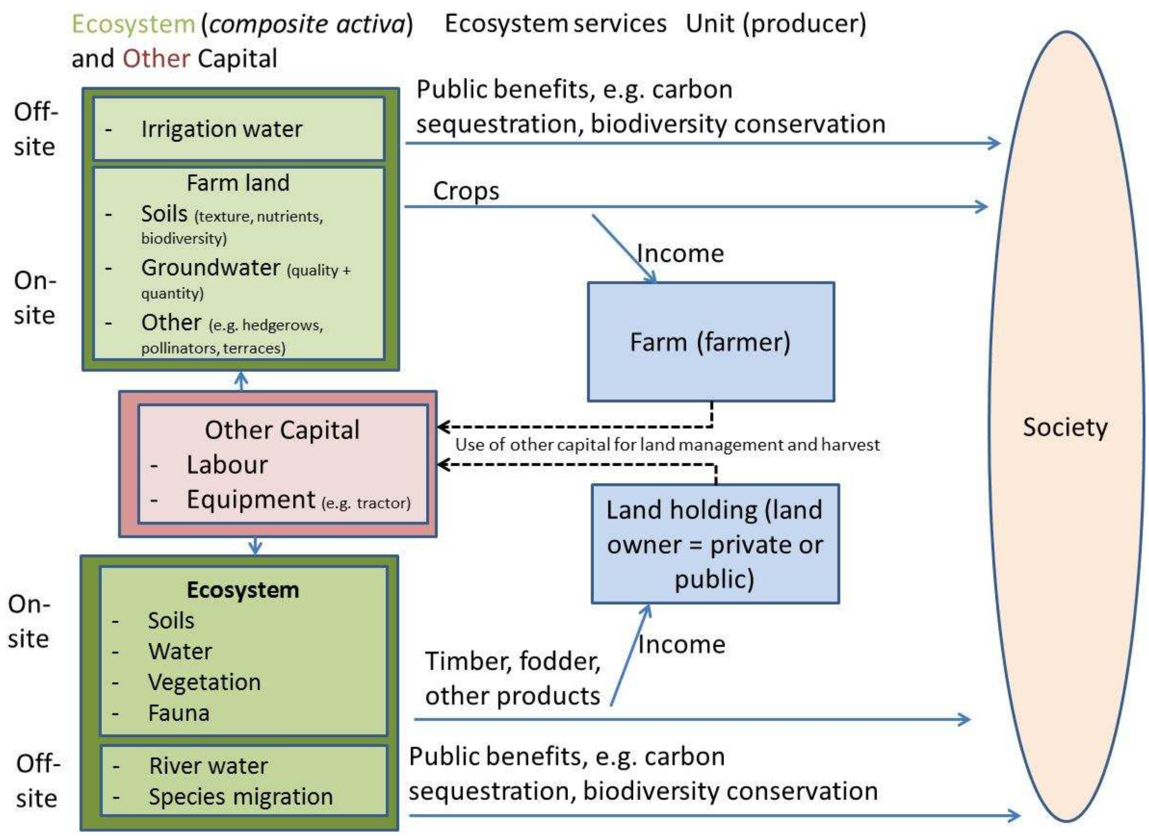


Figure 3. Model 3, Ecosystems are seen as assets, ecosystem services result from the combined use of ecosystem capital and other types of capital.

In Model 3, ecosystem services are defined as outputs of economic activities undertaken by institutional units (corporations; households; government) in which ecosystem assets (E) are used, often in combination with other assets (K, L). Model 3 differs in this respect from Model 2 described above as well as the proposals of Boyd and Banzhaf (2007) and Boyd and Krupnick (2009). Boyd and Banzhaf (2007) for instance state that "ecosystem services should be isolated from non-ecological contributions to final goods and services. For example, recreational benefits and commercial harvests are not ecosystem services because they arise from the combination of ecosystem services with other inputs". By contrast, according to Model 3, ecosystem services are in almost all situations the result of human interaction with nature, and it would be artificial to try to isolate ecosystem services produced with and without human involvement. The overwhelming majority of ecosystems on this planet is managed in some way or another by people, even though the supply of some ecosystem services may be a by-product (e.g. carbon sequestration) rather than an intended produce from the ecosystem (e.g. timber).

Two practical advantages of Model 3 are that (i) the model is better aligned with ecological models on ecosystem use; and (ii) the model requires much less data and less assumptions regarding the specific ecological processes or components and how they influence the outputs from ecosystems. For instance, there is no need to analyse the different ecosystem components of soils, nutrients, water availability in ecosystems because they are no longer ecosystem services but instead ecosystem components or attributes. Selected key (but not all) components may be measured to quantify the 'stock' of ecosystem capital, reflecting it's capacity to supply ecosystem services.

Disadvantages of the model are that it requires a different perspective to ecosystem capital than the perspective applied in the SEEA Central Framework. In addition, farm land is also seen as an ecosystem, which leads to a need to redefine produced assets versus ecosystem assets (in Model 3, an orchard may also qualify as ecosystem asset rather than as produced asset). Third, contrary to Model 1but in line with Model 2, in Model 3 the services generated by an ecosystem are transferred to different stakeholders including the owner / user of the ecosystem and society at large.

Recording ecosystem degradation, in the three models. Recording degradation is a challenge in all three models, in particular with regards to identifying indicators for ecosystem degradation that can be measured at a national scale and are meaningful. In principle, ecosystem degradation reflects a reduced capacity of the ecosystem to supply ecosystem services, either at present, at some point in the future, or in case of a disturbance of the ecosystem (i.e. degradation may also reflect a loss of resilience of the ecosystem not immediately apparent from outputs of the ecosystem, consider for instance the case of Sahelian grasslands where productivity is dramatically reduced on degraded grasslands in years of drought, but hardly reduced in years with good rainfall).

If the ecosystem is considered as an asset, it follows that in accounting terms the recording of the assessment of the condition of ecosystems and changes in that condition is presented in asset accounts. Determining the time periods over which such assessments should be completed is challenging since natural processes will vary in their rate of occurrence/impact and little may change from one year to the next.

On the measurement and recording of degradation, there is for many ecosystem types an understanding of key indicators that may typically reflect the health of the ecosystem because they are linked to many other ecosystem properties as well as to ecosystem productivity (e.g. Soil Organic Matter in soils of African maize/bean farming systems, or the Rain Use Efficiency of semi-arid rangelands). Perhaps the most sophisticated example of an attempt to link individual ecosystem properties (*in casu*: soil properties) to the productive capacity of ecosystems is described in Rutgers et al. (2012). In this paper, building on the Dutch Soil Monitoring network operational since the mid 1990s, four farms were analysed in detail regarding soil properties and ecosystem services supply. The examined ecosystem services were 'nutrient retention and release', 'soil structure', 'natural disease surpressiveness', 'resistance and resilience', 'adaption, land use change options', 'fragmentation and mineralization SOM' , 'natural attenuation, clean groundwater' , 'water retention', 'climate functions', 'biodiversity and habitat function, i.e. soil biodiversity'. These services were linked to 50 different soil properties, including soil organic matter content, abundance of earthworms, biomass bacteria, pH, diversity earthworms (# of taxa), diversity of nematodes (# of taxa), metal concentrations, etc. This linking was done based on expert judgement. The most relevant soil properties were measured, and an index value reflecting the potential capacity of the soil to provide soil ecosystem services as defined above was produced, for these 4 farms.

This innovative research points to a number of issues for EEA. First, the relation between soil properties and the soil ecosystem services defined in this study (which correspond to the ecosystem services interpretation of Model 1 and Model 2) could from an ecological perspective only be quantified by using an Index value and Best Professional Judgement. The Index Values provide a relative score allowing comparison of 4 farms in relatively similar soil types, but convey no meaningin an absolute sense. Underneath the Index is an expert judgement of the relative importance of the 50 soil properties. Two, the relation between ecosystem services supply – as defined in this paper - and farm production has not been examined. A potential way of doing so is to regress the Index Value 'ecosystem service performance index' against farm production while accounting for different inputs in labour and equipment (perhaps in an econometric model). However, critically, the Index value as defined in this paper is highly site dependent. For other soil types and production environments the relative importance of the various properties will be different. Hence, although perhaps feasible using methods still to be developed, this method would be highly data intensive. Third, further research is required to link soil ecosystem services as defined in this paper not only to agricultural production but also to the supply of other ecosystem services such as carbon sequestration in agricultural soils. In Model 3, the approach to measure degradation would be somewhat simpler, because farm productivity and other ecosystem services generated in the farm could be directly regressed against soil properties without consideration of the intermediate step of soil 'ecosystem services'. Perhaps such an approach would entail an index based on an aggregation of measurable soil properties rather than an index reflecting (difficult/impossible to measure) soil ecosystem services. This index should perhaps not be a linear aggregation of different soil properties but reflect that if one key property (e.g. pH) is highly unfavourable, the total productivity fo the land may be impaired.

For forest land, the discussion on degradation should be broadened not only reflecting soil properties (and for some ecosystems perhaps not reflecting soil properties at all), but rather those ecosystem components (e.g. biomass, NPP, species composition) that can most directly be linked to ecosystem services provision, in all Models.

3.Principles of Monetary Valuation

3.1 Basic Concepts

In neo-classical welfare economics, value is related to the price of the good or service in an open and competitive market, as a function of demand and supply. Accordingly, for traded ecosystem services, under perfect market conditions, market price reflects the marginal economic value of the service. The total economic value related to the supply of an ecosystem service (or any other good) is the sum of the consumer and the producer surplus(Freeman, 1993). The individual consumer surplus equals the willingness-to-pay of a consumer for a good minus the price the consumer faces for that good. The aggregate consumer surplus reflects the surpluses obtained by different consumers at a given market price. Consumer surplus is not included in SEEA and therefore there is a need to disentangle the consumer surplus from valuation estimates resulting from the application of different ecosystem valuation approaches.

The producer surplus indicates the amount of net benefits a producer gains, given his production costs and the (market) price he receives for his products. In the valuation of ecosystem services, the producer surplus needs to be considered if there are costs related to "producing" the ecosystem good or service, such as for example the costs related to collecting or harvesting forest products (Hueting et al., 1998). In case an ecosystem services approach is used to analyse activities such as agriculture or fisheries, clearly, the full production costs of the fisherman (boat, equipment, labour, etc.) or farmer (land, machinery, inputs, labour, etc.) need to be accounted for, consistent with SEEA.

The concepts of consumer and producer surplus in the context of ecosystem services can be illustrated with the example of the pollination service. Insect pollination is required for a range of crops including apples, oranges, almonds, etc. Insect pollination can be achieved by bringing in bee hives, or can be performed by naturally occurring bees or, for some crops, other animals. In the latter case, pollination is an ecosystem service, in particular, a regulating service required for agricultural production. In the valuation of pollination, it is necessary to consider the scale at which pollination is studied. For instance, in case the value of pollination in one particular farm is studied, there will probably be no price effects since the production of this farmer is likely to be small compared to the overall market supply. In this case, changes in the producer surplus can be estimated on the basis of multiplying physical changes in ecosystem services supply with net revenues generated per unit of ecosystem service. For example, Rickets et al. (2004) relate the value of the pollination service supplied by forest patches on a Costa Rican coffee farm (which serve as habitat for pollinating bees) to the impact of pollination on the coffee yields, the total area of coffee plants pollinated, and the net benefits obtained from the sale of coffee (off-farm price minus variable production costs). However, in case pollination declines at the national scale, price effects for pollinated crops become increasingly likely, because the supply of the affected crops is reduced while demand, presumably, is not affected. Valuation of pollination services at the national scale, therefore, needs to consider that prices may not be constant. In this case, demand and supply curves have to be constructed to analyse changes in the producers and consumers surplus as a function of changes in the supply of the pollination service.

There are several types of economic value, and different authors have provided different classifications for these value types (e.g. Pearce and Turner, 1990; Hanley and Spash, 1993; Munasinghe and Schwab, 1993; and Millennium Ecosystem Assessment, 2003). In general, the following four types of value can be distinguished: (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value.

(i) Direct use value arises from the direct utilisation of ecosystems, for example through the sale or consumption of a piece of fruit. All provisioning services, and some cultural services (such as recreation) have direct use value.

(ii) Indirect use value stems from the indirect utilization of ecosystems, in particular through the positive externalities that ecosystems provide. This reflects the type of benefits that regulating services provide to society.

(iii) Option value relates to risk. Because people are unsure about their future demand for a service, they are willing to pay to keep the option of using a resource in the future – insofar as they are, to some extent, risk averse. Option values may be attributed to all services supplied by an ecosystem. Various authors also distinguish quasi-option value, which represents the value of avoiding irreversible decisions until new information reveals whether certain ecosystems have values we are not currently aware of. Although theoretically well established, the quasi-option value is in practice very difficult to assess.

(iv) None-use value is derived from attributes inherent to the ecosystem itself. Hargrove (1989) has pointed out that non-use values can be anthropocentric, as in the case of natural beauty, as well as ecocentric, based upon the notion that animal and plant species have a certain 'right to exist'. Kolstad (2000) distinguishes three types of non-use value: existence value (based on utility derived from knowing that something exists), altruistic value (based on utility derived from knowing that somebody else benefits) and bequest value (based on utility gained from future improvements in the well-being of one's descendants). The different categories of non-use value are often difficult to separate, both conceptually and empirically. Nevertheless, it is important to recognize that there are different motives to attach non-use value to an ecosystem service, and that these motives depend upon the moral, aesthetic and other cultural perspectives of the stakeholders involved.

In principle, the four value types: direct use, indirect use, option and non-use value are exclusive and may be added. The sum of the direct use, indirect use and option values equals the total use value of the system; the sum of the use value and the non-use value has been labelled the 'total economic value' of the ecosystem. If all values have been expressed as a monetary value, and if the values are expressed through commensurable indicators (e.g. consumer and/or producer surplus), the values can be summed.

3.2Ecosystem valuation methods

A first step in developing a valuation approach applicable to EEA is to define the specific object of valuation. Supporting services, as identified in the Millennium Assessment can be defined as intermediate services and should be accounted forthrough impacts on other services and therefore not valued separately. Otherecosystem services will be intermediate or final services depending on their relationship withfinal goods and services that are valued by consumers. For example as described in the UK National Ecosystem Assessment, for anglingwater quality (an aspect of natural capital) is an intermediate service in the provision of fish,but so will be other capital inputs such as human capital (the skills of the fisherman and thetime invested) and man-made capital (the fishing gear). By contrast for drinking water, clean water of sufficient quality to be used as raw material for drinking water production is a final ecosystem service.

Figure 4 presents a basic framework for analysing the economic value of ecosystem services. The framework involves four subsequent steps: (i) definition of the spatial and temporal boundaries and object of the (eco)system and identification of the services to be studied; (ii) quantification of ecosystem services in biophysical terms; (iii) valuation of ecosystem services; and (iv) aggregation or comparison of values of different services.

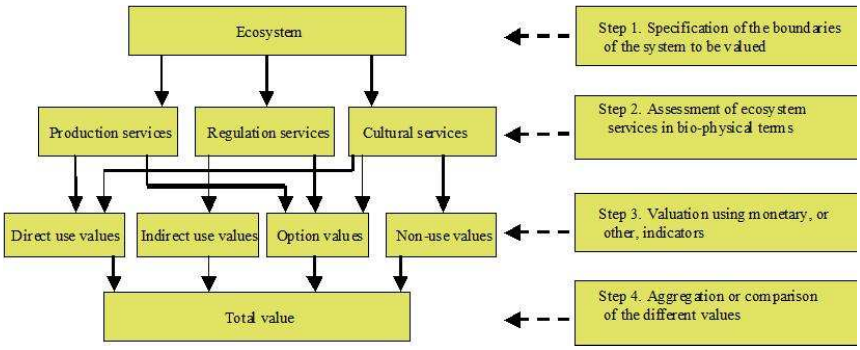


Figure 4. General flowchart for valuing ecosystem services

A range of economic valuation methods for non-market ecosystem services have been developed. For public goods or services, the marginal willingness to pay can not be estimated from the direct observation of transactions, and the demand curves are usually difficult to construct. Two types of approaches have been developed to obtain information about the value of public ecosystem services: the revealed and stated preference approach (Pearce and Howarth, 2000).

The revealed preference approaches use a link with a marketed good or service to indicate the willingness-to-pay for the service. There are two main types of revealed preference approaches:

- Physical linkages. Estimates of the values of ecosystem services are obtained by determining a physical relationship between the service and something that can be measured in the market. The main approach in this category is the damage-function (or dose-response) approach, in which the damages resulting from the reduced availability of an ecosystem service are used as an indication of the value of the service. This method can be applied to value, for instance, the hydrological service of an ecosystem.

- Behavioural linkages. In this case, the value of an ecosystem service is derived from linking the service to human behaviour – in particular people's expenditures to offset the lack of a service, or to obtain a service. An example of a behavioural method is the Averting Behaviour Method (ABM). There are various kinds of averting behaviour: (i) defensive expenditure (a water filter); (ii) the purchase of environmental surrogates (bottled water); and (iii) relocation. The travel cost method is another example of an indirect approach using behavioural linkages.

With stated preference approaches, various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important approaches are the Contingent Valuation Method (CVM) and related methods. In the last decades, CVM studies have been widely applied (see e.g. Nunes and van den Bergh, 2001 for an overview). It is the only valuation method that can be used to quantify the non-use values of an ecosystem in monetary terms. Information collected with well-designed CVMs has been found suitable for use in legal cases in the U.S. - as in the case of the determination of the amount of compensation to be paid after the Exxon Valdez oil spills (Arrow et al., 1993). Nevertheless, various authors question their validity and reliability - both on theoretical and empirical grounds. There are two main points of criticism against CVM. First, CV estimates are sensitive to the order in which goods are valued; the sum of the values obtained for the individual components of an ecosystem is often much higher than the stated willingness-to-pay for the ecosystem as a whole. Second, CV often appears to overestimates economic values because respondents do not actually have to pay the amount they express to be willing to pay for a service (see e.g. Hanemann, 1995).

Scales. Important in the development of SEEA is also consideration of the spatial scale at which ecosystem services are analysed. For instance, pollination can be a regulating service in the sense that it provides an input in the production of fruit in an orchard. If the object of the valuation is the orchard, adding the value of pollination and the value of the fruit will lead to double counting. In this case, pollination can be seen as one of the inputs required for producing the fruit (and could be attributed part of the value of fruit production using the production factor method). If however the object of valuation is the forest adjacent to the orchard that provides habitat to populations of wild pollinators, this pollination service can be seen as one of the specific services provided by the forest. If the forest would be converted to a greenhouse, this service and the value it generates would be lost, having an adverse effect on nearby fruit production. Such a distinction is meaningful for a local scale analysis. At the national scale, the physical relation between insect pollinators and fruit production remains the same; pollinators residing in natural landscape elements close to farms contribute to pollination. However the specific impact of this effect is difficult to analyse at this national scale, given that it is the aggregate contribution of a large number of forest patches. Moreover, at the national scale, consideration of pollination and fruit production would lead to double counting. The role of pollination, when analysed at the national scale, may instead be seen as akin to that of ancillary services included in SNA.

Valuation at the national scale. Given the relative scarcity of valuation data on non-market ecosystem services, value transfer may be needed in the application of EEA. In specific situations therefore valuetransfer (also known as 'benefits transfer') can be a cost effective additional tool in support of EEA. Value transfer techniques consist of applying estimates of the value ofecosystem services to a different geographical and policy context from the specific context inwhich they were developed, but a context that is nevertheless sufficiently similar for thetransferring of (suitably adjusted) values to be meaningful (see e.g. Pittini, 2012). In many cases however, the influence of environmental factors on the value of the ecosystem service is substantial, and care needs to be taken in the application of value transfer. For wetland services, meta analyses show insights in how value estimates vary as a function of the difference in local biophysical and economic conditions (Brander et al.).An interesting, novel approach in this regard is the Simulated Exchange value approach that is being applied to estimate the benefits from forestry in Andalucia, Spain (Campos et al., 2007). This approach aims to measure income that would occur in a hypothetical market for ecosystem services based on the construction of supply and demand curves and assuming hypothetical price levels reflecting the price that would be charged by a profit maximising 'seller' of the ecosystem service.

Further lessons can be drawn from currently ongoing or completed national level ecosystem assessments, in particular the UK National Ecosystem Assessment (NEA) (UK NEA, 2011) which is comprehensive and sophisticated of the supply and value generated by selected ecosystem services at the national scale.The NEA used market prices (e.g., for provisioning services), surrogate markets (e.g. for valuing amenity) and stated preference techniques in case other approaches would not be viable (e.g., for non-use values of biodiversity), see also the paper by Pittini (2012). In addition, new pricing mechanisms for ecosystem services (Eigenraam, 2012) are relevant in revealing value information on non-market ecosystem services.

3.3 Next steps

In general, there is a broad literature on valuation methods that can be used as a basis for the respective parts of the text in the EEA. In addition, the paper by Pittini (2012) provides an excellent starting point, with important information on uncertainties described by Simpson (2012) and information on novel pricing methods by Eigenraam (2012). As part of the EEA preparation process, there is a need to further consolidate available material and make sure that the text on valuation methods is aligned with the model adopted to analyse ecosystem services (Chapter 2 of this report). Specific issues that require further consideration are to analyse how consumer surplus can be excluded from value estimates, to examine how thresholds and resilience of ecosystems can or cannot be reflected in EEA, and how localised value estimates of ecosystem services can be translated to national scale value indications suitable for the EEA. Perhaps a section can be added to the EEA dealing with valuation methods for specific ecosystem services.

4. Sustainability

The World Commission on Environment and Development (the 'Brundtland report') defined sustainable development as: "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987). Subsequent to the Brundtland report, many studies have further examined the sustainability concept. A main issue in the interpretation of sustainable development is the assumed degree of substitutability between natural and man-made capital. For instance, Pearce et al. (1989) and Daly (1990) assume a low degree of substitutability between natural and man-made capital. Pearce et al. (1989) state that sustainable development invokes maximisation of the benefits of economic development subject to maintaining the services and quality of natural resources over time. Along this line of reasoning, Daly (1990) argues that sustainability requires that: (i) harvest rates of renewable resources (e.g. fish, trees) not exceed regeneration rates; (ii) use rates of non-renewable resources (e.g. coal, gas, oil) not exceed rates of development of renewable substitutes; and (iii) rates of pollution not exceed the assimilative capacities of the environment.

However, substitutability was assumed to be much higher in, for instance, Beckerman (1994). If substitutability is assumed to be high, the well-known Hartwick rule offers some guidance on the maintenance of consumption levels under resource depletion: under many circumstances in a closed economy with non-renewable resources, the rent derived from resource depletion is exactly the level of capital investment that is needed to achieve constant consumption over time (Hartwick, 1977).

An intermediate position on the interpretation of sustainability is that natural and man-made capital can be either substitutes or complements depending upon the characteristics of the economic system and the specific natural and man-made capital involved (e.g. Georgescu-Roegen, 1979; and Cleveland and Ruth, 1997). In this view, the rate of substitutability depends, among others, upon the type of ecosystem service involved. For instance, the regulation of climate and biochemical cycles, as well as several cultural services can only to a very limited extent be replaced by man-made capital. Solow (1993) also follows a more intermediate position. Solow argues that it is not possible to preserve the full stock of natural capital, and suggests a weaker definition of sustainability where partial substitution of human-made and natural capital is allowed.

Based upon the assumed rates of substitutability, Carter (2001) classifies the different definitions of sustainability into four main categories: (i) very weak; (ii) weak; (iii) strong; and (iv) very strong sustainability. Very weak sustainability allows for infinite substitution between natural and other capital (human and economic). In weak sustainability, it is recognised that certain life supporting ecosystem services can not be replaced, but otherwise it allows for the substitution between different types of capital. Strong sustainability states that the total natural capital stock should not be further reduced, but that limited replacement of one type of natural capital with other types of natural capital is possible (e.g. reforestation may offset clear-cut of forest in other locations, or even the destruction of a certain amount of coral reefs). Finally, very strong sustainability implies that no reduction of the stock and composition of natural capital is allowed. Other authors have linked sustainability to the maintenance of the integrity of the world's ecosystems. In this approach, particular attention is given to the dynamic relations between and among ecosystems, and the importance of the life-support services of ecosystems. In this perspective, sustainable management is interpreted as management that maintains the resilience of ecosystems (Common and Perrings, 1992).

Implicit in applying the sustainability concept is a view on the long-term, dealing with environmental change in the time span of several generations. Hence, assessing sustainability will normally involve the modelling of the impact of ecosystem management on the state and the stability of the ecosystem, and its capacity to supply ecosystem services, over a prolonged time period.

The analysis of sustainability requires selecting a reference situation. It is usually not straightforward to select a reference situation, for which there are three basic options:

- The present. The Brundtland definition was formulated with a focus on assessing sustainability at coarse scales (e.g. at the national or global level). This allows for the degradation of some ecosystems if this is compensated by rehabilitation in other places (WCED, 1982). However, application of the concept at the scale of the ecosystems become problematic since it is unclear if national or global trends require rehabilitation of the particular ecosystem involved, or if there is room to allow degradation while maintaining national or global sustainability. The alternative is then to consider the ecosystem in isolation, and to assume that sustainability requires maintaining the qualities of the ecosystem compared to its present state. However, in this case, if the ecosystem is currently in a heavily degraded state due to recent ecosystem changes, it would, according to most commonly used definitions of sustainability including the one above, be sustainable to leave the ecosystem in its degraded state. This would be contrary to the general perception that restoring recently degraded ecosystems would contribute to sustainability.

- A historical situation. An alternative that would circumvent the risk described above is to compare sustainability with the ecosystem quality in a year in which the ecosystem has a desired environmental quality. For instance, for water quality in Northwest European waterbodies, 1960 can be taken as a reference year. At this point in time, nutrient and agro-chemical pollution levels were generally low, water was relatively unpolluted, and biodiversity and potential to supply ecosystem services were high. In this case, restoring water quality to the 1960 level can be interpreted as environmentally sustainable. However, clearly, the choice of the reference year may be perceived as arbitrary. Since the large majority of the world's ecosystems has undergone gradual or rapid change as a function of human management during centuries or millennia, the selection of a reference year without human disturbance is generally not feasible.

- Defining a reference situation based on ecosystem properties. A different approach is to define sustainability at the ecosystem level on the basis of the properties of the ecosystem itself (e.g. its biodiversity, capacity to provide services, resilience, habitat for specific species, etc.). For instance, sustainable forest management could in specific cases be defined as forest management that conserves the species diversity of the forest, or the numbers and diversity of specific, highly threatened species. For instance, remaining forest patches in Kalimantan tend to have orang-utan densities above their long-term carrying capacity because they serve as a refuge for displaced orang-utan from nearby forests converted to oilpalm plantations. In this case, sustainable management could be interpreted as management that support the forests in harbouring these orang-utan populations, for instance by reforesting degraded spots in the forest with trees that provide forage for the animals.

Hence, defining a reference situation to assess sustainability at the ecosystem level requires consideration of the properties and management history of the ecosystem, and will often require stakeholder involvement in order to select the appropriate reference basis. In addition, the selection of a reference situation needs to consider that there may be temporal variations in ecosystems which cause fluctuations of the ecosystems qualities between years. For instance, the productivity and species composition of semi-arid rangelands depends strongly on annual rainfall, and the reference situation needs either to correct for the impact of annual fluctuations, or to use an average over a number of years to define a reference situation.

In the perspective of the EEA, sustainability can be linked to the capacity of the ecosystem to supply services. A constant or increasing capacity of ecosystems to supply services implies sustainability *sensu*Pierce et al. (1989). An advantage of the EEA approach is that sustainability can be analysed at different spatial levels. Enhancing the capacity to supply ecosystem services in some parts of a country while experiencing a loss in this capacity in other areas may still be sustainable (except when a very strong sustainability criterion is applied). A key task in the context of EEA will be to develop a conceptual understanding of indicators reflecting ecosystem capital or stock, or ecosystems' capacity to supply services, and analyse how they can be identified and quantified. Challenges are the diversity of ecosystems worldwide, dealing with thresholds and ecosystem resilience, and identifying approaches to define sustainability in the face of natural variability.

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