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Mapping Ecosystem Services' Values: Current Practice and Future Prospects

By **Jan Philipp Schägner** and **Joachim Maes**, Institute for Environment and Sustainability (IES), Joint Research Centre (JRC), European Commission, Italy

Luke Brander, Institute for Environmental Studies (IVM), VU University Amsterdam,The Netherlands

Volkmar Hartje, Technische Universität Berlin, Institute of Landscape Architecture and Environmental Planning, Germany

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Summary

Mapping of ecosystem services' (ESS) values means valuing ESS in monetary terms across a relatively large geographical area and assessing how values vary across space. Thereby, mapping of ESS values reveals additional information as compared to traditional site-specific ESS valuation, which is beneficial for designing land use policies for maintaining ESS supply. Since the well-known article by Costanza et al. (1997), who mapped global ESS values, the number of publications mapping ESS values has grown exponentially, with almost 60% being published after 2007. Within this paper, we analyse and review articles that map ESS values. Our findings show that methodologies, in particular how spatial variations of ESS values are estimated, their spatial scope, rational and ESS focus differ widely. Still, most case studies rely on relatively simplistic approaches using land use/cover data as a proxy for ESS supply and its values. However, a tendency exists towards more sophisticated methodologies using ESS models and value functions, which integrate a variety of spatial variables and which are validated against primary data. Based on our findings, we identify current practices and developments in the mapping of ESS values and provide guidelines and recommendations for future applications and research.

Keywords: Ecosystem Service Assessment, Ecosystem Service Mapping, Ecosystem Service Valuation, Ecosystem Service Modelling, Value Transfer, Land use Policy Assessment

JEL Classification: Q5

Address for correspondence:

Jan Philipp Schägner
Institute for Environment and Sustainability (IES)
Joint Research Centre (JRC), European Commission
Via E. Fermi 2749, TP460
I-21027 Ispra (VA)
Italy
Phone: +39033278 9266
Fax. +390332 78 5819
E-mail: Philipp.Schaegner@jrc.ec.europa.eu

Mapping Ecosystem Services' Values: Current Practice and Future Prospects

Jan Philipp Schägner[a], Luke Brander[b], Joachim Maes[a], Volkmar Hartje[d]

a. Institute for Environment and Sustainability (IES), Joint Research Centre (JRC), European Commission, Via E.Fermi 2749, TP460, I-21027 Ispra (VA), Italy

b. Institute for Environmental Studies (IVM), VU University Amsterdam, The Netherlands

d. Technische Universität Berlin, Institute of Landscape Architecture and Environmental Planning, EB 4-2, Straße des 17. Juni 145, 10623, Berlin, Germany

Corresponding author: Jan Philipp Schägner, Institute for Environment and Sustainability (IES), Joint Research Centre (JRC), European Commission, Via E.Fermi 2749, TP460, I-21027 Ispra (VA), Italy, Tel. +39-0332-78 9266 Fax. +39-0332-78 5819, Email: Philipp.Schaegner@jrc.ec.europa.eu

Abstract

Mapping of ecosystem services' (ESS) values means valuing ESS in monetary terms across a relatively large geographical area and assessing how values vary across space. Thereby, mapping of ESS values reveals additional information as compared to traditional site-specific ESS valuation, which is beneficial for designing land use policies for maintaining ESS supply.

Since the well-known article by Costanza et al. (1997), who mapped global ESS values, the number of publications mapping ESS values has grown exponentially, with almost 60% being published after 2007.

Within this paper, we analyse and review articles that map ESS values. Our findings show that methodologies, in particular how spatial variations of ESS values are estimated, their spatial scope, rational and ESS focus differ widely. Still, most case studies rely on relatively simplistic approaches using land use/cover data as a proxy for ESS supply and its values. However, a tendency exists towards more sophisticated methodologies using ESS models and value functions, which integrate a variety of spatial variables and which are validated against primary data.

Based on our findings, we identify current practices and developments in the mapping of EES values and provide guidelines and recommendations for future applications and research.

Keywords: ecosystem service assessment; ecosystem service mapping; ecosystem service valuation; ecosystem service modelling; value transfer; land use policy assessment

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1. Introduction

The framework of ecosystem services (ESS) is widely used for communicating links between ecosystems and human well-being (MA 2005). Manifold studies aim to integrate ESS assessments into decision making processes (TEEB 2010; UK NEA 2011). The economic value (i.e., contribution to human welfare) of an ESS is, as with any good or service, determined by its supply and demand. The supply side of an ESS is largely determined by ecological processes and characteristics (e.g., functioning, fragmentation, productivity, resilience or climate) that may be influenced by human activities, either deliberately or inadvertently. The understanding and modelling of the supply of ESS has largely been taken up by natural scientists (e.g., ecologists, geographers, hydrologists). The demand side is largely determined by the characteristics of human beneficiaries of the ESS (population, preferences, distance to resource etc.). The understanding and modelling of the demand side has largely been taken up by economists. It has been recognised that the determinants of both, the supply and demand of ESS, are spatially variable, which makes the assessment of ESS values inherently spatial. In recent years, a growing body of literature assesses ESS spatially by producing digital maps either of ESS supply or its value (Troy and Wilson 2006; Maes, Paracchini, et al. 2011). In this paper we review studies that map values of ESS. We define mapping of ESS values as the valuation of ESS in monetary terms across a relatively large geographical area that includes the examination of how values vary across space.[1] Thereby, mapping of ESS values reveals additional information as compared to traditional site-specific ESS valuation, which is beneficial for designing efficient policies and institutions for maintaining ESS supply.

To some extent spatial issues have been disregarded in environmental and resource economics, including ESS valuation, but have attracted increasing attention with the emergence of advanced GIS technology in the 90's (Bockstael 1996). The first studies to map ESS values, examine recreational values for Welsh forests (Bateman et al. 1995) and multiple ESS across a protected area in Belize (Eade and Moran 1996). A milestone in

[1] The literature that we examine does therefore not only include studies that produce graphical value maps but also includes analyses that explicitly address spatial variability in values.

this development is the well-known paper by (Costanza et al. 1997), in which global ESS values are mapped. This paper raised a lot of attention and initiated a debate on value mapping in general and on the meaningfulness of aggregate global values. Since then, the number of publications mapping ESS values has grown exponentially, with almost 60% being published after 2007 (see Figure 1). The methodologies applied in these studies differ widely, in particular with respect to how spatial variations of ESS values are estimated. The precision and accuracy of mapped ESS values has been questioned, and accordingly the utility for policy guidance. However, no consensus has been reached on which methods can and should be used to inform specific policy contexts (de Groot et al. 2010). Until now, no comprehensive review of the literature on mapping ESS values has been conducted.

Within this paper, we review all peer reviewed journal articles published before 2012 that map ESS values. Articles were obtained by searching the SCORPUS, Science Direct and *Google scholar* databases with various key word combinations and by scanning the references of all relevant papers. In total, we obtained 384 articles of which 143 mapped ESS. We excluded all studies from the review that map only ESS supply (54) and non-monetary ESS values (20). We analysed the remaining 69 articles and reviewed them according to the methodologies used for ESS quantification and valuation, the ESS assessed, study rational and case study area characteristics. The purpose of this review is to identify current practices and developments in the mapping of EES values with a view to providing recommendations for future applications and research.

The paper is organised as follows: In section 2 we give an overview of the rational and contribution of ESS value mapping to ESS research and policy making. Section 3 gives a quantitative review of general study characteristics, such as location, scale of analysis, and ecosystems and ESS addressed. In section 4, different methodologies used for mapping ESS values are analysed and studies are classified within a methodology matrix. We discuss evidence on the accuracy of current value mapping exercises and evaluate the different methodologies. In section 5, we give an outlook on future prospects and avenues for development. Finally, section 6 provides some conclusions.

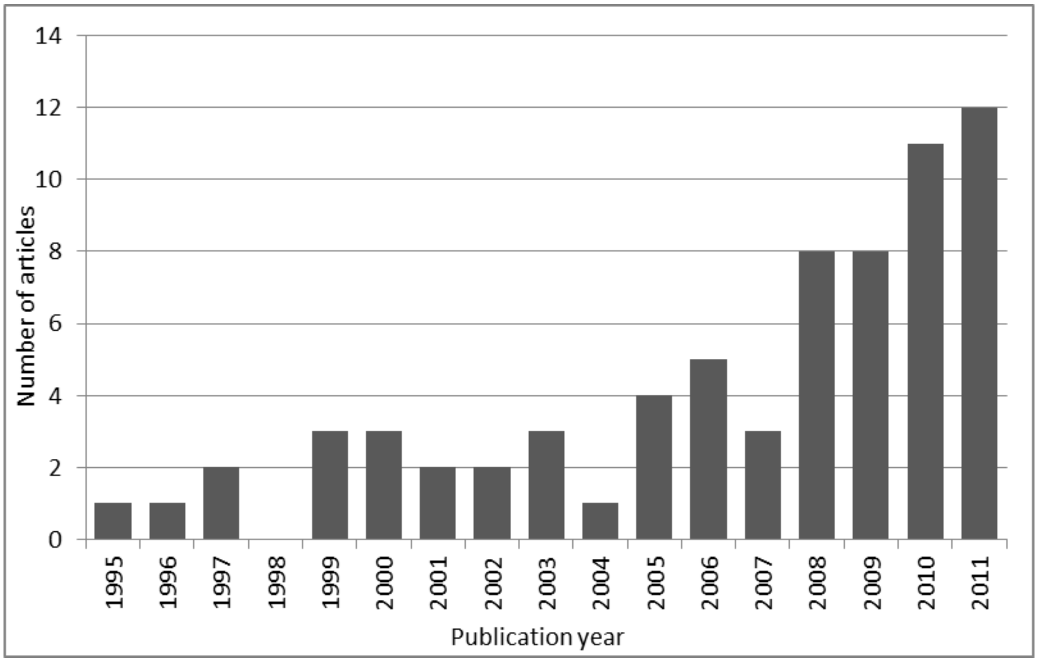

Figure 1: Published articles per year.

2. Why Map Values?

Natural ecosystems produce various ESS, which strongly contribute to human well-being (TEEB 2010; MA 2005). Nevertheless, due to the public good characteristics of many ecosystems and their vulnerability to externalities, such as air, soil and water contamination, the costs of ecosystem degradation are not sufficiently incorporated into individual or public decision-making. As a result, ecosystems in all parts of the world are being degraded to a suboptimal extent, causing loss of ESS supply. Various national and supranational policies have been introduced to protect natural ecosystems, which have only been partially effective (e.g. Ramsar Convention on wetlands of international importance; Convention on Biological Diversity 2010 target). Reversing the degradation of ecosystems requires "*significant changes in policies, institutions, and practices that are not currently under way"* (MA 2005).

One of the main challenges in designing effective policies derives from the complexity of integrating the multidimensional environmental impacts into decision making processes. Typically, decisions are based mainly on information that is well understood and known with high certainty, for example information on readily observable financial or market transactions. Ecological externalities are typically insufficiently considered because of uncertain estimates regarding expected impacts, difficulties in interpreting results from various disciplines and difficulties in translating impacts into changes of social welfare. Monetary valuation of ESS is a method to overcome such difficulties. It enables the aggregation of multidimensional costs and benefits of alternative measures within a one-dimensional welfare measure (Pearce et al. 2006). Although the practice of monetary valuation and its underlying framework are subject to debate and criticism (Spash and Carter 2001; Sagoff 2004), the concept of monetary valuation and cost-benefit analysis is widely accepted and subject to intensive research activity.

The estimation of accurate ESS values, however, is not straightforward, in part due to spatial heterogeneity in biophysical and socioeconomic conditions. The spatial perspective of variation in ESS values is relatively new and has not been extensively researched. Insufficient knowledge exists about how ESS values differ across space and what their spatial determinants are (Bockstael 1996; Bateman et al. 2002; Plummer 2009; de Groot et al. 2010). With the development of advanced GIS technology, mapping of ESS values emerged and became an important research issue in recent years.

As compared to traditional site-specific ESS valuation, mapping reveals additional valuable information. Besides communication and visualisation, it makes site specific ESS values available on a large scale. Thereby, it allows policy makers to extract estimated values easily from a database at any scale and for any site of interest in order to evaluate potential policy measures. Time consuming primary valuation or value transfer studies may not be necessary. Thereby, spatially explicit ESS value maps have specific advantages for several policy applications including:

(1) Green Accounting, (2) land use policy evaluation, (3) resource allocation and (4) payments for ESS. Figure 2 presents the frequency with which specific policy applications are mentioned as the potential end-use of value data in the ESS mapping literature.

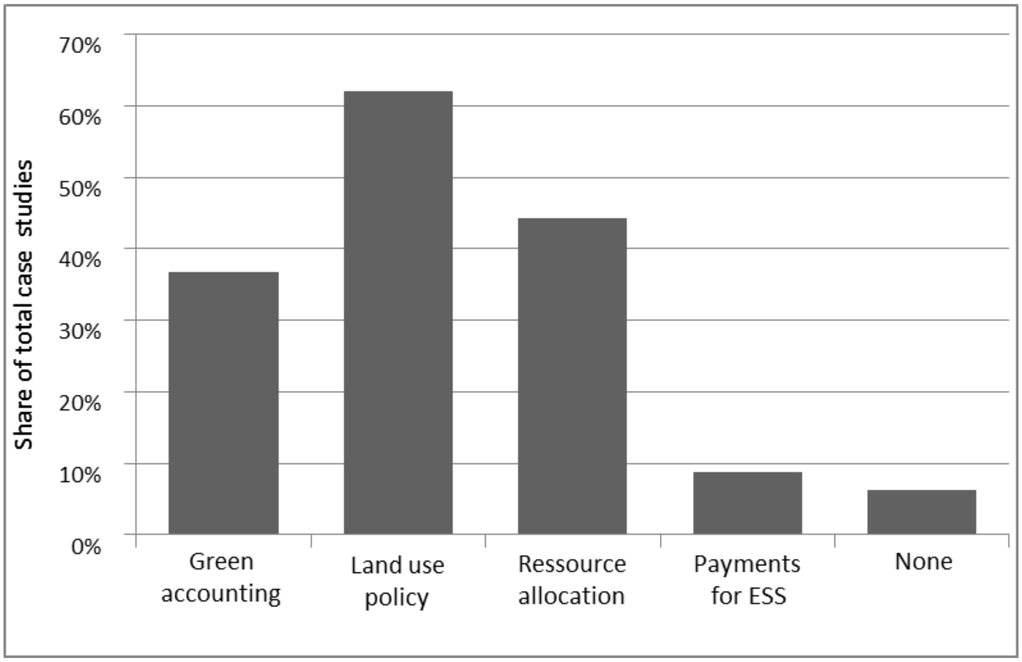

Figure 2: Citation of policy applications in ESS mapping literature

(1) Green Accounting: Mapping of ESS values allows for estimating a green GDP at different spatial scales, by summing up total ESS values across the region of interest (TEEB 2010). (2) Land Use Policy Evaluation: Mapping of ESS values allows for the evaluation of broad land use policies at a regional or even supranational level. Typically, land uses are multifunctional and therefore provide multiple services. ESS value mapping displays trade-offs and synergies in ESS values, which may result from land use change. (3) Resource Allocation: Mapping of ESS values does not only support decisions on whether or not to conduct a policy measure, it also tells where to conduct a policy measure. It allows identifying locations in order to minimize negative or maximize positive ecological side effects. For example, by identifying ESS hot spots for conservation it allows assessing "*synergies and trade-offs in conserving biodiversity and ecosystem services"* (Naidoo et al. 2008). (4) Payments for ESS: By making ESS values spatially explicit, schemes of payments for ESS can be designed more accurate in order to allow for more efficient incentives across providers of ESS.

3. Quantitative Review of Studies Mapping ESS Values

In total we analysed 69 publications, which include 79 separate case studies. Studies differ strongly with respect to their spatial scope, the ES and ESS assessed and the methodologies applied. Case study areas are mainly located in three continents, with 34% in Europe (mainly UK), 24% in North America (mainly USA) and 22% in Asia (mainly China). Figure 3 shows the spatial distribution of the case studies across the world. The colour indicates the number of studies covering each country. The minimum for each country is five as there are five global case studies. The continental, national and subnational case studies are then added for each country.

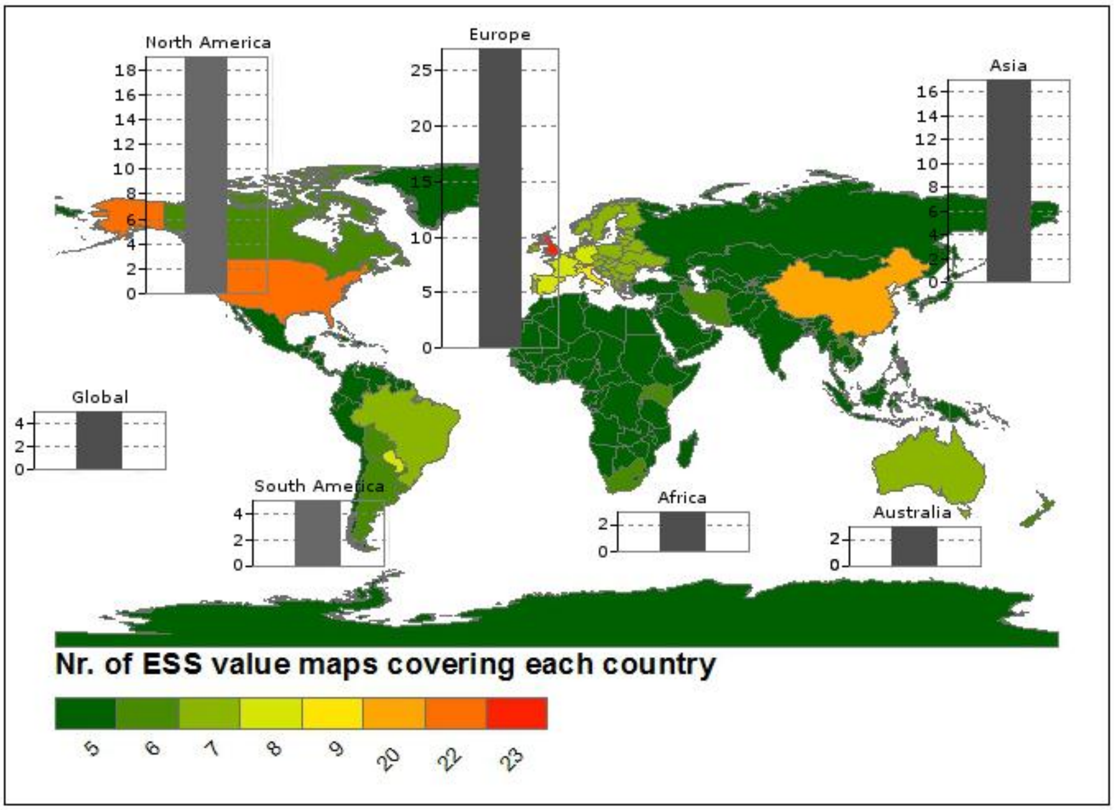

Figure 3: Spatial distribution of case study areas.

Study areas differ in size, ranging from global to local assessments (see Figure 4), with the smallest case study area comprising a 550ha forest in the surrounding of Gent, Belgium (Moons et al. 2008). Approximately 20% of all studies are 'local' applications with a case study area smaller than 1,000 km^2. Typically, they focus on a single protected area, a single forest or an urban area. Approximately 23% focus on case study areas between 1,000 km^2 and 100,000 km^2. Most of them are defined by the borders of an administrative region. Study site areas from 100,000 km^2 up to 1,000,000 km^2 comprise 24% of all studies. They contain mainly regional to national assessments. About another 24% of all study areas are mainly continental, supra national or global ESS value assessments with study areas above 1 million km².

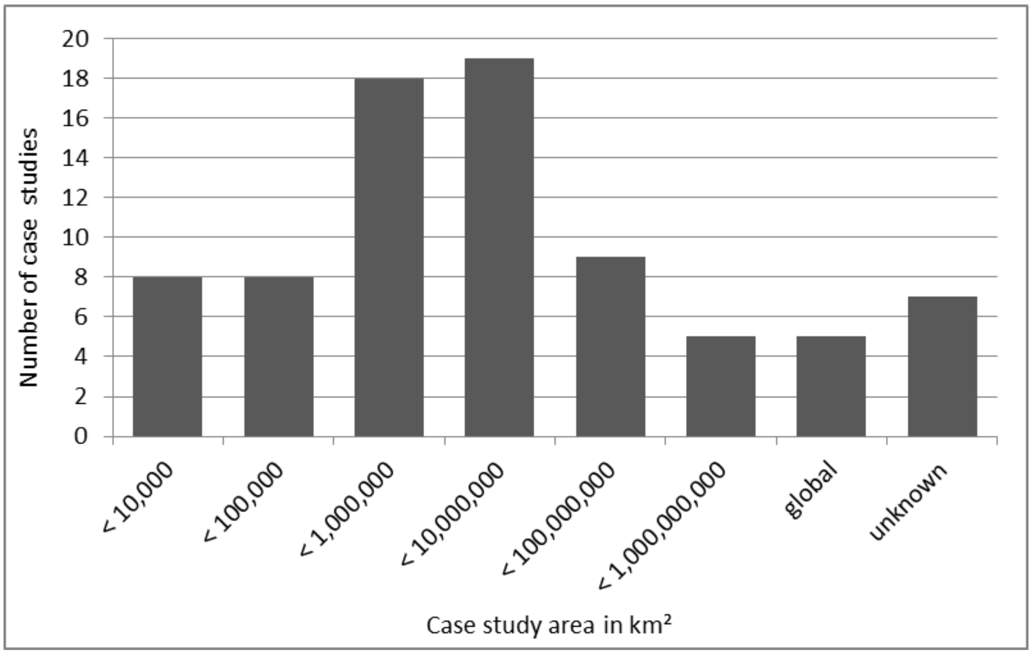

Figure 4: Study site area size.

Most study area definitions depend on political boarders, such as administrative regions (37), countries (16%), urban areas (3%) or protected areas (9%). Study areas defined by some geomorphological features are mainly related to river features (13%) such as basins or watersheds or are coastal areas (11%), such as a bay or an estuarine (see Figure 5).

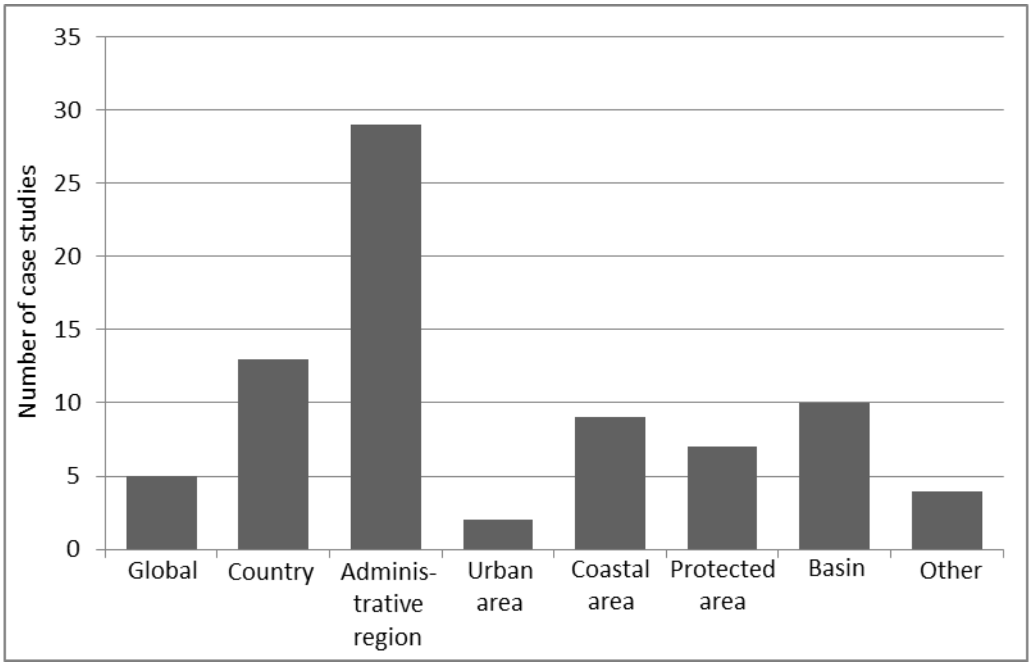

Figure 5: Types of study areas.

Most studies focus on more than four (multiple) land cover or land use (LCLU) (see Figure 6), which is expected given that values are generally mapped across larger areas. Some smaller case studies, however, focus on specific landscapes involving only one to four LCLU. Some studies map values of only one land cover within a larger area, for example all forests in Wales (Bateman, Lovett, et al. 1999).

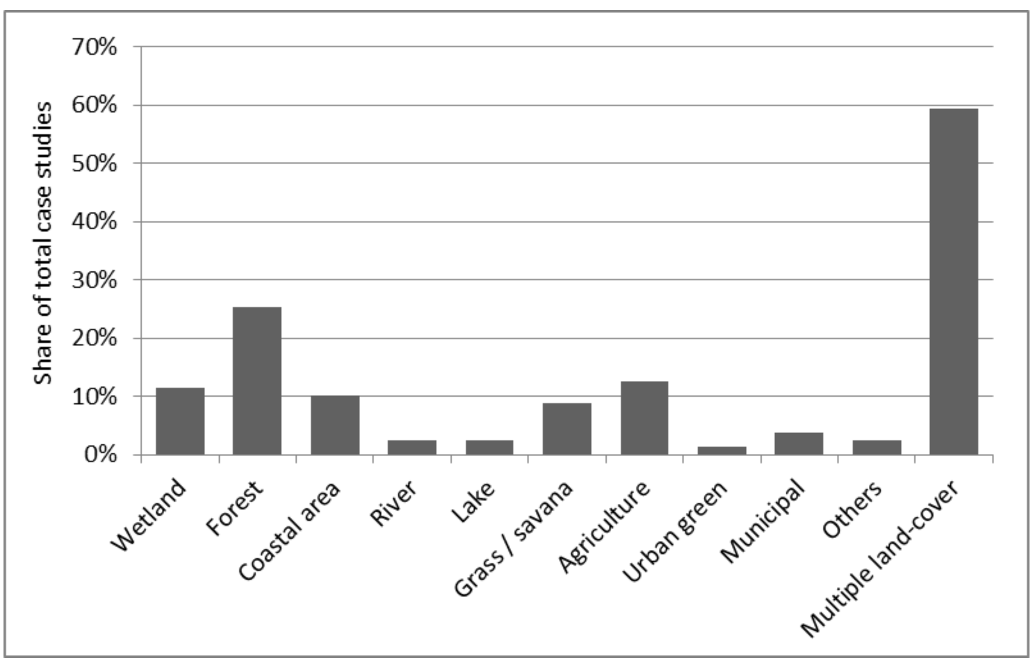

Figure 6: Ecosystems assessed.

On average, each study maps values for seven ESS. However, many studies focus only on one single ESS (28%) and about 50% map three or less ESS. At the other end of the scale, 18% of all studies follow the approach of (Costanza et al. 1997) and accordingly map 17 ESS (see Figure 7).

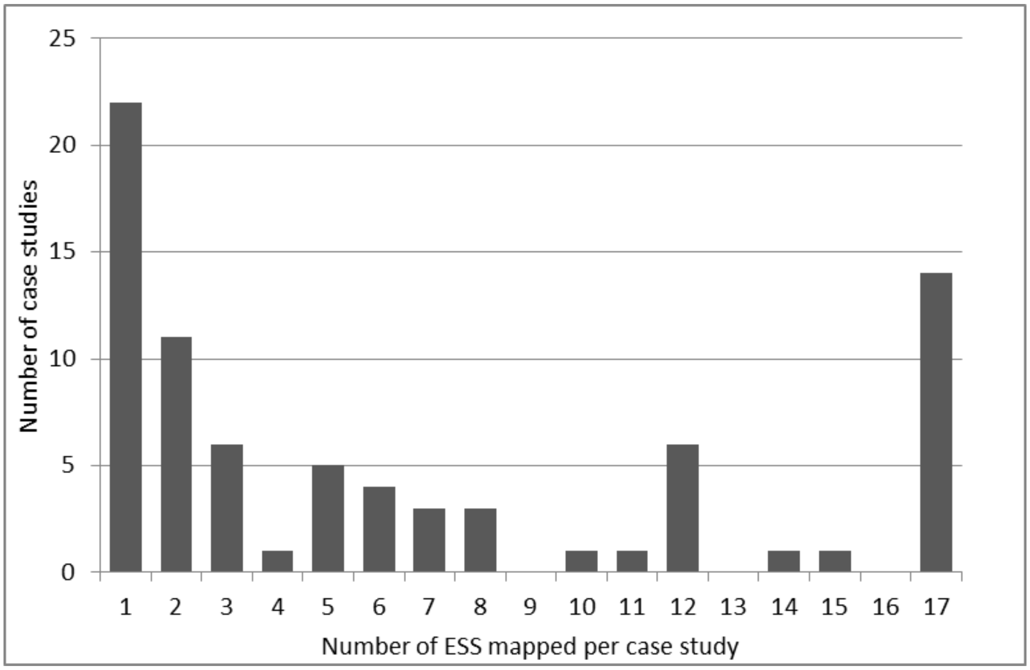

Figure 7: Number of ESS mapped per case study.

The set of ESS mapped by (Costanza et al. 1997) are mapped frequently, as their approach has been replicated several times. In total, recreation is the most frequently mapped ESS with 50 case studies, followed by the control of greenhouse gases (mainly carbon sequestration). The frequency with which each ESS has been mapped is shown in Figure 8.

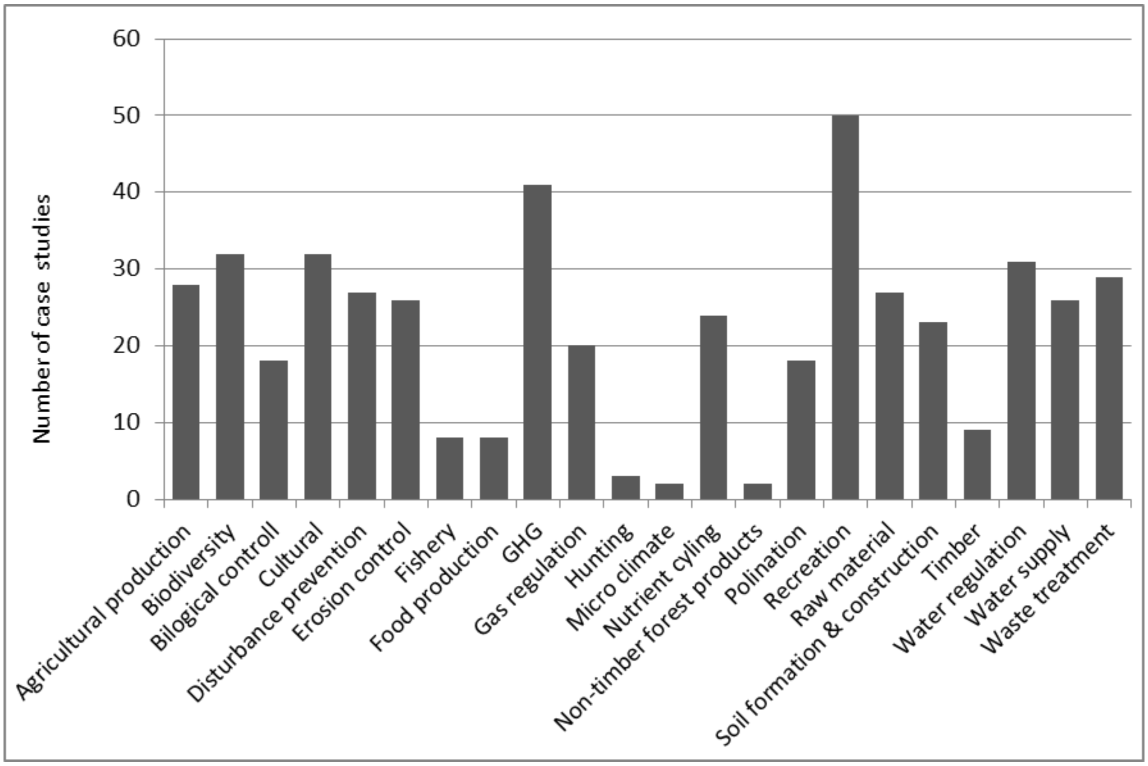

Figure 8: Frequency with which each ESS is mapped.

Many studies do not give any information on the resolution at which values are mapped. For studies that do provide such information, the range is from 1 meter to 10,000 meter resolution (see Figure 9).

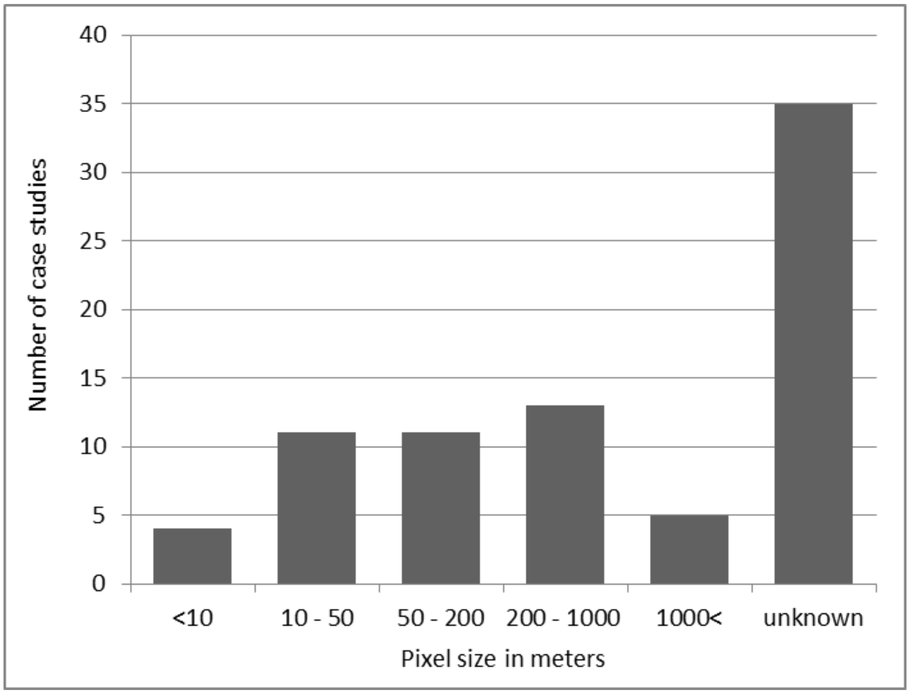

Figure 9: Resolution of ESS value map.

4. Methodologies for Mapping ESS Values

ESS valuation applications involve two dimensions: (1) A biophysical assessment of ESS supply and (2) a socioeconomic assessment of the value per unit of ESS. If ESS values are mapped, variations in ESS values across space are either assessed by mapping spatial variations of ESS supply, by mapping spatial variations of the value per unit of ESS supply or by a combination of both dimensions.

In the reviewed literature, we identified five different methodologies used for mapping ESS supply (Eigenbrod et al. 2010a) and, in analogy to environmental value transfer, four different methodologies of attaching a value to the ESS supply. In this section we first describe the different methodologies used for assessing ESS supply and its value (section 4.1 and 4.2). We then give an overview of how these methodologies are used in combination in order to map ESS values (section 4.3). Thereafter, we discuss evidence on the accuracy and precision of ESS value maps (section 4.4). Based on our findings we then discuss and evaluate the different methodologies (section 4.5).

4.1. Mapping of Ecosystem Service Supply

Methodologies used for mapping ESS supply can be divided into five main categories: (1) One-dimensional *proxies*, (2) *non-validated models*: ecological production functions (or models) based on likely causal combinations of explanatory variables, which are grounded on researcher or expert assumptions, (3) *validated models*: ecological production functions, which are calibrated based on primary or secondary data on ESS supply, (4) *representative samples* of the study area and (5) *implicit modelling* of ESS supply within a monetary value transfer function. Figure 10 shows the share of studies using a certain methodology for assessing ESS supply.

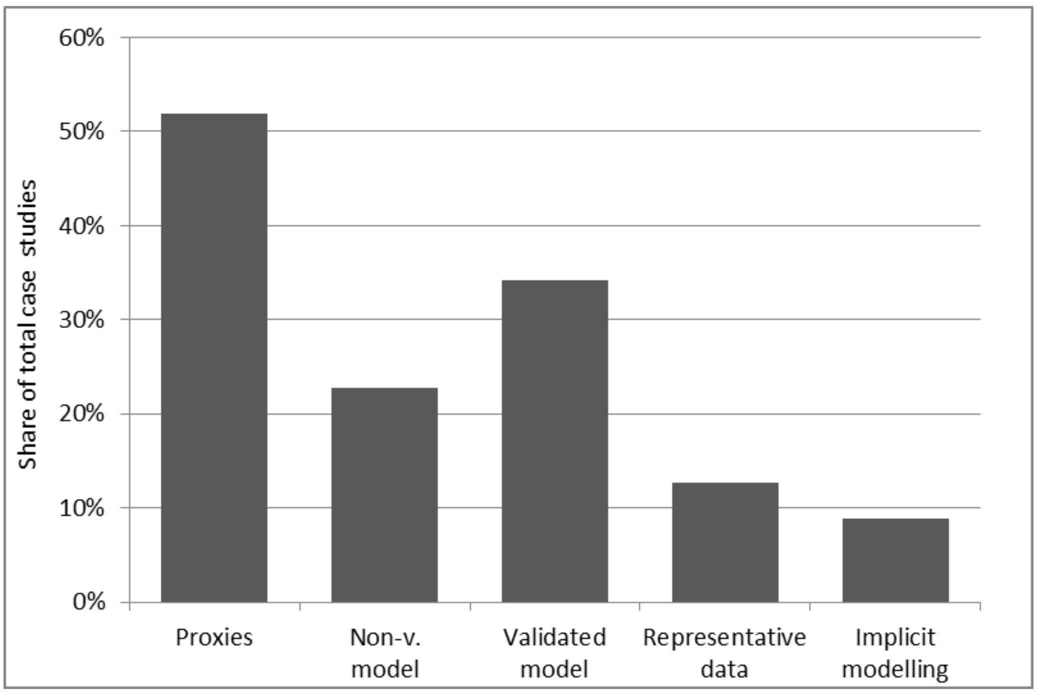

Figure 10: Share of studies using a specific methodology for mapping ESS supply.

(1) Most common are ESS maps, which are based on easily available *proxies*, mainly LCLU data. About 52% of all studies map a minimum of one ESS based on *proxies*. The use of ecological production functions in mapping ESS, which model the supply of ESS based on spatial explanatory variables, has been an important area of development. (2) In the absence of any primary data on ESS supply for model calibration, researchers tend to build models based on likely causal combinations of explanatory variables (23% of all studies). Causal combinations are grounded on researcher or expert assumptions or on information taken from literature. (3) *Validated models* use primary or secondary data on

ESS supply in order to calibrate the model parameters. This approach is used by 34% of all studies. However, the intersection between models that are calibrated based on primary or secondary data and those that are based on the researchers' assumption is smooth. Almost every complex ESS model relies on some kind of assumption. In the absence of data on ESS supply within the study area, some studies use data for calibration, which were derived in a different spatial context and for different purposes. (4) The use of representative samples for mapping ESS supply is limited and exists mainly either for small study areas or in coarse resolutions (Eigenbrod et al. 2010a). About 13% of all studies map at least one ESS based on *representative data*. (5) Some studies – typically with a strong environmental economic background – model ESS supply implicitly within monetary value transfer functions (9%). Such studies use (*meta-analytic*) *value functions* to extrapolate ESS values per unit of area. However, the *value function* contains more biophysical variables than just LCLU *proxies*, which have a causal relationship with quantitative ESS supply. The model can then be interpreted as modelling ESS supply and its value per unit at the same time, even though the ESS supply is not displayed explicitly.

4.2. Mapping of Ecosystem Services' Values

Mapping ESS values requires that monetary values are assigned to mapped ESS provision. Typically, this is done by up-scaling values from one or multiple primary valuation study sites to all relevant ecosystem sites within the case study area. This can be viewed as a specific form of value (benefit) transfer in that values are transferred to the entire stock of an ecosystem within a geographic area (Brander et al. 2011). However, some studies, typically with small case study areas, conduct primary valuations on the entire ESS supply of the case study area. The value estimate is than distributed across the study area. In total 42% of all studies conduct primary valuation, whereas 84% use up-scaling for at least one ESS.

In analogy to the value transfer literature, we distinguish between four different methodologies for distributing values across the study area: (1) *Unit values* (2) *adjusted unit values* (3) *value functions* and (4) *meta-analytic value function transfers*.

(1)In the *unit value approach,* a constant value per unit of ESS supply is applied across the study area. Thus, variations of ESS value across space result only from variations in

ESS supply. *Unit values* are the predominant methodology for valuing ESS within the value-mapping literature (78% of all studies). (2) The *adjusted unit values* approach adjusts values across the study area using simple variables in order to account for spatial value variations. Typically, such variables are population density, income levels or consumer price index. Such adjustments thereby account for the number of beneficiaries of a certain ESS or for the impact of income levels on willingness to pay. About 5% of all studies use *adjusted unit values* for ESS value mapping. (3) *Value functions* map values across the study area based on a function, which may contain multiple spatial variables. The *value function* is typically estimated within one primary valuation study, which may be conducted within or outside of the study area. It is then applied to the entire study area by plugging in site-specific parameter values into the value function. About 20% of all ESS value mapping studies uses *value functions*. (4) The *meta-analytic value function transfer approach also* transfers (or scales up) values to the entire study area by plugging in site-specific characteristics into a transfer function. In this approach, however, the function is estimated by statistical regression analysis of a number of primary valuation studies. About 4% of all case studies use this methodology.

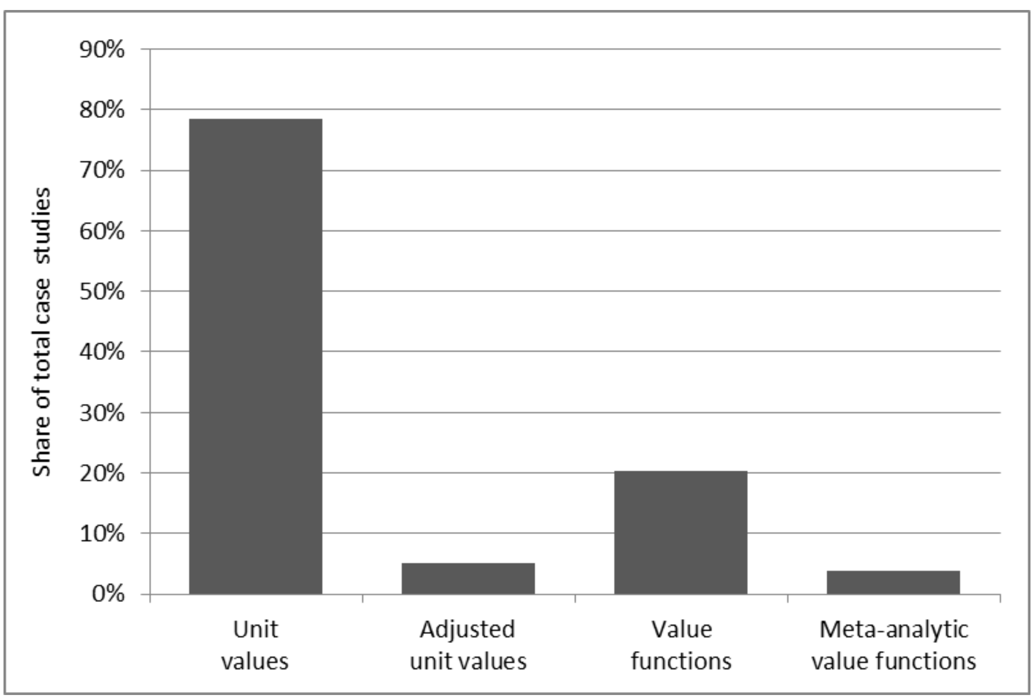

Figure 11: Share of studies using a specific methodology for valuing ESS.

4.3. Combinations of Methodologies Applied in Literature.

By combining the two dimensions of ESS value mapping, we draw a methodology matrix and allocate all reviewed studies within this matrix (see Table 1)[2]. Almost half of all studies combine LCLU *proxies* with *unit values* (46%).[3] With reference to the well-known publication of (Costanza et al. 1997), this is also referred as to the "Costanza Approach". Within this study, global ESS values are mapped by attributing mean values of multiple ESS per LCLU class from a number of primary valuation studies to a global LCLU data set. Their approach has been replicated in similar ways, multiple times at local to global scales and by using different valuation and LCLU data sets (Sutton and Costanza 2002; Troy and Wilson 2006). Besides that, several studies use LCLU in combination with *unit values* in order to complement their findings on ESS, which they investigate more in depth. Typically, such studies map one or a few ESS by *validated* or *non-validated models* in combination with different valuation methods. Further ESS values are then included by the rather simple combination of LCLU and *unit values* in order to allow for a more comprehensive assessment of ESS value.

Validated models in combination with *unit values* are used by about 25% of all studies. For example, Guo et al. (2001) value forest water flow regulation by its positive effect on electricity production in a downstream hydropower plant. The total value estimate is distributed across the study area in accordance with the contribution to water flow regulation, which is modelled based on vegetation, soil and slope angle. The model is calibrated based on "*in-situ surveys and field experiments*". (Brainard et al. 2009) model carbon sequestration in Welsh forests for live wood, wood products and soils. Carbon sequestration differs spatially due to variation in tree species and yield classes which are modelled based on several spatial variables such as climate data, soil types and legal status. The model is calibrated based on multiple forest records, but includes also some assumptions based on the researchers' best judgments. Carbon is valued by one uniform value estimate per ton sequestered carbon. Simonit & Perrings (2011) model the impact of wetlands on the water quality in Lake Victoria. Data for model calibration is not taken

[2] The classification of some studies was difficult (mainly the differentiation between validated and *non-validated models*), as not all relevant information is available in the published article. In such cases, we searched for further information within mentioned references.
[3] Note that a number studies use different methodologies for mapping values of different ESS.

from the study area itself, but from "*closely allied systems*". A uniform value is estimated per unit of nutrient retention based on an estimated impact on fish catch in the downstream lake.

Also the combinations of *non-validated models* with *unit values* (19%) and *representative data* with *unit values* (10%) are used relatively frequently. (Eade and Moran 1996) map recreational values based on the assumptions that the ESS distributes across the study area are based on "*distance and visibility from tourist areas*". The total recreational value estimate for the entire study area is then distributed in accordance to the mapped ESS distribution. (Crossman et al. 2010) map agricultural production values based on yield statistics combined with constant farmer net returns for each LCLU type. (O'Higgins et al. 2010) map values of recreational clamming in a 1,800ha bay in Oregon, US. Recreational use was in a quantified spatially explicit manner based on a comprehensive survey of the study area. A constant WTP value is attributed to each recreational user.

Besides *unit values*, *value functions* are the only valuation method used relatively often, mainly in combination with *validated models* (10%). (Polasky et al. 2008) model yields and net revenues of agricultural and timber products for a 3 billion hectare basin in Oregon, US. A number of studies use validated models to map recreational use, which is then valued based on *travel cost models* (Moons et al. 2008; Termansen et al. 2008; Bateman, Lovett, et al. 1999). (Grêt-Regamey et al. 2008) model the impact of forest on avalanche protection based on avalanche probability and run out zone models. Values are a function of avalanche risk reduction and property and human lives at risk.

Methodology	Unit Values	Adjusted unit values	Value functions	Meta-analytic value functions
Proxies	**AP:**16, 26, 31, 32, 36, 40, 41, 49, 52, 53, 55, 60, 64, 69; **B:** 9, 16, 26, 29, 31, 32, 35, 37, 40, 41, 47, 49, 51, 52, 53, 54, 55, 58, 59, 60, 64, 69; **BC:** 9, 16, 26, 31, 32, 40, 41, 45, 47, 49, 52, 53, 55, 59, 60, 64, 69; **CUL:** 9, 16, 26, 29, 31, 32, 40, 41, 45, 47, 49, 51, 52, 53, 55, 58, 59, 60, 64, 69; **DP:** 9, 16, 21, 26, 31, 32, 40, 41, 47, 49, 51, 52, 53, 55, 58, 59, 60, 64, 69; **E:** 9, 16, 26, 29, 31, 32, 40, 41, 45, 47, 49, 51, 52, 53, 54, 55, 59, 64, 69; **F:** 39; **FO:** 29, 45, 47, 51, 58, 59; **GHG:** 9, 16, 21, 24, 26, 31, 32, 35, 37, 40, 41, 45, 47, 49, 51, 52, 53, 55, 58, 59, 62, 64, 69; **GR:** 9, 16, 24, 26, 31, 32, 40, 41, 47, 49, 52, 53, 55, 59, 64, 69; **Hun:** 35; **MC:** 45; **NC:** 9, 16, 26, 29, 31, 32, 40, 41, 45, 47, 49, 52, 53, 55, 59, 60, 64, 69; **P:** 9, 16, 26, 31, 32, 40, 41, 45, 47, 49, 52, 53, 55, 59, 60, 64, 69; **R:** 9, 16, 26, 29, 31, 32, 35, 39, 40, 41, 47, 49, 51, 52, 53, 55, 59, 60, 64, 66, 69; **RM:** 16, 26, 29, 31, 32, 40, 41, 45, 47, 49, 51, 52, 53, 55, 59, 60, 64, 69; **SF:** 9, 16, 26, 31, 32, 40, 41, 45, 47, 49, 51, 52, 53, 55, 58, 59, 64, 69; **T:** 13, 23, 35, 37; **WR:** 9, 16, 26, 31, 32, 40, 41, 45, 47, 49, 51, 52, 53, 55, 58, 59, 60, 64, 69; **WS:** 9, 16, 26, 29, 31, 32, 40, 41, 45, 47, 49, 51, 52, 53, 55, 59, 60, 64, 69; **WT:** 9, 16, 26, 31, 32, 40, 41, 47, 49, 51, 52, 53, 55, 58, 59, 60, 64, 69	**CUL:** 18; **Non-T:** 14; **R:** 18; **T:** 14	**CUL:** 11; **R:** 11, 24	**CUL:** 62; **B:** 8, 14, 62; **F:** 8; **Hun:** 8; **R:** 14; **RM:** 8; **DP:** 8, 62; **WT:** 8, 62; **WS:** 8, 62
Non-validated models	**AP:** 27, 57, 65; **B:** 28, 57; **CUL:** 57; **DP:** 28; **E:** 20, 57, 61; **GHG:** 3, 6, 20, 28, 54, 61, 62; **GR:** 30, 57; **NC:** 20, 30, 54, 57; **R:** 12, 13, 21, 22, 57; **RM:** 20, 30, 54, 57, 61; **SF:** 57, 61; **T:** 27; **WR:** 20, 28, 54, 57, 61; **WS:** 28**; WT:** 20, 28	**CUL:** 34	**AP:** 63; **R:** 2; **T:** 12	
Validated models	**AP:** 15, 56; **B:** 23; **GHG:** 3, 6, 12, 18, 34, 38, 62; **GR:** 56; **E:** 21, 24; **F:** 1, 39; **Hun:** 37**; MC:** 34; **NC:** 56; **R:** 5, 12, 10; **WR:** 24, 25, 33, 34**; WT:** 18, 24, 48	**WT:** 34	**AP:** 4, 38, 42; **DP:** 23; **R:** 2, 7, 35, 50; **T:** 38, 42	**R:** 62
Representative data	**AP:** 13, 18, 19; **B:** 21; **GHG:** 14; **F:** 39, 46; **Non-T:** 21; **R:** 39, 46; **RM:** 22; **WS:** 22	**R:** 44	**AP:** 35	
Implicit modelling			**AP:** 62**; CUL:** 23, 43, 62; **R:** 43, 62; **DP:** 17	**CUL:** 8; **R:** 8, 67

AP: Agricultural production, **B:** Bidiversity, **BC:** Biological Control, **CUL:** Cultural (including Amenity), **DP:** Disturbance Prevention (including storm protection, flood protection and avalanche protection), **E:** Erosion Control, **F:** Fisheries, **FO:** Food Production, **GHG:** Green House Gasses Regulation, **GR:** Gas Regulation (atmospheric chemical composition), **Hun:** Hunting, **MC:** Micro Climate Regulation, **NC:** Nutrient Cycling, **Non-T:** Non-Timber Forest Products, **P:** Pollination, **R:** Recreation, **RM:** Raw Material, **SF:** Soil Formation, **T:** Timber, **WR:** Water Regulation, **WS:** Water Supply, **WT:** Waste Treatment (including soil, air and water quality)

1. (Armstrong et al. 2003), 2. (Baerenklau et al. 2010), 3. (Bateman and Lovett 2000), 4. (Bateman, Ennew, et al. 1999), 5. (Bateman et al. 1995), 6. (Brainard et al. 2009), 7. (Brainard 1999), 8. (Brander et al. 2011), 9. (Brenner et al. 2010), 10. (Bateman, Lovett, et al. 1999), 11. (Campbell et al. 2009), 12. (Chan et al. 2011), 13. (Chen et al. 2009), 14. (Chiabai et al. 2011), 15. (Coiner et al. 2001), 16. (Costanza et al. 1997), 17. (Costanza et al. 2008), 18. (Crossman et al. 2010), 19. (Crossman and Bryan 2009), 20. (De-yong et al. 2005), 21. (Eade and Moran 1996), 22. (O'Farrell et al. 2011), 23. (Grêt-Regamey et al. 2008), 24. (Guo et al. 2001), 25. (Guo et al. 2000), 26. (Helian et al. 2011), 27. (Holzkämper and Seppelt 2007), 28. (Ingraham and Foster 2008), 29. (Isely et al. 2010), 30. (Jin et al. 2009), 31. (Konarska et al. 2002), 32. (Kreuter et al. 2001), 33. (Mashayekhi et al. 2010), 34. (McPherson et al. 2011), 35. (Moons et al. 2008), 36. (Naidoo and Adamowicz 2006), 37. (Naidoo and Ricketts 2006), 38. (Nelson et al. 2009), 39. (O'Higgins et al. 2010), 40. (Petrosillo et al. 2009), 41. (Petrosillo et al. 2010), 42. (Polasky et al. 2008), 43. (Powe et al. 1997), 44. (Rees et al. 2010), 45. (Sandhu et al. 2008), 46. (Scheurle et al. 2010), 47. (Seidl and Moraes 2000), 48. (Simonit and Perrings 2011), 49. (Sutton and Costanza 2002), 50. (Termansen et al. 2008), 51. (Troy and Wilson 2006), 52. (Williams et al. 2003), 53. (Yoshida et al. 2010), 54. (Yu et al. 2005), 55. (Yuan et al. 2006), 56. (J. Zhang et al. 2011), 57. (M. Zhang et al. 2011), 58. (W. Zhang et al. 2007), 59. (Zhao et al. 2004), 60. (Zhao et al. 2005), 61. (Zhiyuan et al. 2003), 62. (Bateman et al. 2011), 63. (Naidoo and Adamowicz 2005), 64. (Viglizzo and Frank 2006), 65. (Anderson et al. 2009)66. (Ghermandi et al. 2010), 67. (Ghermandi et al. 2011), 68. (Wei et al. 2007), 69. (Liu et al. 2010)

Table 1: Matrix of methodologies used in literature for mapping ecosystem service values.

Other methodology combinations show relatively few applications. *Implicit modelling* of ESS supply within *value functions* is used by 6% of all studies. (Costanza et al. 2008) map wetland values for storm protection. The *value function* for modelling marginal wetland values includes biophysical variables of storm probability, wind speed, storm swath and wetland area. (Powe et al. 1997) use a hedonic pricing model for mapping recreational and amenity values of forests. The model includes forest characteristics in form of an access index. About 4% combine *value functions* with *non-validated models*. For example, Baerenklau et al. (2010) map recreational values within a protected forest assuming that recreational use within the forest distributes equally from the access points and that landscape value is dependent on its visibility. Values are a function of visitor numbers, visibility and travel costs estimated for each access point. *Meta-analytic value functions* are still relatively rarely used within ESS value mapping, even though they have gained increasing attention within traditional individual site specific value transfer. About 4% of all studies use *meta-analytic value functions* in combination with *proxies* and about 3% conduct *implicit modelling* within the *meta-analytic value function*. For example, (Bateman et al. 2011) map multiple wetland ESS values based on a *meta-analytic value function*. The only biophysical variable causing values to differ spatially is the distinction between inland and coastal wetlands. Within the *meta-analytic value function* used by (Ghermandi et al. 2011) in order to map global coastal recreational values, multiple biophysical variables are used such as climate, biodiversity and accessibility. *Proxies* in combination with *value functions* are used by 3% of all studies. (Guo et al. 2001) map recreational values by using a travel cost model for valuation. However, the only biophysical feature affecting spatial value distribution is LCLU. Roughly, 1% of all studies use *non-validated models* in combination with adjusted unit value transfer. McPherson et al. (2011) map amenity values of urban trees by assuming that amenity depends on tree size. However, no primary or secondary data is used for calibration or validation of this relationship. A value per large tree is taken from one hedonic pricing study, which is then adjusted further by the number of beneficiaries in terms of residential housing density.

We identified some correlations between the methodology used and other study characteristics. However, due to the limited amount of studies for some methodology combinations, it is sometimes difficult to conclude an overall trend. Typically, studies using a combination of *proxies* and *unit values* map values of multiple ESS at the same time (mean 10), whereas more complex methodologies result in lower mean numbers of about 1 to 2.[4] All ESS values, which are mapped frequently, are commonly mapped by the combination of *proxies* and *unit values*. This results from the predominant share of this methodology combination and its high mean number of ESS values mapped per case study. Besides that, we could only identify few concentrations of certain methodology combinations being used for mapping values of a specific ESS. Recreational values are relatively frequently mapped by a variety of different methodology combinations other than *proxies* and *unit values*. Some studies use *validated models (8)*, especially in combination with *unit values* (3) or *value functions* (4). Some applications use *non-validated models* in combination with *unit values* (5) and also *implicit modelling* within (*meta-analytic*) *value function* exists (4). Some case studies map waste treatment by *validated* (3) or *non-validated models* (2), both in combination with *unit values*. Water regulation (4/5) and GHG (7/7) are mapped frequently by validated or *non-validated models*, always in combination with *unit values*. Also erosion is mapped by *non-validated* (3) and *validated models* (2) in combination with *unit values.* For raw materials we found five case studies using *non-validated models* in combination with *unit values.* Agriculture has some applications of *non-validated* (4) and *representative data* (4); mainly in combination with *unit values* but also some applications of *validated models* (5), mainly in combination with *value functions*. Also the different policy applications show some patterns with respect to the methodology used in the studies. Green Accounting is dominantly mentioned within studies using *unit values*, either in combination with *proxies*, *non-validated models* or *representative data*. RA and LUPE are mentioned frequently within studies using *unit values* or *value functions*, both with any ESS quantification methodology. Study area sizes seem to be rather small for studies using *value functions* and for studies using *validated* or *non-validated models*. The largest mean study areas are found for studies using *proxies*. Finally, we identified a temporal trend towards the application of more sophisticated methodologies. Only 47% of all studies

[4] Only *meta-analytic value functions* in combination with proxies show a higher mean number of about 5 ESS mapped per case study. However, only three case studies were found for this methodology combination.

published after 2007 use *proxies* or *non-validated models* combined with *unit values* or *adjusted unit values*. For the sum of all other methodology combination, this share amounts to 75%.

4.4. Accuracy and Precision in ESS Values Mapping

An important and insufficiently assessed issue in mapping of ESS values is the accuracy and the precision of such maps. If ESS value maps are used to support policy decisions, policy-makers need to know how reliable the mapped values are. How close are the estimated values to the real ESS values? Does the value map provide accurate and precise site-specific value estimates, or does it display coarse trends at landscape level, or does it only give a rough estimate of total ESS values in the case study area?

Reviewing the literature, we found that about one third of all studies does not address the question of accuracy and precision of their mapped values at all, even though existing evidence show that errors in value mapping may be high. About 58% of all studies at least discuss potential value mapping errors qualitatively. However, only a minor part of all studies gives quantitative information on error margins of their results either by displaying parameter estimates from the statistical analysis, by estimating boundaries within which the actual values may most likely lie, by conducting sensitivity analysis or by comparing predictions with real world observations (see Figure 12). Due to the limited number of studies quantifying error margins, it is not possible to draw conclusions on which method may deliver the most accurate and precise value maps. However, some conclusions can also be drawn from the value transfer and ESS modelling literature.

Errors in ESS value mapping may result from inaccurate/imprecise mapping of ESS supply and their values. Both of them can be subdivided into four sources of errors: (1) *Errors in the primary ESS supply and value estimates*, (2) *uniformity*, *generalisation* or *interpolation errors*, (3) *sampling* or *publication errors* and (4) *regionalization* or *extrapolation errors* (Eigenbrod et al. 2010a; Eigenbrod et al. 2010b).

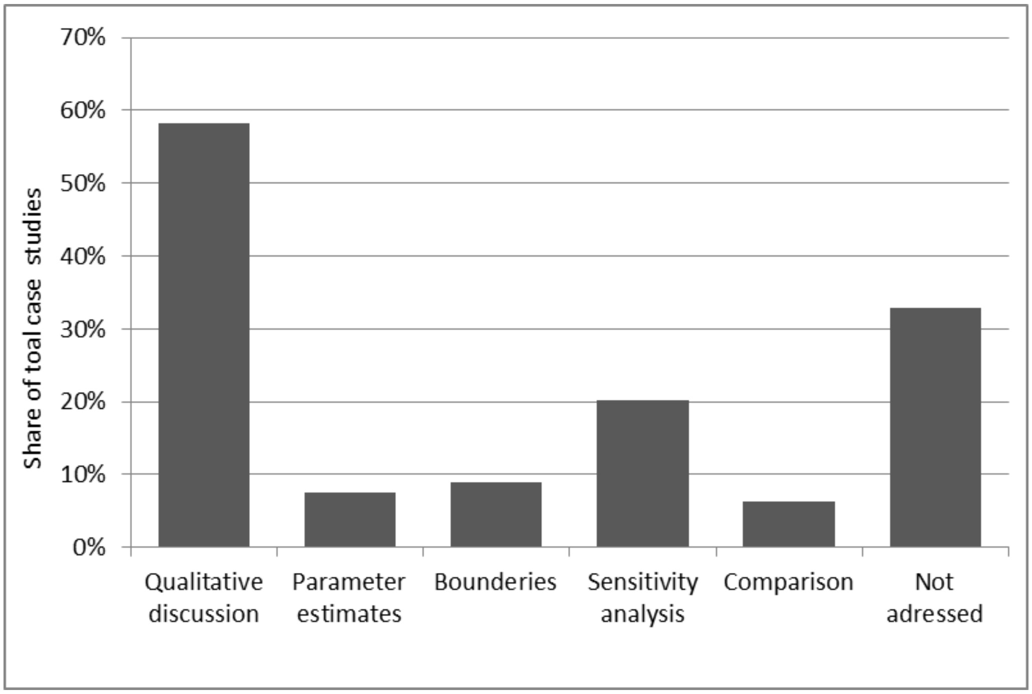

Figure 12: Assessment of results accuracy.

(1) Errors in primary data collection may depend on the methods and care in taking samples. Meta-analyses report that sample results can be statistically significantly different for different primary data collection techniques, both for ESS measurements and primary valuation. (2) *Uniformity, generalisation* or *interpolation* errors result from the fact that ESS supply and its values are considered to be constant across heterogenic ecosystems, even though ESS supply and its values differ due to factors, which are not observable or are not accounted for in the mapping exercise. (3) *Sampling* or *publication bias* errors result from the fact that primary data may not be representative for the study area. Reasons for this are for example higher publication rates of statistically significant and prior expectation supporting results and non-representative study site selection due to researchers' interests and research funding policy (Stanley and Rosenberger 2009; Rosenberger and Johnston 2009). (4) *Regionalization* or *extrapolation* errors may occur when values are transferred between different areas that are characterized by different ESS supply and demand. Due to limited data availability, primary data may often be taken from samples outside of the study site and therefore, their transferability may be limited (Eigenbrod et al. 2010b; Eigenbrod et al. 2010a; Rosenberger and Phipps 2007; Johnston and Rosenberger 2010).

The few studies quantifying accuracy of their mapped values show considerable errors. (Konarska et al. 2002) use LCLU *proxies* and *unit values* to compare how different resolutions of LCLU datasets influence the results of total ESS values in the US. The total value estimate increased by a factor of two for the finer resolution, because the share of high value and highly fragmented LCLU increased. Using *MAFV* for mapping wetland values across the EU, (Brander et al. 2011) report 95% confidence intervals of the total wetland value predictions per country. The lower bound differs to the upper bound up to a factor of two. (Costanza et al. 1997) conduct sensitivity analysis on the ESS value estimates, which they attribute to the different biomes in order to map global ESS values. The total value estimate differs by a factor of more than three. Conducting sensitivity analysis by limiting marginal storm protection values to increase for small wetland areas, total value estimate differed by a factor of almost seven (Costanza et al. 2008). By applying different valuation methodologies for mapping water supply values, (O'Farrell et al. 2011) estimate that total values differ by a factor of about six.

The reported error margins here are the sum of mean errors over large areas and give no information on the precision and accuracy for any site specific estimate. Such errors may be far higher. (Eigenbrod et al. 2010b; Eigenbrod et al. 2010a) estimate errors associated with ESS mapping via land cover *proxies*. Land cover based ESS maps and maps, which are based on representative primary data, show correlations from 0.37 for biodiversity, 0.42 for recreation and 0.57 for carbon storage. Combining their results with *unit values* in order to derive an ESS value map would result in even higher errors, as values per unit of ESS supply may again differ across space. However, for mapping recreation they find that including additional explanatory variables for population and accessibility increased the correlation for recreation to at least 0.50. (Brookshire et al. 2007) assess the impact of uncertainties in economic valuation and biophysical models on the value of water resources in a river basin for agricultural, domestic and conservation use. They conclude that uncertainties result from the valuation and population predictions rather than from the biophysical ESS modelling.

For conventional value transfer most studies find site specific transfer errors between 0 to 100% (Eigenbrod et al. 2010a), but also higher errors are reported. Some authors argue that function transfers may result in lower transfer errors, even though evidence is mixed

(Akter and Grafton 2010). In general, transfer errors tend to increase if study sites and policy sites are more heterogenic. However, due to the potential of (*meta-analytic*) *value function* approaches to make adjustments that reflect site-specific characteristics, these methods tend to be superior to (*adjusted*) *unit values* transfer in cases where sites differ heavily (Eigenbrod et al. 2010a). Some studies compare *meta-analytic value function* transfer with *value function* transfer, but do not reach a consensus on which method is preferable. The accuracy of (*meta-analytic*) *value function* transfer depends on the quality of the primary research being used to calibrate the *value function* and the available explanatory variables (Johnston and Rosenberger 2010). If *meta-analytic value functions* are only based on few observations and explanatory variables, they are likely to produce inaccurate predictions. A potential source of transfer error is that most (*meta-analytic*) *value functions* do not (or insufficiently) include site-specific bio-physical indicators in order to account for differences in ESS supply (Rosenberger and Phipps 2007; Johnston and Rosenberger 2010).

4.5. Discussion of Methodologies

Currently no consensus exists in the literature on which method best to use for a specific purpose and under specific circumstances. Several factors may determine the methodology choice, such as data availability, the ESS assessed, study area characteristics, the available resources and the political and scientific purpose of the study. Advantages and disadvantages of each methodology combinations depend heavily on the quality and the background of the individual study. Anyhow, we evaluate each methodology combination by giving a tentative quality judgement on their advantages and disadvantages (see Table 2).

The different political applications of ESS value mapping may demand different requirements in terms of accuracy and precision. If results are used for G*reen Accounting*, an accurate overall value estimate of the entire study area's ESS supply may be desired. However, precision – meaning to display accurate value estimates for each pixel of the map – may be of minor importance. Also *land use policy evaluation* may rather require accurate total value estimates of the different land use scenarios than precise value maps. In contrast, if results are used for *resource allocation,* for designing spatially explicit *payments schemes for ESS* or for a *valuation database,* accuracy but also precision are of major importance. In any case, if results are used for real policy support, comprehensiveness in terms of ESS assessed is of major importance. If relevant ESS are not covered within the value map, it may change the ranking order of alternative policy options (de Groot et al. 2010).

The advantage of LCLU *proxies* and *unit values* is that such data is easy to obtain. However, their correlation with location specific ESS supply and ESS values may be limited (Eigenbrod et al. 2010a; Eigenbrod et al. 2010b). The assumptions of uniform ESS supply and values across the same land covers, as done by (Costanza et al. 1997) and repeated by many others, can be considered as huge simplification (Plummer 2009; Eigenbrod et al. 2010b). It may hold for small and homogeneous case study areas and for ESS, which by their nature are less prone to spatial variations in their supply and values. For example, it could be considered that spatial variations are low for agricultural yields and that their productions costs within study areas are characterized by relatively similar climate and soil properties. In contrast, recreational use may even differ strongly across a relatively small homogenous forest due to limited diffusion of visitors away from access points. Nevertheless, LCLU *proxies* and *unit values* may still result in an accurate overall value estimate of entire study area's ESS supply, if correct mean values per LCLU are applied. However, it may offer little information for a specific location on the map. Furthermore, if mean values are transferred that were derived within totally different spatial contexts, the information provided may be low; both in terms of precision and accuracy (Plummer 2009; Tallis and Polasky 2009).

Ecological production functions have the advantage that they allow the mapping of ESS supply more precisely across larger and heterogeneous areas by accounting for a number of spatial variables. However, their application may be limited due to complexity and effort in model construction and due to the unavailability of consistent comprehensive ESS indicators, especially for larger study areas. Applied models differ strongly in their complexity and the extent to which they incorporate site-specific characteristics. This may result in a wide range of accuracy and precision. Mapping ESS based on *non-validated models* can be considered as a pragmatic approach that combines best available knowledge (de Groot et al. 2010). However, the quality of *non-validated models* remains intransparent and depends heavily on the researchers' judgment regarding the variables' causal combinations, whereas models based on primary data allow for validity testing. Thereby, researchers have stronger incentives to produce more accurate and precise value maps. The share of studies that do not discuss the issue of accuracy is especially high for studies using *non-validated models* (almost 60%), in particular in combination with *unit values*. 67% of these studies do not give any reference to the accuracy of their results.

Representative data on ESS supply can result in very accurate and precise maps of ESS supply if samples are carefully collected. However, data collection is a very time consuming procedure and therefore its application is limited to small case study areas or coarse resolutions.

Implicit modelling has the advantage that it allows research with a limited ecological background to include bio-physical indicators as explanatory variables into (*meta-analytic) value functions*. Thereby it can account for variations in ESS supply and the value per ESS unit, which may have a higher impact on the areas total ESS value. However, modelling ESS supply and its value at once introduces additional complexity, which may result in less accurate and spatially explicit ESS value maps. The number of variables used within *meta-analytic value functions* is limited by the availability of primary value estimates used for the regression analysis. Thus, rarely valued ESS such as most regulating services can only be assessed by relatively simple *meta-analytic value functions*. Consequently, it may be of advantage to model ESS supply and values separately. If spatial variations in ESS supply are already explained, *meta-analytic value functions* may predict remaining spatial variations of values per unit of ESS supply more efficiently.

Methodology	Unit values	Adjusted unit values	Value functions	Meta-analytic value functions
Proxies	Simple / Low data requirements / Low precision/ Intransparent quality	Simple / Low data requirements / Low precision/ Intransparent quality	Medium complexity / medium data requirements / medium precision/ Intransparent quality	Medium complexity / medium data requirements / medium precision/ transparent quality
Non-validated models	Medium complexity / medium data requirements / medium precision/ intransparent quality	Medium complexity / medium data requirements / medium precision/ intransparent quality	High complexity / medium data requirements / high precision/ intransparent quality	High complexity / high data requirements / high precision/ transparent quality
Validated models	Medium complexity / medium data requirements / medium spatial explicitness, transparent quality	Medium complexity / medium data requirements medium spatial explicitness, transparent quality	High complexity High data requirements high spatial explicitness transparent quality	High complexity Very high data requirements high spatial explicitness very transparent quality
Representative data	Simple / High data requirements medium spatial explicitness intransparent quality	Simple / High data requirements medium spatial explicitness intransparent quality	Medium complexity / High data requirements high spatial explicitness intransparent quality	Medium complexity / Very high data requirements high spatial explicitness intransparent quality
Implicit modelling	–	–	Medium complexity / medium data requirements medium spatial explicitness intransparent quality	Medium complexity High data requirements medium spatial explicitness transparent quality

Table 2: Evaluation of methodologies.

The specific strengths and weaknesses of different value transfer methodologies are discussed widely in value transfer literature and remain similar for ESS value mapping, but with some further specifications. The way they account for value determining spatial characteristics is in particular relevant if values are mapped across large case study areas, as study areas tend to be more heterogenic the larger they get. Furthermore, one main advantage of ESS value mapping results from the possibility of revealing how values differ across space. *Adjusted unit values* allows for adapting values across space by some variables, which have shown strong impacts on ESS values, such as income levels or number of beneficiaries. The value of flood control depends on property values at risk (de Kok and Grossmann 2010). Values of air quality improvements depend on the number of inhabitants living in the improved airshed. However, ESS values may differ spatially due to further spatial circumstances such as the availability of substitutes and different human preferences of dissimilar sociocultural groups. (*Meta-analytic*) *value functions* allow for incorporating such additional value determining factors and may thereby deliver more accurate and spatially explicit ESS value maps, especially for heterogenic study areas (Bateman and Jones 2003; Johnston and Rosenberger 2010; Rosenberger and Phipps 2007; Nelson and Daily 2010). However, they are more complex and time consuming to develop and they require comprehensive data sets of the explanatory variables across the entire study area, which can be a limiting factor for their application.

Typically, *value functions* are estimated for a specific location. However, parameters of the variables may be different in other locations, especially, if values are transferred across national or cultural borders (Johnston and Rosenberger 2010). This may limit the accuracy and precision of *value functions* for larger case study areas. An advantage of a *meta-analytic value functions* is that they are based on multiple primary estimates, which can be collected across a large area and which use diverging valuation methodologies. Thereby, *meta-analytic value functions* allow capturing the impacts of a greater heterogeneity within site and context variables and, it can be accounted for impacts of methodologies in primary valuation studies (Bateman and Jones 2003; Brander et al. 2010). Some evidence support that *meta-analytic value functions* outperform other value transfer techniques, if sites differ strongly and if the number of samples is large (Rosenberger and Phipps 2007). This points out that *meta-analytic value functions* may be favourable for value mapping and that its potential may increase due to a growing body of primary valuation studies. Furthermore, *meta-analytic value functions* allow for comparing predictions with real world observations and thereby for quantification of prediction errors. However, meta-analysis require a broad and qualitative database on primary value estimates, which is a time consuming procedure and which may limit its application for rarely valued ESS. Even though *meta-analytic value functions* gained a lot of attention in traditional value transfer literature, it has only rarely been used for mapping ESS values.

5. Future Prospects in ESS Value Mapping

There are several issues within ESS value mapping being of interest for future research. The challenge is to make ESS value maps more accurate, more precise and more comprehensive and link them to issues of political concern. Finally, the role of biodiversity and ecosystems resilience is yet insufficiently understood.

Barriers in accurate highly spatial ESS value mapping are manifold. Mapping of ESS and their values is depending on qualitative, comprehensive and high resolution input data, both, for models calibration and as explanatory variables for extrapolation. With improved remote sensing technologies and with continuous sampling, this data pool can be expected to grow in quantity, quality and spatial resolution. Efforts are required to harmonize available data and to construct online meta-databases in order to allow access for as many researchers as possible, such as within initiatives of the "*The Ecosystem Services Partnership*" and "*Earth Economics"*. Quality and reporting standards for primary data collection have been suggested for several times in order to allow easier statistical assessments (Eigenbrod et al. 2010a; Rosenberger and Phipps 2007; Johnston and Rosenberger 2010). Furthermore, still little is known about many spatial determents of ESS supply and its values, for example, how values differ across space due to different institutions and attitudes (Kotchen and Reiling 2000; Spash and Vatn 2006; Pritchard Jr. et al. 2000), how different ESS are interlinked and how biodiversity contributes to ESS supply (Nicholson et al. 2009).

Accounting for the determents of both, ESS supply and their values, requires a deeper integration of the involved disciplines (Bockstael et al. 2000). Still, many studies have rather an economic or an ecological background, but only a limited number combines the strengths of both perspectives. Whereas ecologically dominated studies tend to come up with sophisticated ESS models, they tend to value them by rudimentary *unit values* methodologies. In absence of qualitative valuation data, several studies combine quickly derived value estimates, such as expenditure data, replacements costs and market prices for different ESS, but without any reference to the meaning and accuracy of such different value measures. On the other hand, economic dominated studies may focus on the valuation process, but tend to rely on LCLU *proxies* or *implicit modelling* for ESS quantification. Within ESS modelling, attention needs also to be given to the definition and distinction of different ESS in order to avoid double counting and in order to fit model results into environmental economic valuation metrics.

Covering values of all relevant ESS is of great importance for policy decision support as only comprehensive value maps allow for identifying desirable policy measures (Tallis and Polasky 2009; de Groot et al. 2010). However, comprehensive ESS value maps trade off with accuracy and precision. Typically, studies mapping values of multiple ESS combine simple LCLU *proxies* with *unit values*. The creation of meta ESS models – including the feedbacks and linkages between different ESS – is a great challenge, not only due to limitations in computer processing power, but especially in harmonizing input and output variables of different models (Tallis and Polasky 2009; Nicholson et al. 2009).

Furthermore, policy orientation of many studies is still poor. Only about 35% of all studies evaluate some kind of scenario, which may allow for policy evaluation and conclusions. For giving guidance for policy makers, ESS value maps need to be linked to future policy assessments. Quantification and reporting of error margins in mapped values is still poor. If policy makers want to base their decision on ESS value maps, they need to know about the uncertainties and error margins related to such maps. Therefore, validating mapped values against real world observations is indispensable (de Groot et al. 2010).

Finally, still little is known about the role of biodiversity and ecosystem resilience. The recent attempts of employing the concept of ESS for arguing in favour of biodiversity protection have only partly been successful. Evidence on short term correlations between biodiversity and ESS supply are mixed (Maes, Braat, et al. 2011; Maes et al. 2012). However, the contribution of biodiversity to ecosystem resilience (its capacity to resist to disturbances) and its insurance values (the value of ensuring future ESS supply) are yet hardly quantified. The often non-linear and multi-scale relations between measurable bio-physical quantities, ESS and biodiversity are not yet sufficiently understood. When and how drivers and pressures on ESS and biodiversity hit tipping points, beyond which ecosystems shift into a less desirable state, is a critical question in ESS mapping and valuation. Ecosystem resilience and biodiversity's insurance values are yet hardly quantified. The rate of substitutability between different ESS and man-made capital, which is implied by their derived monetary values, changes drastically if thresholds are reached. Their incorporation into environmental valuation and policy scenarios analysis is of critical concern for ensuring sustainable policy recommendations (de Groot et al. 2010; Nelson and Daily 2010).

ESS value mapping is gaining increased attention in current research and there are a number of initiatives progressing in ESS value mapping. The TEEB project is mapping global ESS values based on LCLU *proxies*, but transferring values based on *meta-analytic value function* (TEEB 2010). Similar, the AIRES project develops value up-scaling methodologies further in order to derive more accurate ESS value maps. The UK NEA maps ESS values of agricultural and timber product, carbon storage and recreation across the UK. It combines different methodologies of mapping ESS supply, from comprehensive agricultural production data to validated production functions for timber, carbon storage and recreation (Bateman et al. 2010). The InVest tool aims at combining the capacities of researchers with different disciplinary backgrounds, in order to map multiple ESS supply and their values by combining different models and valuation methodologies (Tallis and Polasky 2009).

6. Conclusion

With the emergence of advanced GIS technology, spatial issues in environmental valuation gained increasing attention and the importance of spatial relationships in ESS valuation became more and more recognized. Studies mapping ESS values by displaying how ESS values differ across space grew exponentially in recent years. As compared to traditional site-specific valuation, ESS value mapping offers additional beneficial information, such as evaluating broad land-use policies by displaying trade-offs and synergies of different policies and identifying preferable locations for policy measures.

However, studies differ widely within their spatial scope, their purpose, their disciplinary background and by the ESS assessed. A great variety exists in the methodologies used for revealing how ESS supply and ESS values differ across space. Spatial variations in ESS values can be assessed by estimating spatial variations in ESS supply, the value per unit of ESS supply or by both of them. In this paper, we developed a matrix for classifying studies with respect to the methodologies applied for ESS mapping and its valuation. Methodologies for ESS supply mapping include one-dimensional *proxies*, *validated* and *non-validated models*, *representative data* and *implicit modelling* within (*meta-analytic*) *value functions*. ESS valuation methodologies include *unit value*, *adjusted unit value*, *value function* and *meta-analytic value function*. However, until now, no consensus exits about which methodology to use best for which purpose.

Accuracy and precision are issues of great concern in ESS value mapping, which is yet insufficiently addressed in literature. Only a minor part of all studies assess this issue in a quantitative manner, even though evidence show that error margins can be large. Due to coarse assessments and large uncertainty within mapped values, some studies may hardly qualify for any site-specific policy suggestions. The “Costanza approach” of combining LCLU *proxies* with *unit values*, which were derived in specific contexts, may display coarse trends at landscape level, but may give only little information for site-specific assessments. The study of (Costanza et al. 1997) may have been a step stone at its time, but its limitations have been widely discussed in literature and mainly been recognized within the study itself. The current challenge within research is to develop spatially explicit ESS models supply combined with spatially explicit (*meta-analytic) value* *functions*; both validated on real world observations in order to allow for accuracy assessment and discussion. Some promising initiatives exist, such as UK NEA, AIRES, INVEST or TEEB. However, still most studies focus either on the spatial distribution of ESS supply or on the spatial distribution of its value per ESS unit, but only few studies undertake efforts to incorporate both dimensions in a highly sophisticated manner. Mapping of ESS value is a very interdisciplinary exercise and it requires the integration of researchers with ecological and economic backgrounds in order to utilize their specific strengths in assessing either the spatial biophysical or socioeconomic dimension of ESS values.

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