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Expert Meeting on Ecosystem Valuation in the Context of Natural Capital Accounting

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Expert Meeting on Ecosystem Valuation in the Context of Natural Capital Accounting

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Cover picture: View of the Southern Rhine from the Drachenfels (L. Kümper-Schlake).

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The expert meeting was organised by the German Federal Agency for Nature Conservation (BfN) in collaboration with the United Nations Statistics Division (UNSD) and the United Nations Environment Programme (UN Environment) as one of the activities of the European Union funded project '"Natural Capital Accounting and Valuation of Ecosystem Services".

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Table of Contents

Preface

Human well-being depends on manifold values and services of nature. There is nothing new about this insight. But much remains to be done to bring this into the mainstream. One way to uncover the values of nature and its multiple achievements is the economic perspective. Taking a closer look at the economics of nature opens a way of reasoning that might be especially helpful for decision-makers. One prerequisite for good governance is to realize and appreciate nature´s different services and their contributions and values for society.

After decades of limitation of production factors to labour and capital, economic theory has rediscovered the capacity of nature as an additional production factor that provides humans with goods and services to meet physical, social and ethical needs. This also has implications for the central economic reporting systems. In contrast to man-made capital, natural capital always has multiple functions - a hedge provides pollination, pest control, a favourable growth climate, erosion reduction, while preserving biodiversity. Integrating all these functions into economic accounting is, of course, a great challenge. It gives hope that this challenge will now be taken up.

The increasing awareness that nature is a source of ecosystem services and the fact that these ecosystem services are limited and not arbitrarily resilient can be interpreted as a reversal of earlier popular economic views that still have a clearly noticeable imprint on our reporting systems. The integration of the value of ecosystems and their services into the national accounting system may help us. The development of such a tool is an ambitious goal, the implementation of which will require significant resources, but the benefits are manifold and deliver added value.

Therefore, the German Federal Agency for Nature Conservation (BfN) initiated the "Expert Meeting on Ecosystem Valuation in the context of Natural Capital Accounting", which took place from the 24th to 26th April 2018 in Bonn, Germany. It was organized in collaboration with the United Nations Statistics Division (UNSD) and the United Nations Environment Programme (UNEP) as one of the activities of the European Union funded project '"Natural Capital Accounting and Valuation of Ecosystem Services". The meeting aimed to contribute to the achievement of the Convention of Biodiversity's Aichi Target 2 and Action 5 of the European Union Biodiversity Strategy to 2020, which calls for countries to map and assess the state of ecosystems and their services, including economic valuation and integration into accounting and reporting systems. The meeting also advanced the research agenda on valuation in the context of the revision of the System of Environmental Economic Accounting Experimental Ecosystem Accounting (SEEA EEA).

The Federal Agency for Nature Conservation is an institution that works at the interface of science and politics. Our task is to generate and integrate policy-relevant scientific findings from various disciplines and to translate them into a language suitable for political communication. The starting point for the international conference corresponded exactly to this approach: building bridges between different scientific communities in the field of environmental economics and accounting to develop accounting approaches that meet the needs of environmental policy to obtain reliable information on nature´s contributions to people – at present and in the future.

Our aim was to contribute to the exchange of scientific findings, best practice examples and experiences among and between scientists, policy-makers and practitioners. We were delighted to see that an intended small-scale expert meeting developed in the course of preparation into a major international event. The organizers, as well as the participants of the expert meeting, had the impression that we achieved the objectives of the conference and took another important step towards the integration of natural capital into the accounting systems. The vivid discussions between the participants from all over the world with various backgrounds gave this conference an extraordinary interdisciplinary character.

In the name of the organizing team, I would like to thank the speakers, session chairs and all participants for their excellent contribution, inspiring debates and the engagement in the various sessions.

The present conference proceedings are an attempt to reflect the value of the presentations as well as discussions and may serve as an impulse from the view of nature conservation to various stakeholders.

Prof. Dr. Beate Jessel

German Federal Agency for Nature Conservation

Summary of the Meeting

United Nations Statistics Division (UNSD),
United Nations Environment Programme (UN Environment),
German Federal Agency for Nature Conservation (BfN)

The Expert meeting on Ecosystem Valuation in the context of Natural Capital Accounting brought together around 100 participants, including policy makers, economists, and statisticians, from about 25 countries, to discuss valuation of ecosystem services and natural capital assets.

The expert meeting was organized by the German Federal Agency for Nature Conservation (BfN), in collaboration with the United Nations Statistics Division and the United Nations Environment Programme, as one of the activities of the European Union funded project "Natural Capital Accounting and Valuation of Ecosystem Services". Financial support was provided by BfN and the European Union.

There is a strong policy demand for the valuation of ecosystems and their services, as evidenced by the Convention of Biodiversity's Aichi Target 2 and Action 5 of the European Union Biodiversity Strategy to 2020, which calls upon countries to map and assess the state of ecosystems and their services, including economic valuation and integration into accounting and reporting systems.

The revision of the System of Environmental Economic Accounting Experimental Ecosystem Accounting (SEEA EEA) was recently launched, with the objective of reaching consensus on concepts, methods and classifications of ecosystem accounting by 2020 and as a result drop the work "experimental" from the title. The revision process presents an opportunity to advance the research agenda on valuation and address the policy demands on valuation of ecosystem assets and services.

Plenary sessions showcased key approaches and best practices on valuation to achieve policy mainstreaming. Through parallel sessions, in depth discussions were held on the valuation of specific ecosystem services, as well as a wide range of issues ranging from projecting future ecosystem service flows, wealth accounting, ecological debt and degradation. Panel discussions were held to foster dialogue and understanding between the various areas of expertise represented at the meeting.

The meeting provided a platform to share best practices on ecosystem valuations building on experiences from different communities, advance the research agenda on ecosystem valuation and foster enhanced collaboration between various communities on ecosystem valuation. A program of work was developed as a result of the meeting to contribute to the revision process of the SEEA Experimental Ecosystem Accounting. Considering the tight timeline of the revision process, priorities will need to be set to ensure that key issues are resolved for inclusion in the revised SEEA EEA.

Key findings

The meeting was structured around the identified revision issues, and a paper was prepared to frame the discussion. A number of key technical and contextual findings emerged during the discussions:

Technical findings

1. SEEA has to date focused on exchange values, whereas the environmental eco-

nomics literature uses welfare values. This is a fundamental difference to resolve since welfare values can be many times higher than exchange values, which are consistent with the market valuation principles of the SNA. This is particularly true where markets are incomplete and where natural capital has been treated as "free". There is a demand for these welfare values to be presented in complementary accounts in addition to exchange values to provide insights in a broader range of values:

a. It was agreed that it is critical that we articulate the relationship between valuations based on exchange values and the measurement of welfare. Current discussion focused on the differences between simple monetary values using either exchange or welfare value concepts was not sufficient, and a more nuanced discussion is needed that takes into account aspects such as changes in real terms, shadow prices, and income measures that adjust for the cost of capital. For instance, it was shown that if the focus is change over time, there may be minimal differences between changes in welfare values and changes in volume terms, like deflated income, based on exchange values.

b. A crucial element to take into account is the assumptions made regarding institutions / market mechanisms, when doing non-market valuation. In traditional national accounts, exchange values typically represent the outcome of markets under existing governance and property rights scenarios; estimating welfare values often requires assuming conditions such as perfect competition amongst sellers (i.e. no resource monopoly rent) or conversely, open access to the ecosystem service and zero marginal cost. Values are sensitive to these assumptions and therefore the decision to estimate values under real versus hypothetical institutional or market arrangements must be consistently applied across time and across ecosystem services. For instance, the discrepancy between exchange values and welfare values can be driven to zero when we assume that the seller (e.g. the ecosystem) has perfect knowledge of the buyer's willingness-to-pay and so drive consumer surplus to zero. The simulated exchange value method also needs assumptions about institutions. It was suggested that the range of valuation outcomes may be described as a function of the assumed mechanisms, as a way to bridge.

c. There is a need to better explain the uses of exchange values and how they relate to welfare values. It was suggested that information on the broader range of values, may inform the potential welfare gains from investments in conservation.

2. It is noteworthy that the values available in valuation databases do not provide full coverage vis-à-vis the valuation required for SEEA mainstreaming. Although the ecosystem services valuation literature has developed a lot over the past decades, there is a general bias of studies towards interesting/attractive areas. Secondly, a lot less studies have been undertaken for developing countries, with the result that the literature is often not representative enough for what is needed for accounting.

3. The ecosystem accounting approach provides added value. It not only imposes discipline on the debate by providing clear definitions and concepts, but is also able to avoid issues such as double counting. By looking at both the supply of ecosystem services and the condition of the underlying ecosystem, ecosystem accounting will detect situations where the value of a specific ecosystem service increases due to a specific management regime which favors higher yields, at the detriment of the condition of the underlying ecosystem condition.

4. The meeting showcased that cost based approaches have progressed a lot, and

should no longer be seen in opposition to service based approaches (as during the SEEA 1993 and 2003), but rather as complementary:

a. The potential of ES valuation to inform the motivation and underpinning of policy was recognized as important, and restoration costs need not only be perceived in relation to a former reference state but may better be considered in as being forward looking reflecting the costs required to reach socially agreed desired states (based for instance on international environmental agreements such as the Paris Agreement).

b. Restoration cost approaches may even be instrumental in obtaining a valuation of specific ES such as carbon sequestration.

c. There appear to be various uses of different definitions of costs, ranging from replacement cost and damage costs to avoided costs and restoration costs. The differences between these and other definitions of cost (e.g. opportunity costs) should be clearly defined.

5. There is general support for the net present value approach towards valuing ecosystem assets. This approach is applied widely by countries as well as in wealth accounting approaches such as the Wealth of Nations of the World Bank and the Inclusive Wealth Index of UNEP. In some instances, more sophisticated methods are used, such as dynamic biophysical models that take into account issues such as scarcity and feedback loops when projecting future prices. The potential to adapt dynamic methods should be evaluated.

6. A key challenge in ecosystem accounting has been determining the appropriate approach to allocate ecosystem degradation to economic units. The fundamental question is whether degradation should be allocated to the unit affected by it through loss of income; or whether it should be allocated to the unit causing the degradation. There was broad agreement that the allocation should be to the unit causing the degradation, notwithstanding the acceptance that this may be difficult in some circumstances. This outcome provides a strong starting point for future work in this area in the course of the SEEA EEA revision.

7. An important aspect related to understanding changes in welfare in an accounting context is determining an appropriate recording of ecosystem disservices. These arise not through a mutually agreed transaction but when environmental processes and changes impact negatively on economic units and people. While ecosystem disservices fit very directly into an externality and welfare change perspective, the lack of an observed transaction makes recording challenging for accounting. Nonetheless, it is clear that for ecosystem accounting to be considered most useful, it is necessary for information about ecosystem disservices to be meaningfully organized, and more generally for ecosystem disservices to be effectively placed in context.

8. The meeting expressed support for using time-use information to assess services such as nature based tourism or recreation. The issue of the extent to which time-use information can be used to place a value on such services should get more prominence in the revision process, as there are concerns regarding consistency with national accounts principles such as the production boundary.

Context and process findings:

9. While many conceptual and measurement challenges were identified, given the experiences built up in many disciplines and in countries, there is an excellent foundation for describing concepts and methods that will be appropriate for ecosystem accounting.

10. A fundamental issue is that the purpose of ecosystem accounting needs to be made much clearer. For those new to the SEEA community it was unclear what type of question ecosystem accounting was trying to answer and hence it was difficult for them to ensure their responses were appropriate. This speaks to both the spatial scale at which ecosystem accounting focuses and the assumptions concerning non-market valuation. It was agreed that a short note be drafted for discussion that aims to clarify the main purpose of ecosystem accounting.

11. There was a clear benefit in discussing the issue of ecosystem valuation using a focus on individual ecosystem services. To this end it was proposed that the future development of technical guidance on valuation be structured around individual services, and that such an approach would also be useful during the SEEA EEA revision process.

12. There is a need to engage broadly as part of the revision process to better understand the users demand and clearly articulate how the SEEA can answer these demands while at the same time making clear the purpose and boundary of the SEEA, being closely related to the SNA, and ensuring the priority issues are addressed within the time frame of the revision process.

Ultimately, the ecosystem accounts should become the “go-to” dataset for biophysical and valuation data, being multi-year and comparable across countries, driving a virtuous cycle of engagement with policy.

The meeting generated a lot of enthusiasm among participants, and succeeded in bridging between the various disciplines. Overall, this meeting was an excellent commencement to the revision of the SEEA EEA.

1. Foundations of the integration of ecosystems and ecosystem services into the Environmental-Economic Accounts in Germany[1]

Karsten Grunewald[2], Rachel Pekker[3], Roland Zieschank[4], Jesko Hirschfeld[5], Burkhard Schweppe-Kraft[6], Ralf-Uwe Syrbe[7]

Summary

Our economic and social activities are constantly putting pressure on our ecosystems, changing their condition and their capacity to produce the services we desire in a sustainable manner. Against this background, the integration of ecosystems and their services in the national economic accounts seems necessary since it offers considerable potential for improving political steering capacities. This paper explains the theoretical basic conditions, methodological foundations and challenges of such an "ecosystem accounting" and outlines case studies of a corresponding German pilot study.

Keywords: Biodiversity, Natural capital, Economical assessment, Ecosystem accounting

1. Introduction

The loss of biodiversity is emerging as one of the major unsolved environmental problems of the 21st century. This development is threatening the integrity of the biosphere, also in view of the *planetary boundaries* (Steffen et al. 2015), and ultimately requires a rethinking of the patterns of production and consumption.

Also, nature is being disregarded as a “productive factor” and is thus at risk of being underestimated. Soil fertility, clean air and potable water are only a few of the contributions of intact ecosystems; and it is advisable to remember that neither an apple nor an interconnected community of plants and animals can be manufactured industrially. Ecosystem services (ES) are therefore to be integrated into the societal reporting systems in future (EU 2011). This includes the development of a perspective which acknowledges that societal prosperity is not only based on human labour and capital and is thus more comprehensive than the gross domestic product (GDP). In order to strengthen society's perception of the "natural capital", the systems of national accounting must also become more “inclusive” and must adequately capture the services of ecosystems and the associated biological diversity for people and society. Integrating these values into the national accounting systems (NAS) and in particular into the systems of environmental-economic accounting (SEEA) is meant to contribute to supporting policymakers and businesses in making decisions on corre-

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sponding measures and on financing mechanisms for the conservation of natural resources.

Against this background, the Federal Agency for Nature Conservation (*Bundesamt für Naturschutz*, BfN) initiated the pilot project "Integration of ecosystems and ecosystem services into the Environmental-Economic Accounting" ("*Integration von Ökosystemen und Ökosystemleistungen in die Umweltökonomische Gesamtrechnung*"), which is being implemented by the authors of this paper. The project takes the form of a feasibility study. In the present contribution, we discuss the conditions and foundations as well as some challenges for a corresponding accounting in Germany and outline some of the study's example cases.

2. Foundations

2.1 Considerations on the conceptual framework for including natural capital and ecosystem services in societal reporting systems

The interest in capturing ES can be derived from three lines of arguments, which sometimes overlap in political practice.

First, there is an extended discussion on alternative welfare concepts, which derives from criticism of the gross domestic product (GDP) that still dominates economic policy. A key milestone was the conference "Beyond GDP" that was convened in 2007 by the EU and other international institutions. The spectrum of publications ranges from the report "The Changing Wealth of Nations" by the World Bank (2011) to a variety of comprehensive welfare indicators. A broader understanding of welfare and well-being that also includes human and social capital as well as the natural capital is crucial (Diefenbacher, Zieschank 2010; EDI, BFS 2016).

The second discursive thread deals with the significance of nature and natural capital in the context of economic thought. Of particular prominence are the publications in 2009 and 2010 of the international TEEB Initiative (The Economics of Ecosystems and Biodiversity) as well as of the "National Capital Committee" in the United Kingdom, which has meanwhile led to comprehensive assessments of the natural capital; significantly supported by the Office for National Statistics (ONS 2015).

The third line of argument, which is more strongly committed to nature conservation and the preservation of species diversity, is associated with the international Convention on Biological Diversity (CBD). As animal and plant species can only be preserved together with their habitats, the regular inventories and spatio-temporal assessments of their state not only document nature as an economic factor but are also an essential instrument for stopping the decline of biological diversity. Actors include UNEP and the World Conservation Monitoring Centre, as well as the European Environment Agency and the MAES working group in Europe (Maes et al. 2013).

Many of these initiatives are active in elaborating and further revising the "System of Environmental-Economic Accounting – Experimental Ecosystem Accounting" (SEEA EEA) supported by the UN, so this is an instrumental focal point.

In the context of concepts that span disciplines and sectors, it is important to establish a terminology that is equally understood and accepted by economists, ecologists, sociologists, practitioners and politicians. Precise communication about what is meant by natural capital, ecosystem and service of the ecosystem and about how this is to be measured is not trivial. The various international approaches are quite heterogeneous in this respect;

moreover, the correspondence between English and German terms is not automatically unambiguous. Also, the central question of how ES (e.g. natural soil fertility) can be distinguished from the results of human actions (agricultural products) not only conceptually but also metrologically has yet to be answered in a unified way.

2.2 International initiatives and the current state in Germany

CBD and EU Biodiversity Strategy

The so-called 'Aichi targets' of the Convention on Biological Diversity (CBD) are crucial to the international efforts to preserve biological diversity. Target 2 provides that by 2020 biodiversity indicators and assessments shall be included in national accounting and reporting systems as appropriate. In a similar manner, Action 5 (Target 2) of the EU Biodiversity Strategy 2020 aims to improve knowledge of ecosystems and their services. The member states, with the assistance of the commission, are to map and assess the state of ecosystems and their services in their national territory by 2014, and to promote the integration of ecosystems and their services into accounting and reporting systems at EU and national level (EU 2011).

SEEA EEA

In 2014, the UN, the European Commission, the Food and Agriculture Organization of the UN, the OECD and the World Bank published a handbook on "Systems for Environmental-Economic Accounting - Experimental Ecosystem Accounting" (UN 2014). The publication includes chapters for representing ecosystem services ("flows") and ecosystems ("stocks") as well as their valuation, integration and relationship to the quantities of national accounting.

Whereas in the existing SEEA framework the accounting starts from the economic perspective and then puts environmental information on natural resources and environmental impacts in relation to economic actors or sectors, the SEEA EEA places the focus on ecosystems and their relationship to economic and other human activities. Nature is viewed not in the form of individual unrelated stocks (soils, wood, fish etc.), but as ecosystems (lakes, forests, city areas etc.). This is based on the overarching understanding that the ecological system and the economic system are to be viewed as one cohesive unit.

The central categories and connections are shown in Figure 1, from the ecosystems and their processes to individual and societal welfare, based on the material and immaterial advantages of the use of ES.

The inclusion of ES through the expansion of the SEEA is to occur in a form analogous to and compatible with the existing accounting logics. A complete implementation would have to capture all aspects of an environmental-economic model, from the inclusion of nature as stock or capital through ES and goods produced (with them), to emissions, waste flows and the degradation or recreation of natural capital by society. In analogy with national accounting, the ambition here is to capture not just the most important, but generally all ecological systems as spatial measurement units (e.g. all rivers, lakes and groundwater reservoirs) and to describe their physical and monetarily valuated exchange with the economic system (e.g. water quantity, quality, wastewater, recreational use, extraction of drinking water, purification costs).

Currently, experimental ecological accounting is undergoing a worldwide, participatory revision process, which is to be concluded by 2020 under the leadership of the UN and with the participation of EUROSTAT in particular.

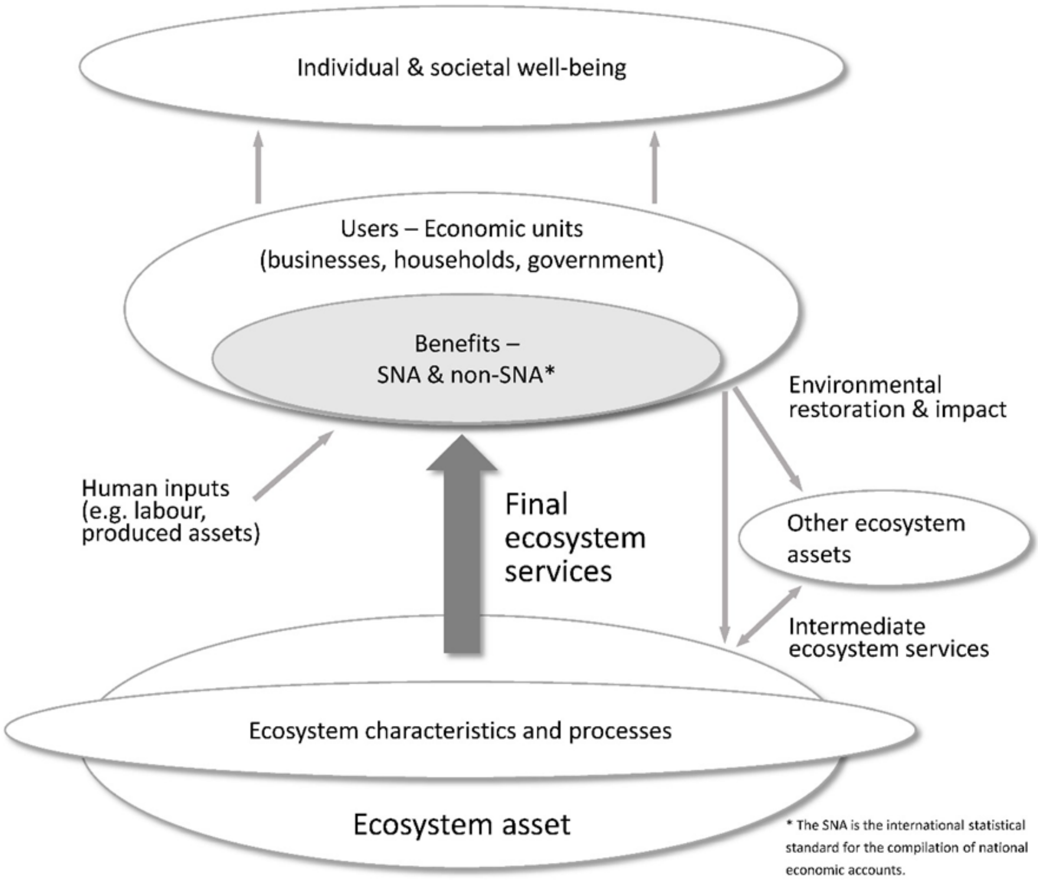

Figure 1. Ecosystem and ecosystem services accounting framework (adapted from UN SEEA EEA 2017)

KIP-INCA

At the European level, the “EU Knowledge Innovation Project on an Integrated System for Natural Capital and Ecosystem Assessment” (KIP-INCA) that the European Commission is establishing together with further partners is of great importance. INCA is testing and promoting the SEEA-EEA and by 2020 will establish both physical and monetary accountings for an entire range of ES, from water pollution control and pollination services to carbon capture and habitat preservation as well as the ecosystem contribution to recreation. The goal is an accounting system at the EU level. The participants view the inclusion of ES in the standardized national accounting systems as an essential approach for making nature and biodiversity part of macroeconomic and political processes (EC/EEA 2016).

WAVES

The Worldbank’s initiative “Wealth Accounting and Evaluation of Ecosystem Services” (WAVES) is also making a key contribution at the international level. The initiative combines goals of sustainable economic development with the preservation of biological diversity and natural capital and the actors are in close contact with the United Nations Development Programme (UNDP) as well as the CBD. There are now various studies on ecosystem accounting from developing countries that were supported by WAVES.

The current state in Germany

In 2010, the Federal Statistical Office (Destatis) presented conceptual considerations by O'Connor/Schoer (2010) on an increased integration of components of environmental assets and their degradation. Currently, internal work is being done on the prerequisites for linking ES with the SEEA.

The Federal Agency for Nature Conservation, which also took part in the development of the SEEA EEA together with the UBA Vienna (Environment Agency Austria) and the Swiss Federal Office for the Environment, commissioned a first research project in 2010 on the implementation of the above-mentioned Target 2, Action 5 of the EU Biodiversity Strategy. The results included recommendations for developing a first national indicator set for capturing and valuating ecosystem services (Marzelli et al. 2014; Albert et al. 2015). This work is being built on and being supplemented by indicators for the state of ecosystems. As of December 2017, 50 indicators have been proposed for 20 ES classes, including for the provisioning service wood, for the regulation service flood protection through floodplains, for avoided water erosion as well as for the accessibility of urban green spaces as cultural ES (Grunewald et al. 2017b).

On this basis and with the advice of the Federal Statistical Office, the project mentioned in the introduction is now to take first concrete steps towards an ecosystem accounting also in Germany.

2.3 Economic valuation methods and accounting

Any national accounting consists of a consideration of amounts and a consideration of values. The monetary valuation of goods and services offers a common metric that allows for aggregation and comparison, including comparisons of ecosystems and ES with capital quantities and services that enter into the NAS. The theory of the economic valuation of ecosystems and ES is based on an anthropocentric approach. Ecosystems and ES can be assigned a value through the direct or indirect satisfaction of human needs (TEEB 2010). The approach for valuating ES is based on the analysis of the change in benefit depending on marginal changes of the quantity or quality of goods or services. From an economic perspective, ecosystems can be viewed as part of our natural capital, and the flow of ES can be viewed as the goods that are produced with this capital.

The *Total Economic Value* (TEV) is employed as a methodological framework for determining the total economic value of ecosystems and ES (Pearce 1993; De Groot et al. 2002). It forms the basis on which the project analyses the compatibility of the current standard valuations of ES with the rules of the accounting and subsequently develops proposals for integration into the SEEA and NAS.

A variety of methods for valuating ES have been developed over the past decades (e.g. Garrod, Willis 1999; Chee 2004). A comprehensive account of all current valuation methods of ES and their link to the TEV can be found e.g. in the TEEB-Local and Regional Policy Makers (2010); for an overview see Table 1. Up to now, the focus has been on welfare-economic procedures that are employed in the context of cost-benefit analyses and capture the change in the value of an amount of a good in case of an increase in sales through the continuous change in the willingness to pay per unit of the good between two quantities of sales ("consumer surplus"). By contrast, in accounting the quantities and current prices at two different points in time are shown multiplied.

Table 1. Valuation methods under the TEV (total economic value) approach

Economic valuation methods	Method
Market analysis	Price based • Market prices Cost based • Avoided damage costs • Replacement costs • Restoration costs • Opportunity costs Production based • Production Function • Net income method
Revealed preference methods	• Hedonic price approach • Travel cost method
Stated preferences methods	• Contingent valuation • Choice Modelling • Deliberative group valuation
Benefit transfer	• Average • Adjusted mean • Utility function

Whereas the system of accounting is based on nominal market prices (exchange values), the economic (welfare) value is based on a person's willingness to pay for a good. This can be higher than the market value. Depending on the situation and the allocation of property rights, the willingness to pay can be measured as the willingness to pay or the willingness to accept.

In contrast to the TEV, accounting frameworks and methods are traditionally not designed to capture the total economic value of a national economy in the sense of the combined benefit of all goods and services including e.g. external effects. By contrast, the goal of integrating ecosystems and ES is to expand the national accounting by estimating a completed range of services and assets (Nordhaus 2006; Obst 2018). Valuation methods and their suitability for valuating ES according to SEEA were analysed and discussed in the SEEA EEA Technical Recommendations (UN 2017) (Table 2). These are taken into account in the project.

Table 2. Valuation techniques and their use in ecosystem accounting (adapted from UN SEEA EEA 2017)

Evaluation method	**Suitability for the evaluation of individual ecosystem services**
Unit resource rent	In principle, appropriate
Production function, cost function and profit function methods	Reasonable, under certain conditions
Payments for Ecosystem Services (PES)	Maybe appropriate
Hedonic prices	Basically appropriate in certain circumstances
Replacement costs (or replacement costs)	Reasonable, under certain assumptions
Avoided damage costs	Reasonable under certain assumptions
Defensive behaviour	Maybe appropriate
Restoration costs	Probably inappropriate
Travel costs	Maybe suitable
Expressed preference	Unsuitable (no exchange values, but possibly suitable for generating a demand function)
Limits of demand functions	reasonable

3. Sample implementation

3.1 Analysis of the development of the area of various ecosystems in Germany (*ecosystem extent account*)

Ecosystem mapping is the first step in implementing Target 2 of the EU Biodiversity Strategy 2020. This basis is propaedeutic for the valuation of the ecosystem states and services (Blasi et al. 2017) and goes beyond the land use statistics as integrated in the SEEA as "area by type of actual use". Accordingly, approaches were developed and coordinated with the BfN in order to allow for a complete, non-redundant description of land and water areas. Challenges exist in particular in the degree of thematic detail in the ecosystem mapping, mainly with respect to whether and which functional features should be included that can be suitable for supporting the attainment of the respective goals (e.g. protecting biodiversity, prioritizing the restoration of damaged ecosystems, ecosystem extent account) at the national level.

The development of the ecosystem typology was guided by the following premises:

- clear, coherent structuring principle for ecosystem types (ET): by land cover (vegetation/use)
- derivable from existing data sources
- compatible with international systems (such as MAES / SEEA)
- time sections available (monitoring): changes in the various stocks quantifiable (Which ET was replaced by which other one?)

Accordingly, three hierarchical levels were proposed: 5 main ET, 14 sub-ET and further differentiation into 37 CLC classes for Germany (Table 3). They are based on the European classification of Corine Land Cover (CLC), which is evaluated based on the digital land cover model of Germany (*Landbedeckungsmodell Deutschlands*, LBM-DE). The proposed classification is based – as far as possible – on the European biotope classification EUNIS (European Nature Information System) of the European Environment Agency (EUNIS 2007). For this purpose, similar biotope types from different CLC classes were sometimes combined into the ET.

Federal evaluations are to be carried out primarily on a 1 x 1 km grid basis (INSPIRE grid) in order to ensure compatibility with other data bases. The output of results can be implemented based on the "area proportion" in the form of individual maps for theme-specific propositions.

The time sections of the calculation primarily refer to 2012 and 2015, corresponding to the LBM data. For these two reference years (and in future every third year), population movements and changes in the types can be represented. Older time sections cannot be compared exactly due to changes in the data acquisition methods.

More finely differentiated ecosystem types in the framework of capturing and assessing habitat types according to the FFH directive and the mapping of high-nature-value farmland as well as differentiated state information from the federal forest inventory are assigned to the above comprehensive ecosystem types and used for describing and quantifying changes in their quality.

Table 3. Proposed classification system of ecosystem types (EST) for Germany (CLC – Corine Land Cover)

Main EST	Sub EST	CLC code	CLC class name
1 Semi-natural open areas	11 Grassland and heath-land	321	Natural grassland
		322	Moors and heathland
	12 Wetlands	411	Inland marshes
		412	Peatbogs
		421	Coastal salt marshes
		423	Intertidal flats
	13 Open spaces with no or little vegetation	331	Beaches, dunes and sand plains
		332	Bare rock
		333	Sparsely vegetated areas
		334	Burnt areas
		335	Glaciers and perpetual snow
2 Forest and grove areas	21 Forest	311	Broad-leaved forest
		312	Coniferous forest
		313	Mixed forest
	22 Grove	324	Transitional woodland/shrub
3 Agricultural land	31 Arable land	211	Non-irrigated arable land
		221	Vineyards
		222	Fruit tree and berry plantations
	32 Grassland	231	Pasture, meadows and other permanent grasslands under agricultural use
	33 Heterogeneous agri-cultural area	242	Complex cultivation patterns
		243	Land principally occupied by agriculture, with signifi-cant areas of natural vegetation
4 Water	41 Streams	511	Water courses
	42 Inland water bodies	512	Water bodies
	43 Marine waters	521	Coastal lagoons
		522	Estuaries
		523	Sea and ocean
5 Settlement and artificial modified areas	51 Buildings and trans-portation area	111	Continuous urban fabric
		112	Discontinuous urban fabric
		121	Industrial and commercial units
		122	Road and rail networks and associated land
		123	Port areas
		124	Airports
		133	Construction sites
	52 Mining and dump sites	131	Mineral extraction sites
		132	Dump sites
	53 Urban vegetated areas	141	Green urban area
		142	Sport and leisure facilities

3.2 Case studies for the sample implementation

The case studies of the pilot project (Textbox 1) were selected according to the criteria of exemplary coverage of important ES areas, potential compatibility with the SEEA, practicability (e.g. usability of data/groundwork from other projects) as well as processing capacity. An important point is that the case studies follow a systematic approach. They should all be based on common classifications and a uniform assessment system.

Textbox 1. Outline of two planned case studies

[1] "Accessibility of urban green spaces" as a cultural ES

The establishment of empirical foundations and action targets for a green infrastructure in our cities is crucial for their sustainable development, as "Green in the City" has a decisive influence on the quality of life. An accounting of urban ecosystems and services would be new in the SEEA.

For this purpose, the ES indicator "Accessibility of urban green spaces" (Grunewald et al. 2017a) is employed, which was developed by IÖR/BfN and was proposed as a core indicator for formulating action targets for city green and sustainable building by the Federal Institute for Research on Building, Urban Affairs and Spatial Development / Federal Office for Building and Regional Planning (*Bundesinstitut für Bau-, Stadt- und Raumforschung / Bundesamt für Bauwesen und Raumordnung*, BBSR/BBR) (BBSR 2017).

In 182 German cities with more than 50,000 inhabitants, geodata on green spaces and water areas are analysed with respect to their quality (size, usability) and the reachability of these areas (distance from users). 74.3% of the inhabitants of the cities studied can reach both smaller (≥ 1 ha) green spaces and water areas within a straight-line distance of up to 300 m (≈ 500 m by foot) and larger ones (≥10 ha) within a straight-line distance of up to 700 m (≈ 1 km by foot).

This indicator is an understandable, robust and reproducible measured quantity. It can be supplemented by parameters such as the proportion of green spaces, the green area per inhabitant, the proportion of sealed areas and the cost of upkeep per unit area (BBSR 2017). The indicator is thus a combination of supply (of green spaces) and potential demand (number of inhabitants living nearby).

The monetary valuation is based on economic assessment studies on urban green spaces in Germany. So far, methods of *revealed preference* (*hedonic pricing*) and *life satisfaction* have primarily been used for this (e.g. Bertram, Rehdanz 2015; Krekel et al. 2016). The value of a green space is determined by its quality and accessibility, expressed through its use type (forest, garden, water body etc.) and its reachability in terms of the location relative to the inhabitants under the assumption of average usage behaviour.

[2] Biodiversity as a cultural ES and as part of the capital value of an ecosystem

The "biotope values" used in nature conservation law to regulate interventions are interpreted as physical indicators for the service of an ecosystem for preserving biological diversity and for the biodiversity itself (existence value of the cultural ES) as part of the capital value. The calculation is based on the willingness to pay for national nature conservation programmes, using benefit transfer per biotope value point. Payment systems like PES (*Payments for Ecosystem Services*) are viewed as a suitable valuation basis in the framework of accounting; for when biotope value points are actually traded on "markets", they have an exchange value character.

Alternatively, mean restoration costs (including "time costs") for implementing the areal goals of the FFH directive are offered per additionally created biotope value point. By converting the restoration costs into annuities and the willingness to pay into capital values, the results of the two procedures can be compared both in terms of services and in terms of assets. Updating this work with randomized surveys on the costs of the development and upkeep of biotopes or acquiring data on special expenses for environmental protection would be helpful for improving the empirical content.

3.3 Challenges/Limits of the accounting

If the differentiated standards of some methodological considerations (UN 2014; La Notte et al. 2017) are taken seriously, an accounting becomes quite demanding overall:

- For the beginning of a reporting period, the initial stocks must be determined (*opening stocks*). At the end, the expansions or reductions in the area or number of an ecosystem must be identified and quantified.
- Ecosystem services should be captured in analogy with the production of goods or services in the framework of the NAS or SEEA. For the exchange between the ecological and the economic system, an *exchange value* for the service/the good must be determined, and ultimately it can only be reasonably determined with respect to the advantages for a *concrete* user or actor (*ecosystem services supply and use accounts* in physical and monetary forms).

Both conceptual and methodological challenges can arise in the monetary valuation of ecosystems and ES (see e.g. Obst 2018 for a more detailed consideration). In order to protect the estimated economic values against misinterpretations, it is important to clearly describe and communicate the economic assumptions underlying the values.

Environmental systems are usually very complex, spatially heterogeneous and sometimes marked by non-linearity. For these reasons, the value of large changes cannot always be inferred on the basis of the valuation of small changes (a key term in this context is *tipping points*). The spatial heterogeneity of environmental goods complicates the transfer of values and thus impedes benefit transfer studies and meta-studies that can serve as a substitute for expensive primary studies. Estimating the values also raises the challenge of decisions on the precision of the values in view of complexity and feasibility. There is also a need to clarify which method of benefit transfer (average, adjusted average or benefit function) is to be employed.

As ecosystems and ES are public goods, their economic valuation also raises questions of distributive justice. It is certainly questionable whether the valuation of public goods should depend on the users' income. A precise analysis and an understanding of the underlying political and analytical question is necessary for being able to decide whether an exchange value concept is adequate. When *exchange values* are used in order to combine ecosystem data and economic data into integrated accounts, it can also prove to be appropriate to estimate supplementary *welfare-based* valuations, using the same underlying biophysical information. Thus, two different monetary values could be listed side by side in order to point out and represent the differences between the exchange value and the welfare value.

Moreover, it must be taken into account that e.g. values obtained from real estate prices are already included in the NAS and the values are "only" being reassigned. In any case, a double counting must be avoided.

4. Outlook

Germany principally intends to increasingly integrate ecosystems and ES in the SEEA/NAS in future. Some methodological foundations for taking ecosystems and their services into account in the SEEA were presented, as well as principles of an accounting, but they form a dynamical field at the interface of interdisciplinary science and statistical reports for policymaking. A rigorous framework for this with compatibility with existing systems is only now emerging, both internationally and nationally.

Thus, suitable access points for bringing in the proposed ES indicators and their monetary valuations are to be identified in coordination with the Federal Statistical Office. Building on this, recommendations are developed and discussed with experts on how a future accounting could be fleshed out in Germany in the style of the SEEA EEA – but also going beyond it with respect to the available information – in order to provide the informational foundations for an expanded view of societal prosperity and the role of nature.

Acknowledgements

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2. INCA - The EU Knowledge Innovation Project on an Integrated System for Natural Capital and Ecosystem Services Accounting[1]

Lisa Waselikowski[2]

Background

Eurostat presented the INCA project at the expert meeting on 'Ecosystem valuation in the context of natural capital accounting' that took place in Bonn, 24-26 June 2018. The presentation outlined the objectives of INCA, project partners and the planned INCA accounting system for 2020. It further illustrated how INCA contributes to a community of practice in the EU member states as well as at a European level and provided an overview of publicly available INCA output.

INCA objectives

The INCA project aims to design and implement an accounting system for ecosystems and their services in the European Union by 2020, integrating existing georeferenced EU data and data from member states reporting exercises. It builds on results achieved under the EU initiative on Mapping and Assessment of Ecosystems and Services (MAES), as well as on categorizing ecosystem services through the Common International Classification of Ecosystem Services (CICES). Its work is further based on output of other relevant projects such as ESMERALDA, OPERAS and OPPLA.

INCA provides added value in a range of specific policy contexts and addresses key EU policy objectives of the EU Biodiversity strategy to 2020 and the EU 7th Environmental Action Plan.

By applying the guidelines of the UN System of Environmental Economic Accounting – Experimental Ecosystem Accounting (UN SEEA EEA) at European level, INCA tests and advances the SEEA EEA methodology and contributes to its further development. Alignment with the SEEA framework and its current revision process is ensured though Eurostat's close collaboration with the SEEA EEA process: Eurostat is a signing institution of SEEA CF and SEEA EEA and a member of the UN Committee of Experts on Environmental-Economic Accounting. Since 2017 Eurostat chairs the SEEA EEA Technical Committee that advances the revision of the SEEA EEA.

INCA partners

INCA is a project of the European Commission and the European Environment Agency and consists of five partners who all hold distinct roles and responsibilities:

- <u>Eurostat:</u> coordinates INCA, provides data, and ensures alignment and testing with SEEA EEA alignment.
- <u>EC Joint Research Centre (JRC)</u>: operates information systems, has vast expertise in modelling ecosystem services and develops ecosystem services accounts.
- <u>DG Research and Innovation (DG RTD)</u>: ensures coordination between INCA and EU

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[1] Summary of the presentation 'INCA - The EU knowledge innovation project on an Integrated System for Natural Capital and Ecosystem Services Accounting', at the expert meeting on 'Ecosystem Valuation in the context of Natural Capital Accounting', Bonn 24 – 26 June 2018.

[2] Eurostat

research activities.

- DG Environment (DG ENV): provides policy context, manages MAES, is a principal user of INCA outputs.
- European Environment Agency (EEA): develops shared data platform and ecosystem extent and condition accounts, provides data.

INCA accounting system

Figure 1 describes the overall set-up of the INCA accounting system which is planned for 2020. Data, models and a shared spatial data platform constitute the basis of all the accounts which, in turn, deliver user-oriented outputs in the form of accounting tables, maps and geospatial information, and indicators which describe the state and trends of natural capital.

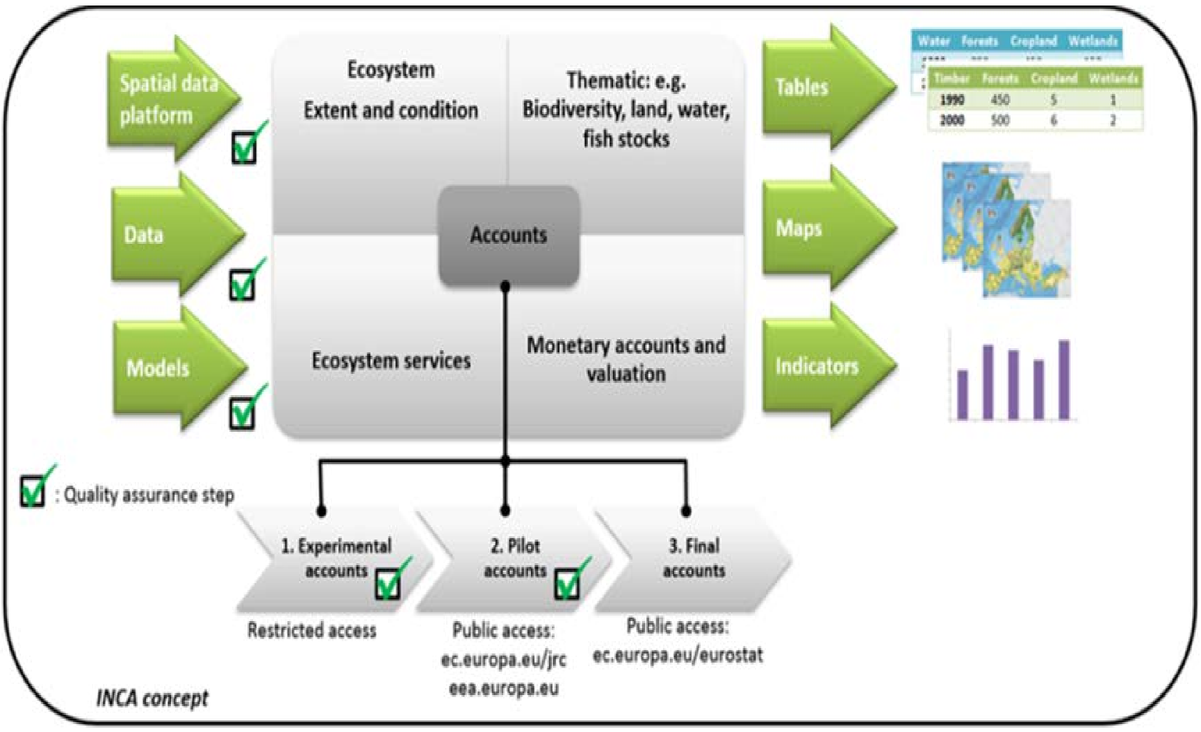

Figure 1. INCA accounting system (own representation)

Supporting a community of practice

The INCA partners contribute to a community of practice both on member state and EU level through a range of measures:

- Eurostat grants to support ecosystem accounting in statistical offices of EU member states. The first grants were awarded in 2017.
- Courses on ecosystem accounting under the European Statistical Training Programme for the statistical offices of EU member states.
- Dedicated research action on natural capital accounting under H2020 WP 2018 – 2020. Project work will start at the beginning of 2019 and will directly feed into the further development of the INCA accounting system.

Public INCA output

All INCA output is published on a central website. It features reports published by INCA partners such as the INCA phase I report on progress and planned output under the project, as well as technical documents such as the JRC report on outdoor recreation and crop pollination accounts. The website further displays INCA support contract studies on various ecosystem accounting issues such as valuation for Natural Capital and Ecosystem Accounting, pilot marine accounts and others.

3. Theoretical developments of the comprehensive (or "green") national accounting literature[1]

Geir B. Asheim[2]

Abstract

I present a definition of income which is compatible with an important line of theoretical literature on comprehensive national accounting.

1. Introduction

What is the income of a national economy, of each individual in an economy, or of each sector of an economy?

In practical applications, income has often been measured by wealth-based measures, whereby the present value of consumption (or the cash flow from a sector) is estimated, and income is equated with the interest on wealth determined in this manner (see Aslaksen et al., 1990, and Brekke, 1997, Sections II.C and IV in the case of sectoral income).

In contrast, a line of theoretical literature – from Hicks (1946, Chapter 14) via Samuelson (1961) to Sefton and Weale (2006) – has taken a quite different route by associating income with the present value of real interest on future consumption and savings with the present value of future consumption changes.

In this note I provide an exposition of this line of theoretical literature on comprehensive national accounting. I start in Section 2 by presenting a short survey of relevant literature and the income concepts presented in these contributions. Then, in Sections 3 and 4, I present a definition of real income at the national level in line with Sefton and Weale (2006), while generalizing their welfare results slightly. How this definition can easily be extended to individual and sectoral income is the topic of Asheim and Wei (2009) and will not be treated in any detail here.

2. What is income?

At a national level, in particular in the context of a closed economy with a stationary technology, income can be derived from net national product, measuring the value of the flows of goods and services that are produced by the productive assets of society. National income derived in this way has also welfare significance, as established by Weitzman (1976) and later references (e.g., Aronsson et al., 1997, 2004; Asheim and Weitzman, 2001). At a sectoral level, it is however hard to determine what a sector's "net product" is, since much of the return on the sector's assets may derive from expected capital gains.[3] In particular, the remaining deposits of a non-renewable resource is not productive as a stock but yields its owners positive returns by being moved closer to the time of depletion. This motivates a brief survey of relevant literature on income concepts.

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[1]This note contains extracts from Asheim and Wei (2009). It is part of the research activities at the Centre for the Study of Equality, Social Organization, and Performance (ESOP) at the Department of Economics at the University of Oslo. ESOP has been supported by the Research Council of Norway through its Centres of Excellence funding scheme, project number 179552.

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[3]See, however, an interesting attempt to do so in Sefton and Weale (2006, Section 6.2).

Income in the tradition of Fisher (1906) and Lindahl (1933, Section II) is associated with interest on wealth, where wealth is the present value of future consumption. If, at each point in time, national consumption equals the sum of the cash flows from the different sectors of the economy, this definition allows national income to be split into sectoral income so that sectoral income summed over all sectors adds up to national income.

Hicks (1946), in Chapter 14 of Value and Capital, suggests that “the practical purpose of income is to serve as a guide for prudent conduct” by giving “people an indication of the amount which they can consume without impoverish themselves” (both quotes from Hicks, 1946, p. 172).

Hicks (1946, p. 174) points out that income as interest on wealth is not an indicator of prudent behaviour if the real interest rate is expected to change. This observation is nicely illustrated by the Dasgupta-Heal-Solow model (Dasgupta and Heal, 1974, 1979; Solow, 1974) of capital accumulation and resource depletion where the real interest rate is decreasing along a path where capital is accumulated and resource flow diminishes. In this model, income as interest on wealth exceeds both net national product and consumption along an efficient path with constant consumption. Hence, in this setting, the consumers of the economy would impoverish themselves if they were consuming the interest on their wealth.

If we instead use Hicks’s (1946) suggestion to obtain alternative income concepts, then we must operationalize what is meant by “the amount which they can consume without impoverish themselves”. Hicks (1946) himself offers the following operationalization, referring to the corresponding concept as “Income No. 3”:

“Income No. 3 must be defined as the maximum amount of money which the individual can spend this week, and still be able to spend the same amount in real terms in each ensuing week” (Hicks, 1946, p. 174). “The standard stream corresponding to Income No. 3 is constant in real terms . . . We ask . . . how much he would be receiving if he were getting a standard stream of the same present value as his actual expected receipts. This amount is his income. (Hicks, 1946, p. 184)”

Hence, income is associated with the “stationary equivalent of future consumption” (Weitzman, 1976, p. 160).

In an economy where well-being depends on a single consumption good, this concept of income can be defined as the constant level of consumption with the same present value as the actual future stream of consumption. Such wealth equivalent income can be determined at both a national and sectoral level in such way that sectoral income summed over all sectors adds up to national income. Moreover, the concept is designed to be an indicator of prudent behaviour (although wealth equivalent income is only hypothetically sustainable if interest rates are changed when consumption is transformed into a constant and efficient path).

Unfortunately, as pointed out by Asheim (1997) and Sefton and Weale (2006, Section 3.1.2) (and discussed in Appendix B of Asheim and Wei, 2009), such wealth equivalent income at the national level does not equal net national product, even in a closed economy with stationary technology, unless the interest rate is constant (which is the case analysed by Weitzman, 1976) or the consumption level is constant (in which case Hartwick's rule[4] implies that net national product equals this constant level and, thus, the result follows). Moreover, this concept is hard to generalize to the empirically relevant case of multiple consumption goods, since determining an amount constant in real terms leads to an indexing problem if relative consumption prices are changing.

However, Hicks's "amount which they can consume without impoverish themselves" can be interpreted in an alternative manner. Hicks (1946, p. 172) writes that "it seems that we ought to define a man's income as the maximum value which he can consume during a week, and still expect to be as well off at the end of the week as he was at the beginning." One attractive possibility, suggested by Pemberton and Ulph (2001) and Sefton and Weale (2006), is to associate "as well off" with the level of dynamic welfare.

It is an insight first pointed out by Samuelson (1961, pp. 51–52) that the present value of future consumption changes measures welfare improvement in a market economy following an optimal path. This gives a welfare foundation for interpreting the present value of future consumption changes as national savings. Adding current consumption to this notion of savings (measured in the same numeraire) leads to a concept of national income with nice properties:

(1) It follows from Samuelson's insight that such a concept of national income is an indicator of prudent behaviour, since the present value of future consumption changes is positive—and thus, dynamic welfare improves—if and only if consumption is smaller than national income.

(2) It follows through integration by parts that such a concept of national income can be expressed as the present value of real interest on future national consumption.

(3) It follows from the analysis of Sefton and Weale (1996) and Weitzman (2003, Chapter 6) that such a concept of national income equals net national product in a closed economy with a stationary technology.

In Sections 3 and 4 (and backed up by the results of Appendix A of Asheim and Wei, 2009) I establish formally properties (1)–(3) under assumptions more general than those imposed by Sefton and Weale (2006); in particular, we do not assume that a discounted utilitarian welfare function is maximized, and we do not require that the technology satisfies constant-returns-to-scale.

In Sections 5 and 6 of Asheim and Wei (2009) we pose the question of how to split this concept of national income into sectoral income in such way that sectoral income summed over all sectors adds up to national income. We do so first, in Section 5, by splitting national income into individual income, building on analysis presented by Sefton and Weale (2006), and then, in Section 6, by defining sectoral income by considering the contributions to individual income that the sectors give rise to. Throughout (and in line with the analysis of Sefton and Weale, 2006), consumer price indices play a central and natural role when turning nominal into real prices.

[4] Cf. Hartwick (1977) and Dixit, Hammond and Hoel (1980).

3. Defining national income

Consider a national economy, where c is a comprehensive vector of consumption flows, implying that all determinants of current well-being are included in c.

Let $\{\mathbf{c}(t)\}_{t=0}^{\infty}$ be the path of consumption flows in this economy and let $\{\mathbf{p}_c(t)\}_{t=0}^{\infty}$ be the corresponding path of market (or calculated) present value prices of consumption. The term “present value” reflects that discounting is taken care of by the prices.

In particular, if relative consumption prices are constant throughout and there is constant real interest rate R, then it holds that $\mathbf{p}_c(t) = e^{-Rt}\mathbf{p}_c(0)$. However, we will allow for non-constant relative consumption prices and will return to the question of how to determine real interest rates from $\{\mathbf{p}_c(t)\}_{t=0}^{\infty}$ in this more general case.

Differentiation of $\mathbf{p}_c(t)\mathbf{c}(t)$ yields

$$\tfrac{d}{dt}\big(\mathbf{p}_c(t)\mathbf{c}(t)\big) = \dot{\mathbf{p}}_c(t)\mathbf{c}(t) + \mathbf{p}_c(t)\dot{\mathbf{c}}(t)\,.$$

Integrating on both sides under the assumption that $\mathbf{p}_c(\tau)\mathbf{c}(\tau) \to 0 \text{ as } \tau \to \infty$, leads to the following equation:

$$-\mathbf{p}_c(t)\mathbf{c}(t) = \int_t^\infty \dot{\mathbf{p}}_c(\tau)\mathbf{c}(\tau)d\tau + \int_t^\infty \mathbf{p}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau\,.$$

By rearranging this equality we obtain

$$\underbrace{\int_t^\infty \big(-\dot{\mathbf{p}}_c(\tau)\big)\mathbf{c}(\tau)d\tau}_{\text{National income}} \quad = \quad \mathbf{p}_c(t)\mathbf{c}(t) \quad + \quad \underbrace{\int_t^\infty \mathbf{p}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau}_{\text{National savings}}\,. \qquad (1)$$

Here, we will interpret the l.h.s. as national income at time t and the second term on the r.h.s. as national savings at time t. As we will argue next, these interpretations can be supported in both a welfare and a productive perspective.

In line with Samuelson (1961, pp. 51–52), one can argue that $\int_t^\infty \mathbf{p}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau$ measures welfare improvement in a market economy following an optimal path. A precise and more general statement of this result is proven in Appendix A of Asheim and Wei (2009, Proposition 4). In particular, we need not assume that the dynamic welfare is discounted utilitarian. Moreover, by allowing for the possibility that the prices are calculated, we need not assume that the economy implements a welfare maximizing path of consumption flows through an intertemporal market equilibrium.

Thus, Proposition 4 of Asheim and Wei (2009) gives a welfare foundation for interpreting $\int_t^\infty \mathbf{p}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau$ as national savings. Then, if national income is to serve as a guide for prudent conduct in the sense that dynamic welfare improves if and only if national consumption is smaller than national income, we obtain that national income equals $\mathbf{p}_c(t)\mathbf{c}(t) + \int_t^\infty \mathbf{p}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau$, which by (1) can be transformed to $\int_t^\infty \big(-\dot{\mathbf{p}}_c(\tau)\big)\mathbf{c}(\tau)d\tau$.

If an economy implements a path with constant instantaneous well-being and the vector of consumption prices $\mathbf{p}_c(t)$ is at any time proportional to the contributions that the various consumption flows make to instantaneous well-being, then it follows that $\mathbf{p}_c(t)\dot{\mathbf{c}}(t) = 0$ at all times. Hence, national income equals the value of consumption and shows that this concept of income serves as a guide for prudent conduct also in this special case.

Under the assumptions of the technology being stationary and the economy realizing a competitive equilibrium, then it follows from Dixit, Hammond and Hoel (1980, proof of Theorem 1) that

$$\mathbf{p}_c(t)\dot{\mathbf{c}}(t) + \frac{d}{dt}\left(\mathbf{p}_k(t)\dot{\mathbf{k}}(t)\right) = 0 ,$$

where $\{\mathbf{k}(t)\}_{t=0}^{\infty}$ is the path of the vector of capital stocks in this economy and $\{\mathbf{p}_k(t)\}_{t=0}^{\infty}$ is the corresponding path of market (or calculated) present value prices of net investment flows.

Integrating on both sides under the assumption that $\mathbf{p}_k(\tau)\mathbf{k}(\tau) \to 0 \text{ as } \tau \to \infty$, entails that the following equation holds for all t:

$$\int_t^{\infty} \mathbf{p}_c(\tau)\dot{\mathbf{c}}(\tau)dt = \mathbf{p}_k(t)\dot{\mathbf{k}}(t)$$

Combined with (1) we obtain:

$$\underbrace{\int_t^{\infty} \left(-\dot{\mathbf{p}}_c(\tau)\right)\mathbf{c}(\tau)d\tau}_{\text{National income}} = \underbrace{\mathbf{p}_c(t)\mathbf{c}(t) + \mathbf{p}_k(t)\dot{\mathbf{k}}(t)}_{\text{Net national product}} \quad (2)$$

Hence, national income as defined through (1) equals net national product under the assumptions of the technology being stationary and the economy realizing a competitive equilibrium.

A precise and more general statement of the result that the value of the net investment flows equals the present value of future consumption changes is proven in Appendix A of Asheim and Wei (2009, Proposition 5). In particular, one need not assume that the economy implements a competitive equilibrium. By allowing for the possibility that the prices are calculated, it is sufficient that the path of consumption flows and capital stocks is implemented by a stationary resource allocation mechanism (as introduced by Dasgupta and Maler, 2000; Dasgupta, 2001; Arrow et al., 2003).

Example: Cake-eating economy. It is instructive to illustrate this definition of national income in the setting of a cake-eating economy, faced with the problem

$$\max_{\{c(t)\}_{t=0}^{\infty}} \int_0^{\infty} e^{-\rho t}u(c(t))dt \quad \text{s.t.} \quad \int_0^{\infty} c(t)dt \leq S(0) \ \text{ and } \ c(t) \geq 0 \ \text{ for all } t \geq 0$$

for some twice differentiable and strictly concave utility function $u : [0, \infty) \to \mathbb{R}$ satisfying $\lim_{c \to 0} u'(c) = \infty$, utility discount rate $\rho > 0$ and initial cake $S(0) > 0$.

The optimal path, $\{c(t)\}_{t=0}^{\infty}$, is differentiable and satisfies

$$p_c(t) := e^{-\rho t}u'(c(t)) = u'(c(0)) \ \text{ for all } t \geq 0 .$$

Since $\dot{p}_c(t) = 0$ for all t ≥ 0, national income at each time t equals zero:

$$\int_t^{\infty} \left(-\dot{p}_c(\tau)\right) c(\tau)d\tau = 0 .$$

Moreover, it follows from (1) that the positive value of consumption at each time t exactly cancels the negative present value of the future consumption changes, the latter term measuring the change in dynamic welfare as the remaining cake vanishes:

$$p_c(t)c(t) + \int_t^{\infty} p_c(\tau)\dot{c}(\tau)d\tau = p_c(t)c(t) + p_c(t)\int_t^{\infty} \dot{c}(\tau)d\tau = p(t)\left(c(t) - c(t)\right) = 0$$

since $p_c(\tau) = p_c(t)$ for all $\tau \geq t$ and $\lim_{\tau\to\infty} c(\tau) = 0$.

4. Expressions for real national income

To find real (rather than present value) prices, consider the Divisia consumer price index $\{\pi(t)\}_{t=0}^{\infty}$ defined by $\pi(0) = 1$ and

$$\frac{\dot{\pi}(t)}{\pi(t)} = \frac{\dot{\mathbf{p}}_c(t)\mathbf{c}(t)}{\mathbf{p}_c(t)\mathbf{c}(t)} \quad (3)$$

for all t ≥ 0. Define the path of market (or calculated) real prices of consumption $\{\mathbf{P}_c(t)\}_{t=0}^{\infty}$ by

$$\mathbf{P}_c(t) = \mathbf{p}_c(t)/\pi(t) \quad (4)$$

for all t ≥ 0. Define the path of market (or calculated) real consumption interest rates $\{R(t)\}_{t=0}^{\infty}$ by

$$R(t) = -\dot{\pi}(t)/\pi(t) \quad (5)$$

for all t ≥ 0. Then, by applying (3) – (5),

$$\big(- \dot{\mathbf{p}}_c(t)\big)\mathbf{c}(t) = -\frac{\dot{\pi}(t)}{\pi(t)}\mathbf{p}_c(t)\mathbf{c}(t) = \pi(t)R(t)\mathbf{P}_c(t)\mathbf{c}(t) \quad . \quad (6)$$

Hence, it follows from (1) that real national income, $\int_t^{\infty} \big(- \dot{\mathbf{p}}_c(\tau)\big)\mathbf{c}(\tau)d\tau/\pi(t)$, is equal to the present value of real interest on future national consumption, as stated in the following definition.

Definition 1 Real national income at time t is determined as

$$Y(t) := \int_t^{\infty} \frac{\pi(\tau)}{\pi(t)} R(\tau)\mathbf{P}_c(\tau)\mathbf{c}(\tau)d\tau \quad .$$

By using (4) in (1), we can express real national income as the sum of current real national consumption and the real national savings, as stated in Proposition 1 below. Furthermore, by differentiating Y (t) w.r.t. t, we obtain as the second part of the proposition that $\dot{Y}(t) \geq 0$ is equivalent to $\mathbf{P}_c(t)\mathbf{c}(t) \leq Y(t)$ and $\int_t^{\infty} (\pi(\tau)/\pi(t))\,\mathbf{P}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau \geq 0$ if the real interest rate R(t) is positive; hence, $\dot{Y}(t) \geq 0$ can serve as an alternative guide for prudent behaviour.

Proposition 1 Real national income at time t can be expressed as

$$Y(t) = \mathbf{P}_c(t)\mathbf{c}(t) + \int_t^{\infty} \frac{\pi(\tau)}{\pi(t)}\mathbf{P}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau \quad .$$

Furthermore,

$$\dot{Y}(t) = R(t)\left(Y(t) - \mathbf{P}_c(t)\mathbf{c}(t)\right) = R(t)\left(\int_t^{\infty} \frac{\pi(\tau)}{\pi(t)}\mathbf{P}_c(\tau)\dot{\mathbf{c}}(\tau)d\tau\right).$$

Example: Cake-eating economy (continued). In the case of the cake-eating economy introduced in Section 3, $p_c(t) = p_c(0)$ for all t ≥ 0. It follows from $\pi(0) = 1$ and (3) that

$\pi(t) = 1$ for all t ≥ 0. Furthermore, by (5), R(t) = 0 for all t ≥ 0. Hence, by applying Definition 1, we obtain that real national income, Y (t), in a cake-eating economy equals zero for all t. Furthermore, since the real interest rate equals zero for all t, Y (t) ≥ 0 cannot serve as a guide for prudent behaviour. This is caused by the fact that a cake-eating economy has only one asset, the "cake", which is unproductive as a stock. In the respect, a cake-eating economy represents an extreme case, which corresponds neither to the models that economists usually analyse nor to real economies.

Definition 1 and Proposition 1 yield expressions for income that can be used at a national level also if the technology is not stationary, and it also facilitates the definition and expression of income for individuals and in different sectors of a national economy. Such definitions are given by Sefton and Weale (2006) and Asheim and Wei (2009).

5. Concluding remarks

In this note I have presented a definition of national income which is compatible with an important line of welfare-based theory of comprehensive national accounting in the tradition of Hicks (1946), Samuelson (1961), Weitzman (1976) and Sefton and Weale (2006). The definition yields a concept of national income which

- is a guide to prudent behaviour in the sense that dynamic welfare improves if and only if consumption is less than national income, and
- equals net national product in a closed economy with a stationary technology.

This concept of national income can easily be decomposed into individual and sectoral income such that individual income summed over all individuals and sectoral income summed over all sectors add up to national income.

As noted in Section 2, one need not require that a discounted utilitarian welfare function is maximized. Rather, the formal analysis builds on the assumption that the economy's actual decisions are taken according to a resource allocation mechanism (as introduced by Dasgupta and Maler, 2000, Dasgupta, 2001; Arrow et al., 2003). The resource allocation mechanism is allowed to be inefficient, due to, e.g., externalities, monopolistic competition, or distortionary taxation. This is relevant in a world facing serious environmental problems caused by uninternalized externalities. In particular how is the income of an economy's petroleum sector affected by the fact that both petroleum extraction and petroleum use cause greenhouse gas emissions? It is not trivial to calculate the relevant accounting prices under such conditions. Guidelines for practical calculation of accounting prices are outside the scope of the present note. Arrow et al. (2003) discuss problems that arise within such a framework and are a useful reference.

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4. Exchange values, consumer surplus, avoided cost and ecological liabilities – A real income compatible "green-box" would strengthen policy relevance of ecosystem accounting[1]

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Abstract

Political decisions like the Aichi Biodiversity Targets of the CBD (CBD 2010: Target 2) or the European Biodiversity Strategy (EU 2011: Target 2, action5) are asking for the integration of ecosystems and ecosystem services into accounting. Behind this is the idea that information on ecosystems in national accounts could improve decision making by continuously delivering data on the value of ecosystem services, natural assets, degradation etc.

The notion that ecosystem accounting could improve knowledge for an improved policy on ecosystems, regarding their protection, maintenance and restoration, may result from studies on the economic impacts of ecosystem degradation on welfare such as the TEEB-study and from proposals to report these impacts on a regular basis in the accounting system (TEEB (2011: Ch. 10, p. 11)

However, accounting for ecosystem services with the standard accounting methods that are based on exchange values (Obst et al. 2016: 5) result in ambiguous information that can hardly be used in political communication. Ecosystem service losses can have a positive effect on the overall value of the services, service increases can lower their value. Whether the effect is positive or negative depends on the real or simulated price elasticity of service demand. It is quite easy to understand that a message drawn from accounting data like "Due to continued political efforts the value of ecosystem services fell dramatically" although economically absolutely reasonable, for reducing scarcity lets prices drop, is completely inappropriate for a targeted political communication to enhance the protection of our natural capital.

Furthermore, a major part of the data on the economic effects and the relevance of ecosystem degradation for the economy and well-being consist of consumer surplus changes or deals with changes of production cost. In standard accounting procedures consumer surplus, however, has to be disregarded and cost changes have no direct effect on nominal GDP. Therefore, welfare economists, who have done the most work on ecosystem service valuation so far, still see some problems to deliver data on ecosystem service values that are compatible with accounting rules on the one hand and, after integration into the accounts, still show relevance for political decision makers on the other hand.

This paper wants to increase awareness that there is already a feasible way to bridge political demand, welfare economic methods and accounting procedures. This solution would require a stronger focus on the effects of ecosystem services, degradation and restoration

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on the change of real (deflated) GDP. Due to the efforts of welfare economy to express welfare changes in monetary terms, as changes of income and accounting to make price indexes consistent with underlying utility functions, there is a strong functional similarity between the application of price indexes and the calculation of welfare change (s. e.g. Hicks 1939a, b, Diewert 1976, Asheim & Weitzman (2001).

On the basis of this relationship a “green-box” as an addition to the common exchange-value-based accounts is proposed, where consumer surplus and cost-based values for ecosystem services, following concepts of “defensive expenditure”, “avoided cost” or “ecological liabilities” (Leipert & Pulselli 2008, Vanoli 2015) can be presented comparable to real national income.

Introduction

The System of National Accounts (SNA) is in particular designed for documenting the level and structure of economic activities in a country, which is, in short, the input of labour and capital in the different sectors of the economy. It is therefore of fundamental importance for policies regarding growth, counter-cyclical macroeconomic policy and structural changes and imbalances. Although strong references already exist, its prior goal is not to account for welfare.

The System of Environmental Economic Accounting and the guidelines for Experimental Ecosystem Accounts (United Nations 2014) are traditionally developed in a way very close to the SNA, though their prior aim is quite different. The SEEA broadens the system by including the environment as a supplementary asset and shall gather and provide data that assist policy makers to tackle problems of non-optimal use of resources that cause welfare loss due to environmental degradation. Such problems are usually analysed by using welfare economic methods and not with macroeconomic concepts like those underlying SNA. However, tackling problems of resource allocation in an SNA-like framework works well, as long as the environmental problems in focus can be analysed on the basis of a comparison of costs. Taking market prices (transaction values), which are at the heart of the value concept of SNA, are, in most cases, the best method to capture the value of resources that are used for one or the other activity. Therefore, for instance, it makes sense to compare the additional cost to reduce air emissions with thereby reduced health care cost on the basis of market prices including those for labour and capital. The attempt to document effects of political decisions in the accounts in a way that provides policy with additional information to decide whether it performed right or wrong, however, finds its limitation as soon as the decision is not between costs but between different consumer goods, such as the additional amenity, recreational and nature values of agricultural landscapes after enrichment with trees, hedgerows and flowering strips versus the loss of agricultural production or its negative effect on income. In such cases, it is likely that the restriction to exchange values will create the risk of ambiguous and misleading messages.

The authors of the TEEB-study (TEEB 2011: Ch. 10, p. 11) and subsequent political initiatives (Aichi Biodiversity Target 2; EU-Biodiversity Strategy 2020, target 2, action 5) ask for integrating ecosystem services into the accounts, obviously hoping to hereby strengthen efforts towards a more effective conservation of natural capital.

However, a growth in natural services does not need to have positive effects on the exchange values reported in the accounts. Just the opposite can happen: An increase in natural capital reduces the scarcity of the ecosystem services it delivers. More services and less scarcity let people´s desire for any additional service fall. In market simulations these decreasing marginal values are taken as proxies for the price paid, if services would be sold. Whenever the price reduction is stronger than the increase in quantity, the market value of the total quantity of goods decreases although volume rises. Likewise, a decrease of services can give rise to their overall value. Both happens as soon as the price elasticity of the demand for ecosystem services – the ratio between the percentage (marginal) change of demand and the percentage (marginal) change of the price, either estimated on market reaction or simulated on the basis of hypothetical markets, is between - 1 and 0, called (relatively) “inelastic” in economic terms.

So, the above quoted hypothetical message on falling ecosystem service values due to increased conservation and restoration efforts is far from being nonsense. It makes sense for those who know about the effects of using standard accounting methods. But there should be some doubt, of course, whether such data and statements, would be helpful in a broad political communication on the benefits of halting the loss of natural capital and reaching environmental goals by rehabilitating ecosystems.

Things would be much easier to understand and communicate if the accounts would also deliver numbers on the contribution of natural capital to welfare. Welfare growth is always positive or at least zero, but never falls when the level of ecosystem services rises. In contradiction to the figures generated with the common, exchange value-based accounting rules, welfare values would represent a simple story that may not only be understandable by experts: every increase of the level of ecosystem services would – ceteris paribus – be connected with a rise of welfare and the other way around. Therefore a “green-box” should be established that presents welfare values on ecosystem services and other selected environmental issues in addition but in close connection to the common accounting procedures.

There seems to be, somehow, a lack of awareness in both scientific communities, in the accounting community as well as among environmental economists, that the accounts already include an anchor for such a “green-box”. This anchor is real national income or – in other words – deflated income. The efforts to develop monetary welfare measures commensurable with income change, on the one hand, and price indexes compatible with underlying utility functions on the other hand, led to converging concepts in both the welfare economics and the accounting community. In 2001 Asheim & Weitzman published an article, where they could prove in a highly sophisticated and elegant way “that changes in real Net National Income mirror accurately changes in dynamic welfare” (Asheim & Weitzman 2001: 233) if a Divisia price deflator is applied, that is (theoretically) based on continuous-time series for prices and quantities.

High correspondences between measures of welfare and real income change can be demonstrated not only for a Divisia index but also for indexes that are in practical use. This can help to extend the accounts in a deliberate way to obtain higher relevance for the discussion on environmental issues such as the benefits of ecosystem services and natural capital and – in a broader context – also for other non-market activities.

In the following, after pointing out the general differences between accounting and welfare based valuation, the high similarity – and also the deviations – between nominal accounting plus deflation on the one hand and welfare calculations on the other hand are demonstrated in quantitative terms with the help of a simple two sector model of the economy, one being a market good and the other being either a market good as well or an environmental consumer good (e. g. an ecosystem service that can be handled like a consumer good).

Comparative calculations between the effects of deflating and welfare calculations are made for different assumptions on the relative size of the environmental good, differences in the amount of changes, different price-elasticities and inflation- and growth-rates. As deflators the Laspeyres, the Paasche and the Fisher index are used. An extension to a three-sector model showed no changes in the results.

Building on this, a proposal for a “green-box” within the accounts is developed that includes welfare-based information on the value of ecosystem service change and can integrate also other concepts like “defensive expenditures” and “unpaid ecological cost” that were developed as alternatives to the standard accounts and would also deliver political relevant information in comparison to real GDP.

Differences between accounting and welfare calculations

The “market” for an environmental consumer good is shown in a stylized way in fig. 1. In order to avoid being purely theoretical and with reference to a recent study (Krekel et al. 2016) the hypothetical market for the ecosystem services of urban parks is taken here as an example. In the mentioned study, the quantity of the service was not measured in park-visits or hours spent in parks but in the area of the parks within a radius of 1km around the home. The price was assessed with the life-satisfaction approach, which measures the willingness to pay for an additional amount of park space on the basis of two statistical relations: (1) park space and life satisfaction, and (2) income and life-satisfaction. With this approach not only the recreational use value of parks is measured but also the option and the amenity value. The hypothetical price elasticity of demand for additional park space from citizens with an average park supply and average income was found to be - 0,46. This means that the hypothetical transaction value of parks is likely to fall if park space rises.

Presentations as in fig. 1 and underlying analyses can be made in a similar way for all ecosystem services that are consumer goods, including for instance landscape amenities, recreational fishing, deer hunting etc.

$\overline{AB}$ is the estimated demand curve for urban green. $\overline{CD}$ is the long-term supply curve of urban green, that shows the capital and maintenance cost per unit area. $\overline{EF}$ is the current supply with urban green. In this case it is a vertical line, but the analysis would also work with an inclined supply curve. The price for an additional ha of urban green that citizens would pay under the current conditions would then be “p”. The marginal cost including capital cost of a municipality for supplying with an additional ha of park space is “c”. If the park ecosystems would act as producers on a competitive market than they could get the price “p” from each citizen for the bundle of services (e.g. shadow spent on a hot summer day by a fully grown tree) if they would be able to exclude non-paying consumers from the benefits the parks produce.

The ecosystem park can only exist with all its services with an input from society. The marginal value of these inputs is “c”, and its complete costs are ☐ CIEO. If these municipality inputs would be provided by a market under competitive conditions, then the sales of these inputs would have the higher value ☐ JIEO. If the parks could sell their services and had to pay the municipality for the inputs then ☐ GHIJ or ☐ GHIC, respectively, would be the producer surplus of the parks.

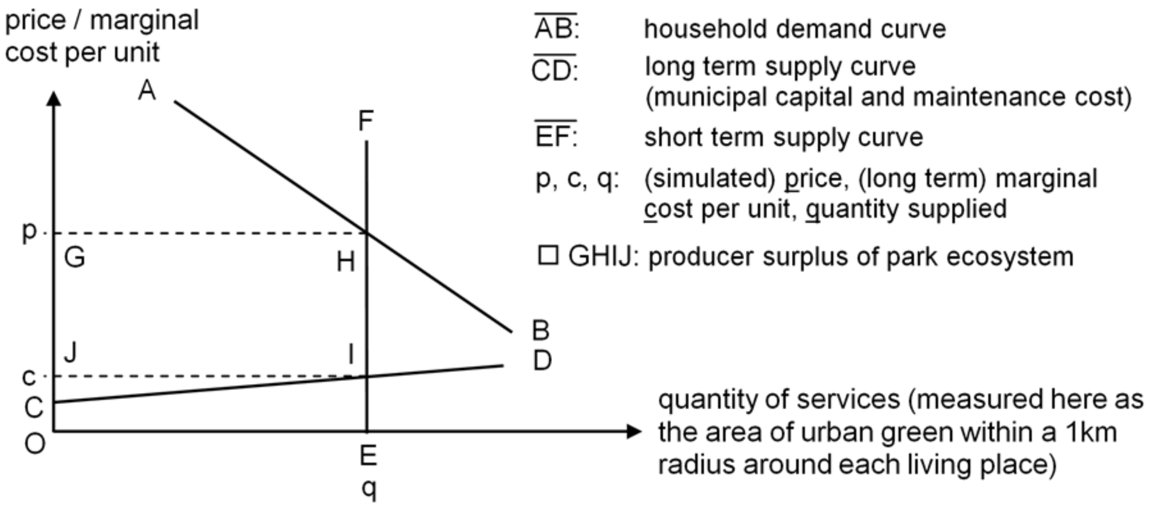

Figure 1. The "market" for an environmental consumer good, here: ecosystem services of urban green (Own illustration)

Figure 2 shows the situation after a rise of ecosystem capital that leads to the higher supply $\overline{E'F'}$. In this case, the hypothetical price for a service unit drops from p_0 to p_1. A proxy for the welfare-effect of this change of quantity is □ HH'I'L. It consists of the elements ∆ HH'K and □ KH'I'L which are the sum of the changes of consumer and producer surpluses (+ □ GHH'G' + □ KH'I'I - □ GHKG' - □ J'LIJ'). In accounting, however, only the change of the simulated producer surplus (+□ KH'I'L - □ GHKG' - □ J'LIJ') can be taken as a further element of the new nominal national income. The difference between the two measures is □ GHH'G'. This part of welfare gain gets lost in the nominal accounts that are based on exchange values only.

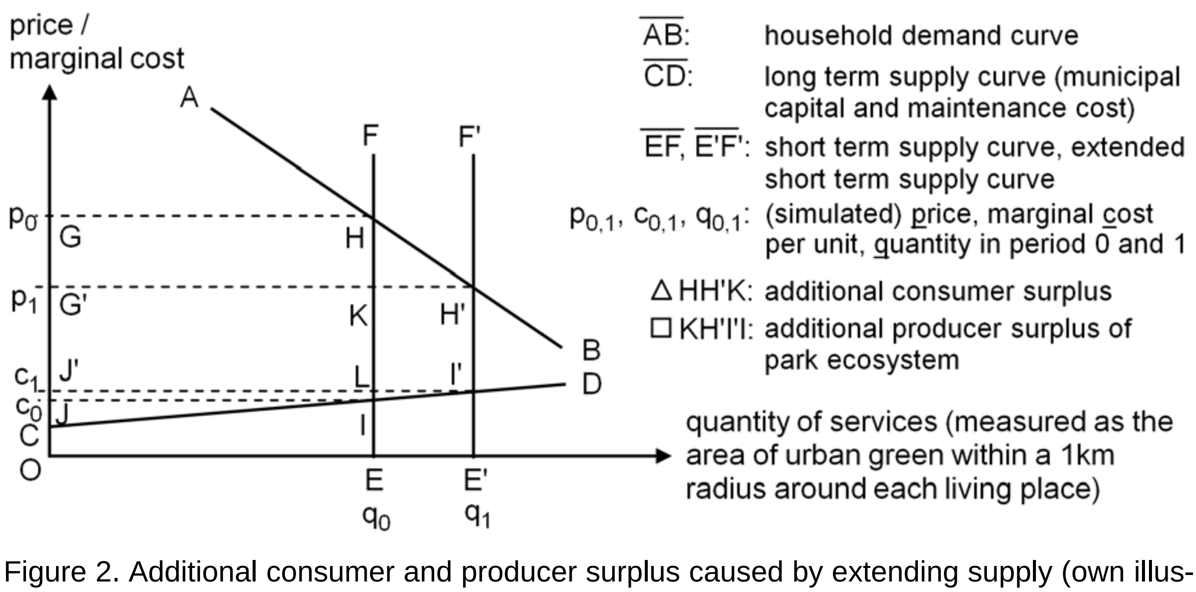

Figure 2. Additional consumer and producer surplus caused by extending supply (own illustration)

The additional cost for municipality inputs □ II'EE' are of course another factor that contributes to national income. They are, however, produced with factors that were removed from other parts of the economy. So, the production of these inputs is outweighed by production losses elsewhere.

Whenever a producer surplus of ecosystems is detected, which would be the case also when standard accounting methods are applied, this is a clear signal for policy that there is additional scope for increasing welfare by further extending the supply with ecosystem services. But this signal might be not enough to satisfy the aspiration of welfare economists that all the values they detect, are taken notice of. And, in fact, the difference between the two measures, welfare based and accounting values, is quite evident.

Comparison of consumer surplus change and change of deflated income

Campos & Caparrós (2006) and Caparrós et al. (2017) show that the relation between the welfare value of a certain ecosystem service supply and the correspondent transfer value depends on the assumptions about the underlying hypothetical market: competitive, monopolistic etc. (also called “institutional settings”). The differences are relatively highest if a competitive market is assumed, and lower if the price simulation is based on monopolistic competition with differentiated goods or a complete monopoly. The deviation vanishes completely when the monopolist is able to turn the whole consumer surplus into his own profit by perfect price discrimination.

However, there are no clear criteria for deciding what kind of the above mentioned institutional settings should be applied. Countervailing effects in terms of welfare values and transaction values and questionable assumptions on the distribution of net benefits complicate the decision. Welfare values would be highest and transaction values lowest, if an ecosystem service volume would be supplied that would also be provided by perfect competition. If a monopoly is assumed, the supply would fall under the optimum level. Transaction values would be higher than under perfect competition, but the effect on welfare is lower. The monopoly option would imply a non-optimal behavior of the government if the service is a public good. If perfect price discrimination is assumed, a change in ecosystem service supply would affect a series of individual prices. All individual market value changes would enter the accounts so that transaction value changes and welfare effect would have the same amount. However, perfect price discrimination means that the net benefit of a service to the consumer is zero, which is opposite to reality for a public good.

Instead of looking for the institutional setting that could perhaps be the most appropriate one or that delivers hypothetical prices that are near to welfare values, the present text argues in a different way. It tries to show that both accounting and micro-economy have already developed concepts that produce comparable data. In accounting this is the real GDP and in microeconomics it is compensating and equivalent variation, which can be approximated by consumer surplus (s. among others Levin & Milgrom 2004: 29).

Indeed, the differences between accounting measures and welfare values pointed out above, diminish or at least shrink when changes in welfare values are compared with real (deflated) GDP instead of nominal GDP. This is shown, in the following, on the ground of common deflators with a simple two sector model. In this model “R” is the sales value of the “rest of the economy” while “p” and “q” are respectively the price and the quantity of a certain good that is assumed to be the ecosystem service; that could, however, also be a market good.

Changes in production cost as in fig. 2 are not taken into account here in order to simplify the calculations. That means that the ecosystem service does not need any further inputs by the economy (s. fig. 3). This assumption, however, does not weaken the validity of the following conclusions. More complex calculations including input cost would come to the same results.

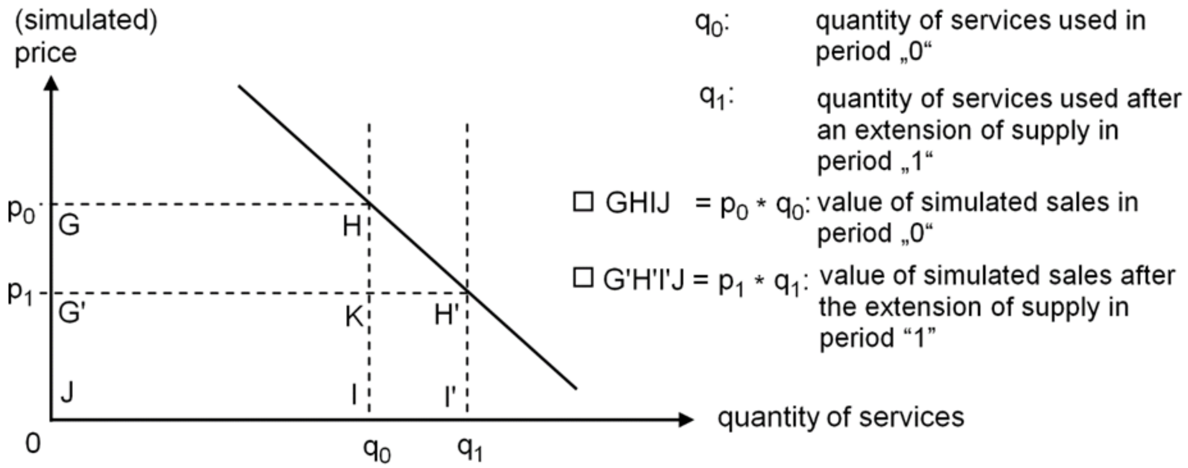

Figure 3. Simplified market for ecosystem services (own illustration)

The general structure of the following analysis is shown in fig. 4. It is assumed that within one year, from t_0 to t_1 the rest of the economy “R_0” changes with a real growth rate “gr” and an inflation rate “ir”. The ecosystems services rise, in the special case, without any additional human intervention (e. g. by growing trees spending additional shadow and cooling of local temperatures) from q_0 to q_1.

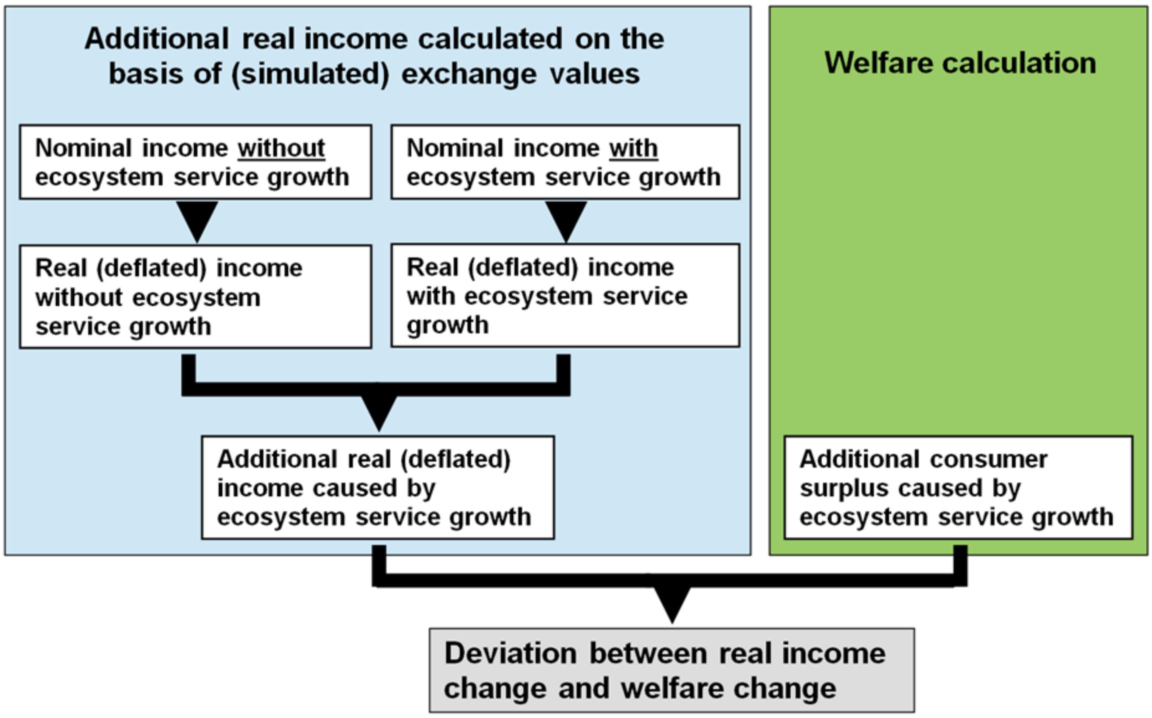

Figure 4. Flow chart of the comparison between real income change and additional consumer surplus caused by ecosystem service growth (own illustration)

The formula for the national income (Y) in t_0 then is: $Y_0 = p_0 * q_0 + R_0$

If ecosystem services quantity in national income had not changed (“Y_{SU}”), we find Y in t_1 as:

$Y_{SU_1} = p_0 * q_0 + R_0 * (1+ir+gr)$. If they have changed, Y in t_1 is:

$Y_1 = p_1 * q_1 + R_0 * (1+ir+gr)$

The Laspeyres price index for t_1 with the basis year t_0 is calculated under these conditions as:

$P_{LSU} = \big((p_0 * q_0) + (1+ir) * R_0\big) / \big((p_0 * q_0) + R_0\big)$ and

$P_L = \big((p_1 * q_0) + (1+ir) * R_0\big) / \big((p_0 * q_0) + R_0\big)$ respectively.

The rise in real (deflated) national income (Y_r) from t_0 to t_1, caused by the increase of eco-system services then is:

$\Delta Y_r = Y_1 / P_L – Y_{SU_1} / P_{LSU}$

The (deflated) welfare change due to rising ecosystem services (ΔW) between t_0 and t_1, calculated as the change of (deflated) consumer surplus (☐ HH'I'I) is:

$\Delta W = p_1 * (q_1 – q_0) + (p_0 – p_1) * (q_1 – q_0) / 2 / (1+ir)$

The last formula is valid only for linear demand curves. Non-linear curves, however, can be approximated by a sequence of linear elements. For a continuous, yearly, step to step measurement as applied in the accounts the above formula should therefore be a good proxy.

The welfare change (ΔW) is then compared with the corresponding change of real income (ΔY_r) by calculating the percentage deviation (D) of deflated income change from deflated welfare change:

$D = \big[(\Delta Y_r / \Delta W) – 1\big] * 100$

This deviation can range from “0” (no differences) to ± ∞. A value of + 100 means that real income change is twice the welfare change; - 50 means that welfare change is twice the change of real income.

The deviation “D” was calculated for different conditions regarding:

- the relative size of the ecosystem service sector: $(q_0 * p_0) / (q_0 * p_0 + R_0) * 100$ = 1 %; 10 %; 20 %
- the price elasticity of (simulated) demand for the ecosystem service: - 0.11; - 0.12; - 0.15; - 0.2; - 0.3; - 1; - 4 (- 1 meaning that demand increases with x % if the price falls at the same percentage, s. also fig. 5)
- the extent of increase of the ecosystem service: 1 %, 5 %, 10 %, 25 %
- the inflation and growth-rate of the “Rest of the economy” (R_0): both 1 %, 2 %, 4 %.

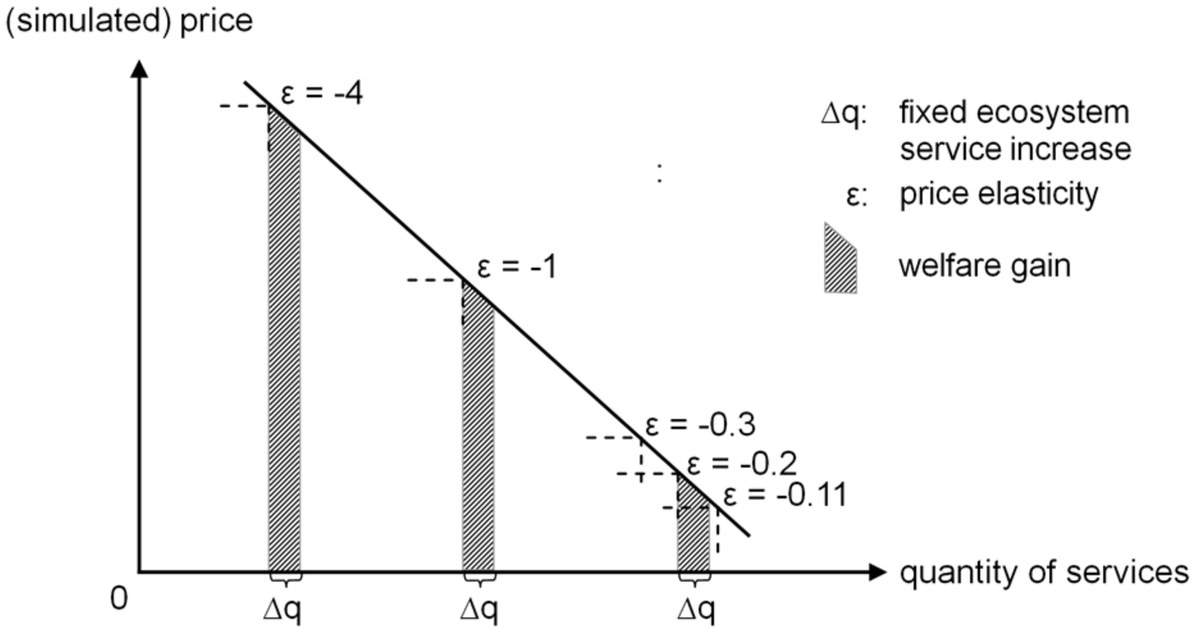

Figure 5. Welfare gain of a fixed ecosystem service increase in different (simulated) market conditions with varying price elasticities (own illustration)

Additionally, it was tested, whether the number of sectors has any influence on the results. The calculations were made for the Fisher, the Laspeyres and the Paasche index (s. Infobox 1). It is well known that the Laspeyres index overestimates inflation and by that underestimates real income. With the Paasche index it is just the opposite (Schultze 2003). The Fisher index is the geometric mean between Laspeyres and Paasche and by that tries to avoid over- and underestimation. Within the three mentioned, the Fisher index is the one which is closest to a Divisia price index.

Infobox 1. Price Indices

The Laspeyres and Paasche indices (PL, PP) are the weighted sums of the ratios between the current price (pi,t1) and the price in the reference period (pi,t0). For the Laspeyres index, the weighting factor is the share of sectoral sales in total sales in the reference period. For the Paasche index, the quantities of the current period (qi,t1) are used for the calculation of the weighting factor instead of the quantities of the reference period (qi,t0).
The Fisher Index is the geometric mean between the Laspeyres and the Paasche Index.

$$P_L = \sum_{i=1}^{n} \left(\frac{p_{i,t_1}}{p_{i,t_0}} \cdot \frac{p_{i,t_0} \cdot q_{i,t_0}}{\sum_{j=1}^{n} (p_{j,t_0} \cdot q_{j,t_0})} \right) = \frac{\sum_{i=1}^{n} (p_{i,t_1} \cdot q_{i,t_0})}{\sum_{j=1}^{n} (p_{j,t_0} \cdot q_{j,t_0})}$$

$$P_P = \sum_{i=1}^{n} \left(\frac{p_{i,t_1}}{p_{i,t_0}} \cdot \frac{p_{i,t_0} \cdot q_{i,t_1}}{\sum_{j=1}^{n} (p_{j,t_0} \cdot q_{j,t_1})} \right) = \frac{\sum_{j=1}^{n} (p_{i,t_1} \cdot q_{i,t_1})}{\sum_{j=1}^{n} (p_{j,t_0} \cdot q_{j,t_1})}$$

$$P_F = \sqrt{P_L \cdot P_P}$$

Results

Figure 6 shows the range of deviations between welfare change and the change of deflated income for all analysed cases. The largest deviations, ranging from - 90 % to + 108 % occur when growth rates of 4 % are combined with an inflation rate of 1 % and vice versa and the quantity of ecosystem services is increased by 25 %. If a policy measure is very successful, it could perhaps happen that for some ecosystem services an increase of 10 % can be managed during one year. For a 10 % increase the maximum deviations fall to - 66.5 % and 90.7 % respectively. Normally high growth is combined with higher inflation and vice versa. Looking at the economic development in the last decades, the combination of a growth rate of 2 % with the same inflation rate is beyond all calculated cases the most probable one, which, combined with the 10 % increase, leads to deviations between - 49.9 % and + 87 %.

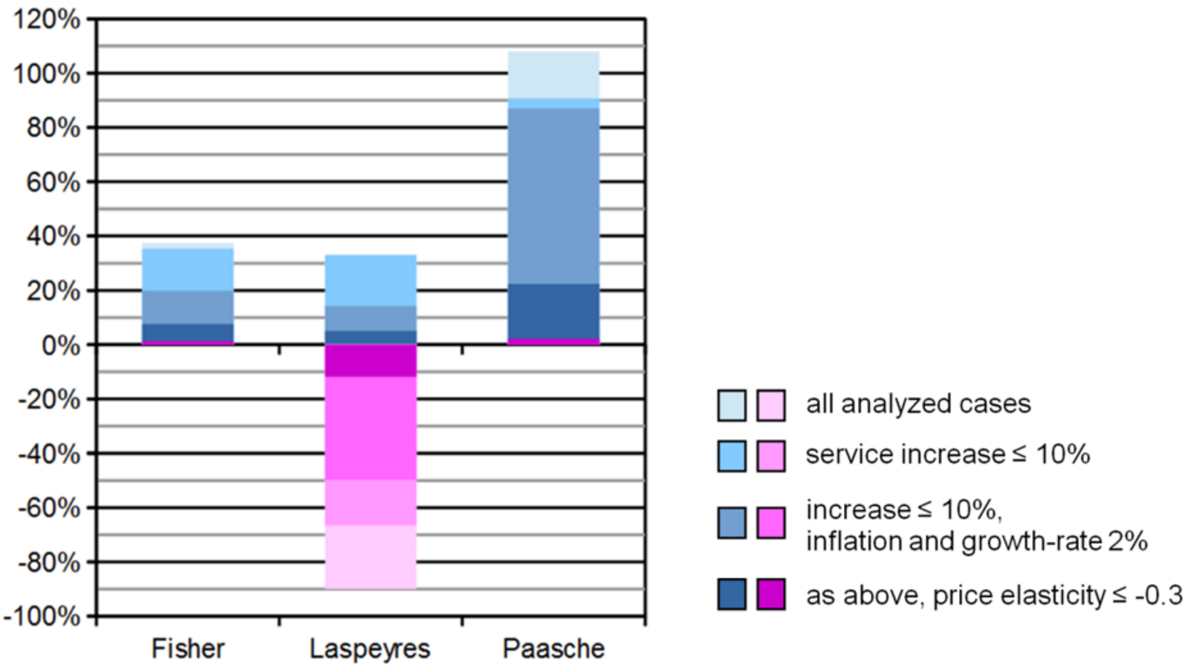

Figure 6. Minimum (pink) and maximum (blue) deviations of real income change from welfare increase (Own calculations)

Furthermore, it should be mentioned that high deviations correlate with high price elasticities (s. fig. 7). High price elasticities mean that demand is near to the saturation point. Welfare gains from a further expansion of ecosystem service supply are then relatively small (s. fig. 5), so that taking one measure (consumer surplus gain) or the other (additional real GDP) does not matter that much for making decisions to optimize welfare. If price elasticities be-tween - 4 and - 0.3 are taken into account only, deviations vary only between - 11.8 % and + 22.5 %.

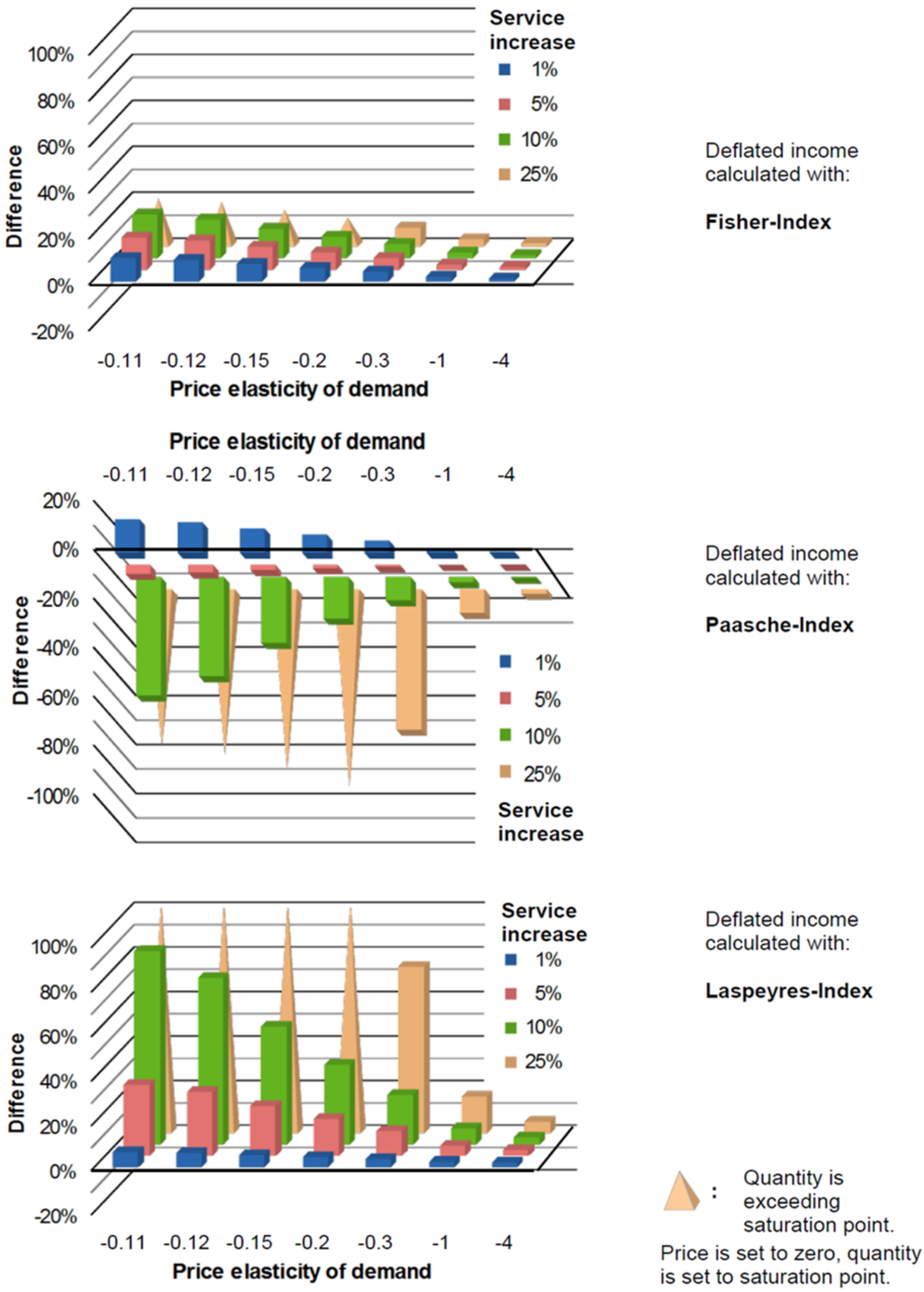

Figure 7. Difference ("D") between a welfare-based valuation of an ecosystem service increase and the correspondent change of real income (Inflation- and growth-rate 2 %), size of ecosystem-service sector 10 % (Own calculations)

The high deviations mentioned above appear only, of course, if deflated income is calculated with indexes that are over- or underestimating inflation or real income respectively: Laspeyres- and Paasche index. When using the Fisher index deviations are far smaller, reaching from 1 % to + 37.4 % for all cases and from + 1.3 % to + 7.6 % if taking into account only the most probable ones (max. 10 % increase; inflation and growth rate both 2 %) and price elasticity is ≤ - 0.3.

Also, the direction of the deviations depends on the kind of index that is applied. Calculations with the Paasche index show an increase of deflated income that is always higher than the welfare increase. If the Fisher index is used welfare effects are also overestimated by deflated income but to a less extent. The Laspeyres index shows a mixed picture (s. fig. 6 and 7). If the increase of the service is small and simulated prices fall considerably (price elasticity > - 1) welfare increase is overestimated. In the other cases it is underestimated.

Generally, for all deflators deviations are smaller if service increase and price elasticity are small, and vice versa (s. fig. 7). Deviations from this rule occur for the Laspeyres index when the further reduction of ecosystem service increase has the effect of switching deviations from negative to positive.

It is interesting that the share of ecosystem services in overall income does only have a marginal impact on the results (s. fig. 8). The introduction of a third good did not change the results, so that they are valid also for a multi sector economy.

To avoid any kind of misunderstanding, it should be mentioned here that the real income change induced by an additional ecosystem service supply, calculated in the way above, is quite different from the change of ecosystem service supply calculated in volume terms. To get from nominal change to a change in volume terms, the nominal change is usually deflated with an aggregate deflator, e. g. the consumer price index.

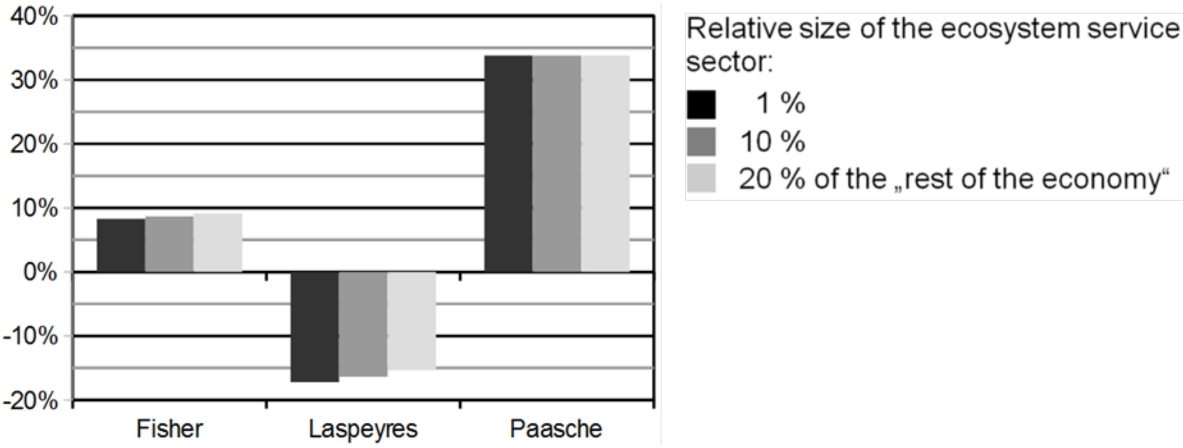

Figure 8. Mean of calculated deviations between deflated income change and welfare increase for different extents of the ecosystem service sector (own calculations)

Nominal change and change in terms of volume both express the contribution of a sector or a good to the activity level of an economy (factor income), the one in current prices, the other reduced by the inflation rate, whereas the contribution to real income change, calculated here with the help of the with-and-without approach shown in fig. 4, expresses the contribution to real income as a welfare indicator. Infobox 2 illustrates this in numbers using a two-sector example.

Infobox 2. Difference between change in terms of volume and contribution to real income as a welfare indicator

Table 1 and fig. 9 show on the basis of a numerical two-sector example, that the increase in terms of volume and the contribution of a sector to real income increase, calculated using the with-and-without approach (s. fig. 4), both add up to the same number (s. Table 1, right column). However, the contribution of every sector to real income differs significantly depending on whether it is calculated as deflated sectoral turnover or by the with-and-without approach. Only the with-and-without comparison is able to measure the contribution of a sectoral change to real income in terms of a welfare indicator. This is also made clear by the third last line. It shows that the with-and-without approach can be very well approximated by calculating the consumer surplus. The change in deflated turnover, on the other hand, is a good indicator of the change in economic activity, the use of capital and labour that takes place in a sector.

Table 1. Welfare change, contribution to real income change and sectoral change in terms of volume in a two sector model (own illustration)

Period	t_0		t_1 without change of S		t_1 with change of S		Aggregated change from t_0 to t_1 (with change of S)
Good ("**R**est of the economy" / "Ecosystem **S**ervice")	**R**	**S**	**R**	**S**	**R**	**S**	
Quantity in units	100.00	40.00	102.00	40.00	102.00	44.00	
(Simulated) price in monetary units	6.00	1.67	6.20	1.67	6.20	1.57	
"Sales"	600.00	66.67	632.40	66.67	632.40	69.02	
Nominal Income	666.67		699.07		701.42		
Price Index (Fisher)	1.0000		1.0300		1.0238		
"Sales" in volume terms	600.00	66.67	613.96	64.72	617.70	67.42	
Real Income	666.67		678.69		685.12		18.45
Consumer surplus change calculated with deflated prices					12.06	6.40	18.45
Contribution of R, S to real income change			12.02			6.43	18.45
Change in volume terms					17.70	0.75	18.45

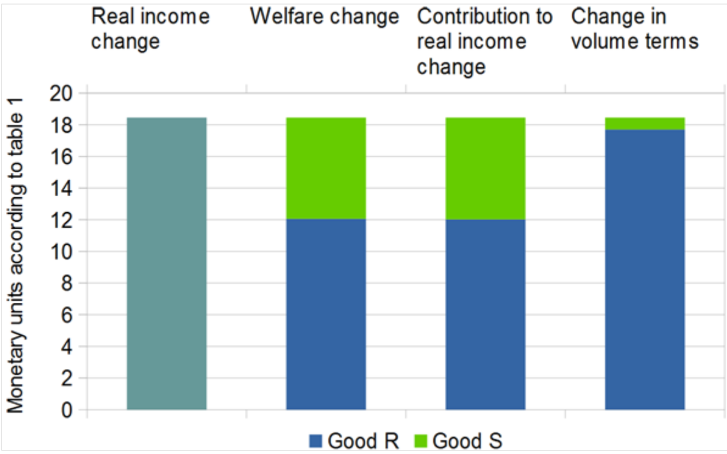

Figure 9. Welfare change, contribution to real income change and sectoral change in volume – results of the two sector model in table 1 (own illustration)

Conclusions for integrating welfare calculations into the accounts

Asheim & Weitzman (2001) showed that there would not be any deviation between deflated income and welfare increase if a Divisia index would be used to correct for inflation. In practice normally Laspeyres (e. g. European Union: HICP, European Commission/Eurostat 2017: 19; US: CPI, Velde 2015) and Fisher-Indexes (e. g. US: PCE, Velde 2015) are used for consumer price indexes. These will stay in practice, of course, for still a long time, for switching to a different index would not only need organizational efforts in the Federal Banks but also broad adjustments in the private sector.

The alternative to switching to a more ideal index would be the application of some pragmatic adaptation rules. As shown, the deviations between welfare and deflated income change follow certain rules that make it possible to define add-ons or deductions to make welfare calculations more compatible to deflated income. This may be required especially if deviations between the both values are considered being too large: e. g. if high relative changes coincide with high values for the simulated price elasticity. In other cases, e. g. small changes ≤ 10 % and low price elasticity ≤ - 0.3 % (s. fig. 5), the deviations between both measures are so small that they would be of low additional relevance, compared with the usual methodological uncertainties of many benefit estimations of public goods.

Changes of the welfare values of ecosystem services could be reported separately, apart from the normal accounts but interconnected, e. g. by using data on simulated prices as exchange in the normal accounts and as the basis for consumer surplus to calculate the separately reported welfare change. When choosing such kind of separated reporting, the requirements for the consistency of "green-box" information with the "normal" accounts could be lowered. Also, the demands on data precision and statistical significance could be reduced for a separate "green-box" to respond to the higher degree of uncertainty that is usually combined with welfare estimations and the valuation of public goods and external effects, compared with market data.

Additional functionalities of a "green-box"

A "green-box" that presents welfare-based values of ecosystem service change compatible to deflated income could also include other kind of information that are relevant to get a more comprehensive picture on welfare and the sustainability of nations economy from the environmental and natural capital perspective:

a) information on "defensive expenditures" or avoided cost spent to compensate environmental impacts and productivity losses of ecosystems, and

b) "unpaid ecological cost" or "ecological liabilities" (Vanoli 2015, SEEA 2015: 113) which may be appropriately defined in this context as:

   environmental costs that are not compensated yet and are still reducing welfare or

   that have to be compensated in the future to reach a sustainable development path.

Defensive expenditures

Ecosystem services like the natural fertility of soils, or the extent and growth-rate of fish populations, can be interpreted as elements of a production function that is constituted by human and natural inputs needed to manufacture a certain end product. Human inputs are for example fertilizers, pesticides, fuels, working and machine ours; natural inputs are the extent of arable land and the qualities of its soil, precipitation, ground-water level, fish populations of different size, age structure, density and spatial distribution etc.

If natural conditions are deteriorating, for instance by erosion, less wild insect pollination, overfishing etc., the same yield – if achievable at all – needs additional human inputs, to be produced. This gives rise to the production cost per unit. In fig. 10 this is shown as an upward shift of the supply function from $\overline{CD}$ to $\overline{C'D'}$. In the new market equilibrium, the smaller amount q_1 will then be sold to the price p_1. The additional cost to produce q_1, ☐ C'EIC, that would have been avoided if there had been no loss of ecosystem services are sometimes also called “defensive expenditures”.

Also, many cultural ecosystem services are just an element of a production function, in this case an individual or household production function. If people look for outdoor-recreation they often use a car or public transport to reach their destination. If such areas are converted to commercial or residential use, additional transportation costs and travel time may arise, both defensive expenditures in the sense of cost that could have been avoided if the recreational qualities of ecosystems had not been deteriorated or destroyed.

“Defensive expenditures” or avoided cost – in the definition above – are only part of the value of the lost ecosystem service. The complete additional cost to produce the amount q_0 of goods that was sold (or consumed) before ecosystem service declined is ☐ C'FGC. These full restoration cost or alternative cost (in the case of e. g. developing other nearby recreational opportunities) are sometimes also called “avoided cost”, however not in the sense above, as cost that would have been avoided if the ecosystem service loss had not happen but in the meaning of cost that have to be spent to reach the same level of production again. Such restoration costs are alternative costs to reach the same level of production as before are sometimes also taken as a proxy for the value of a service. However, if people can react to price changes by substituting with other goods, these approaches can overestimate welfare loss, whereas the additional cost to produce the new amount would underestimate it. A good proxy for the right value gives the sum of ☐ C'EIC (“defensive expenditures”) and Δ EGH; the latter being the consumer surplus loss above the quantity change.

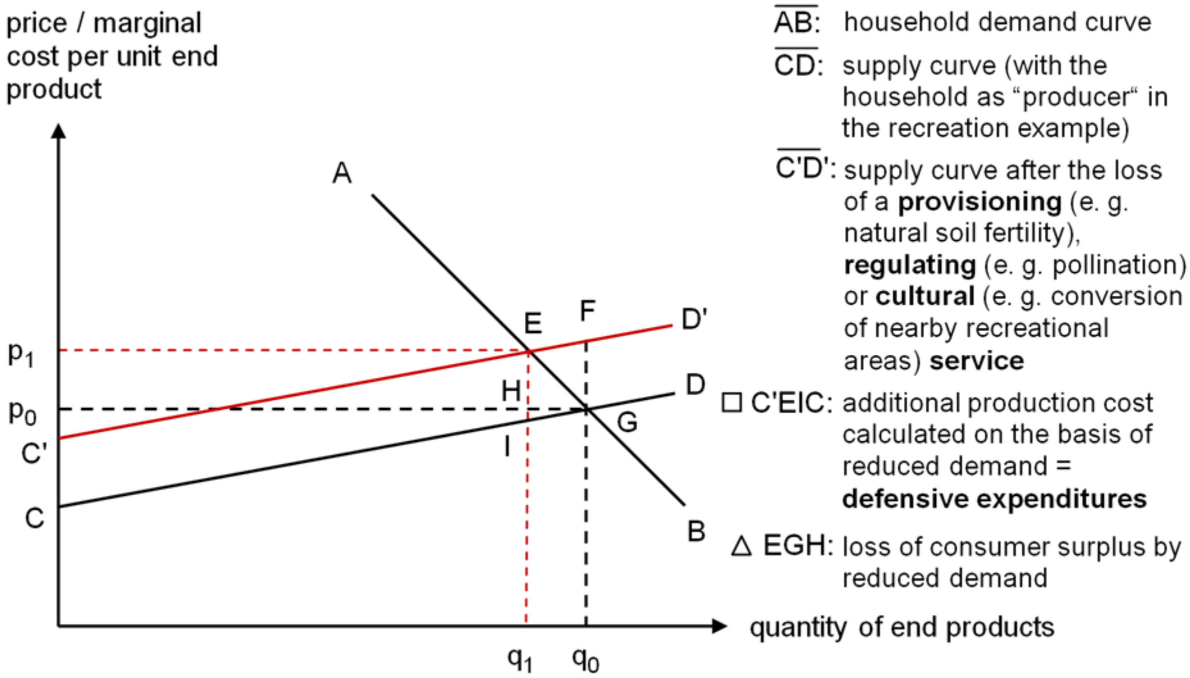

Figure 10. "Defensive expenditures" as compensation for ecosystem service loss (Own illustration)

This consideration shows how the different concepts and approaches that can be used for calculating the values of a service change (consumer surplus, defensive expenditures/avoided cost, restoration cost, alternative cost and cost that would have been avoided) interact and fit together. A green-box must provide clear rules to take these interrelations into account and avoid double counting.

Sometimes it is argued that "defensive expenditures" have to be made explicit because they contribute positively to GDP whereas the reason why they are spent is only for compensating a loss. This argument, however, is not always true. Defensive expenditures can have the effect that products are getting more expensive, as shown above. In such a case nominal GDP may keep unchanged but there will be a negative imprint on real GDP through rising prices. In this case "defensive expenditures" must not be subtracted from real GDP because they are already internalized by the effects of deflating. Presenting them in the "green-box" would just inform about the reasons why real GDP was negatively affected in the specified extent, namely due to measures for compensating ecosystem service loss.

The case is different, when "defensive expenditures" mean additional "defensive" end products or are compensated by the public sector. More intensive heat waves in cities, for example, caused by reducing green space for additional housing or commercial developments will increase health care costs. In the accounts these will be regarded as additional end products and not as cost. So, both nominal and deflated GDP will not be negatively affected. The same happens, if negative effects are compensated by the public sector, for public goods are integrated into the accounts on the basis of their cost. In such cases the "green-box" would hint at the fact, that real (deflated) GDP is overestimating welfare and should be corrected by the amount of "defensive expenditures" e. g. for health care.

Irrespective of reducing GDP or not, a quantification of "defensive expenditures" in GDP would be a helpful information for policy-makers to identify policy needs and give support for re-allocating resources to keep natural capital near to a welfare optimizing and sustainable level. The quantification of "defensive expenditures" and avoided cost is however not only a matter of extracting such data from the existing accounts. It would need additional efforts like modelling based on survey data, expert knowledge and it will need assumptions that may produce estimations with a level of uncertainty somewhat unusual for accounting data. Therefore, there may be good reasons to keep such information apart from the rest of the account. A "green-box", where useful information on environmental costs and benefits can be found, that are not completely compatible but more or less comparable with real GDP could be an appropriate place for it.

Unpaid ecological cost

This is true also for the concept of "unpaid ecological cost", which refers to that kind and those part of ecosystem and ecosystem service loss that is not yet compensated. In figure 10 this would be ☐ EFGI, which are the complete costs for compensating the ecosystem service loss less the "defensive expenditures". As pointed out above ☐ EFGI can overestimate welfare loss, when people´s reaction to prices compensates the service loss at least partly (SEEA 2015: 97 Table 8.1: Replacement cost). This is a reason why the concept of "unpaid ecological cost" is normally used only when a full restoration or compensation is to be expected, especially when political targets exist that are so binding that measures to reach a certain environmental goal will be more or less inevitable, only the time of implementing being uncertain. Examples may be greenhouse gas mitigation commitments under the Kyoto Protocol (United Nations 1998), the no loss goal of the Aichi Targets (CBD 2010: target 5) and corresponding regional (e. g. European Union 2011: target 2) and national biodiversity strategies or the aim to reach a good ecological status of water bodies of the European Water Framework Directive (WFD, Art 14).

"Unpaid ecological costs" are ecological liabilities that reduce GDP in the future as soon as they are paid. Such liabilities can be integrated into the asset-accounts if they are caused by the degradation of ecosystems. To take an example: if grassland is converted to cropland a part of the carbon stored in the soil is released to the atmosphere. This reduces the asset value of the ecosystem with regard to its contribution to climate change mitigation and thus reduces NDP (= GDP – depreciation/degradation). It would be double counting, if this asset value loss is also calculated as "unpaid ecological cost" (SEEA 2015: 114). A "green-box", however, would be a place where "double counting" in the sense of extraction and presentation of facts that are already included in the system anywhere else is the very objective in order to make them more explicit. Also, the other elements that are proposed for a "green-box", here, are "double-counting" in the explained sense (see fig. 11): consumer surplus has its double-counting in the effect of price and volume changes on real income, defensive expenditures are already accounted as intermediates or end-products. Therefore, "unpaid ecological cost" would only add a third category to the "green-box": Besides values that are already subtracted and values that have to be subtracted from real GDP to get a more comprehensive view of the welfare change, there will be another category of values that are assumed to be subtractions that have to be made in the future.

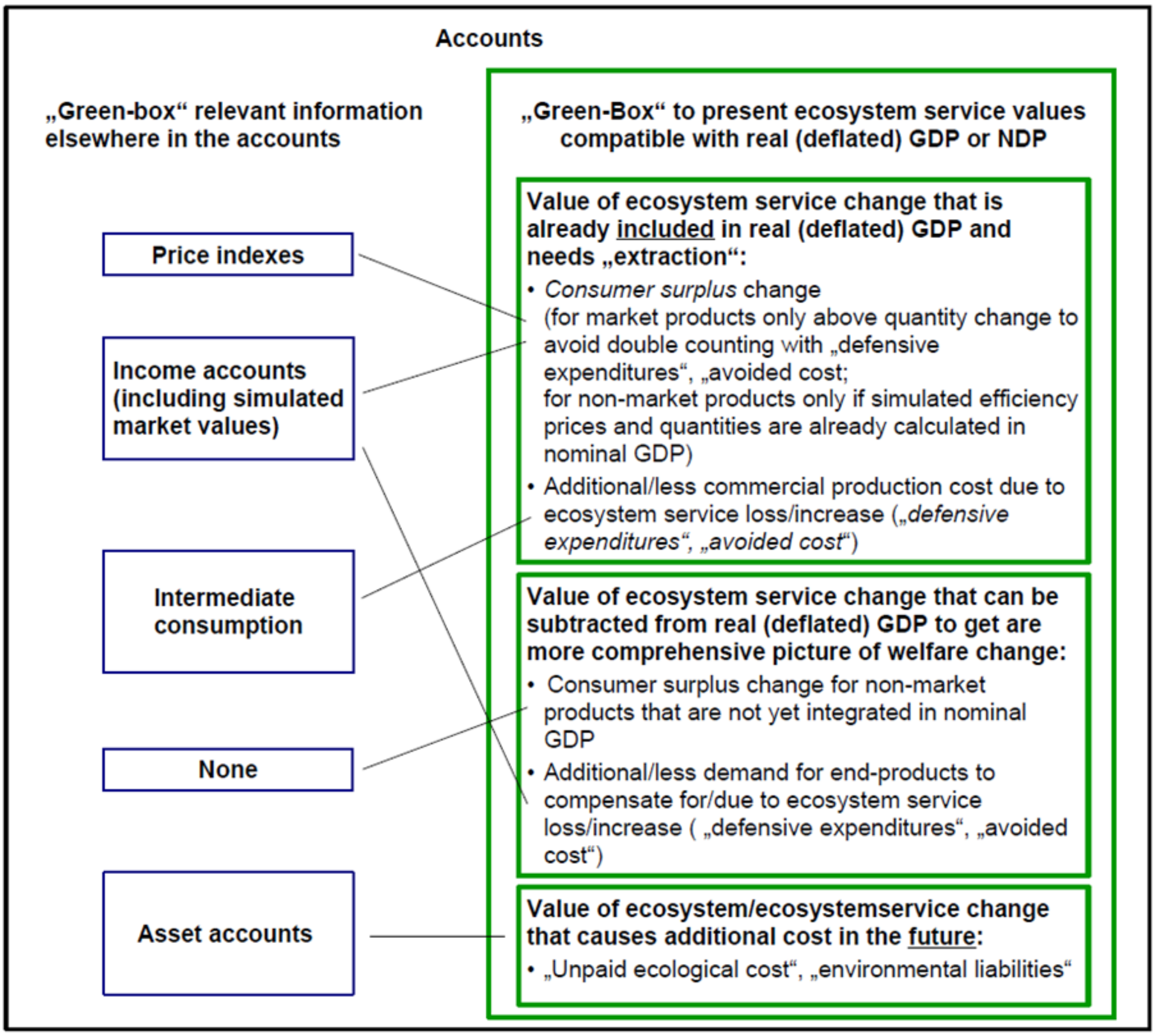

Figure 11. „Green-box“ relevant information at other places in the accounts

Recommendations for the further development of ecosystem accounts

Ecosystem services can be end products or intermediates. Welfare economic approaches normally measure the value of ecosystem service changes by estimating the effects on consumer and producer surplus (as a proxy for equivalent or compensating variation) or on production cost. Applying these kinds of methods positive changes of the quantity of services will always have positive effects on value and vice versa.

Accounting methods use exchange values (prices or simulated prices) to measure the monetary value of goods and services. Else equal, prices fall if quantities are increasing and rise with growing scarcity. Applying accounting methods to value ecosystem services would therefore have the effect that the value of overall (simulated) sales of a certain kind of ecosystem service – (simulated) price multiplied with volume – can decrease while the quantity of services grows and rise if there is a service loss. This ambiguity between quantity (change) and value (change) is one reason why accounting data are not directly applicable for decision making on resource allocation: Does society loose or win if there are additional efforts to preserve or restore ecosystems and their services? The common methods of accounts give little help to answer this question especially if the change of a consumer good is in focus.

The political demand for integrating ecosystem services into accounting, however, has its origin in the desire to get more informed for better decision making on the preservation and restoration of our natural capital – ecosystems and their services. Welfare based information on the value of ecosystem service change, compiled in a “green-box” in the accounts would provide with this kind of information. Politicians could realize what additional welfare, estimated in units of real GDP, the ecosystem service changes have caused and could compare this with e. g. the cost of restoration. Such a “green-box” should not stand alone, but linked to the rest of the accounts as close as possible. One reason for politicians to trust in accounting data is the international process on harmonization of data and statistical methods. Confidence in “green-box” data would gain when they are meeting prerequisites that are defined in the same or similar processes.

The anchor for the link between welfare calculations and accounting is real (deflated) GDP. Deflators like the Laspeyres, the Paasche or the Fisher index work with prices and quantities in such a way that the effects of price and volume changes on real GDP have a high similarity with assessing the same effect with welfare calculations on the basis of consumer surplus. Welfare based values compatible with real GDP offer the opportunity to compare the impacts of cultural, regulating and provisioning services on consumer surplus and production cost directly with economic growth and with the money spent for preserving or restoring natural capital. That's what politicians expect or hope from integrating ecosystems and their services into the accounts. A “green-box” with values of ecosystem service changes compatible with real GDP would help to satisfy these needs.

Furthermore, this kind of “green-box” would offer a place for the integration of other concepts describing environmental cost and degradation such as “defensive expenditures” and “non-paid ecological cost”. They were partly developed as an alternative for the theoretically preferable but rather challenging method of integrating degradation into the capital accounts by estimating changes in the net present value of future benefits. Both concepts “defensive expenditures” and “unpaid ecological cost” fit rather well to the intention of a “green-box” to deliver real GDP compatible information, thus contributing to the aim to make the accounts more policy relevant without changing basic structures and rules.

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5. Learning from 30+ years of non-market valuation of ecosystem services[1]

Dr. Luke M. Brander[2]

Introduction

The objective of this paper is to provide background on the environmental economics literature on the value of natural capital and ecosystem services and draw observations on the directions this field of research is taking. The economic importance of natural capital and the ecosystem services it provides has long been recognised (Pearce and Turner, 1990; Costanza et al., 1997; Balford et al., 2004; MA 2005; TEEB 2010); albeit expressed using varying and evolving concepts, terminology and classifications (Fisher and Turner, 2008; Gómez-Baggethun et al., 2009; Haines-Young and Potchin, 2012). It is also widely recognised that much of the economic value of ecosystems services does not appear in (market focused) conventional accounts due to the public good characteristics of many natural resources, which precludes them from being traded in markets. Without private incentives to manage such natural resources and without public recognition of their importance to society, natural capital is often over-exploited and ecosystem services are under-supplied. A response from several disciplines to this sub-optimal use of natural resources has been to produce information on the value of ecosystem services in order to inform private and public decision makers and enable them to take account of these values in their decisions. The development of the System of Environmental-Economic Accounts (SEEA) framework is part of this broad undertaking to provide information to support decision-making.

During the past 30+ years, the discipline of environmental economics has attempted to produce information on the economic value of ecosystem services, i.e. their contribution to human wellbeing (Pascual et al., 2010). For the most part, the unit of measurement of economic value is money, since this is a widely understood medium of exchange and enables direct comparison with other values in the economy. The body of knowledge on the economic value of ecosystem services now comprises thousands of studies covering all regions of the world and all ecosystem services (de Groot et al., 2012). This information may potentially be used as input into ecosystem service accounts and represents an important resource for the implementation of SEEA (Day, 2013). This paper focuses on providing an overview of the environmental economics literature on the value of ecosystem services. It recognises, but does not directly address, the commensurability of different concepts of value (i.e. welfare values that are estimated in many economic studies and exchange values that are used in the System of National Accounts).

The structure of this paper is as follows: Section 1 provides a brief introduction to the non-market economic valuation methods that are widely used to estimate values for ecosystem services, including primary valuation and value transfer methods; Section 2 describes the past 30+ years of valuing ecosystem services in terms of the number of studies by location, method and measure of value; Section 3 illustrates current research efforts with two case studies representing the use of different methods at different scales of analysis (local and

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global); Section 4 discusses the future directions that non-market valuation of ecosystem services can take to enhance the information that is provided to decision makers.

Non-market valuation

A variety of methods have been developed for estimating the economic value of ecosystem services that are designed to span the range of valuation challenges raised by the application of economic analyses to the complexity of the natural environment (Freeman, 2003). Figure 1 provides a representation of the available economic methods for valuing ecosystem services. A key distinction is between methods that produce new or original information generally using primary data (primary valuation methods) and those that use existing information in new policy contexts (value transfer methods).

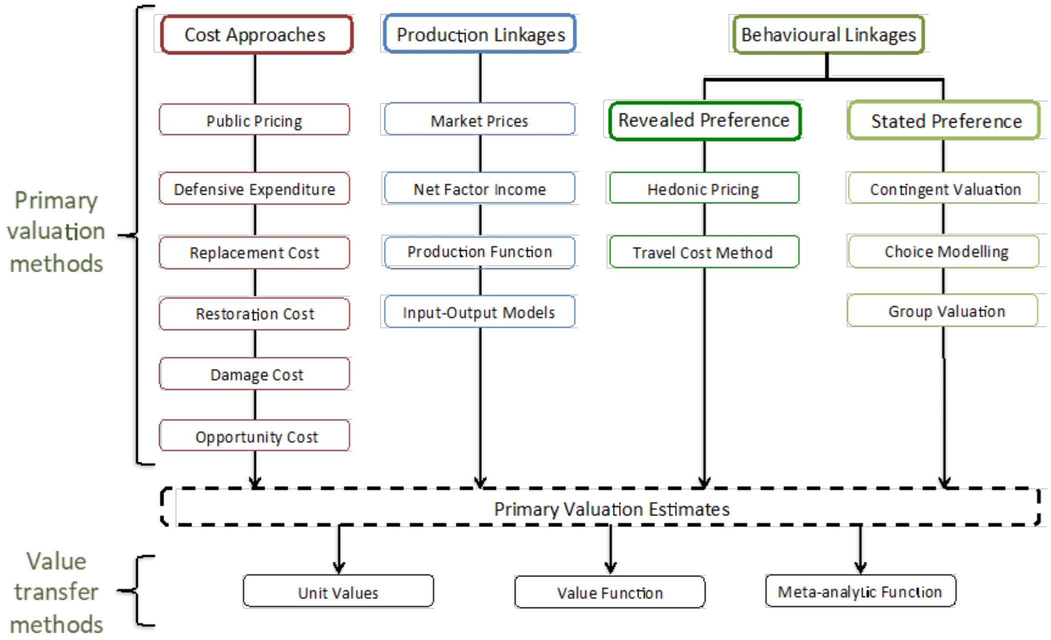

Figure 1. Available economic methods for valuing ecosystem services (Brander et al., 2018a)

Primary valuation methods can be divided into three categories: 1. Cost-based approaches that use some measure of the costs associated with an ecosystem service as a proxy for the value of the service; 2. Methods that estimate the value of ecosystem services as inputs into production; and 3. Methods that use consumer behaviour to measure the value of ecosystem services. This third category can be further usefully divided between revealed preference methods (those that observe actual behaviour of the use of ecosystem services to elicit values) and stated preference methods (those that use public surveys to ask beneficiaries to state their preferences for, usually hypothetical, changes in the provision of ecosystem services). Revealed preference methods may be favoured since they reflect actual behaviour but are limited in their applicability to only a sub-set of ecosystem services. Stated preference methods on the other hand rely on responses recorded in surveys or experiments but are more widely applicable to a larger set of ecosystem services, including biodiversity as a source of both use and non-use values to people (Mace et al., 2012).

Decision-making often requires information quickly and at low cost. New ‘primary’ valuation research, however, is generally time-consuming and expensive. For this reason, there is interest in using information from existing primary valuation studies to inform decisions (and as input into ecosystem service accounts). This transfer of value information from one context to another is called value transfer.

Value transfer is the use of research results from existing primary studies at one or more sites or policy contexts (“study sites”) to predict welfare estimates or related information for other sites or policy contexts (“policy sites”) (Navrud and Ready, 20??; Brander, 2013; Johnston et al., 2015). Value transfer is also known as benefit transfer but since the values that are transferred may be costs as well as benefits, the term value transfer is more generally applicable.

In addition to the need for expeditious and inexpensive information, there is often a need for information on the value of ecosystem services at a different geographic scale from that at which primary valuation studies have been conducted. So even in cases where some primary valuation research is available for the ecosystem of interest, it is often necessary to extrapolate or scale-up this information to a larger area or to multiple ecosystems in the region or country. Primary valuation studies tend to be conducted for specific ecosystems at a local scale whereas the information required for decision-making, and indeed the SEEA, is often needed at a regional or national scale. Value transfer therefore provides a means to obtain information for the scale that is required.

Value transfer can potentially be used to estimate values for any ecosystem service, provided that there are primary valuations of that ecosystem service from which to transfer values. Value transfer methods have been employed widely in national and global ecosystem assessments (e.g. UK NEA, 2011; Hussain et al., 2011), value mapping applications (see Schägner et al., 2013) and policy appraisals. The use of value transfer is widespread but requires careful application. The alternative methods of conducting value transfer are described here.

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

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

Meta-analytic function transfer uses a value function estimated from the results of multiple primary studies representing multiple study sites in conjunction with information on parameter values for the policy site to calculate the value of an ecosystem service at the policy site. A value function is an equation that relates the value of an ecosystem service to the characteristics of the ecosystem and the beneficiaries of the ecosystem service. Since the value function is estimated from the results of multiple studies, it is able to represent and control for greater variation in the characteristics of ecosystems, beneficiaries and other contextual characteristics. This feature of meta-analytic function transfer provides a means to account for simultaneous changes in the stock of ecosystems when estimating economic values for ecosystem services (i.e. the “scaling up problem”). By including an explanatory variable in the data describing each “study site” that measures the scarcity of other ecosystems in the vicinity of the “study site”, it is possible to estimate a quantified relationship between scarcity and ecosystem service value. This parameter can then be used to account for changes in ecosystem scarcity when conducting value transfers at large geographic scales (see Brander et al., 2012a, for a more detailed explanation of this method).

The following section shows that the number of primary studies on the value of ecosystem services is substantial and increasing rapidly. This means that there is a growing body of information to draw on for the purposes of transferring values. With an expanding information base, the potential for using value transfer is continually improving.

Past: 30+ years of valuing ecosystem services

In this section we aim to provide an overview of past research efforts to estimate economic values for ecosystem services. For this purpose, we present a summary of studies contained in the Environmental Valuation Reference Inventory (EVRI), which is currently the most comprehensive database of non-market valuation studies (www.evri.ca). EVRI contains records on over 4600 studies that value ecosystems services. Nevertheless, this database is partial in its coverage and not geographically representative. There is an understandable over-representation of North American and European studies given the partners that initiated and host this inventory and the higher visibility/publication rates of studies from these regions. As such, our overview of the existing literature is also partial and not globally representative. Given that data on the actual population of economic valuation studies is not available, it is not possible to quantify how complete our overview is, but we expect is represents the tip of the iceberg (i.e. approximately one 10th of the total number of studies). To give some sense of proportionate coverage, we compare the EVRI data with a recent review of forest valuation studies for Peninsular Malaysia (Brander and Yeo, 2018), which itself may of course be incomplete. For this specific context, EVRI contains 7 studies and the review by Brander and Yeo (2018) contains 50, i.e. the proportionate coverage is 14%. Bearing this limitation in mind, we provide the following overview of the literature.

Figure 2 represents the cumulative total number of valuation studies for ecosystem services from 1970-2017. From the year 2000 onwards there has been a steady flow of new valuation studies (150-250 studies per year). Figure 3 represents the number of valuation studies by country. As mentioned previously, the data set is dominated by studies conducted in North America and Europe, but this is also likely to reflect the spatial distribution of valuation research applications. The implication of this spatial distribution of primary valuation studies is that for the purposes value transfer, the information base is relatively limited for the rest of the world and particularly for developing countries.

Figure 4 represents the number of valuation studies by valuation method. Stated preference methods have been the most heavily used with relatively few applications of cost-based or production linkage methods, with the exception of direct market prices. It is also notable that a large number of studies apply multiple valuation methods, which is often the case when estimating values for diverse ecosystem services that require the use of different methods. Figure 5 represents the number of studies by the type of value concept that they estimate. Most studies estimate consumer willingness to pay, which is potentially applicable in an accounting framework if these values can be translated into a simulated market price or exchange value.

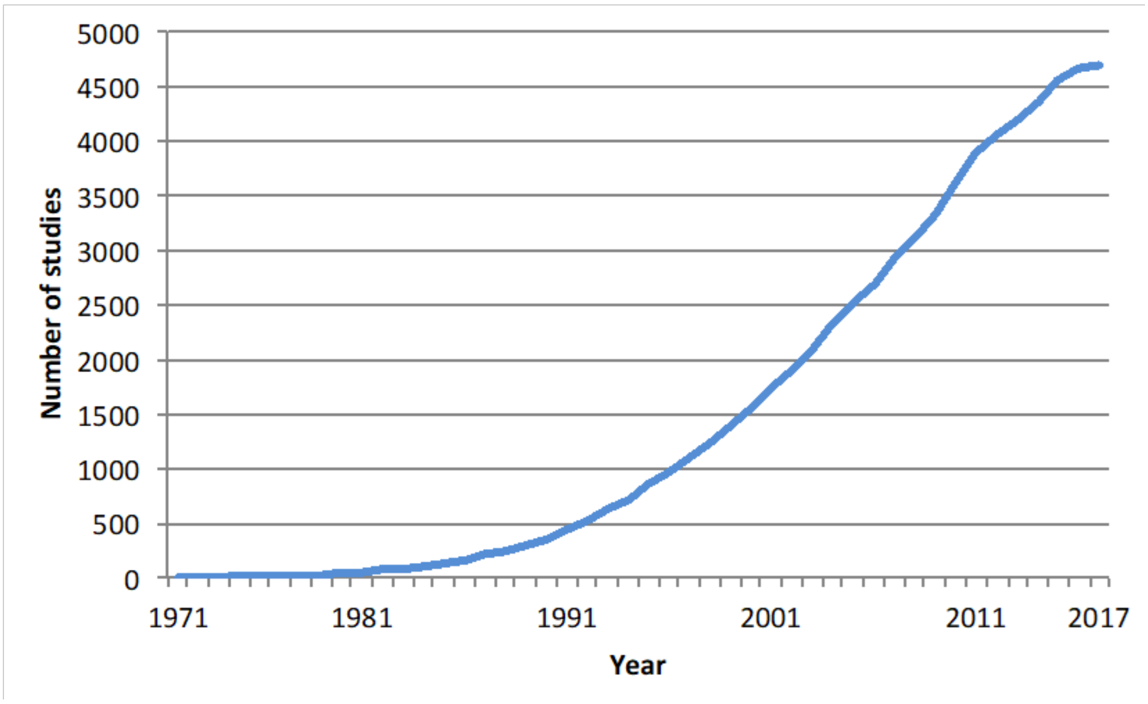

Figure 2. Cumulative number of ecosystem service valuation studies sourced from the Environmental Valuation Reference Inventory, www.evri.ca (Modified from Christie et al., 2008)

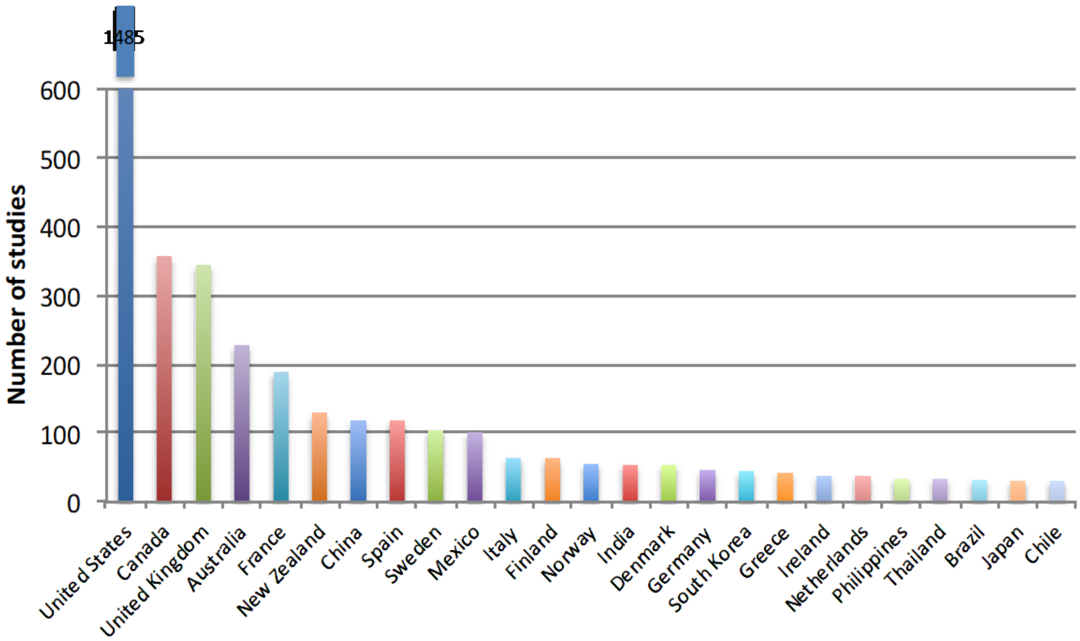

Figure 3. Number of ecosystem service valuation studies by country (Source data: Environmental Valuation Reference Inventory, www.evri.ca)

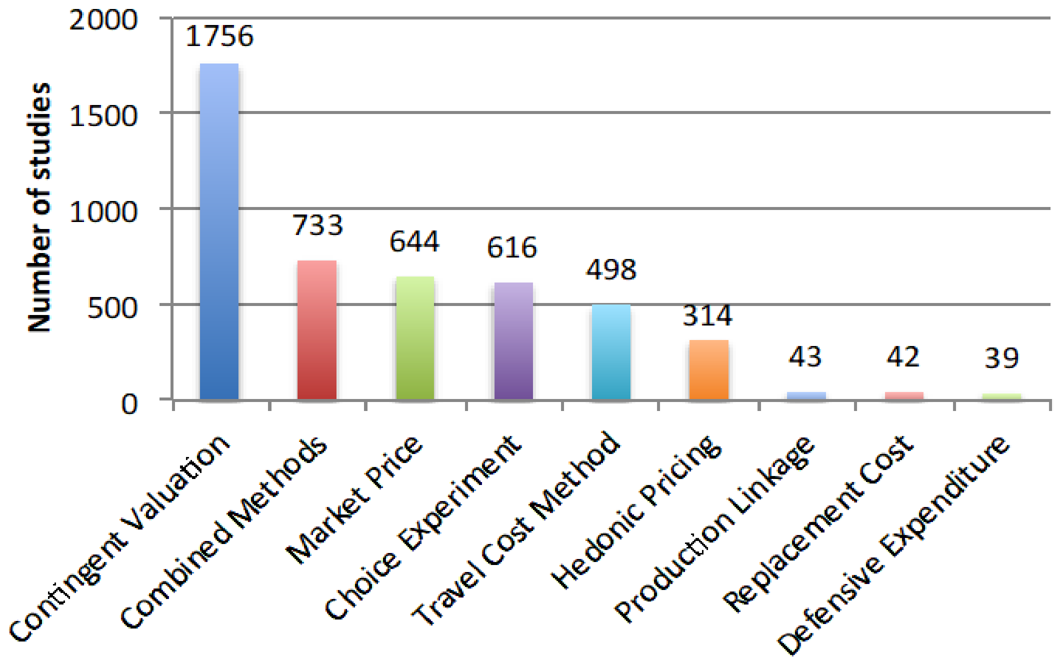

Figure 4.. Number of ecosystem service valuation studies by valuation method (Source data: Environmental Valuation Reference Inventory, www.evri.ca)

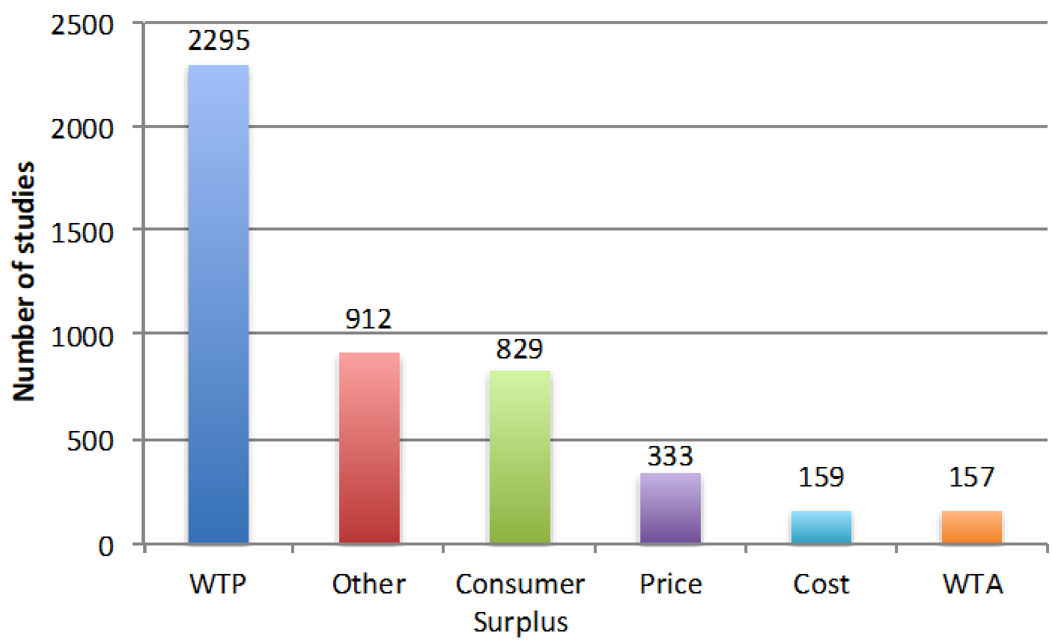

Figure 5. Number of ecosystem service valuation studies by value concept (Source data: Environmental Valuation Reference Inventory, www.evri.ca)

Present: Example valuation studies at different scales

Providing a comprehensive overview of present efforts to estimate economic values for ecosystem services is not attempted in this paper. Here we simply describe two contemporary valuation studies that represent the use of different methods at different spatial scales.

The intention is to give an impression of the variation in valuation methodologies (primary valuation and value transfer) and scales at which they can be applied. The first case study is a watershed-scale valuation of ecosystem services to local communities using a choice experiment application. The second case study is a global valuation of changes in ecosystem services resulting from the expansion of marine protected areas using value transfer methods.

Case study 1: The value of ecosystem services to long house communities in the Baleh watershed, Sarawak, Malaysia. This study attempts to estimate the economic value of several key ecosystem services and development options in a remote watershed in Sarawak, Malaysia. The location of the Baleh watershed (1.24 million hectares) is indicated in Figure 6. The study conducted a survey of 237 households living in long house communities within the watershed during November-December 2017. The survey collected information on household use of natural resources and used a choice experiment to elicit preferences for changes in five attributes: 1. Availability of wild bush meat and fish; 2. Access to traditional hunting grounds; 3. Availability of freshwater; 4. Road access to the long house; Monthly income from agriculture. An example choice card is represented in Figure 7. Survey respondents were asked to choose their preferred option on each of a set of choice cards, thereby implicitly stating the rates at which they would exchange each attribute. By specifying one of the attributes in monetary units (agricultural income) it is possible to compute respondents' willingness to pay (WTP) for changes in the other attributes. The willingness to pay results are represented in Figure 8. The highest valued attribute is road access to the long house, for which households are on average willing to pay US$ 120 per month, followed by water and bush meat availability. Households did not express positive WTP for access to traditional hunting grounds. Full details of the choice experiment design and analysis can be found in Brander et al. (2018).

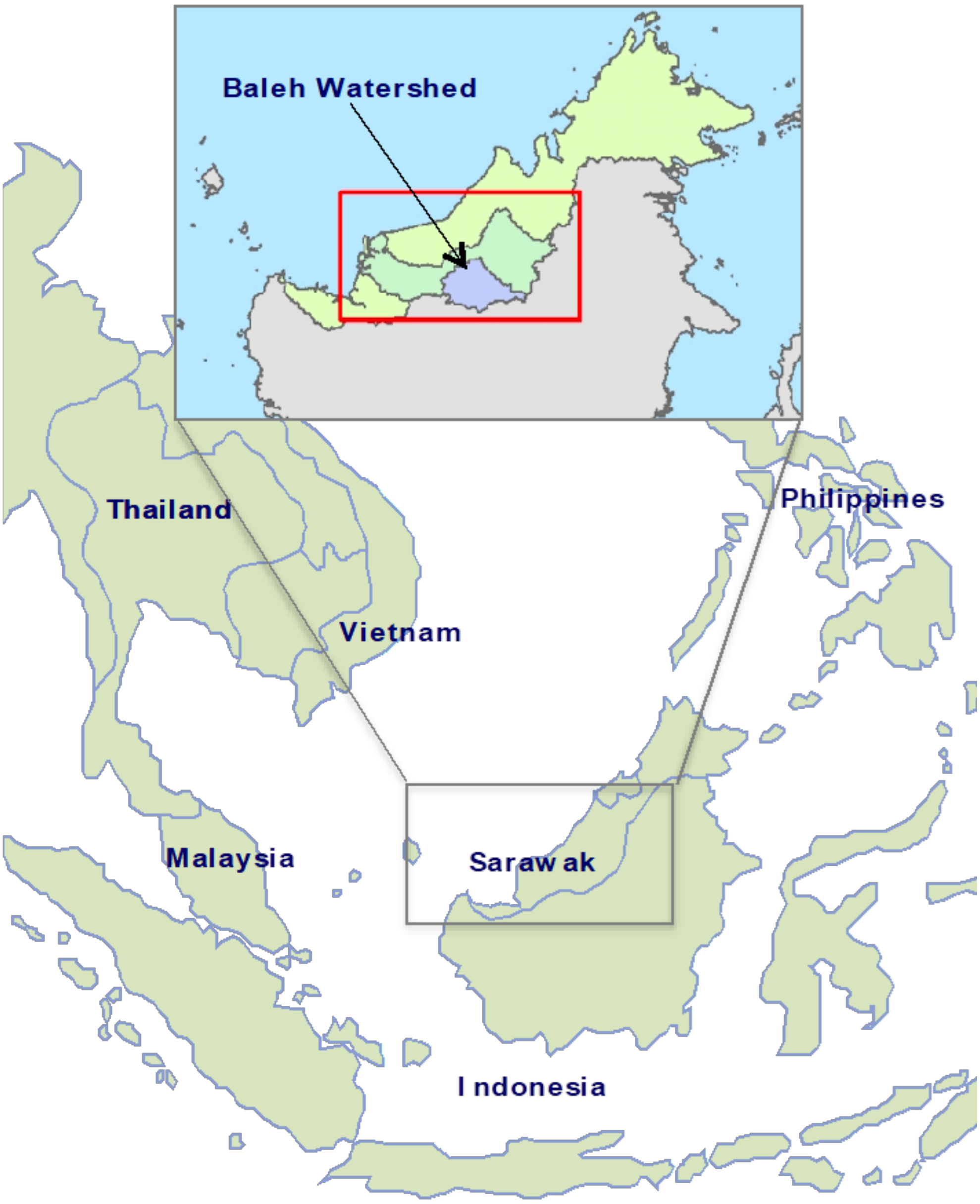

Figure 6. Location of the Baleh watershed, Sarawak, Malaysia (Brander et al., 2018b)

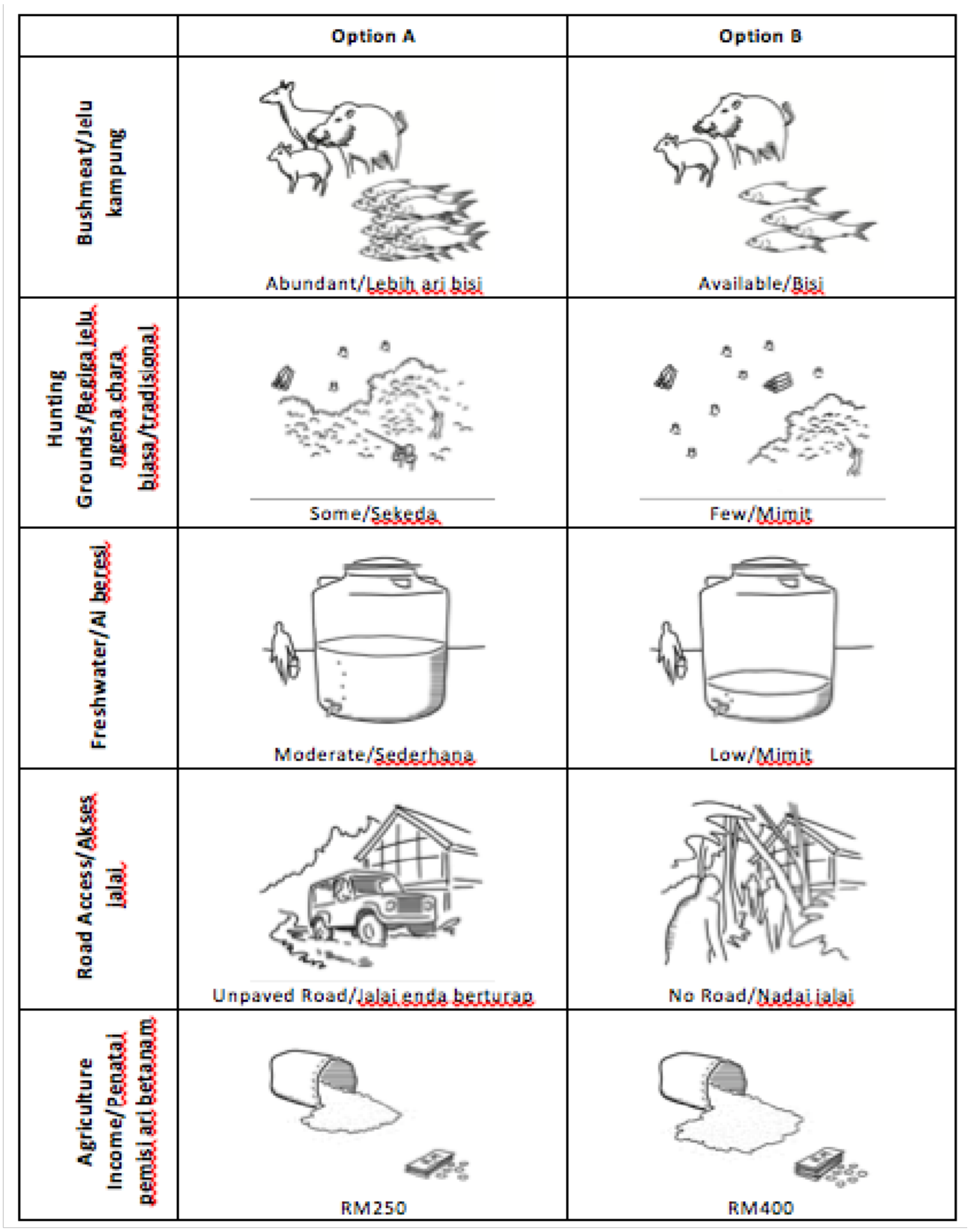

Figure 7. Example choice card (Brander et al., 2018b)

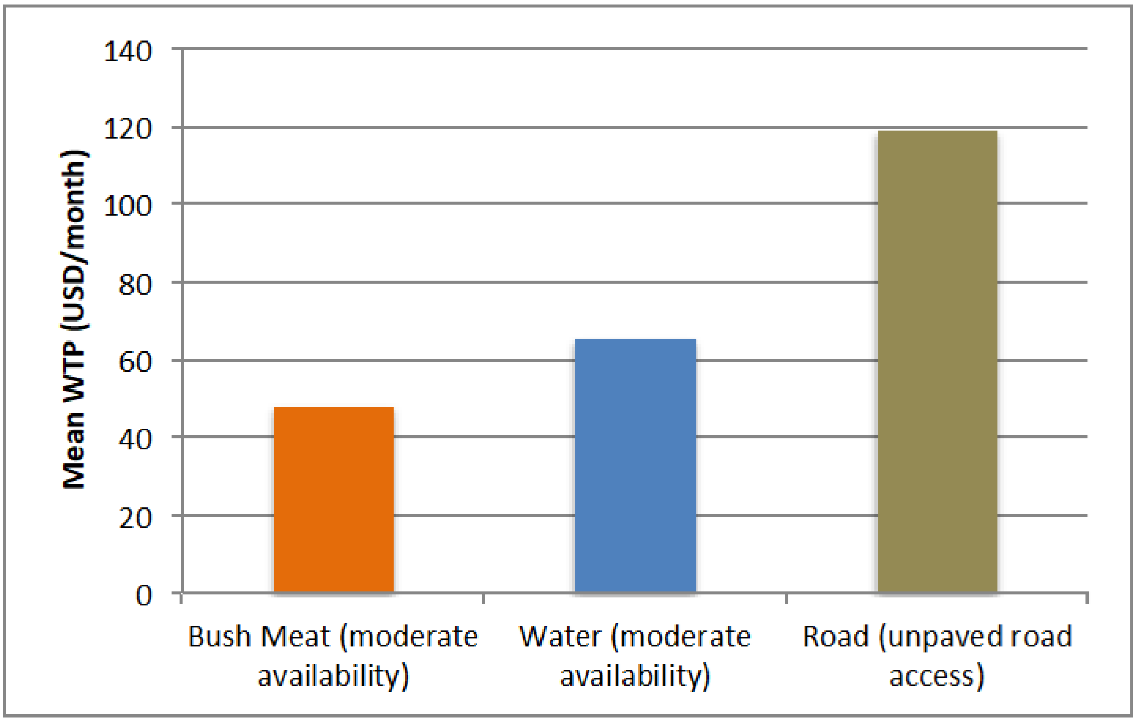

Figure 8. Estimated willingness to pay for changes in environmental and development characteristics in the Baleh watershed (Brander et al., 2018b)

Case study 2: The global economic benefits of expanding marine protected areas. This study examines the economic case for expanding marine protected areas by conducting a cost-benefit analysis of alternative global scenarios for expanding their coverage. Currently, 3.4% of global marine area is designated as marine protected area (MPA), with 0.59% established as no-take MPAs. The location of existing MPAs is represented in Figure 9. The Convention on Biological Diversity (CBD), Aichi Target 11 and the Durban Action Plan call for an expansion of MPA coverage to 10% and 30% of global marine area respectively. To assess the economic rationale for MPA expansion, Brander et al. (2015) conduct a cost-benefit analysis to estimate the net benefits of expanding global marine protected areas (MPAs) to 10% and 30% coverage of total marine area. The study developed a set of six mapped scenarios for the global expansion of MPAs (see Figure 9). The scenarios vary along two dimensions: 1. The coverage of MPAs as a proportion of total marine area; 2. The characteristics of target locations for MPAs in terms of biodiversity and degree of human impact. By targeting locations that are characterised by high biodiversity and high human impact, the MPAs serve to mitigate damage: the “Protect to Mitigate” (P2M) scenario. Alternatively, targeting areas with high biodiversity and low human impact provides protection to intact ecosystems from potential future human impact: the “Protect to Preserve” (P2P) scenario. Targeting areas with low biodiversity and low human impact identifies locations that are currently not exploited and do not have biological resources that may be exploited in the future: the “Easy to Expand” (E2E) scenario. These three variants of target location are combined with the two targets for areal extent to give six mapped scenarios represented in Figure 9: P2M-10%, P2M-30%, P2P-10%, P2P-30%, E2E-10% and E2E-30%.

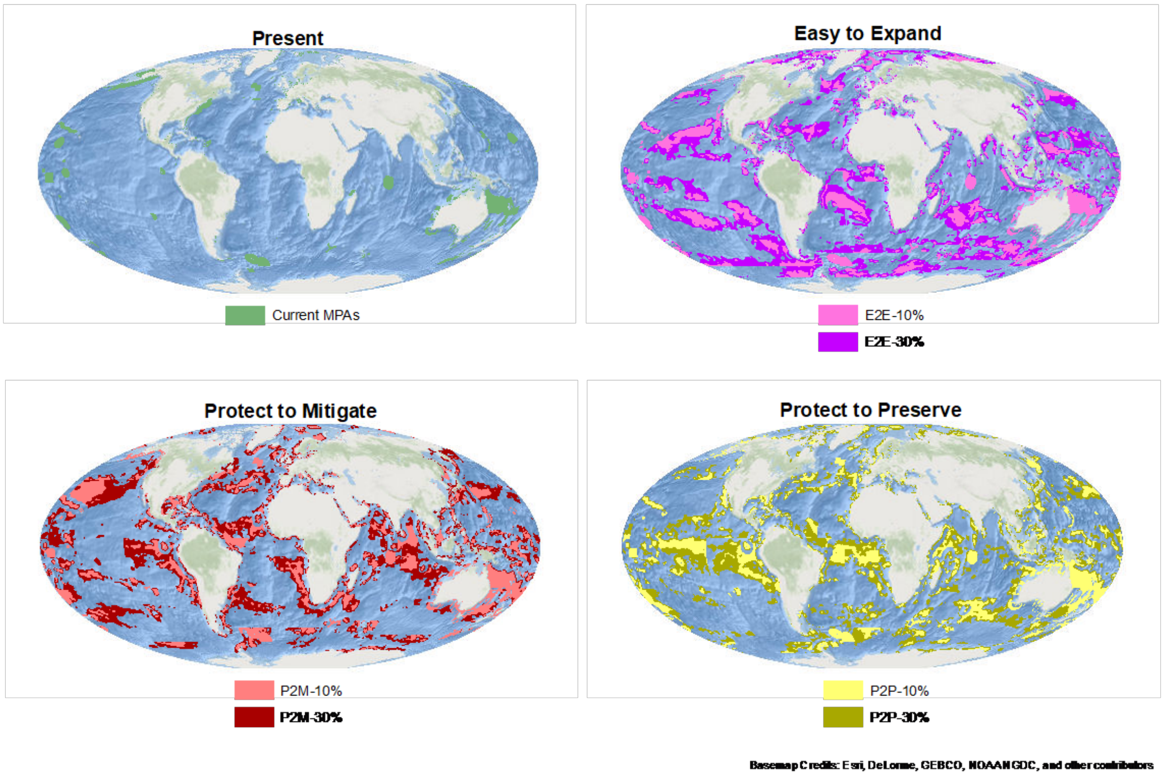

Figure 9. Current and future global distributions of marine protected areas (Brander et al., 2015)

The economic benefits of expanding MPA coverage are the maintained or enhanced flows of ecosystem services that are provided by protected marine ecosystems (Sala et al., 2013; Potts et al., 2014; Pascal et al., 2018). The marine ecosystems included in our assessment are coral reefs, coastal wetlands and mangroves. The marine ecosystem services assessed are the provision of food for subsistence or commercial use; tourism and recreation; coastal protection; carbon sequestration; and biodiversity.

Spatial data for coral reefs, coastal wetlands and mangroves are obtained from global maps (Burke et al., 2011; Lehner and Döll, 2004; Giri et al., 2011). Differences in ecosystem extent between a baseline scenario, representing spatially variable continuing trends of ecosystem loss, and each MPA expansion scenario are modelled using estimates on MPA effectiveness obtained from the literature. Marginal values for changes in ecosystem extent are subsequently estimated using value functions for coral reef, wetland and mangrove ecosystem services that have been estimated through meta-analyses of the relevant economic valuation literatures (Hussain et al., 2011; Brander et al., 2012b). The value of avoided carbon emissions and additional sequestration by mangroves that are protected by MPAs is estimated separately using unit value transfer of the social cost of carbon and parameters obtained from the literature (Pendleton et al., 2012). The aggregated present values of benefits of improved provision of marine ecosystem services for each scenario are presented in Figure 10 and range between US$ 622 billion under E2E-10% and US$ 1,145 billion under P2P-30%. The estimated benefits of MPA protection are substantial, reflecting both the high economic value of marine ecosystem services and the high rates of loss in the absence of additional protection. The results also show very large differences in the yield of benefits across scenarios. The spatial distribution of MPAs under the P2P scenario, i.e. targeting areas with high biodiversity and low human impact, delivers considerably higher returns.

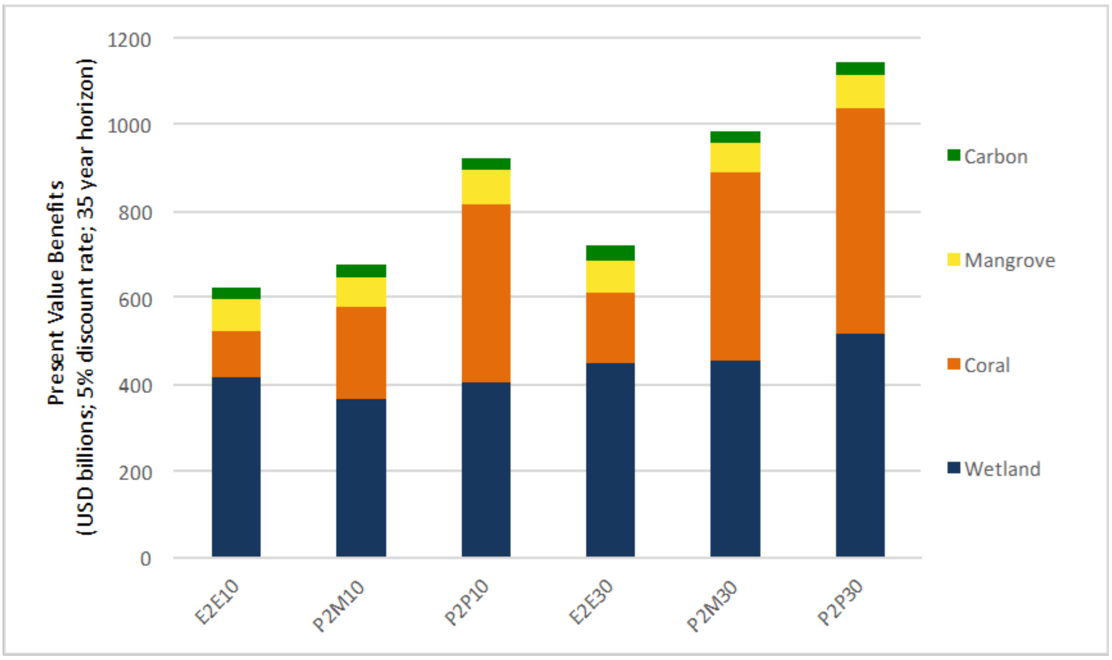

Figure 10. Global benefits of expanding marine protected areas (present values over 35 years; USD billions; 5% discount rate). (Brander et al., 2015a)

Future: On-going and future developments

Non-market valuation of ecosystem services has undergone substantial conceptual and methodological development during the past 30+ years and continues to do so (Oleson et al., 2018). Here we discuss the on-going and future developments in this field of research. Generally, the development of non-market valuation is characterised by increased integration in several dimensions.

Regarding valuation methods, there is increasing integration of the results of multiple methods within individual studies. Integrating multiple valuation methods serves several purposes: 1. To enable comparison and validation of values for an individual ecosystem service estimated using different methods (e.g. using stated and revealed preference methods to estimate willingness to pay for biodiversity conservation); 2. To estimate separate components of value for an individual ecosystem service (e.g. using a revealed preference method to estimate consumer surplus for outdoor recreation and a production linkage method to estimate producer surplus); 3. To estimate the value of multiple services derived from the same ecosystem.

Regarding ecosystem services, there is increasing integration in terms of the range services valued and the linkages between values. There is recognition that individual services provided by an ecosystem are not perfectly separable outputs but in some cases are jointly produced (e.g. mangroves providing support to fisheries and coastal protection) or in other cases are mutually exclusive to some extent (e.g. forests providing habitat for rare species or recreational use). Valuation studies therefore need to continue developing more complete bio-economic models to assess complements and trade-offs in ecosystem services.

A continuing development in non-market valuation, which has already been underway for over 20 years (Bateman et al., 1996; Bateman et al., 2002) but shows promise for further development, is the integration of spatial data using geographic information systems (GIS). The estimation of accurate values for ecosystem services requires that account is taken of spatial heterogeneity in biophysical and socioeconomic conditions. Spatial factors that affect the supply of ecosystem services include among others: ecosystem area (possibly characterised by a non-linear relationships and thresholds), networks, resilience, biodiversity, fragmentation, disturbance, and accessibility. Spatial factors that affect demand for ecosystem services include: the number of beneficiaries, culture and preferences, ecosystem area, distance to the ecosystem, and the availability of substitutes and complements. See Bateman et al. (2002), Hein et al. (2006), and Schaafsma (2015) for more detailed discussions of spatial determinants of ecosystem service demand and supply. The integration of spatial data can therefore produce more accurate value estimates and reveal additional information. Besides communication and visualisation, value mapping makes site-specific ecosystem service values available on a large spatial scale, which may be useful for several policy applications including ecosystem service accounting, land use policy evaluation, conservation planning, targeting land restoration activities and designing payments for ecosystem services (Brander et al. 2015b). Integration of spatial variables in both primary valuation and value transfer continues to be a promising avenue for future development.

A related but broader integration is across disciplines. Economic methods for valuing ecosystem services primarily focus on measuring changes in human welfare following changes in the availability of ecosystem services, often driven by biophysical changes in ecosystem extent, condition and functioning. Any economic valuation therefore fundamentally relies on inputs from biophysical measurement or modelling of changes in ecosystem service availability. It is also the case that economic valuations use inputs from socio-cultural methods, for example to define the scope of an assessment using participatory GIS, or to develop scenario storylines using participatory scenario planning. The flow of information from one disciplinary set of methods to another can also travel in the other direction, with results from economic methods used as inputs in biophysical and socio-cultural assessments. The reality is that methods defined by disciplinary boundaries are to a large extent complements rather than substitutes in providing information on the importance of ecosystem services in decision-making. There is a need for further development on linking of biophysical, economic and socio-cultural methods.

A final avenue for future development is the compilation and synthesis of value estimates. Existing efforts to organise data from the expanding number of primary valuation studies include the Environmental Valuation Reference Inventory (EVRI) and the Ecosystem Service Valuation Database (ESVD)[3] developed by the TEEB initiative. The EVRI is currently the most comprehensive global database of studies, i.e. each row in the data describes an individual study in terms of its location, methods, results etc., and is thereby a very useful starting point for the purpose of conducting literature reviews. The ESVD is currently the most comprehensive database of standardised valuation estimates, i.e. each row in the data describes a value estimate that has been converted to a common currency and year of value, which is a very useful starting point for the purposes of conducting value transfers, meta-analyses, or potentially as input into ecosystem service accounts (de Groot et al.,

[3]https://www.es-partnership.org/services/data-knowledge-sharing/ecosystem-service-valuation-database/

2012). These two databases are complements and might benefit from being functionally linked. All such databases, however, require continuous, or at least periodic, updating to remain useful. There is a need to organise and fund the continued upkeep of such information sources; and also to develop efficient methods of doing so given the volume of new data on the value of ecosystem services that is being produced.

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6. Simulated Exchange Values and Ecosystem Accounting[1]

Alejandro Caparrós[2]

Summary

Ecosystems provide many goods and services that are relevant to society, and there is an increasing interest in scientific and political arenas to extend the System of National Accounts (SNA) to explain this contribution. Although there is a range of supply side valuation methods, valuation of non-market goods and services produced by ecosystems has traditionally focused mainly on the demand side, by using non-market valuation techniques. These methods allow the estimation of the different Hicksian variations (which are equivalent to the consumer surplus if the income effect is small). Hicksian variation estimates are relevant in cost-benefit analysis; however, for ecosystem accounting, one needs to distinguish the part that could, in fact, be internalized in terms of prices multiplied by quantities. Therefore, several authors have proposed focusing exclusively on the part that can be internalized because this is the only part that is consistent with current estimates in the standard System of National Accounts (Ahmad et al. 1989, Caparrós et al. 2003, Cairns 2003 and Obst et al. 2016).

To translate this into applications, Caparrós et al. (2003, 2017) proposed the Simulated Exchange Value (SEV) method. Briefly, the method consists of using demand functions that are estimated using non-market valuation methods to simulate the entire market (demand, supply and competitive environment) to obtain the market value that one could obtain from a given ecosystems service if it were internalized. To fix ideas, the discussion below uses free access recreation in terrestrial ecosystems as an example; however, other non-market amenities could be treated in a similar manner.

1. The Simulated Exchange Value method

Let us assume that we are interested in estimating the SEV for free access recreation in a region or country. There are a given number of recreational areas and, although in principle new areas can be established, this needs a considerable amount of time. Thus, in the short run, the number of recreational areas is fixed and each area would have a site-specific demand function. This corresponds to a market under monopolistic competition[3]. In the long run, monopolistic competition assumes that new entries are possible and that this will drive benefits to zero. The standard model assumes that every new entry reduces the demand for all. However, in the case of nature based recreational areas, new entries are difficult even in the long-run. In addition, the production account in SNA does not regard the long-run, and we are not going to be working with the total demand of the sector, but with the site-specific demand. Thus, it is more reasonable to use the values estimated for the short-run.

Let us assume that there is a non-market valuation study available for each recreational area (or one that allows estimating different values for each area). To simplify, assume fur-

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[1]This paper essentially summarizes the article Caparrós et al. (2017).

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[3]As noted by Obst et al. (2016) national accounts do not only include values obtained in perfect markets. Hence, simulated exchange values are not restricted to perfect markets either.

ther that all are single-bounded contingent valuation studies, where visitors were faced with two alternatives: visit the area (i=1), at a given price, or remain at home (i=0). The utility of visiting the area for an individual n is then given by:

$$U_{ni} = \beta' x_{ni} + \varepsilon_{ni} = \hat{\beta}' \hat{x}_{ni} + \beta_p p_{ni} + \varepsilon_{ni},$$

where x is a matrix that includes all the explanatory variables, and β is a vector that includes all the associated coefficients (to focus on the price, we distinguish, within x and β, the price, p, and the price coefficient, β_p, reserving the notation $\hat{x}$ and $\hat{\beta}$ for the remaining explanatory variables, i.e., $x = \hat{x} \cup p$ and $\beta = \hat{\beta} \cup \beta_p$).

What the non-market valuation study ultimately estimates is the probability that an average visitor accepts to pay a given amount of money. For the standard logit model, the probability that the individual n chooses the alternative i is (Train, 2009):

$$\Pr_{ni} = \frac{e^{\beta' x_{ni}}}{\sum_{i \in I} e^{\beta' x_{ni}}}$$

Assuming that the number of visitors at a zero price (initial situation) is known for the particular recreational area considered ($q(0) = Q$), that the price is the same for all visitors, i.e., that $p_{ni}=p$, and normalizing the parameter values in β such that the utility of not visiting the area is zero, the inverse demand function, $q(p)$, for one particular recreational area can be written as (setting $\hat{x}_{ni} = \overline{x}$, i.e., assuming that the model is well specified and that the sample average represents the population well):

$$q(p) = Q\Pr_i = Q\frac{e^{\hat{\beta}'\overline{x} + \beta_p p}}{e^{\hat{\beta}'\overline{x} + \beta_p p} + 1}, \quad \text{or} \quad p = P(q) = \frac{\ln(q/(Q-q))}{\beta_p} - \frac{\hat{\beta}'}{\beta_p}\overline{x}.$$

Finally, assume that the manufactured total costs (essentially cleaning and warden costs) to provide the free access recreational service are given by $C(q)=c_1+c_2q+((c_3)/2)q^2$, where q are the number of visitors, C are total costs and c_i (for i=1,2,3) are parameters. Taking all this into account, the short-run monopolistic competition quantity and price are implicitly given by $R'(q)=C'(q)$, or:

$$\frac{1}{\beta_p}\left(\ln\left(\frac{q}{Q-q}\right) - \hat{\beta}'\overline{x} + \frac{Q}{Q-q}\right) = c_2 + c_3 q \quad \Rightarrow \quad q^{ML}, p^{ML}.$$

If the owner of the terrestrial ecosystem (forest) has market power, as assumed in monopolistic competition in the short run, he/she could, in theory, charge each visitor a different price and internalize all the Hicksian variation of the site-specific demand (or all the consumer surplus if the income effect is small). As noted by Caparrós et al. (2003), this is an excessively strong assumption if the objective is to simulate a real market, as it implies that the forest owner is able to identify the maximum willingness to pay of each visitor, and to charge each of them a different price.

2. Application to free access public recreation in Andalusian forestlands

Andalusia is a region of approximately nine million hectares located in the south of Spain. Because there are important costs involved in providing these recreational services, an economic assessment of the income generated has empirical relevance. Access to protected areas is generally free of charge in Andalusia, at least in public lands. On private land, owners can prohibit access, and visitors have only the right to use public roads and stock driveways. Therefore, a scenario in which visitors would need to pay to access recreational areas is credible.

The site-specific demand functions for public recreation in the different terrestrial ecosystems in Andalusia are estimated using two surveys that were conducted in 2010 (Caparrós et al., 2017). We assume that all 27 areas in which public recreation was observed would operate under monopolistic competition in the short run, with each one encountering its own site-specific demand function. For the 9 areas investigated directly, the demands were derived by using the site-specific dummy variables and the mean from our sample of visitors for the remaining variables. For the remaining 18 sites, we used the site dummy variable of the closest site investigated (or the most similar site) and the mean from our sample of visitors for the remaining variables.

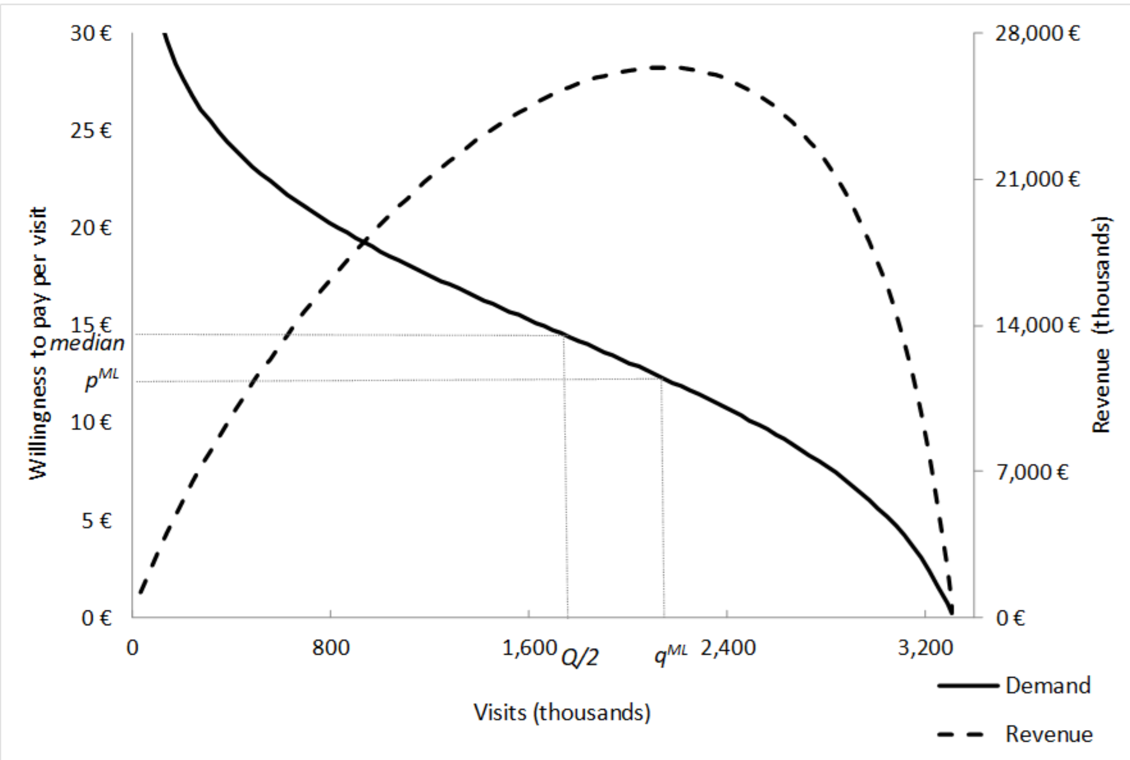

Figure 1. Demand and revenue for recreation in Cazorla (Caparrós et al, 2017)

Concerning costs, only costs paid by the government have been considered. The government assumes direct (using its own production factors) or indirect (through subcontractors) expenditures on activities that affect the supply of private and public products in Andalusian forests. The study focuses on those activities supported by the regional and Spanish governments that are primarily concerned with the provision of public recreational services to free-access visitors. As we were unable to estimate how costs change with the number of recreational visits, because we had only information on costs and number of visits for one year, we assumed that the cost function has only a fixed component and no marginal costs (i.e. $c_2 = c_3 = 0$).

Figure 1 shows the demand function used to estimate the price for free access recreation in Cazorla (one of the areas considered). The figure also shows the values of the price, p^{ML}, and the quantity, q^{ML}, simulated for short-term monopolistic competition (as well as the median, the value accepted by half the population, $Q/2$). The equivalent figures for the remaining 8 recreational areas analysed are qualitatively similar.

Table 1 presents the values for free access recreation using the Hicksian variation and using the SEV estimates; in the latter case, the price that maximizes the revenue under short-term monopolistic competition is used.

Table 1. Aggregated values for free access recreation in Andalucía (year 2010) (Caparrós et al., 2017)

Model and estimated values	**Per visit (€)**	**Number of visits(1)**	**Aggregated values (€)**	**€/ha(2)**
Logit (bid)				
Compensating variation	12.91	26,782,831	345,723,904	78.82
Simulated exchange value *(median as proxy)*	12.91	13,391,416	172,861,952	39.41
Simulated exchange value *(short-term monopolistic competition)*	11.38	15,624,900	177,865,907	40.55
Log-logit (log bid)				
Compensating variation	38.52	26,782,831	1,031,783,830	235.22
Simulated exchange value *(median as proxy)*	15.14	13,391,416	202,712,988	46.21
Simulated exchange value *(short-term monopolistic competition)*	25.31	8,570,506	216,934,005	49.46

(1) Number of visitors are estimated based on our survey to 3,214 adults (≥ 18 years old) from Andalusian households and 836 adults from households located in the rest of Spain (see section 3.1).

(2) The area considered to estimate per hectare values is that of the 27 *monte* sites where we identified free-access visitors (1,867,092 ha).

As Table 1 shows, the Hicksian (compensating) variation implies larger absolute values and, particularly, a larger number of visitors because the SEV only considers the number of visitors that would, in fact, accept a payment. That is, the compensating variation is estimated for the current number of visitors (the observed amount of visitors Q in the current situation without payments), while the SEV values are estimated for the simulated number of visitors (the equilibrium *q*). For the SEV, we propose two measures: one based on the shortcut discussed above of directly using the median and one in which revenues are maximized. For a linear demand function these two values would coincide by definition; however, our results show that for a logistic function both estimations are also similar. For the logit model, this is true both for the price per visit and for the aggregated (or per hectare) values. Although for the log-logit the median and the price that maximizes revenues are not similar, the relevant values from an accounting perspective are the aggregated values, which are relatively similar.

Table 1 also shows that, using the SEV, aggregated (or per hectare) values are also relatively robust to a change in the model used because all the values estimated are approximately between 40 and 50 euros per hectare. In contrast, the estimations based on the compensating variation are very sensitive, at nearly three times higher with the log-logit model than with the logit model.

Table 2 integrates the costs identified and the estimated final production for free access recreation in the production account of the Agroforestry Accounting System (AAS). For details on the AAS, see Caparrós et al. (2003), Campos and Caparrós (2006) or Campos et al. (2017).

Table 2. Ecosystem production account for free access recreation in Andalusia for the year 2010 (in Euros) (Caparrós et al., 2017)

1. Final product	182,849,249
1.1 Public environmental goods and services	177,865,907
1.2 Manufactured fixed capital formation	4,983,342
2. Total cost	44,158,623
2.1. Intermediate consumption	16,020,512
2.2 Labour	20,870,229
2.3 Consumption of fixed capital	7,267,883
3. Net Operating Margin (1 – 2)	138,690,626
3.1 Environmental (ecosystem services)	132,650,688
3.2 Manufactured	6,039,938
4. AAS net value added (2.2 + 3)	159,560,854
4.1 SNA net value added (2.2)	20,870,229
4.2. Non-SNA net value added (3)	138,690,626

3. Discussion

The use of the SEV is considered to be an improvement compared to the alternative of excluding any value for free access recreational services, except the labour cost. At least in the context of national accounting, the SEV is also preferable to the option of estimating welfare measures for both commercial and non-commercial goods and services, in a cost-benefit analysis setting, because this diverges from the national accounting framework. The figures obtained, an estimation of the market value that one could obtain from a given ecosystems service if it were internalized, are also relevant in the policy debate.

One difficulty with the SEV is that part of the visitors would not accept the simulated price. One can simply consider that the number of visitors would be reduced in the simulated market (as we have done above). However, to maintain consistency with physical accounts that may account for the number of visits, one could also consider that the remaining visits have no economic value.

One advantage of the SEV (not shared by the Hicksian variations) is that it is robust to changes in assumptions and models. Table 1 has shown that this statement is true in our application; however, the result will hold more generally because the tails of the probability distribution, in which models differ more, have a relatively small influence on the values obtained by applying the SEV (in contrast to Hicksian variations, which may tend to infinity if the integral does not converge).

Applying this framework at national scale should not be problematic, as the application to Andalusia is as complex as an application to the entire area of Spain would be. Another relevant question is how difficult it would be to reproduce these estimations on an annual basis. A reasonable strategy would be to keep the valuation functions estimated constant for at least five years and run shorter surveys on an annual basis to know the total number of visitors to each area, and their main characteristics (the information on the number of visitors may be available in other countries, but it is not currently available in Spain). Plugging this information into the valuation functions would yield different simulated exchange values for each year.

A consistent application of the SEV for recreation in different countries would also have the advantage of permitting international comparisons. Although certain countries, such as Spain, do not generally charge visitors to national parks and other publicly owned recreational areas, other countries charge visitors for access to their national parks. Moreover, of those charging prices, some apply prices that are designed to extract the largest possible rent from the visitors, as the SEV simulates, whereas others charge symbolic prices that are essentially determined politically. These three different approaches render data collected on the incomes generated by recreational activities difficult to compare. Applying the SEV in all three cases would yield comparable results and, if the estimation is correct, changing from a system in which visitors do not pay to one in which they do should have no impact on the national accounts.

Acknowledgements

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7. Valuing natural capital assets in the context of environmental accounting

Eli P. Fenichel[1]

Clean air, clean water, outdoor leisure, and many other attributes of nature are not privately owned and are not exchanged in markets. Therefore, accounting for their value is challenging. But, few would disagree that nature is valuable, and most would agree the nature's grandeur has value, even if it does not provide a consumptive service (Krutilla, 1967). Moreover, nature provides inputs to many consumptive services, and nature provides some final goods (Barbier, 2013; Daily et al., 2000). However, trying to gage the full value of nature is likely a fool's errand, but valuing changes in stocks and flows from nature is imperative given the importance of macroeconomic indices for evaluating past performance of policy and for social benefit-cost analysis for evaluating specific projects on a microeconomic scale.

Ecosystems and Production

It is undeniable that ecosystems produce goods and services that are essential for life (Daily et al., 2000). Without clean air and water people would perish. This led World Bank economist Mike Toman (1998) to call Constanza et al.'s (1997) total valuation of earth's ecosystems, "a gross underestimate of infinity." The point is that what we can measure, and what we experience, are changes in the quality and quantity of the services that nature provides. The total value is irrelevant, which is good because it is often impossible to measure. On the other hand, asking if economic policy leads to more wealth stored in a given ecosystem or if a specific environmental project will change the real income (Fisher, 1906) of a group of people are important questions. It is the change in value that must be measured. This requires measuring changes in quantities and changes in prices and combining them.

When ecological systems are changed, their capacity to produce goods and services changes. Consider how ecosystems produce goods and services. Boyd and Banzhaf (2007) provide the example of a recreational fishery, an example of outdoor leisure, which is an important ecosystem service. I will build on their example to clarify a key challenge in valuing ecosystems services within an accounting framework. Recall that accounting systems, like the system of national accounts (SNA) or the system of experimental environmental accounts (SEEA), use tables to match quantities to prices. This implies additive separability. However, additive separability is a stumbling block. Now, consider the example of a recreational fishery. The environmental service may be the added value to leisure time. The value of leisure time, in this context, is likely a function of the number and size of fish caught and other attributes of the experience. Certainly, if there is no chance to catch a fish, then allocating time to recreational fishing provides no value (or one is not fishing). On the other hand, a trip where an angler catches four fish is not necessarily twice as good as a trip where she catches two. The marginal value of the fish likely declines as she catches additional fish. The same could be said for the size of the fish, and the "exchange rate" between size and number is likely non-constant. To complicate things further the number (or size) of fish caught is not only a function of the number of fish in the lake. Catching fish requires other inputs, such as a fishing pole, a boat, and knowledge. The sum value of the

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trip is not the sum of the individual parts. Cutting nitrogen pollution in the lake by half will not double the fish population, and even if it did that would not double the number of fish caught. Even if the number of fish caught doubled, the value of the fishing trip would not double. This creates the first problem from imposing a linear structure on ecosystem accounting. Further complicating the additive separability challenge are the kinds of measurements that are often available. Methods for valuing services flows rely on weak complementarity with the overall leisure experience (Freeman, 2003; Phaneuf and Requate, 2017). Seldom is there sufficient variation in the data to estimate higher-order terms that inform nonlinear relationships. This leads to localized constant marginal values – but these values are local and difficult to transfer to other situations.

The second challenge imposed by the linear structure is that one stock, or asset, can produce many services. Consider a reduction in nitrogen pollution in a lake. This can enhance fishing services and swimming services. It is not possible to say that 30% of the nitrogen reduction is allocated to fishing services and 70% is allocated to swimming services. While these challenges are common in business accounting and "finance" (e.g., allocating revenue from a plane ticket that involved a connection across two flights), it is not clear that such an exercise helps solve any of the problems for which we wish to value the environment.

To summarize, the first challenge is that a single service often requires multiple inputs. This is further complicated because the service may require multiple ecological inputs and built or human capital inputs. The second challenge is that a single stock may produce multiple services. The solution to this problem is to focus on the marginal contribution of a capital stock to what Fenichel et al. (2018) call "ecosystem income" – the marginal change in real income associated with the change in the underlying natural capital stocks. Marginal ecosystem income can be measured using a wide array of techniques that respond to changes in quantity, e.g., resource rents, travel costs, hedonics, and even certain types of stated choice measures (see Phaneuf and Requate, 2017 for a detailed discussion of these methods). Focusing on the marginal change in real income derived from a change in the quantity or condition of an ecosystem allows the analyst to measure the asset price and change in asset value of ecological stocks (Fenichel and Abbott, 2014; Fenichel et al., 2016a; Fenichel et al., 2018; Yun et al., 2017). It is possible to develop a seemingly linear index (Dasgupta, 2007) for assets if not for services. Though in fact, the index is only linear in prices but not in quantities.

Prices and Price Curves

Value comes from exchange. Exchange can be between people at a point in time, within an individual through time, or between people across time, often with the use of a trustee for future individuals (Fenichel et al., 2018). It is exchange through time that is the correct notion of exchange to use for pricing capital assets, including natural capital (Fenichel et al., 2018), though sometimes it is helpful to imagine a trustee for future individuals (when an actual trustee does not exist, though it could be argue that this is a legitimate role of government).

The net present value rule provides the value of an annuity that does not change and can be held in perpetuity. The rule can be extended to prices for an annuity that has a constant marginal price, such that the marginal price does not change with the amount of the stock.

If stocks (and their complements and substitutes) are not changing, then there is little reason to be concerned with their value. When stocks are changing the net present value rule needs to be modified for capital gains (when capital gains reflect changes in scarcity rather than arbitrage profits) and for asset appreciation or depreciation. Prices must be adjusted for the marginal effect of an additional unit of stock on real income – in the case of ecosystems this is the marginal ecosystem income. Fenichel and Abbott (2014) generalized Jorgenson's (1963) approach to pricing invested assets to natural capital assets and develop a numerical approximation approach. Yun et al. (2017) extend Fenichel and Abbott's (2014) approach to allow for interacting ecological stocks and show how ecological interactions can affect the asset value of natural capital.[2] Importantly, the extension showed that because components of an ecosystem interact, ecosystem management provides excess returns ("alpha") beyond a single species management benchmark. Indeed, nonlinear ecological interaction and nonlinear human-environmental interactions influence the change in value and return on investment. These interactions can be captured by asset prices.

There are two important features of the Fenichel and Abbott approach to pricing and valuing natural capital. First, the Fenichel and Abbott approach generates a price curve rather than a single point estimate of a price. Changes in the area under the price curve can be used to calculate changes in value associated with changes in the value of asset holdings. Fenichel and Abbott price curves are welfare theoretic and provide exchange prices. This makes Fenichel and Abbott price curves appropriate for SEEA and other national accounting applications and for social-benefit cost analysis. There is no difference between welfare and exchange prices. I will return to this point below.

Second, the Fenichel and Abbott approach goes beyond a conceptual approach to valuing natural capital. It provides a theory of measurement that can be operationalized computationally using data. Yun et al. (2017) introduced the R package capn (capital asset pricing for nature), https://cran.r-project.org/web/packages/capn/index.html for implementing numerical techniques to measure natural capital asset prices.[3]

A key challenge with ecosystem service valuations is that concepts of potential services from ecosystems greatly exceed data. The analyst needs to observe at least one management decision per ecosystem service to measure the value of the flow of ecosystem services from data. Moreover, the multiple decisions and services must be uncorrelated. While the same is true for natural capital, the challenges of multiple products per ecosystem type makes valuing ecosystem stocks substantially more tractable than valuing service flows piece meal. Moreover, because ecosystem services are produced and enjoyed in ways that are not additively separable, focusing on changes in the underlying wealth stored in the stock seem less likely to lead to double or under counting.

Welfare triangles vs exchange rectangles

The challenge of measuring values when quantities and prices can change is the problem of creating a Divisia Index. Fleurbaey and Blanchet (2013) review index number theory within the context of accounting for non-market goods and services.[4] There is no absolute

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[2]Current research efforts are extending these methods for resource dynamics that involve correlated and uncorrelated stochastic dynamics (Abbott et al., 2018).

[3]Currently two worked examples are included with capn and an additional example is available at https://github.com/efenichel/capn_stuff.

[4]Fleurbaey and Blanchet also provide a review of attempts to include the environment and nonmarket goods

value index that solves this problem, but indices can be helpful in measuring changes from a baseline. The national accounts use the notion of exchange value (Obst and Vardon, 2014), which is a price (p) that comes from exchange times the quantity (q) exchanged to create an index of value.

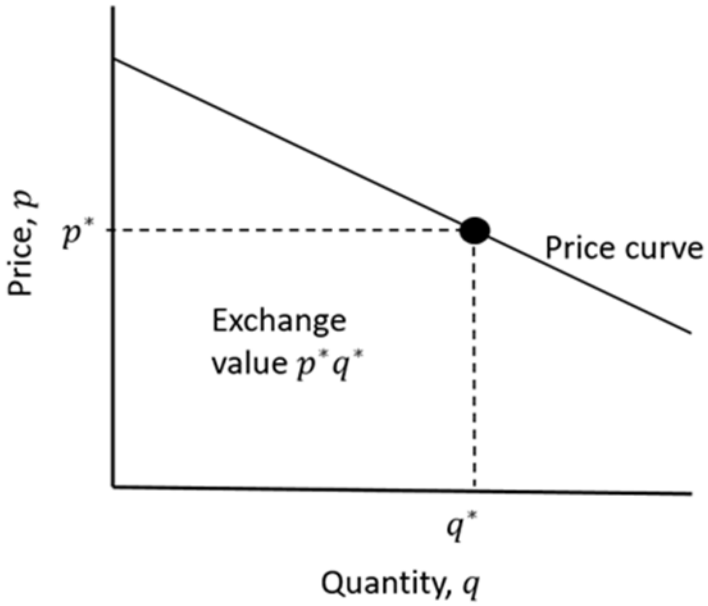

Figure 1. Graphical representation of exchange value (own representation)

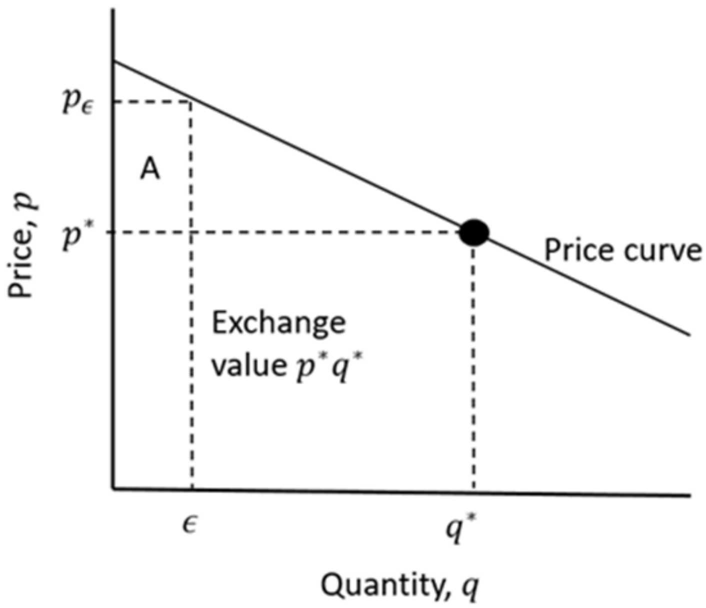

Figure 2. Graphical representation of the problems with total exchange value (own representation)

The exchange value index often takes the prices and quantities at a point in time, such that

and services into national accounts. They argue the problem is not finding a substitute for GDP, but that there are many substitutes for GDP without a clear leader or compelling purpose.

exchange value (EV) can be written $EV_t = p_t \times q_t$, creating a rectangle (Fig 1). But, the real question is the value exchanged over a specific accounting period (Fisher, 1906). The question is how to account for changes in prices and quantities over the same period. $\Delta EV \neq \Delta p \times \Delta q$; graphically the difference of two rectangles (Fig 2). A heuristic explanation of why this is the case is gained by letting the change, $\Delta \to 0$. $\Delta EV \neq \Delta p \times \Delta q$, because the product rule from calculus would imply $\Delta EV = \Delta p q + p \Delta q \neq \Delta p \times \Delta q$. The way to address this is to hold either prices or quantities constant over the period. By convention, the goal is to find a price that allows us to account for the change as if $\Delta p = 0$. However, the result is more complicated because the price of q measures the scarcity of q. Therefore, p is a function of q, perhaps along with the quantities of other stocks (see Yun et al., 2017 for more details on this point). Consider a change from q^* to $q = \epsilon > 0$ (Fig 2). Notice that price of q at a quantity of ϵ is much greater than the price of q and q^*. In order to lose all of q, one must first lose $q - (q - \epsilon)$ and then lose ϵ. But, $p^* q^* < p^*(q^* - \epsilon) + p_\epsilon \epsilon$. Area A in Fig 2 must be included, but the fact that we re-measure along the path of losing all q should not impact the total value of q. Yet, this is what would be required for ΔEV to be $p_t q_t - p_{t+\Delta} q_{t+\Delta}$. The exchange value index concept, as an absolute index – where prices and quantities are measured at exactly time t and $t + \Delta$, is flawed. It is not possible to construct a time series of absolute exchange value indices and track how they change through time, which is a key objective of accounting.

Obst and Vardon (2014) , the SEEA central framework, and the SEEA ecosystem accounts contrast welfare values with exchange values. Welfare values, WV, are defined as the area under the price curve (Fig 3).[5] If the good is provisioned free of charge up to point q^*, then the welfare value is a triangle (areas A+B in Fig 3) plus the exchange value rectangle. Total consumer surplus is interpretable as a gain in welfare from $q = 0$ to q^*. The graphical approach to welfare values does not suffer the same consistency problems as the approach to exchange values. However, the SNA and SEEA accounts focus on exchange values not welfare values seemingly creating an alignment challenge. However, it is important to remember that there is no conflict between the exchange and welfare price concepts.

Change in exchange value is equal to change in welfare value under certain conditions (Fenichel et al., 2016b). Indeed, exchange price curves and welfare price curves are the same when measured correctly (Fenichel et al., 2018). The challenge is not the price concept but the value index. To make change in exchange value and change in welfare value indices equivalent recognize that the change must be valued at a weighted average price, $\bar{p}$, before and after the change – that is accounting is done at constant prices so that $\Delta p = 0$ over an accounting period (Arrow et al., 2003). In Fig 3, let p_c be the choke price of q, which is the price associated with the first unit of q and $\bar{p} = \frac{1}{2}(p^* + p_c)$, which is the arithmetic mean of p^* and p_c. In this case, the correct measure of the change in exchange value is the notion of the exchange value at time t plus the area B+C. This is also the correct change in welfare value because area A and C are identical.

[5]Technically, welfare values are the area under the Hicksian or compensated demand (price) curve (Freeman, 2003). Consumer surplus is often a good approximation of welfare values (Freeman, 2003). But, consumer surplus is only an approximation because consumer surplus is the area under the uncompensated, Marshallian, demand (price) curve. Harberger (1971) points out that a change in consumer surplus is a second-order approximation to a welfare change.

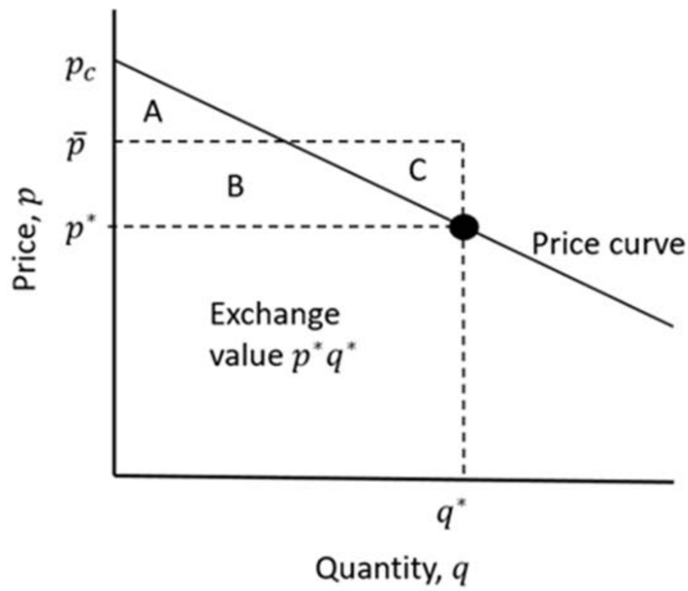

Figure 3. Graphical representation welfare value and changes in exchange and welfare value (own representation)

If q is an essential good, then the price of q goes to infinity as the quantity goes to zero (Fig 4). This means that there is no choke price so $\bar{p}$ for the total value is undefined, but changes in value between two finite quantities can still be found. For example, if the q is drawn down from level q^* to level ϵ, then an approximation of $\bar{p} = \frac{1}{2}(p^* + p_\epsilon)$. In this case $\Delta EV > \Delta WV$, because area C > area A. However, this still might be a reasonable approximation (see Yun et al., 2017 for a worked example of such a case) depending on the curvature of the price function and size of the change in q, which are empirical questions. These properties led Harberger (1971) to call net national product a first-order welfare approximation (using mean prices) and consumer surplus a second order approximation. Of course, we might be interested in finding the point where the first order approximation equals the second order approximation. Fenichel et al. (2016b) explain that with information about the price curve, i.e., its elasticity, it is likely possible to find the weights on the before and after prices to improve the match between ΔEV and ΔWV. If the price curve is convex to the origin, then the change in exchange value will always exceed the change in welfare value. Perhaps this is an argument for allowing changes in welfare value associated with changes in stocks of natural capital to be used in the accounts, where consumer surplus is otherwise excluded.

Discussion

The greatest challenge with merging macroeconomic indicators and national accounts for understanding wellbeing is that the chief indicator from the national accounts is gross rather than net domestic product (Barbier, 2013).[6] Kuznets (1973) knew that Gross National Product (GNP) or Gross Domestic Product (GDP) would be woefully inaccurate for econo-

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[6]It is well known that GDP is not a measure of social wellbeing (Nordhaus, 2006; Stiglitz et al., 2010) and most of the theory has focused on when NDP can be a measure of social wellbeing or welfare (Dasgupta, 2007; Weitzman, 1976).

mies that relied heavily on the environment, but what Kuznets likely failed to appreciate is how dependent even industrialized economies can be on the environment (Carbone and Smith, 2013).

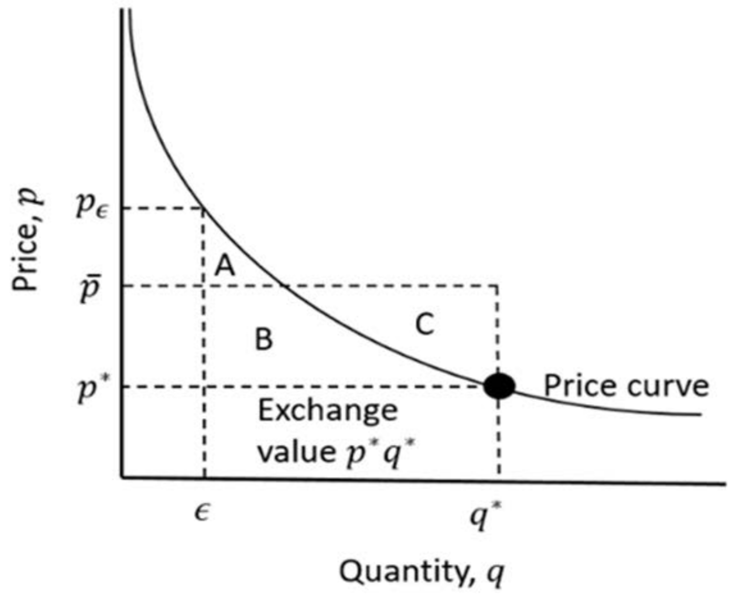

Figure 4. Graphical representation changes in exchange and welfare value for an essential good (or a good with a price curve that is nonlinear) (own representation)

A shift toward net accounting, e.g., Net National or Domestic Product, would be a substantial advance. The primary challenge in a shift towards net accounting is adjusting for capital depreciation or appreciation. Fortunately, there are robust methods for measuring the changes in the value of natural capital assets. This is fortunate because directly measuring the changes in ecosystem income can be challenging because of the additive separability challenges imposed by accounting frameworks. The remaining challenge is to accept that we will not know the total value of the environment – or rather we already know it is infinite. But, that does not help us make decisions or track performance. No matter how much society pollutes the water, the value of the last drop of clean water is infinite. But, we would like to know how much better off society is or how much wealthier society is when the water is cleaner – a change not a total.

The issue of changes in wealth is complicated by general equilibrium effects associated with environmental management (Fenichel et al., 2018). In most market transactions, markets are thick and the actors are price takers. However, management of the environment affects everyone and the whole purpose of managing the environment is to affect scarcity of the environment and hence prices – along with quantities. Thus, society is not a price taker when it comes to the environment. Therefore, a change-in-value index must be capable of handling price and quantity changes.

In this paper, I have addressed critical concepts related to valuing the environment and specifically natural capital. But, what do we expect prices for natural capital assets to look like? My experience is if we currently think the state of natural capital is a state of disrepair, then we would expect nature's revealed asset price to be low, perhaps surprisingly low. This is my experience with fisheries (Fenichel and Abbott, 2014) and groundwater (Fenichel et al., 2016a). These low values signal the potential for high returns on investment, much like a rundown house in a nice neighbourhood. On the other hand, high values may signal overinvestment, too strict conservation, or a large value at risk. It is inconsistent to think there is a need for massive ecological restoration and current asset prices for the environment are relatively high. Rather, the intuition must be there exists an opportunity for a large return on investment, which would likely be realized through changes in prices and quantities.

Benefit-cost analysis for specific projects or programs drove the development of environmental valuation. Values for accounting only reflect valuations of the current and past states of the environment or changes between realized time periods. The point of accounting is to create a time series to assess progress. Thus, we must be careful with respect to what is being valued. Most environmental valuation values a change in a policy that impacts the environment not the environment itself (Freeman, 2003). The values in environmental accounts can be used as a baseline in social benefit-cost analysis, but the change in benefit is often achieved with a hypothetical improved state (assuming the program would pass the benefit-cost test). The point of environmental accounting is not a benefit-cost analysis on a single project, but rather to track how a suite of policies, including macroeconomic policies, are affecting the state of the world. Doing so, will hopefully lead to a more comprehensive policy making process that pays closer attention to environmental impacts.

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8. Cultural Ecosystem Services Assessment within Natural Capital of Novokuznetsk Municipal District, Kemerovo Region, Russian Federation

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Anastasia Mikhailova[4]

Summary

Ecosystems are an essential source of human well-being as they produce a lot of ecosystem services. In the regions with developed mineral resource use, preservation of vital ecosystem services is connected with finding a compromise between two sources of territory development: abiotic services (mineral resources) and ecosystem services, maintaining well-being and a comfortable environment for the local population. In this article, cultural and other ecosystem services of Novokuznetsk district in Kemerovo region of the Russian Federation were assessed and compared with the abiotic services. The article also presents the results of the economic assessment of ecosystem and abiotic services as they are an important element of analysis of sustainable development of the territory.

Keywords: sustainable development, natural capital, ecosystem services, abiotic services, economic evaluation

1. Introduction

The concept of ecosystem services was first implemented as the official basis for sustainability in 1997 by R. Costanza [1] and G. Daily [2]. Nowadays this concept is essential for the development of environmental economics and the sustainable development of territories. An important step towards the recognition of the fact that human communities depend on natural ecosystems was the identification of interrelations between biophysical aspects of ecosystems and human well-being through the concepts of natural capital and ecosystem services [3; 4-6]. This contributed to the fact that ecosystem services were included in the system of environmental-economic accounting (SEEA) for the first time in 2014 [3; 7]. This approach allows the creation of information and analytical support for the solution of two equally important tasks: maintenance of ecosystem structure and functions (the capacity of ecosystems to recover) and reduction in the use of ecosystem resources in production and consumption, as well as reduction in relevant environmental impact [8-12].

Successful integration of these tasks into the decision-making process of territory development requires spatial information about supply and demand for ecosystem services [13-15]. Assessment in monetary terms is used as an essential tool for transferring information on the importance of ecosystems to the decision-makers, thereby increasing their awareness. The reason for this is the inclusion in the management process of those ecosystem services which can be assessed in market prices, while most of the ecosystem services are often not taken into account of the market scope [16-19]. In fact, market failures, related to ecosystem services that are public goods, can lead to increasing pressure, providing short-

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term economic benefits to some stakeholders at the expense of the long-term decline in the well-being of the majority of others [20; 21].

Unlike other ecosystem services, cultural ecosystem services are non-material benefits people obtain from ecosystems through "aesthetic enjoyment, recreation, artistic and spiritual fulfilment, and intellectual development." [22] Therefore, the distinctive feature of cultural ecosystem services is intangibility, which is considered the reason for the difficulty of their assessment [22, 23, 24].

In the Russian Federation, most studies relate to the assessment and analysis of ecosystem services in biophysical indicators [25-30]. Research experience of evaluation of ecosystem services in Russia is mainly attributed to the evaluation of cultural ecosystem services of specially protected natural areas [31-34], as well as to accounting and monetary assessment of environmental resources of the Russian Federation within SEEA [35; 36].

As ecosystem services are generally closely interrelated, optimizing the use of one type of service may affect other services [37]. That's why any ecosystem management options in a territory inevitably are connected with compromises. This study presents an attempt to develop mechanisms for the search of such compromises and to integrate results of the economic assessment of ecosystem and abiotic services into the processes of strategic territory development planning.

2. Initial data and methods

2.1 Scope of the research

Figure 1 presents the general information on the evaluated area of Novokuznetsk municipal district, Kemerovo region, in the context of the main types of ecosystems.

Within the research, ecosystems of the area were divided by cultural ecosystem services they provide (table 1).

Table 1. Ecosystems, cultural ecosystem services and benefits

Ecosystems	**Services**	**Benefits**
Forest lands, floodplain areas and water bodies	Outdoor recreation	Possibility for fishing, hiking, swimming, etc.
Forest lands, farmlands, meadows, floodplain areas	Hedonistic values	Environmentally favourable location of residential property and human habitation.

All ecosystem services and abiotic services together equal to the natural capital of the territory [3; 7]. The importance of accounting all those services results from the need to determine the balance of interests between ecosystem and abiotic services, evaluate alternative land use options and choose directions of territory use that are more relevant to the objectives of its sustainable development.

The distinction between these two types of benefits leads to the difference in approaches to assessing the economic value of the ecosystem and abiotic services [3; 7].

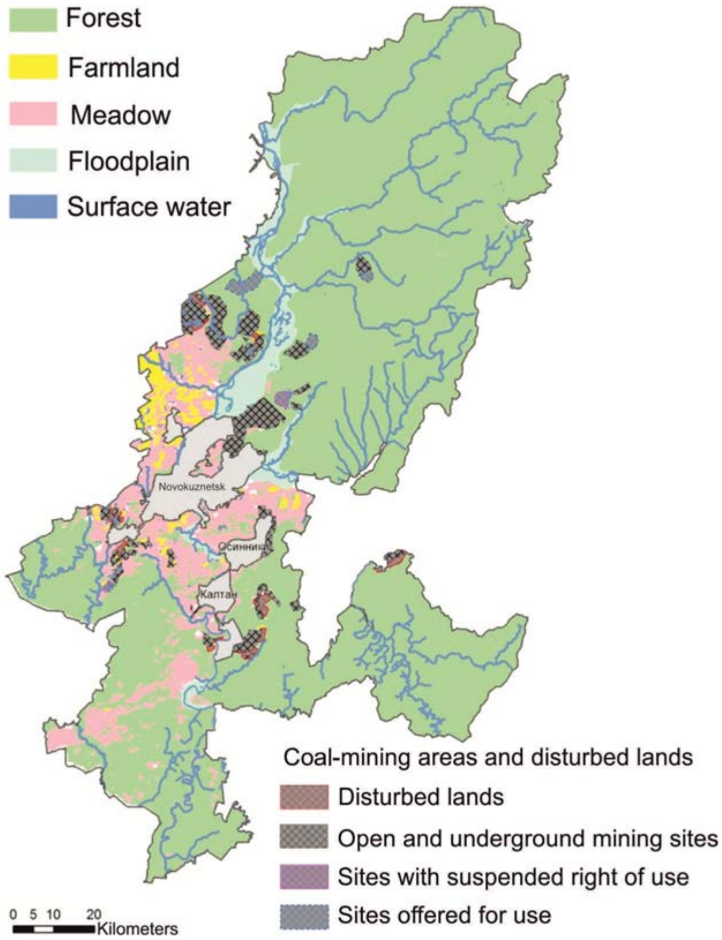

Figure 1. Main ecosystems and coal-mining areas in the Novokuznetsk district ([38]).

2.2 Assessment structure

In terms of the economic assessment of the impact on human welfare, the benefits from ecosystems can be divided into the gains from services:

— that are used or controlled by economic units and sold in markets (e.g., food, water, clothing, housing services, non-timber forest products, recreational services etc.);

— that are directly used by consumers (individuals) and that are not included in the services controlled by economic units (e.g. clean air).

2.2.1 Economic value of services, whose use is somehow connected with purchase and sale (provisioning ecosystem services and abiotic services – coal mining), was calculated either as producer's profit or as the value of consumer surplus.

The value of the producer's profit was calculated by the formula:

$$PP = MP - (PS - P_{pr}), \qquad (1)$$

where:

PP — producer's profit[5] from the service;

MP — market price for a service used by a consumer;

PS — producer's spending on service delivery to the consumer;

P_{pr} — payments by a producer in favour of the resource owner (the State) for the actual resource use.

The value of consumer surplus was calculated by the formula:

$$CS = WP - CE, \qquad (2)$$

where:

CS — consumer surplus, i.e. the consumer surplus for ecosystem services in the form of savings, which he would be willing to pay for the service, but for which he actually didn't have to pay;

WP — the sum of consumer willingness to pay for to use the service;

CE — actual consumer expenditure for using the service.

The value of WP received by the subjective assessment method, based on surveys in which people are invited to say how much they would be willing to pay for specific ecosystem services [39]. Value of CE is determined by expert method, using the results of population surveys.

2.2.2 Ecosystem services, whose use is not connected with purchase and sale (cultural and regulating ecosystem services), were evaluated using such methods as:

— estimates of consumer surplus (CS). In this case, the value of CS is equivalent to the value of WP, i.e. the sum of the willingness of the consumer to pay for saving the opportunity to use and/or for the use of the evaluated service. The value of WP is calculated by the results of generalization and the analysis of the data obtained by subjective evaluation [37; 40; 41];

— transfer value, when the values of ecosystem services or ecosystem assets can be extrapolated to other territories [3; 7; 42; 43]. The source data for the transfer values were based on the results of prior empirical studies of the economic value of ecosystem services. As the quality of the initial research always determines the overall quality and boundaries of the final assessment [44], the main attention was paid to studies that have been conducted in regions with similar researched area geographical conditions.

[5] Under this scenario Producer refers to the legal entity providing the conditions for use of ecosystem services by the consumer (for example, a wood supplier, a recreation organizer, a fish seller etc.).

Table 2 contains information on the main features of cultural ecosystem services assessment.

Table 2. Main features of cultural ecosystem services assessment

Services	**Value type**	**Nature of benefits**	**Assessment method**
Outdoor recreation	Value of indirect use	Non-market benefits	Value judgement method. The assessment on the basis of the data analysis of the people's willingness to pay for maintaining the possibility of using recreational functions of the district's ecosystems. The initial data were obtained from a questionnaire survey of households in rural settlements.
Hedonistic values	Value of indirect use	Market benefits	Value transfer method. The search and analysis of data on ecosystems with similar characteristics and indicators of their hedonistic values, for the development of specific indicators of hedonistic values of ecosystems in the Novokuznetsk region.

2.3 Data sources

Assessment of provisioning ecosystem services by the formulas (1) and (2) were based on the data provided by statistical, natural-resource and sectoral departments of the Administration of Novokuznetsk municipal district of the Kemerovo region, as well as data of the regional markets, results of surveys of the district population, expert assessments.

Assessment of regulating ecosystem services (regulation of climate and air composition, regulation of water resources, assimilation of waste, wildlife conservation, soil formation, pollination), and assessment of cultural ecosystem services (hedonistic values) were based on the value transfer method and specialized online databases: EVRI (http://www.evri.ca); Envalue (http://www.environment.nsw.gov.au/ envalue); Value base Swe (http://www.beijer.kva.se/valuebase.htm); Environmental & Cost Benefit Analysis News (http://envirovaluation.org); Econ Papers (http://econpapers.repec.org).

The assessment of cultural ecosystem services in terms of outdoor recreation by value judgement method was based on the results of data analysis on the local population willingness to pay for conservation of forest and water ecosystems as recreational areas.

The assessment of abiotic services (coal-mining) by the formula (1) was based on the data provided by the Department of Industry, Transport and Entrepreneurship of the administration of Novokuznetsk municipal district of the Kemerovo region.

3. Results and discussion

Table 3 presents the total value of the annual economic value of ecosystem and abiotic services provided in the territory of Novokuznetsk district.

Table 3. Economic value of ecosystem and abiotic services in Novokuznetsk municipal district, million rubles per year ([38])

Source of economic value	Forests	Farmlands	Meadows	Floodplain areas	Surface water bodies	Coal mining sites	Total
Regulating ecosystem services							
Regulation of climate and atmospheric composition	7854.3	-	49.1	1010.9	-	-	8914.3
Regulation of water resources	-	-	36.8	11409.8	-	-	11446.6
Assimilation of wastes	6363	-	785.4	6314	-	-	13462.4
Wildlife conservation	133325	2008.8	-	383.6	-	-	135717.4
Soil formation	696	-	1435.8	-	-	-	2131.8
Pollination	23364.2	44.9	233.2	-	-	-	23642.3
Total	171602.5	2053.7	2540.3	19118.3	-	-	195314.8
Cultural ecosystem services							
Outdoor recreation *	4.6	-	-	-	0.1	-	4.7
Hedonistic values*	13532.1	36.6	257.7	1882.8	414.1	-	16123.3
Total	13536.7	36.6	257.7	1882.8	414.2	-	16128
Provisioning ecosystem services							
Timber*	25.9	-	-	-	-	-	25.9
Non-timber forest resources *	35.8	-	5.8	2.4	-	-	44.0
Water resources	-	-	-	-	0.8	-	0.8
Hunting resources*	0.4	0.01	0.05	0.02	-	-	0.5
Fish resources	-	-	-	-	1.1	-	1.1
Agricultural products	-	117.0	862.8	-	-	-	979.8
Total	62.1	117.01	868.66	2.43	1.9	-	1052.1
Abiotic services							
Coal	-	-	-	-	-	14225.3	14225.3
Total	185201.3	2207.3	3666.6	21003.53	416.1	14225.3	226720.2

*Value of cultural and provisioning ecosystem services (timber, non-timber forest resources and hunting resources) for forests is given excluding Kuznetsky Alatau nature reserve.

The assessment showed that 82% of the annual value of natural capital in Novokuznetsk district is produced by forest lands, more than 9% - by floodplain territories, more than 6% - by coal mining areas. The minimum value of ecosystem services is taken by surface water – 0.2% of the value of natural capital of the area.

Significantly, the value of provisioning ecosystem services and abiotic services in the total economic value of natural capital is 7%, while regulating and socio-cultural ecosystem services are 93% of the economic value of natural capital. Moreover, the value of cultural ecosystem services is comparable to the value of abiotic services.

Spatial visualization of the value distribution for ecosystem and abiotic services of Novokuznetsk municipal district was prepared according to the general plans of rural settlements and the results of interpretation of multispectral satellite imagery and processing of raster maps and vector data in the software package ENVI[6] (figures 2, 3 and 4).

Comparison of figures 2, 3 and 4 shows that the value of ecosystems in undisturbed areas is higher than the value of coal-mine sites by several orders of magnitude.

Ecosystem services are renewable, and while ensuring wildlife conservation, agricultural and forestry development in the district, they perform an important social function of providing households with opportunities for additional employment. At the same time, coal resources are non-renewable, as a consequence of mining, they are gradually depleted, and the ecosystems under mining degrade and lose their capacity to generate ecosystem services (the most shaded areas in figures 2 and 3, and the lightest areas in figure 3, respectively).

The results of the research allowed identification of opportunities and directions for the following tasks: (1) increasing manager's awareness of economic benefits of the ecosystems in the area when making decisions on strategic planning and current management, (2) choosing optimum directions of land use in the district in terms of sustainable development.

Thus, the choice of optimum directions of land use in terms of sustainable development of the district is connected with the recognition that intact ecosystems are of considerable economic value, and their preservation has both environmental and economic benefits for the sustainable development of Novokuznetsk district. Besides, it's necessary to account and analyse ecosystem values within strategic planning of territory development in the framework of the standards of environmental-economic accounting [7].

[6]http://www.harrisgeospatial.com/SoftwareTechnology/ENVI.aspx

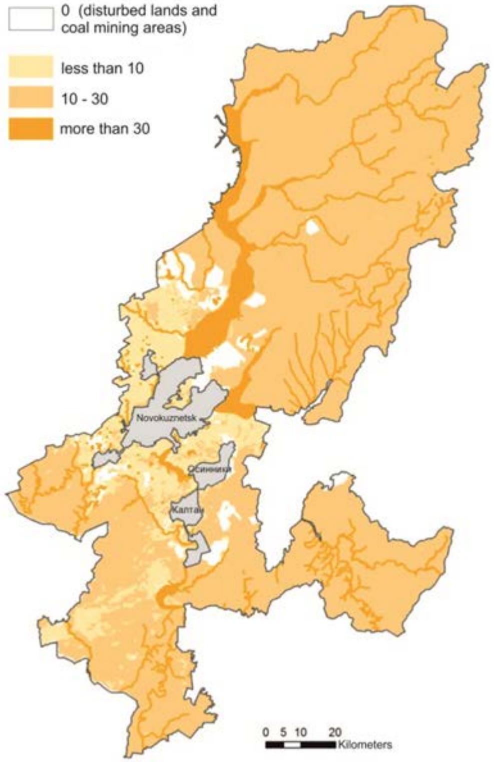

Figure 2. Value of cultural ecosystem services thousand rubles/ha/year ([38])

Figure 3. Value of regulating ecosystem services, thousand rubles/ha/year ([38])

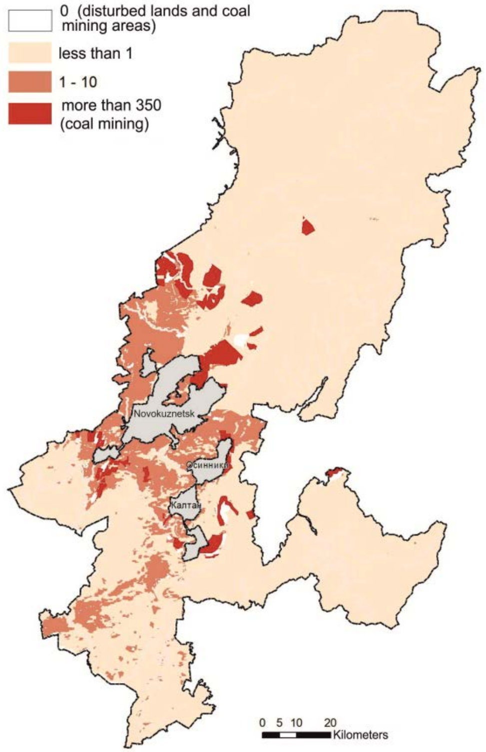

Figure 4. Value of provisioning ecosystem services and abiotic services, thousand rubles/ha/year ([38])

4. Conclusion

The research showed that different ecosystems in Novokuznetsk municipal district, Kemerovo region, provide a wide range of ecosystem services, whose benefits are a large part of natural capital in the area. The comparison of benefits from ecosystem services and abiotic services has been useful for understanding the necessity of a joint search for compromise to ensure the ecosystems conservation of the area and its sustainable development in the conditions of coal mining.

Unlike abiotic services of coal-mining, cultural ecosystem services have more sustainable over time employment potential for the local population and plays an important role in the economy of rural households, maintaining human well-being with local ecosystems. Identification and assessment of cultural ecosystem services increases interest of the local population and authorities in the preservation of intact ecosystems, biodiversity, monuments of nature and culture.

All in all, development of accounting, assessment and mapping of physical and monetary characteristics of ecosystem and abiotic services allows expanding information-analytical framework of decision-making in strategic territory planning, improving their performance in terms of ecosystems conservation and region's sustainable development.

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9. Two valuation approaches in ecosystem services accounting: The cases of crop pollination and water purification in Europe

Alessandra La Notte[1]

1. Introduction

Ecosystem services (ES) are the contribution of ecosystems to human wellbeing and they can end up in SNA (System of National Accounts) and non-SNA benefits. This definition highlights that (i) ES end up in benefits but differ from benefits (for a more detailed analysis ref. La Notte et al. 2017a), and (ii) the relationship with (specific) SNA products varies according to the ES that are assessed.

Accounting for ES (from a national accounting perspective) requires to be consistent with the SNA in both the accounting mechanism and rules, and along the process-product chain. For the sake of consistency, it is thus important: to avoid double accounting where the ES is embedded in SNA products, but also to avoid oversimplification by taking the SNA product as a proxy of the service.

When there is a direct and clear linkage between ES and SNA products, a way to monetize ES might be to disentangle the ES value. No alternative valuation technique would be employed: the market price already recorded in official statistics is the basis from which to assess the ES contribution.

There are other cases where the linkage is neither direct nor clear. When translating biophysical outcomes in monetary terms, SNA compliant valuation techniques should then be applied in order to build accounting tables expressed in a common monetary unit (UN et al 2014, UN et al. 2017).

In this paper two examples are provided: crop pollination is an ES that directly links to SNA pollination dependent crops, and water purification is an ES that (although massively dependent from the agricultural sector) does not directly link to a SNA product. For each ES the valuation approach is briefly introduced, and finally some conclusions are summarized.

2. Ecosystem services that directly link to SNA products: the example of crop pollination

Crop pollination is a regulating ecosystem service defined as the fertilization of crops by insects and other animals that maintains or increases the crop production. There is growing concern that observed declines in insect pollinators may affect production and revenues from pollinator-dependent crops. Knowing the distribution of pollinators, therefore, is crucial to estimate their availability to pollinate crops. This information, in turn, can be used to ensure the maintenance of habitats that support insect pollinators, ultimately safeguarding the long-term provision of crop pollination services.

Accounting for crop pollination requires the assessment of the ecosystem potential to support wild insect pollinators (pollination potential) and the demand for pollination, which, in this case, is defined as the extent of pollinator-dependent crops. Then, the spatial overlap between the pollination potential and the demand for pollination is used to estimate the actual flow of the service.

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The assessment of pollination potential is based on an indicator of the environmental suitability to support wild insect pollinators. The environmental suitability is, then, used to delineate service providing areas (SPA) showing different level of pollination potential (i.e. high, medium, low and none). Pollination potential integrates two different models: an Expert-Based Model for solitary bees (Zulian et al., 2013) and a Species Distribution Model for bumblebees, predicted with observed species records (Polce et al., 2013). Both models are based on land cover, climate data, and on the distance to semi-natural areas. From the environmental variables available to assess the pollination potential, climate is the most important driver of the large-scale occurrence of the groups of pollinators considered here. Land cover is the second most important driver, but its relative importance differs among the taxonomic groups, reflecting their ecological requirements. However, given its importance, there is a large potential of well-designed land management strategies to mitigate the increasingly negative effects of climate change (Potts et al., 2015).

The demand for crop pollination is quantified as the extent of pollinator-dependent crops, following the methodology described in Zulian et al. (2013). Spatial data derived from the CAPRI model (Britz & Witzke, 2014; Leip et al., 2008) are used to quantify the demand as the number of hectares per square kilometre. The crop types benefitting from insect pollination to different extent are ten. The overlap between the pollination potential and demand for pollination is used to quantify the area generating the actual flow of service: the use area. In this way, the use area is defined as the extent of pollinator-dependent crops benefitting from the SPA with different pollination potentials.

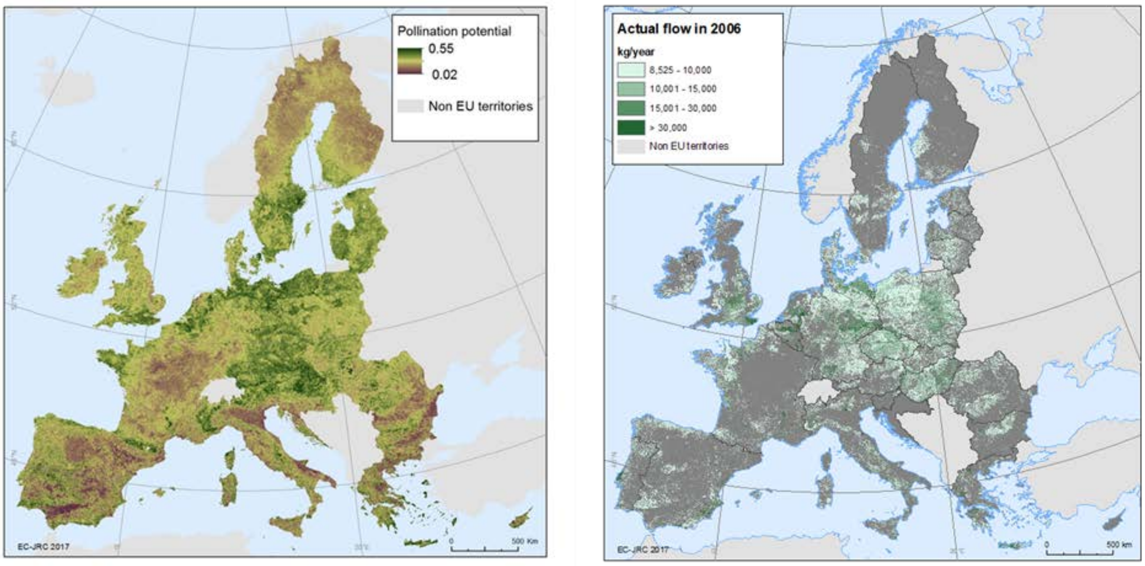


Figure 1. Crop pollination potential and actual flow in physical terms for the year 2006 in EU (Vallecillo et al., 2018)

The starting point for the monetary valuation of crop pollination is the economic account reported for agriculture within the SNA. From the total production expressed in constant monetary values, we estimate the contribution of the ecosystem service (pollination) (i) by separating the pollinator-dependent crop production covered by pollination service from the pollinator-dependent crop production not covered by pollination service, and (ii) by disentangling the contribution of the ecosystem service from the former.

In order to use consistently the official agricultural statistics made available by ESTAT, we first need to move from the actual flow processed using CAPRI data to the actual flow expressed in ESTAT data. There are two sets of information we withdraw from the data processed using the CAPRI model as source: (i) the actual flow, i.e. the tons of met demand multiplied by the dependency coefficients (Klein et al., 2007), (ii) the total production including both met and unmet demand. We obtain a pollination ratio whose amount depends on the way the biophysical side was undertaken (because of the actual flow).

$$Pollination\ Contribution = CAPRI\ Actual\ Flow\ /\ CAPRI\ Total\ Production$$

The pollination contribution expresses how much of the total production depends on pollination: it is not only necessary to have the dependency coefficients, it is also necessary to know how much of the crop demand for pollination is actually met. In fact, when looking at the outcomes obtained by applying the pollination contribution, it becomes clear that the application of the dependency ratio on all production might in some cases results an overestimation of the service that hides sustainability issues.

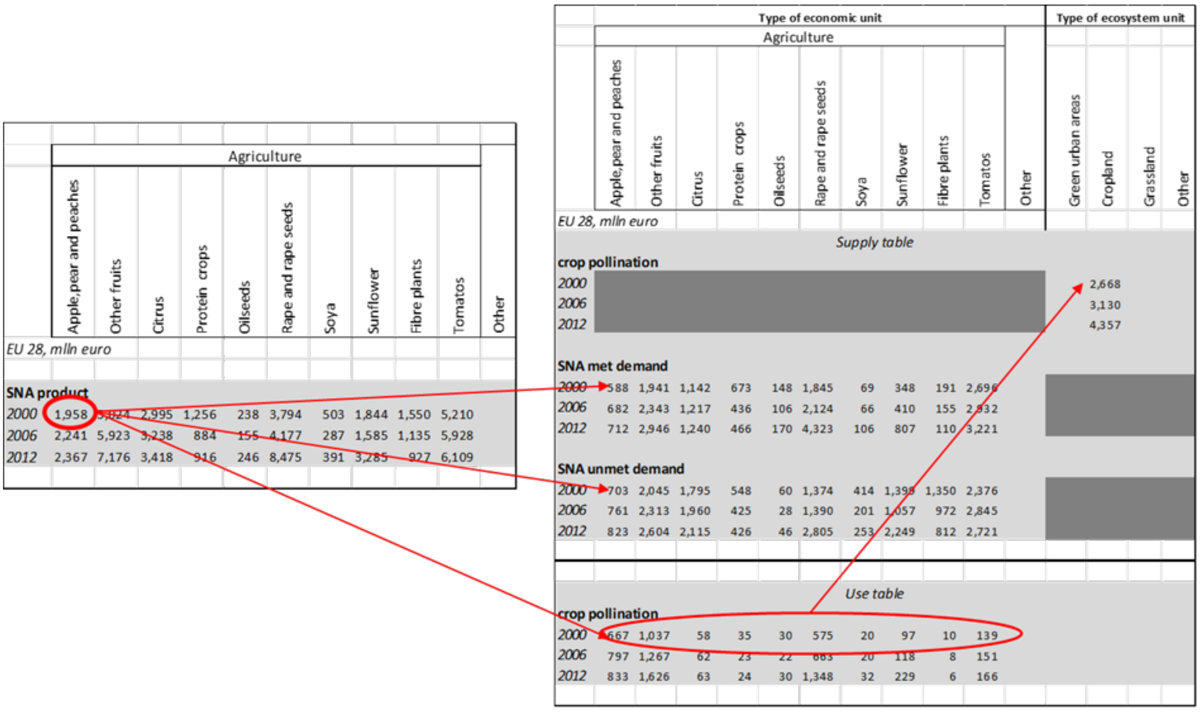
Once the pollination contribution is available, it is multiplied by the agricultural statistics provided by ESTAT in order to estimate the part of met demand, which depends on the action of wild pollinators:

$$ESTAT\ Actual\ Flow = Pollination\ contribution * ESTAT\ Total\ Production$$

ESTAT total production can be calculated in physical terms when tons of yields are considered. In that case, the following step is to multiply the flow by euro/ton.

Figure 2 shows how from the standard agricultural statistics it is possible to first separate the amount of production covered by the pollination service (met demand) from the amount not covered (unmet demand) and second to disentangle the contribution from cropland (the service flow) to the 10 pollination dependent crops.

The unmet demand of crop-pollination highlights that there is room to enhance crop pollination. This could generate higher production and/or more sustainable production practices in countries where pollinator-dependent crops do not receive enough crop pollination service. To invest in creating habitat suitability for crop-pollination could in fact: increase crop production and/or reduce the human factors (especially chemical fertilizers) in the production process by keeping the same amount of production. The two options vary according to the characteristics of different areas and to the current management practices currently in place. For more details on the crop pollination service, please refer to the JRC Technical Report (Vallecillio et al., 2017).



EU 28, mlln euro	Agriculture: Apple,pear and peaches	Other fruits	Citrus	Protein crops	Oilseeds	Rape and rape seeds	Soya	Sunflower	Fibre plants	Tomatos	Other
SNA product											
2000	1,958	3,024	2,995	1,256	238	3,794	503	1,844	1,550	5,210	
2006	2,241	5,923	3,238	884	155	4,177	287	1,585	1,135	5,928	
2012	2,367	7,176	3,418	916	246	8,475	391	3,285	927	6,109	

EU 28, mlln euro	Type of economic unit – Agriculture: Apple,pear and peaches	Other fruits	Citrus	Protein crops	Oilseeds	Rape and rape seeds	Soya	Sunflower	Fibre plants	Tomatos	Other	Type of ecosystem unit: Green urban areas	Cropland	Grassland	Other
Supply table															
crop pollination															
2000													2,668		
2006													3,130		
2012													4,357		
SNA met demand															
2000	588	1,941	1,142	673	148	1,845	69	348	191	2,696					
2006	682	2,343	1,217	436	106	2,124	66	410	155	2,832					
2012	712	2,946	1,240	466	170	4,323	106	807	110	3,221					
SNA unmet demand															
2000	703	2,045	1,795	548	60	1,374	414	1,398	1,350	2,376					
2006	761	2,313	1,960	425	28	1,390	201	1,057	972	2,845					
2012	823	2,604	2,115	426	46	2,805	253	2,249	812	2,721					
Use table															
crop pollination															
2000	667	1,037	58	35	30	575	20	97	10	139					
2006	797	1,267	62	23	22	663	20	118	8	151					
2012	833	1,626	63	24	30	1,348	32	229	6	166					

Figure 2. Crop pollination Supply and Use tables for the EU 28 disentangled from official statistics (2000, 2006, 2012)

3. Ecosystem services that do not link directly to SNA products: the example of water purification

Water purification is a regulating service that consists of all processes occurring in soils, sediments and water bodies that lower and/or decompose pollutants. Since it concerns the mitigation of pollution from economic activities, this service is not about water supply but about "cleaning" of water. There is no SNA product directly linked to the "cleaning process" but there are economic activities indirectly linked to it, i.e. the polluters that in the case of water purification are mainly part of the agricultural sector (i.e. use of chemicals for crop production and manure from livestock).

In the case of water purification, the service is not accounted in the SNA, the external satellite account should be added both in physical and monetary terms by using appropriate valuation techniques. Different approaches can be used for the valuation of ecosystem services. On the one hand, there are monetary valuation techniques that rely on individual preferences through consumer surplus; they value the demand side because it expresses better what is worth to people (Kumar and Wood, 2010). This set of valuation techniques is not considered consistent with the SNA, and thus the use of these techniques is discouraged for satellite accounts (UN et al., 2017). Exchange value techniques are consistent with SNA; moreover, one basic criteria here applied is that the quantification of the ecosystem service is first determined by the biophysical model and then translated in monetary terms by using a valuation technique that is consistent with the biophysical model (La Notte et al., 2015). The ecological model has to explain the trends of the ecosystem services, their functioning and their change; the valuation technique has to translate the outcomes of the biophysical model in monetary terms.

In this application nitrogen (N) retention is taken as proxy for water purification. Excessive nitrogen loading is in fact a leading cause of water pollution in Europe and globally which makes nitrogen a useful indicator substance for water quality (Sutton et al., 2011; Rockström et al. 2009). We define N retention as the process of temporary or permanent removal of nitrogen taking place in the river. This includes the processes of denitrification, burial in sediments, immobilization, and transformation or simply transport (Grizzetti et al., 2015).

For the biophysical assessment the Geospatial Regression Equation for European Nutrient losses (GREEN) model (Grizzetti et al., 2005; 2012) is used to estimate the in-stream nitrogen retention in surface water, which is considered in this paper as the actual flow of service provision. GREEN is a statistical model developed to estimate nitrogen (N) and phosphorus (P) flows to surface water in large river basins. The model is developed and used in European basins with different climatic and nutrient pressure conditions (Grizzetti et al., 2005) and is successfully applied to the whole Europe (Grizzetti et al., 2012; Bouraoui et al., 2009).

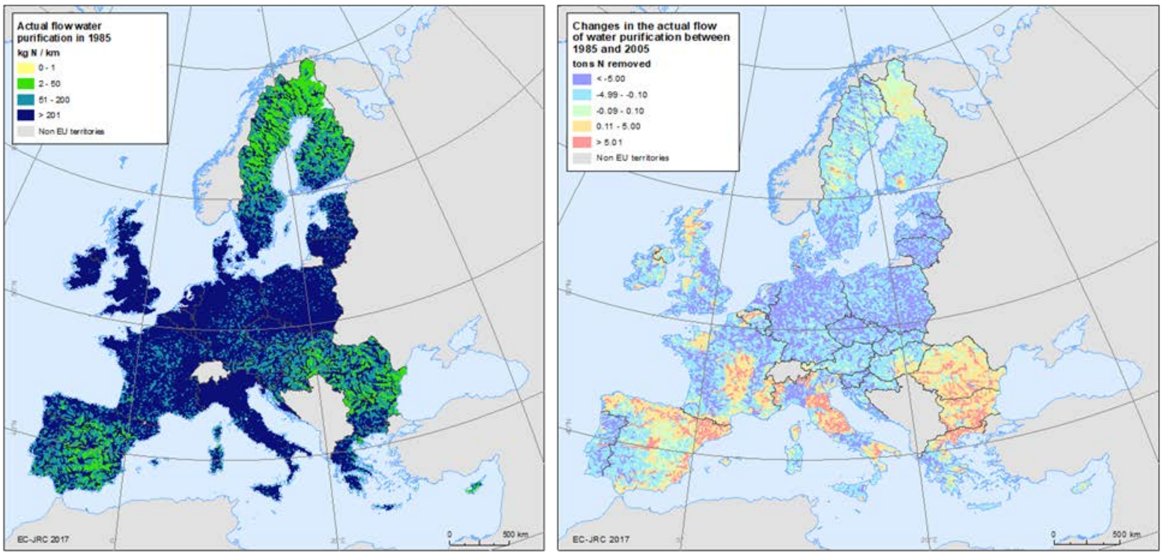


Figure 3. Water purification annual assessment (1985) and its trend over time (2005 – 1985) in physical terms (Vallecillo et al., 2018)

The area of study is divided into a number of sub-catchments that are connected according to the river network structure. For each sub-catchment the model considers the input of nutrient diffuse sources and point sources and estimates the nutrient fraction retained during the transport from land to surface water and the nutrient fraction retained in the river segment. In the case of nitrogen, diffuse sources include mineral fertilizers, manure applications, atmospheric deposition, crop fixation, and scattered dwellings, while point sources consist of industrial and waste water treatment discharges. Diffuse sources are reduced both by the processes occurring in the land (crop uptake, denitrification, and soil storage), and those occurring in the aquatic system (aquatic plant and microorganism uptake, sedimentation and denitrification), while point sources are considered to reach directly the surface waters and therefore are affected only by the river retention. The biophysical model estimates the annual retention of N that constitutes the actual flow of water purification and time series can be built to check how the trend is evolving over time (Figure 3).

After choosing and applying the biophysical model the valuation technique that best translate the results of the biophysical assessment in monetary terms is selected. For this case study the replacement cost technique, in particular the replacement costs of Constructed

Wetlands (CW), is selected. The rationale that justifies this choice is reported in La Notte et al. (2012).

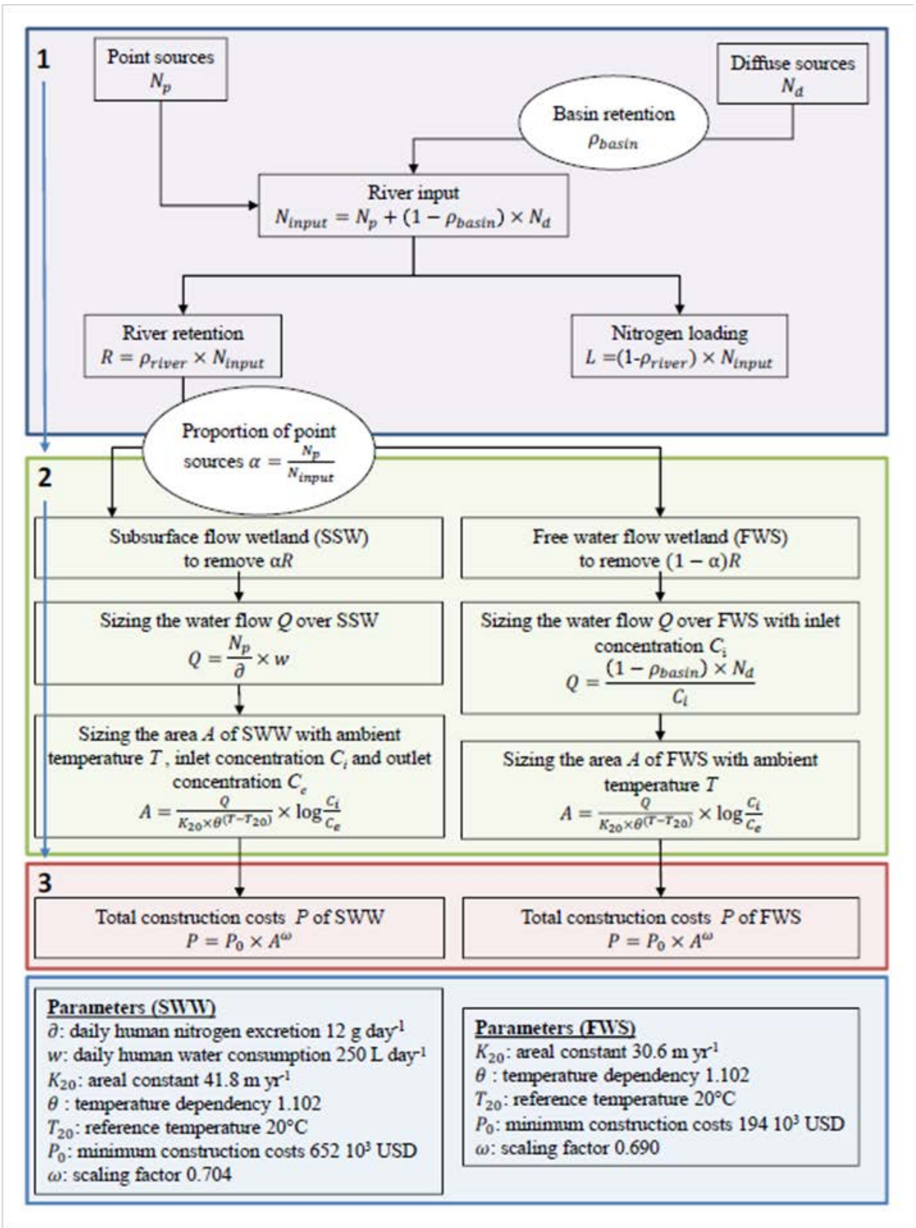


Figure 4. Flow diagram to assess replacement costs of nitrogen retention in river networks (La Notte et al. 2012)

Because GREEN provides the emissions to the river network that originate from diffuse sources (i.e. mineral fertilizers, manure, atmospheric deposition and scattered dwellings) and from point sources of pollution (urban waste water treatment plants, industries and paved areas), it is possible to differentiate the kind of CW costs according to the type of pressure of nitrogen. Wetlands designed for wastewater treatment are different from those designed for agricultural non-point pollution. The costs for Free Water systems (FWS) are applied to diffuse sources, and the costs for Horizontal flow wetland (HF) to point sources. Figure 4 shows the flow chart that from biophysical assessment brings to monetary valuation.

After the biophysical model assesses the tons of N retained (first box), through environmental engineering formula it is possible to move from tons of N to equivalent hectares of CWs (second box) to the replacement cost per CW hectare (third box). As Figure 4 summarizes the procedure in this case is much longer and more complex than in the case of crop provision; a value is in fact created and added rather than disentangled from an estimate that already exists.

All the details about the application of the valuation technique and the building of water purification accounts can be found in dedicated report (La Notte et al. 2012) and articles (La Notte et al. 2017b, La Notte and Dalmazzone 2018).

4. Conclusions

The two ecosystem services here briefly presented show different valuation approaches.

For crop pollination, a fast-track approach is implemented. It starts from the current SNA production and attempts to disentangle from it the contribution of ecosystem service. In this case, the role of the biophysical assessment is crucial to estimate the pollination contribution that defines the "amount" of the ecosystem service itself. In this way, we are able to not only attribute what is provided by ecosystem (as services) but also what of the current production is covered by the ecosystem service and what remain uncovered.

For water purification, where there is no direct linkage with SNA products, ad hoc valuation is performed by linking the outcomes of the biophysical model with an appropriate valuation technique in line with international guidelines. From the tons of N retained, ha of equivalent CWs are estimated and then relevant costs attributed.

In both cases a number of limitations applies (ref Vallecillo et al. 2017 for crop provision and La Notte et al 2017b for water purification). However, the basic concept to be highlighted by this paper is that a difference applies in the approaches used for monetary valuation:

(i) when a direct linked exists with SNA products a fast track approach can be applied. The biophysical model estimates the contribution of the ecosystem to human activities and it is then possible to disentangle it from official statistics already available;

(ii) when there is no direct link with SNA products ad hoc monetary valuation techniques should be applied to build external satellite accounts to be integrated with official statistics; the biophysical model represents in this case only the first step that leads and affects the valuation procedure.

Many other case studies and examples need to be undertaken to further test this methodological proposal. It is in fact a learning by doing approach that has just started.

Acknowledgments

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10. Valuation of Public Green Spaces

Jens Kolbe[1]

Valuation of publicly accessible green spaces can be considered a nontrivial problem in ecological economics. As the term publicly accessible implies, green spaces fulfil the definition of a public good. Hence, there exists no market which would reveal an individual or aggregated willingness-to-pay (WTP) of consumers. In order to solve the problem of missing market values, economists exploit the spatial relationship between consumers' location and public green spaces as a means to derive a WTP.

There are two frequently used approaches which allow for monetary assessment of public green spaces. On the one hand, researcher use hedonic regressions to estimate implicit prices for access to public green spaces and on the other hand, there exists the so-called life satisfaction approach which uses survey data to elicit utility of nearby green spaces.

Both approaches necessitate a spatial measurement of accessibility in order to estimate the utility of public green. Most often, the distance to the next green space and the amount of green spaces within a certain radius around the households' location are used. Those measurements, next to other controls, serve as explanatory variables in a regression model. Both approaches differ in the dependent variable of the regression model. While hedonic regressions typically use property prices as the dependent variable, the life satisfaction approach utilizes the self-reported satisfaction with life of survey participants. The participants determine their level of life satisfaction on a so-called Likert scale ranging from zero to ten. The hedonic method has the advantage that the property price as the dependent variable is measured in monetary terms, hence the coefficients of the regression can be interpreted as implicit prices (i.e. WTP). While the hedonic pricing model delivers the marginal WTP more or less directly, the life satisfaction approach requires a comparison between the ceteris paribus effect of green spaces on life satisfaction and the effect of income (See Ferreira and Moro 2010). The elasticity between both effects is interpreted as the marginal WTP.

Although both approaches use regression techniques to derive values for public green spaces, there are fundamental differences in the design of identification and even in the assumptions of these models. Next to these rather technical issues, there are other reasons which make a direct comparison difficult. First, researchers employ different measurements of accessibility in the regressions models (e.g. distances, coverage, etc.). Second, both methods rely on differing subsamples of the overall population[2]. Looking into the literature, it is not surprising that both approaches show a huge difference in valuing public goods like green spaces.

In general, the life satisfaction approach tends to produce higher estimates of WTP than the hedonic pricing method. For instance, Ambrey and Fleming (2014) conducted an analysis based on the "Australian Houshold Survey". In their analysis, they reveal a household's annual willingness to pay about EUR 63,927 per hectare[3]. In studies for Germany, Bertram

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[2]The life satisfaction approach is usually based on survey data while the hedonic pricing method uses observed sales of real estate. There should be a difference in the composition and hence in the representativity of both groups.

[3]Throughout this summary, the willingness to pay for the life satisfaction approach will be reported per household per hectare per year and in EUR to allow for immediate comparison.

and Rehdanz (2015) and Krekel et al. (2016) found also positive effects of urban green spaces but an essential smaller marginal WTP of EUR 451 and EUR 276 respectively.

In contrast, hedonic pricing models usually produce far smaller figures. Using house and condominium prices for the cities of Cologne and Berlin, Kolbe and Wüstemann (2014) and Wüstemann and Kolbe (2017) estimated a willingness to pay of about EUR 200 and EUR 522 respectively per additional hectare green space. Other studies which only consider the distance to the next park find a significant positive price effect of proximity to green spaces. For example, Morancho (2003) reported a WTP of EUR 1800 for every hundred meters closer to a park. But these values represent only premiums paid once and not a reoccurring annual rate as in the case of the life satisfaction approach. Given a for Germany typical “time of owning” for dwellings of thirty years, effects of the hedonic pricing model become very small.

Frey et al. (2010) investigated on the validity and the reliability of the life satisfaction approach for valuation of public goods. In addition, they compare the hedonic pricing model with the life satisfaction approach and find several reasons for explaining the different results of both approaches. Next to reason why the hedonic methods tend to underestimate the real value of environmental amenities, they give explanations and recommendations for further usage of the life satisfaction approach.

In the end, there is no reason to believe that one method is clearly superior to the other. Both models rely on different assumptions which may be violated if markets are not efficient. Due to the difference in models those market deficiencies may cause opposing effects. While the tendency to underestimate the value of green spaces in the hedonic model may be boosted by market failures, the very same reasons may lead to a higher WTP in the life satisfaction approach. Hence, the results from the life satisfaction approach and the hedonic pricing model may serve as an upper and a lower bound for a marginal WTP.

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Annex

Meeting Agenda

24. April 2018

8:15-9:00	**Registration**
Welcome and introduction	
Moderator: *Eduard Interwies, InterSus*	
Plenary	
9:00-9:20	Natural Capital Accounting – Building bridges between policy, welfare economics and accounting *Thomas Graner, Deputy President of the German Federal Agency for Nature Conservation (BfN)*
9:20-9:30	Aims and structure of the expert meeting *Beyhan Ekinci, BfN and Eduard Interwies, InterSus*
Session 1: Setting the scene	
This session will introduce the policy demand for ecosystem accounting and valuation as well as the current initiatives at international level to respond to these demands. It will also present the System of Environmental Economic Accounting Experimental Ecosystem Accounting (SEEA EEA), the Technical Recommendations in support of the SEEA EEA and elaborate on the research agenda for the revision of the SEEA EEA.	
Moderator: *Eduard Interwies, InterSus*	
Plenary	
09:30-09:45	SEEA EEA: Framework, revision process and research agenda *Alessandra Alfieri,UNSD*
09:45-10:15	Policy demand and current debate surrounding valuation issues *Salman Hussain, UN Environment and Laure Ledoux, EU DG Environment*
Coffee break	
10:45-11:15	Overview of the SEEA Experimental Ecosystem Accounting, with examples from the Netherlands *Sjoerd Schenau, Statistics Netherlands*
11:15-11:30	KIP INCA: The EU Knowledge Innovation Project on an Integrated System for Natural Capital and Ecosystem Services Accounting *Lisa Waselikowski, EUROSTAT*
11:30-12:00	Discussion
Lunch	
Session 2: The valuation of ecosystem services for accounting	
This session will discuss issues on the valuation of ecosystem services, including the relationship between exchange values, following the SNA principles of valuation, and welfare valuation. Theoretical developments in the comprehensive "green" accounting literature will be discussed with implications for measurement. The session will also discuss institutional arrangements that are assumed in case of non-market valuation.	
Moderator: *Salman Hussain, UN Environment*	

8:15-9:00	**Registration**
Plenary	
13:00 - 15:00	Key issues in the valuation of ecosystem services for ecosystem accounting *Carl Obst, Editor revised SEEA EEA, IDEEA Group* Theoretical developments of the comprehensive (or "green") national accounting literature *Geir Asheim, University of Oslo* Compatibility between consumer surplus based values and real income *Burkhard Schweppe-Kraft, BfN* Learning from 30 years of non-market valuation *Luke Brander, VU University Amsterdam and University of Hong Kong* Ecosystem services and asset valuation in the RECAMAN project: integrating market and simulated exchange values *Alejandro Caparrós, Spanish National Research Council*
Coffee break	
15:30-16:00	Panel discussion with discussants *Juha Siikamaki, IUCN; Gerhard Bouwer, Statistics South Africa and speakers*
Parallel sessions on selected cultural and regulating services	
In parallel sessions accounting and welfare valuation-based approaches for selected cultural and regulating services will be discussed with the objective of identifying best practices for valuation methods in different ecological, socio-economic and data environments.	
16:00-17:30	Parallel working groups on regulating services with short keynotes on country examples Air filtration *Rocky Harris, Defra* *Lars Hein, Wageningen University* Erosion control / sediment retention *Takashi Hayashi, Japan Policy Research Institute* *Jane Turpie, University of Cape Town* Pollination *Alessandra la Notte, EU Joint Research Center* *Juha Siikamaki, IUCN/ESAfD* Urban recreation *David Barton, Norwegian Institute for Nature Research Oslo* *Jens Kolbe, Technical University Berlin*

25 April 2018

Session 2: Valuation methods for key ecosystem services, continued	
In parallel sessions accounting and welfare valuation-based approaches for selected cultural and regulating services will be discussed with the objective of identifying best practices for valuation methods in different ecological, socio-economic and data environments.	
Moderator*: Salman Hussain, UN Environment*	
Plenary	
9:00-	Recap of Day 1 including short reports from parallel session chairs, followed by group discus-

Session 2: Valuation methods for key ecosystem services, continued	
10:00	sion. Introduction Day 2
Parallel sessions on selected cultural and regulating services	
10:00- 11:30	Parallel working groups on cultural and regulating services with short keynotes on country examples Nature-based recreation/tourism *Konstantin Loshadkin, STC "Resources and Consulting* *Manuel Woltering, University of Würzburg* Climate regulation (carbon sequestration and storage): *Lars Hein, Wageningen University* *Peter Elsasser, Thünen Institute* Water Purification by streams and floodplains *Mathias Scholz, Centre for Environmental Research* *Alessandra La Notte, EU Joint Research Center* *Jane Turpie, University of Cape Town* Flood control / water flow regulation *Jesko Hirschfeld, Technical University Berlin* *Takashi Hayashi, Japan Policy Research Institute*
Plenary	
11:30- 12:30	Short reports from parallel session chairs followed by discussion
Lunch	
Session 3: Valuing ecosystem wealth, degradation and/or enhancement	
The session will discuss valuation of stocks of ecosystem assets (or wealth), their degradation and/or enhancement over time. Issues include: assumptions on discount rates, asset life, projections of service values when using the net present value (NPV); alternative valuation approaches for estimating wealth; relationship to market values; defining and valuing the degradation of ecosystems in nominal and real terms; cost versus damage-based approaches; attributing degradation costs to economic actors; treatment of defensive expenditures.	
Moderator: *Bram Eden, UNSD*	
Plenary	
13:30- 14:30	World Bank's Wealth Accounting *Raffaello Cervigni, World Bank* Inclusive wealth index *Salman Hussain, UN Environment* Valuing Natural Capital in the Context of Ecosystem Based Management *Eli Fenichel, Yale University*
Coffee Break	
14:45 - 15:45	A «collectively recognized ecological debt» based on a measure of the cost of ecosystem degradation *Yann Kervinio, Ministry for an Ecological a. Solidary Transition (FRA)* Applying the concept of defensive expenditures to ecosystem degradation and enhancement

Session 2: Valuation methods for key ecosystem services, continued		
	Hans Diefenbacher, University of Heidelberg Mexico’s experiences in economic valuation of environment and policies *Raul Figueroa Diaz, National Institute of Statistics and Geography (INEGI)*	
Excursion to Drachenfels and joint dinner (19:15 @ DelikArt Restaurant, Colmantstraße 14 – 16, 53115 Bonn. incl. food and non-alcoholic drinks)		

26. April 2018

Session 3: Valuing ecosystem wealth, degradation and enhancement, continued		
Plenary		
9:00-10:00	Ecosystem wealth, degradation and enhancement in SEEA-EEA - Key issues and including a recap of Day 2 *Carl Obst, Editor revised SEEA EEA, IDEEA Group* The UK-case - projecting future services for ecosystem asset valuation *Rocky Harris, Defra* The corporate natural capital account framework- distinguishing assets and liabilities *Will Evison, PricewaterhouseCoopers/Natural Capital Coalition*	
Parallel Sessions		
10:00-11:00	2 to 3 groups discussing the 2 primary alternative approaches to valuation of ecosystem assets, namely the present value of future benefit flows and restoration cost-based approaches.	2 to 3 groups discussing challenges and solutions to measuring the present value of ecosystem as-sets.
Coffee break		
Plenary		
11:15-12:15	Reporting back by parallel discussion group chairs	
Session 4: Policy, science, accounting and business interface		
The session will discuss a number of issues at the nexus of policy, science, and accounting (public and business) such as best ways to communicate the results of valuation studies, which indicators can inform what policies and what are the enabling conditions to mainstream the valuation exercises within ecosystem assessments as well as accounting.		
Moderator: *Eduard Interwies, InterSus*		
Plenary		
12:15-13:00	Facilitated panel discussion part I Science: *Jan Erik Petersen, European Environment Agency* Policy: *Jakub Wejchert, EU DG Environment; M.P. Johnson - Ministry of Envirronment, Forest & Climate Change India* Accounting: *Sven Kaumanns, Federal Statistical Office of Germany* Business: *Will Evison, Natural Capital Coalition*	
Lunch		

Session 3: Valuing ecosystem wealth, degradation and enhancement, continued	
Plenary	
14:00-14:45	Facilitated panel discussion part II
Session 5: Conclusions and way forward	
This session will revisit the main outcomes of the meeting in light of the objectives and discuss the next steps both in the EU context as well as within the SEEA EEA revision process.	
Moderators: *UNSD /UN Environment / BfN*	
14:45-15:15	Draw conclusions with respect to the objectives of the meeting and next steps
15:15-15:30	Closing remarks *Daniel Van Assche, EU FPI; Alessandra Alfieri, UNSD; Salman Hussain, UN Environment; Beyhan Ekinci, BfN*

List of Participants

Last name	First name	Organisation
Ai	Ruiqing	National Bureau of Statistics of China
Aizpurua	Nerea	DG Research and Innovation. European Commission
Alfieri	Alessandra	UN Statistics Division
Anderson	Hamish	Office for National Statistics
Asheim	Geir B.	University of Oslo
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Barton	David N.	Norwegian Institute for Nature Research (NINA)
Berger	Lars	Federal Agency for Nature Conservation
Bouwer	Gerhardt	Statistics South Africa
Brander	Luke	University of Hong Kong
Bünger	Björn	German Environment Agency
Burkhard	Benjamin	Leibniz Universität Hannover
Caparrós	Alejandro	Institute for Public Goods and Policies / Spanish National Research Council
Caspers	Justus	Destatis
Cervigni	Raffaello	The World Bank
Chow	Julian	United Nations Statistics Division
Cools	Jan	University of Antwerp
de Soye	Yves	UNDP
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Dvarskas	Anthony	Stony Brook University
Edens	Bram	UNSD
Ekinci	Beyhan	Federal Agency for Nature Conservation

Last name	First name	Organisation
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Lopes Batista	Ivone	Brazilian Institute of Geographic and Statistical
Lopez Conlon	Monica	UN Environment
Loshadkin	Konstantin	Group "Recourses and Consulting"
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Nel	Jeanne	SANBI
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Last name	First name	Organisation
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