Spatially Explicit Assessment of Recreational Ecosystem Services

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List of Abbreviations:

AIC	*Akaike Information Criterion*
AIRES	*ARtificial Intelligence for Ecosystem Services*
BIC	*Bayesian Information Criterion*
BOKU	*Universität für BOdenKultur*
CBA	*Cost-Benefit-Analysis*
CDDA	*Common Database on Designated Areas*
CICES	*Common International Classification of Ecosystem Services*
COP 10	*Conference of Parties*
CORINE	*COoRdination of INformation on the Environment*
CSP	*Consumer SurPlus*
CVM	*Contingent Valuation Method*
DOPA	*Digital Observatory for Protected Areas*
ESP	*Ecosystem Services Partnership*
ESS	*EcoSystem Services*
EU	*European Union*
EU DG ENV	*Environment Directorate-General - Environment*
GCP	*Gross Cell Product*
GDP	*Gross Domestic Product*
GHG	*Green House Gas*
GIS	*Geographic Information System*
GLM	*General Linear Regression*
GLMM	*General Linear Mixed Model*
GPS	*Global Positioning System*
InVest	*Integrated Valuation of ecosystem services and tradeoffs*
IPBES	*Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*
IUCN	*International Union for Conservation of Nature*
LCLU	*Land Cover or Land Use*
MA	*Millennium Ecosystem Assessment*
MAES	*Mapping and Assessment of Ecosystems and their Services*
MCA	*Multi-Criteria-Analysis*
MLM	*MultiLevel Modelling*
MMV	*Management and Monitoring Visitors in recreational areas*

NEA	*National Ecosystem Assessment*
NP	*National Park*
NUTS	*Nomenclature of Territorial Units for Statistics*
OSM	*Open Street Map*
PA	*Protected Area*
PEER	*Partnership for European Environmental Research*
PPP	*Purchasing Power Parity*
PRESS	*PEER Research on EcoSystem Services*
R^2	*Coefficient of determination*
REED+	*Reducing Emissions from Deforestation and Forest Degradation in Developing Countries Plus*
RMSE	*Root Mean Square Deviation*
RP	*Revealed Preference*
SAC	*Spatial Auto Correlation*
sd	*standard deviation*
SEA	*System of Environmental Accounts*
SEEA	*System of Environmental-Economic Accounting*
TCM	*Travel Cost Method*
TEEB	*The Economics of Ecosystems and Biodiversity*
TEV	*Total Economic Value*
UK	*United Kingdom*
UNEP	*United Nations Environment Programme*
WCMC	*World Conservation Monitoring Centre*
UNESCO	*United Nations Educational, Scientific and Cultural Organization*
VV	*Value per Visit*
WDPA	*World Database of Protected Areas*
WTP	*Willingness To Pay*
CV RMSE	*Coefficient of Variation of the RMSD*

1 Introduction

"*The most unique feature of Earth is the existence of life, and the most extraordinary feature of life is its diversity*" (Cardinale *et al.* 2012). Its conservation is *"a common concern of humankind*" (UN 1992).

The diversity of life is dramatically affected by human beings (Butchart *et al.* 2010; Sala *et al.* 2000). The current species extinction rate is considered to be 1000 times higher than the natural rate would be without intense human influence on global ecosystems. Besides global warming, nitrogen deposition and biotic exchange, habitat degradation in consequence of land cover change represent the main drivers of biodiversity loss. Nowadays, less than half of tropical forests - the most bio-diverse habitats on the planet - remain (Pimm and Raven 2000; Tollefson 2015). Biodiversity loss is expected to accelerate, if the current rate of habitat degradation persists (Pimm and Raven 2000).

In the last decades, it became more and more recognized, that the loss of biodiversity has fundamental effects on humans and their well-being (Cardinale *et al.* 2012; Díaz *et al.* 2006). Biodiversity is essential for ecosystems to function, producing multiple fundamental services that benefit human beings (Loreau *et al.* 2001). Because of market failures, free market economies tend to overexploit natural capital resulting in biodiversity loss beyond the social optimum.

Politics have responded to species extinction by setting up ambitious and honourable policy goals such as by the Convention on Biological Diversity, ratified after the global summit in Rio de Janeiro (1992) and elaborated in the follow up meetings (Barbault 2011; Leadley *et al.* 2010). However, these goals have typically not been met and biodiversity remains under threat. Still, much action needs to be taken to stop biodiversity from declining (SCBD 2014).

Since the Millennium Ecosystem Assessment (MA), the concept of ecosystem services (ESS) is widely used in order to assess and highlight the contribution of nature to human well-being. Ecosystems provide ESS such as oxygen, water, food, raw materials, recreational opportunities and many more. Just like all living organisms, humans can only exist as a fraction, as a dependent within the universal framework of life on Earth (MA 2005; Müller et al. 2015).

Stressing the value of ecosystems from an anthropocentric point of view, by assssing their ESS supply, became a key component in multiple debates, research and policies on biodiversity and nature protection. If the ESS value can be made explicit, it can be instrumental in making the case for ecosystem and biodiversity protection (Maes *et al.* 2012a).

Since about two decades and the well-known publication of Costanza *et al.* (1997), spatial ESS assessments have become a vibrant research topic and the number of publications has grown exponentially (see also chapter 2). Displaying how ESS and their values differ across space offers great opportunities for nature and biodiversity conservation policies. Besides illustrative purposes, it can contribute for example to the identification of ESS hotspots, to the prioritization of resource allocation, to design location specific conservation policies and to the evaluation of synergies and trade-offs among alternative land-use policies (Crossman *et al.* 2013a; Maes *et al.* 2012a; chapter 2). Recent ESS research focuses for example on the biophysical assessment of ESS (Braat 2013; Volk 2015; chapter 4), on its economic valuation (de Groot *et al.* 2002; Groot *et al.* 2010; Meyerhoff *et al.* 2012), on synergies and trade-offs among ESS and biodiversity (Balvanera *et al.* 2006; Maes *et al.* 2012c; Nelson *et al.* 2009; Power 2010) and on its integration into policy instruments (de Groot *et al.* 2010; Farley and Costanza 2010).

Spatial assessments of ESS also became introduced in recent policy initiatives. The Global Strategic Plan for biodiversity defined at the Conference of Parties (COP 10) 2010, includes the well-known Aichi Targets on biodiversity. These consider ESS as a co-benefit of biodiversity and as a valuable asset for protection. The United Nations call for spatial ESS assessments and valuation to "*be integrated into development plans to ensure that these ecosystems receive the necessary protection and investments*"(UNEP 2013). The European Union (EU) has implemented this commitment within its Biodiversity Strategy 2020. Within Action 5, the EU requires member states to "*map and assess the state of ecosystems and their services in their national territory*" and to "*assess the economic value of such services*" (EC 2011b).

However, there is still no consensus on how to best map ESS and their values for certain applications. A variety of approaches exists, all of them with their specific strengths and weaknesses. The appropriate method depends on multiple aspects, such as the data availability, the ESS assessed, temporal and spatial scales as well as the specific purpose of the study. In recent years multiple reviews, frameworks and guidelines on ESS mapping have evolved (Crossman *et al.* 2013a; Egoh *et al.* 2008; Maes *et al.* 2012a; Martínez-Harms and Balvanera 2012; Paracchini *et al.* 2014; Troy and Wilson 2006; chapter 2). The European Commission has set up a working group to assist the Member States with the spatial ESS assessments and to develop a common framework of understanding and methods (EC 2011a).

The aim of this thesis is to contribute to the methodological development of spatial ESS assessments. An extensive literature review is presented in order to identify recent trends in the mapping of ESS and their values. Existing approaches are compared and their specific strengths and weaknesses are identified. Guidance for ESS mapping exercises and a best-practice approach is provided. The methodological findings are illustrated by a series of case studies, focusing on nature recreation (see chapters 3, 4, 5, 6 and 7). Nature recreation is a substantial ESS of natural ecosystems and protected areas (PA) and can be used as an argument in favour of nature conservation policies, which contribute to the supply of recreational opportunities and biodiversity protection at the same time (chapter 4). Globally, recreational ESS are valued to about $US 21 trillion annually, which is the second most valuable ESS (about 16% of the global TEV of all ESS) (Costanza *et al.* 2014).

The thesis is organised as follows. The subsections of chapter 1 introduce the theoretical foundations, the scientific background of spatial assessments of ESS and their economic values. Section 1.1.1 introduces the framework of ESS, first from an interdisciplinary perspective and then from an environmental economic perspective. In section 1.1.2 the economic concept of ESS valuation is summarised. Subsequently, the emergence of value transfer and ESS value mapping is illustrated. A short introduction in recreational ESS is given, which is used in the later chapters to exemplify best-practice options for spatially explicit ESS assessments. In section 1.2, the research questions of this thesis are deduced and presented. An executive summary of the subsequent six chapters is given. All chapters represent independent articles on the spatial assessments of ESS and their values. Collectively, they form a story line that contributes substantially to solving the raised research question. In chapter 2, a broad literature review is given on the mapping of ESS values. Chapter 3 presents a real world observation database of recreational use within non-urban ecosystems, which serves for the parameterisation and validation of spatial models of recreational ESS. The subsequent chapter 4 presents such a model. The model is used to map recreational visits across European national parks. In chapter 5, spatial modelling of recreational visitors is combined with a meta-analysis to assess the economic value of coral reef recreation in Southeast Asia. Thereby, it is not only accounted for the spatial variations of recreational visitor numbers, but also for spatial variations within the economic value per recreational visit. A similar approach is chosen in chapters 4, 6 and 7, but focusing on nature recreation in Europe and on a finer spatial resolution as well as on comparing alternative ESS value mapping methods. In the final chapter, chapter 8, results of the previous chapters are summarized and it is discussed how they contribute to field of research and how they answer the previously raised research questions.

1.1 Academic Background and Theoretical Foundation

1.1.1 The Concept of ESS

ESS are understood as the benefits humans obtain from nature (MA 2005). Ecosystems provide humans with oxygen, water, food, raw materials, flood protection, recreational opportunities and much more. Human life depends fully on the supply of ESS.

Because of market failures, markets prices do not fully reflect the costs and benefits related to ESS. In consequence, free markets lead to overexploitation and degradation of natural resources and ecosystems. This lead to a decline of ESS supply beyond the social optimum. The ESS concept is used to highlight and communicate the total benefits of ESS that are not accounted for in free markets. Thereby it can increase society's and decision-makers' environmental awareness and can support the design of adequate policies to halt excessive ecosystem degradation.

The concept of ESS was first named and explicitly recognised in a report on *Man's Impact On The Global Environment* in 1970 (SCEP 1970) even though the general understanding of ecosystems' contribution to human welfare had existed for centuries. Plato already recognised that deforestation leads to soil erosion and the drying of springs (Mooney *et al.* 1997). Since at least the MA 2005, the framework is widely used within politics and for communicating links between ecosystems and human well-being (MA, 2005), and the value of ESS has been assessed in numerous studies (Costanza *et al.* 2014; chapter 2). In chapter 4, we estimate the recreational value of European national parks to about € 14.5 billion annually. In chapter 5, we estimate the loss of reef recreation value in South-East Asia due to ecosystem degradation by 2050 to about US$ 128 million (see chapter 5) and in chapter 7, we estimate the recreational value or Europe’s countryside at € 57 billion annually. de Groot *et al.* (2012) estimate global ESS values to range from US$ 490 to 350,000 per year and ha. The total ESS value of world's ESS amounts to US$ 125 trillion annually. The growing interest in ESS valuation has for the most part been motivated by the search for arguments in favour of conservation and biodiversity protection (Salles 2011).

A number of classification schemes exist for ESS, of which the most common one is probably from the MA 2003, which is adopted in many policy initiatives and subsequent studies. However, also the TEEB (The Economics of Ecosystems and Biodiversity, Groot *et al.* 2010) and the CICES (Common International Classification of Ecosystem Services) classification schemes (Haines-Young and Potschin, 2012) are well-known within the scientific community. Within the MA classification scheme ESS are distinguished by four groups: (1) provisioning services, (2) regulating services. (3) cultural services and (4) supporting services (MA, 2005). Figure 1 represents the different ESS classes and how they contribute to human well-being.

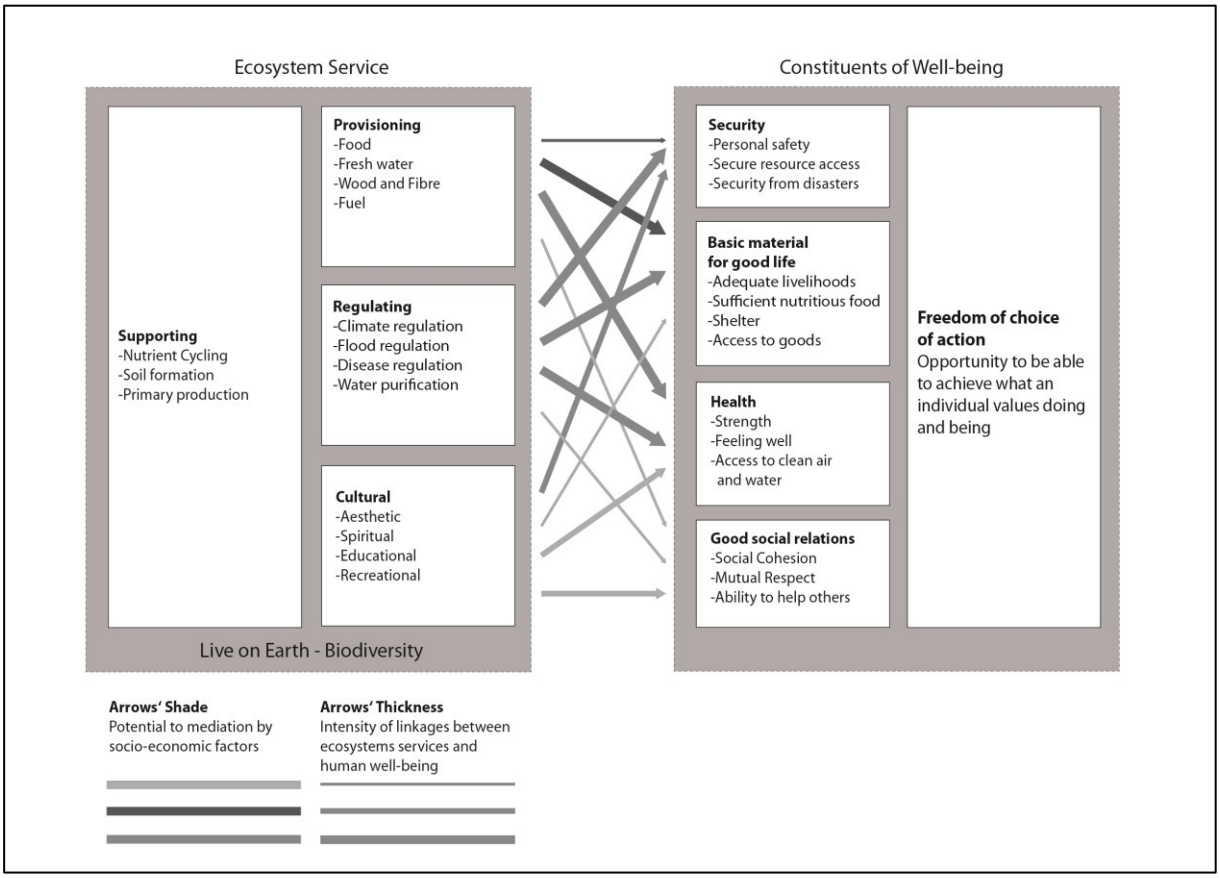

Figure 1: Linkages between ESS and human well-being (MA 2005).

(1) Provisioning services represent goods supplied by ecosystems such as food products, drinking water, raw material, fuels such as wood and fertilizer, genetic resources being used for example in biotechnology, biochemicals such as medicines and also ornamental resources used for example in arts. (2) Regulating services regulate processes of an ecosystem, as for example climate regulation, water and air quality regulation, erosion control, flood protection and pest control. (3) Cultural ESS are nonmaterial benefits. They include for example the aesthetic enjoyment of a landscape, recreation and educational benefits but also spiritual and religious benefits. (4) Supporting services contribute to the supply of other ESS. They include soil formation, photosynthesis and nutrient cycling. The distinction between supporting services and ecosystem functions is not always clear cut. Some classifications may consider supporting services as functions (Fisher and Turner 2008; Turner *et al.* 2000). All ESS contribute to different components of human well-being: the basic material needs, health, good social relations, security. They form the basis for the humans' freedom of choice in order to achieve and be what an individual values.

The contribution of each ESS to human well-being is assessed by different metrics. Whereas Sen (1988) attributes well-being with the freedom of humans' choice, economists tend to assess economic value as indicator for well-being. The assessment of ESS' economic values has two dimensions. The biophysical dimension describes the steps from certain ecosystem structures and processes over ecosystem functions to ESS. The economic dimension describes the process from an ecosystem service to its economic value (chapter 2). A useful framework to access ESS, which has been widely applied and developed further, is the cascade model (Haines-Young and Potschin 2010). It describes the linkages between an ecosystem and the ESS value by a sort of production chain (see Figure 2). The ecosystem is characterised by certain biophysical structures and processes such as geomorphology, climate and biodiversity. Those structures and processes result in certain ecosystem functions. Vegetation for example may slow the passing of water. If the ecosystem function is perceived as useful by humans, it becomes an ESS. Depending on the local conditions, the slowing of water may for example contribute to flood protection or drought prevention. Finally, again depending on the local circumstances, the ESS has a certain economic value. The value of the ESS has a feedback on the habitat itself by stimulating human actions, which may cause pressure on the ecosystem. Policy action is therefore required in order to limit these pressures to protect biodiversity and to prevent ecosystem degradation. Describing the functional form of the linkages between the different steps is a great issue in recent ESS research and one of main research questions of the thesis at hand.

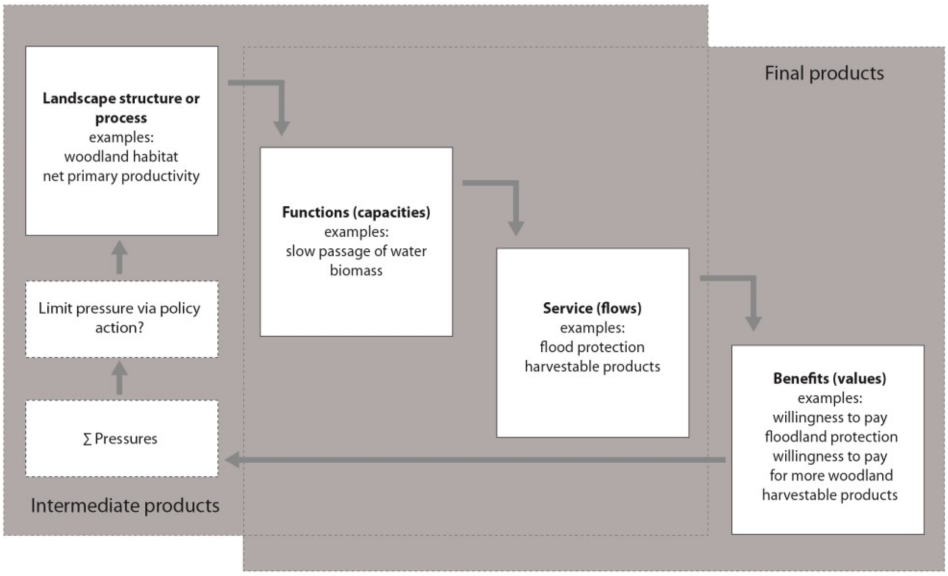

Figure 2: The relationship between biodiversity, ecosystem function and human well-being (modified from Haines-Young & Potschin 2010).

More and more policies integrate the concept of ESS, either explicitly or implicitly. On the international level, a global Strategic Plan for Biodiversity was set up under the umbrella of the International Convention on Biodiversity, which includes the well-known Aichi Targets on biodiversity. Among other aspects, they consider ESS as a co-benefit of biodiversity protection and as a valuable asset worthwhile to be protected on its own. Spatial assessments of ESS and their values are to "*be integrated into development plans to ensure that these ecosystems receive the necessary protection and investments*"(UNEP 2013). Within Action 5, the EU requires Member States to "*map and assess the state of ecosystems and their services in their national territories*" and to "*assess the economic value of such services*" (EC 2011b). The Clean Development Mechanism of Kyoto Protocol allows polluters to pay for emission reductions elsewhere, which may be a result from ESS such as carbon capture from deforestation (UN 1998). A similar approach is followed in the REED+[1] mechanism. The recently established Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is meant to improve the dialog between science and policy makers in order to trigger adequate policy

[1] REDD stands for reducing emissions from deforestation and forest degradation.

responses for environmental protection (Larigauderie and Mooney 2010). Also within national policies, multiple examples exist in order to maintain the flow of ESS. Florida pays farmers to maintain wetlands for water flow regulation. In the Tualatin Basin, Oregon farmers are paid to plant trees along streams in order to cool water flows by affording shade. Seattle invested in natural landscapes to reduce storm water runoff (Scarlett and Boyd 2011). In Germany farmers are paid for reducing grassland cutting in order to protect breeding birds (Hampicke 2001). Nevertheless, it is still controversial how to integrate the ESS concept best into policies and a number of issues are until now debated. The distinction between different ecosystem functions, ESS and benefits as well as their linkages to biodiversity are not yet fully understood and various approaches have been considered in literature. Double counting of ESS benefits may result from the application of differing classification schemes and unclear distinction between services and functions. Non-constant rates of substitution may be substantial for evaluating policies resulting in non-marginal ecosystem changes, but are hardly considered in policy and research. Formalizing safe minimum standards for ecosystem health may be one option to moderate this problem, but they are difficult to define. Trade-offs and synergies between different ESS are still to be explored in detail. Most studies focus only on one or a few ESS. Others use rather rough methodologies resulting in high uncertainties in ESS supply and demand estimations that make trade-off analysis to be a difficult task. Finally, distributional effects of ESS supply and measures for their preservation are a great concern, in particular when ESS producers and beneficiaries are present at different spatial and temporal scales (de Groot *et al.* 2010; Daily *et al.* 2009; Fisher *et al.* 2008).

Several studies and initiatives aim to integrate ESS assessments into decision making processes, such as the TEEB project, national ecosystem assessments or the InVest (Integrated Valuation of ecosystem services and trade-offs) modelling platform (TEEB 2010; UK NEA 2011; NCP 2015; ARIES 2015) and thereby try to contribute to stopping the loss of biodiversity and ecosystem degradation. In order to define efficient ESS policies, accurate assessments of the ESS supply and their values are a precondition. However, there is still no consensus on how to assess ESS and their values for certain applications in the best way and how to design adequate policy measures for their protection.

1.1.2 Valuing Ecosystem Services

Valuing ESS means to assess their contribution to human well-being. Because of market failures such as external effects and public goods, market prices do not sufficiently reflect the costs and benefits related to ESS. In consequence, free markets result in the sub-optimal allocation of resources and thereby lead to excessive ecosystem degradation and biodiversity loss. For the design of efficient policies to moderate this problem, accurate and comprehensive accounting of all effects on ESS values is a precondition. However, this represents one major difficulty in selecting the most desirable policy measures from a set of alternatives. In particular, it is challenging to assess the multidimensional sets of impacts and their relative contribution to human well-being. Complex policy measures, such as a nature conservation project, are characterised by certain project costs, but may also affect biodiversity, greenhouse gas (GHG) emissions, farming income, recreational use, the tourism sector and various other ESS. The multidimensional impacts on the economic, environmental, social system are difficult to assess. A political decision maker may encounter difficulties in assessing such multidimensional impacts, because he may be faced with uncertain estimates regarding the different expected impacts, he may lack knowledge to interpret results from various disciplines and he may fail to weigh the different impacts' contribution to social welfare. In consequence, public administrations base their decisions on simple financial analysis or cost-effectiveness-analysis, which consider none or only one environmental impact dimension (Pearce *et al.* 2006). By valuation, the multidimensional set of impacts can be expressed in one uniform metric of human well-being or welfare.

Different metrics exist to measure the contribution of different goods and services to human well-being (Kakwani 1981; Offer 2000; Sen 1988). Within utilitarianism concepts of welfare economics, the monetary value of goods and services is commonly used as metric of welfare. Monetary values can be aggregated within cost-benefit-analysis (CBA). Thereby, multidimensional costs and benefits stemming from the impacts of alternative policy measures can be compared using a one-dimensional measure of welfare. The result represents the change in the so-called total economic value (TEV), which reflects every impact on human utility (see Figure 3). By valuation of each impact caused by the considered policy alternative, a CBA provides a rational and systematic procedure for decision-makers, considering all costs and benefits of every affected individual from current and future generations.

Turner *et al.* (2000) exemplify this using a framework, similar to the cascade model from Haines-Young and Potschin (2010) presented above, but looking at it more from an environmental economic perspective (see Figure 3). A habitat, characterised by certain boundary conditions, manifests specific ecological structures and processes. These structures and processes may form ESS which represent the interface between the ecological and the economic dimension. The values of ESS are divided by three types: (1) direct use values, (2) indirect use values and (3) non-use values.

(1) Direct use values result from a direct use of the provided goods and services, such as the use of provisioning services but also recreation or the joy of nature's aesthetics. (2) Indirect use values relate to the regulation and support of ESS through the regulation of water flows or the formation of soils. (3) Non-use values relate to values humans perceive although they do not use the ESS. For example, the pure existence and bequest value of certain habitat and species that humans perceive are considered non-use values.

Within this utilitarian concept, the sum of these values constitutes the so-called total economic value as a measure for human well-being. The total economic value is the final indicator to support policy decisions. Following economic theory, an investment project, such as a new nature reserve should be designed in order to maximize the TEV. An environmental tax or a payment for ESS scheme should be designed in order to internalise all external effects. Thus, a polluter should burden the negative external effect's impact on the TEV by an environmental tax. On the contrary, a producer of positive external effects should gain the benefits of all his positive impacts on the TEV by a subsidy.

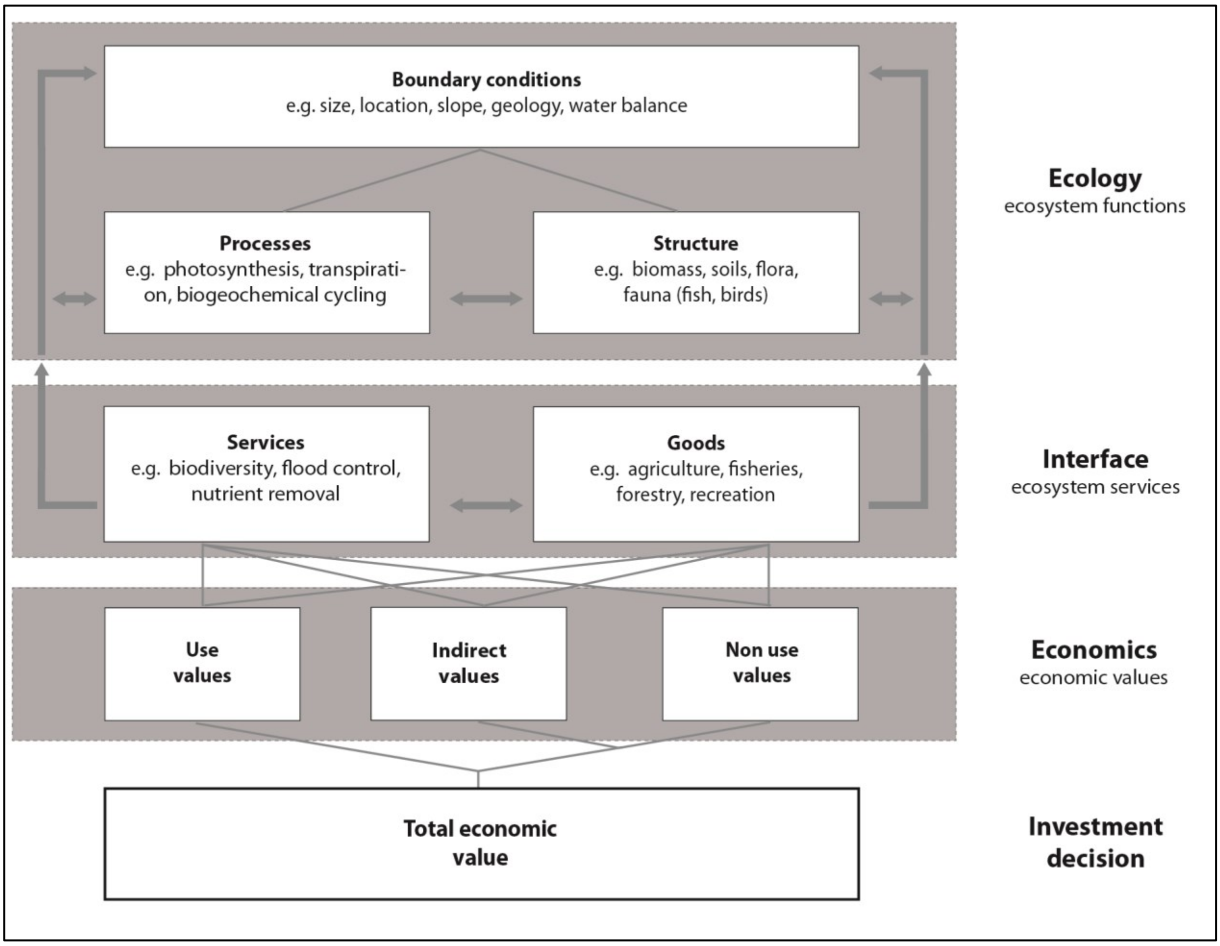

Figure 3: Ecological goods and services and their valuation. (modified from Turner et al. 2000).

The environmental economic value of an ESS is, as with any good or service, determined by its supply and demand. It is composited out of the producer and consumer surplus (minus related costs such as from policy intervention). The consumer surplus is defined as the benefits enjoyed by a consumer purchasing a good for a price that is less than his or her maximum willingness to pay. The producer surplus is defined as the benefits enjoyed by a producer selling a good for a price that is higher than his or her minimum willing to accept. Typically, for tradable goods, the short-term supply curve is a positive function of the price, as producers offer more if the price increases. On the contrary, the demand is a negative function of the price, as consumers demand less of a products if its price rises (also see Figure 4) (Mankiw 2001).

However, many ESS are not offered by producers in the market, but instead supplied by nature for free. In consequence, even though consumers may have a positive willingness to pay (WTP), at a given supply, the producer's surplus is zero. In addition, the supply curve is very inelastic (vertical in Figure 4) as the ecosystem service supply does not increase in response to an increased WTP of the consumers. In addition, the demand curve of many ESS does not have a prohibitive price (a price for a goods consumption at which its demand is zero). Because the consumption of many ESS is only substitutable up to a certain point, as a minimum is a perquisite for human life, the WTP for it becomes infinite as soon as its supply is life threatening scarce. Therefore, the total global ESS value is infinite. However, decision-making processes result typically only in a marginal change of ESS supply. Therefore, only the value of marginal changes in ESS supply is assessed in environmental economic valuation and not the total global ESS value (Costanza *et al.* 1997; Costanza *et al.* 1998; NRC 2005).

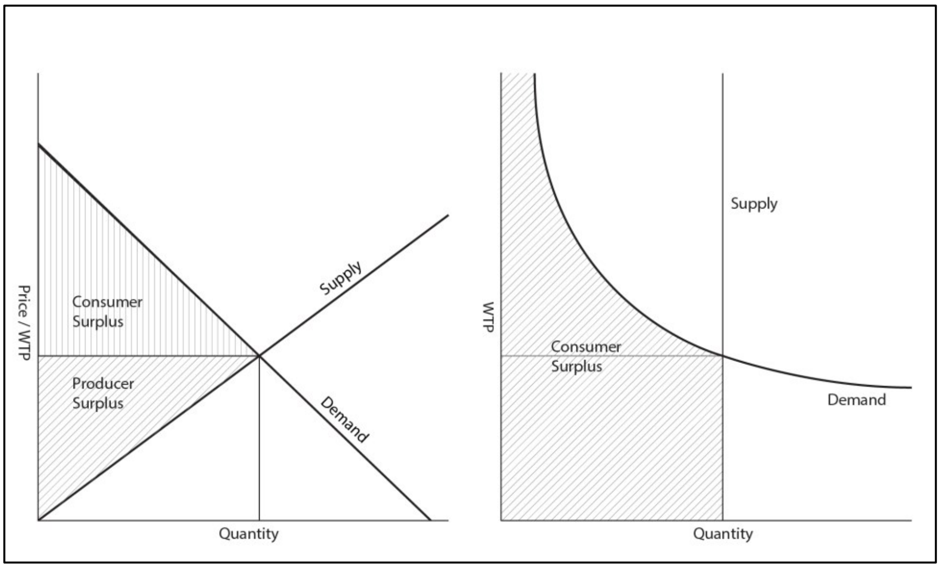

Figure 4: Supply and demand curves of traded goods (left) and ecosystem services (right).

The supply of ESS is determined by ecological processes and structures, which result in a certain quantity of ESS. ESS supply is typically expressed in biophysical units such as tons of carbon captured or tons of soil erosion prevented. The demand side of ESS is determined by human preferences, but also by the availability of substitutes, and complementary goods and services. The demanded quantity of an ESS is the amount consumed given a certain cost or price. If an ESS is abundant relative to its demand, a marginal increase of its supply has only a small effect on human well-being. On the contrary, if the ESS is scarce, a small change in supply can have a significant effect on human well-being (see Figure 4).

The supply side of an ESS is typically assessed by natural scientists whereas the demand side falls into the domain of economists. Methods used to quantify ESS supply, range from measurement or observations, the use of simple indicators such as land-cover, to complex ecological production functions or ecological models (see chapter 2). For assessing the demand side of ESS, environmental economists typically conduct monetary valuation. Several valuation methods have been developed. Revealed preferences valuation techniques analyse the behaviour of individuals in related markets, such as analyses of their WTP for housing with access to an ecosystem for recreation or nice scenic views (hedonic pricing). Another example is the analysis of costs that individuals burden to abate environmental pollution or replace the ESS (replacement cost approach). Stated preferences valuation methods derive ESS values through surveys for market simulations, such as discreet choice experiments. An alternative to primary valuation is to transfer value estimates from another study to the area of interest (see 1.1.3). This method is used for valuing recreational ESS in chapters 4, 5, 6 and 7. A good general overview on the different valuation techniques can be found in Maler and Vincent (2005) and Endres and Holm-Müller (1997).

However, the practice and theoretical foundation of monetary valuation and CBA have been subject to criticism. Due to shortcomings within monetary valuation techniques, CBA results may only be an insufficient indicator for the real impact of alternative measures on human well-being. For many ESS it appears difficult to estimate their change in supply as result of human action as well as their contribution to human well-being in relation to other social, cultural, and economic factors. Valuation techniques that rely on stated preference have especially been criticized. The hypothetical situation of any kind of questionnaire on fictive measures may not allow estimating the real society’s perception regarding environmental values. The assumptions on human behaviour that are the foundation of CBA, such as perfect information, rationality and predefined preferences, do obviously not represent reality (Coase 1984; Hardin 1968; March 1978; Michaelowa and Jotzo 2005; Pigou 1912; Sen 1977; Simon 1978; Sugden 1991).

In addition, monetary valuation cannot capture all dimensions of environmental values, but only so-called instrumental values. Monetary valuation implies certain substitutability among money and different goods and services, which, if exchanged against each other by the rate of substitution, leaves the individual the same as before. However, religious, moral and intrinsic values associated with nature cannot be compensated for by monetary means. Critics mention that monetary valuation supports a culture of environmental commoditization, which implies complete substitutability, although it is not the case. Even the rate of substitution for instrumental values is not constant and may change dramatically for extensive ecosystem changes (Kosoy and Corbera 2010; Nicholson *et al.* 2009; Ring *et al.* 2010).

Another problem is that CBA and monetary valuation can be a very costly and time consuming procedure. Typically, a gap remains between the requirements of the theoretical framework and the common practice of CBA. CBA requires that each impact dimension is at first to be quantified and then valued in monetary terms. Due to the complexity of ecosystems and their interactions, both may cause considerable difficulties (Powe 2007). Almost every CBA study dealing with complex environmental problems does not value all relevant impact dimensions or is comprised of only rough estimates. Especially social side effects, such as income distribution or employment are almost never valued within CBA. Valuation techniques for such impacts are hardly developed (Spash und Carter 2001; Spash 2008). If the difference between the considered options' TEV is small, inaccurate and incomplete, ESS valuation may cause that the ordering of the analysed options implied by the CBA may not reflect their contribution to human welfare. As a result, CBA may not be appropriate to offer a definitive policy recommendation and may leave the decision maker with almost the same problems he had before. Countless choices are made every day all around the globe, which result in environmental side effects, but their comprehensive and accurate assessment remains elusive.

In response to the critics on CBA and monetary valuation, two different schools of thought have emerged. Whereas environmental economists keep on developing more refined methods for monetary valuation, ecological economists tend to refuse the concept of monetary valuation and the TEV (Gómez-Baggethun *et al.* 2010). Instead, they come up with alternative metrics to measure the impact of alternative policies on human well-being. One alternative to CBA for supporting decision making processes are multi-criteria-analysis (MCA). Just like the CBA, MCA approaches have to estimate each option's effect on every impact dimension. Then, a judgment with regard to the relative importance of each impact dimension is necessary. In the CBA, this is done by monetary valuation. In MCA, weights may be defined by stakeholder workshops, experts or public appraisal or citizen jury (Rauschmayer und Wittmer 2006; DCLG 2009). Proponents of such participatory approaches stress that it allows the public to become familiar with the ecosystem's goods and services and thereby helps to develop preferences on the issues at stake. However, the diverse spectrum of MCA is as well no panacea for identifying desirable policy options. The complexity of alternative policies with multidimensional side effects makes them difficult to evaluate, no matter if stake holders, citizens, experts or policy makers are the ones to choose an alternative.

1.1.3 Value Transfer

One option to avoid time consuming and costly primary valuation studies, is to conduct so-called benefit transfer or value transfer[2]. The idea of value transfer is to estimate the values of certain ESS at a certain site of policy interest (policy site) by transferring the values that were estimated by a former primary valuation study at some different site (study site). Value transfer may allow CBAs to consider impacts which would otherwise not be included due to limited time and financial resources. However, value transfer comes with the risk of potential transfer errors. Since the circumstances at the policy site may differ from the study site, ESS supply and demand may differ as well. In consequence, the value estimate from the primary valuation study may be less accurate for the policy sites. Researchers try to overcome this problem by adjusting values from the primary valuation studies to the circumstances at the policy site by some spatial variables. Four different forms of value transfer exist which differ in the way and extent they adjust the transferred values to the circumstances at the policy site: (1) unit value transfer, (2) adjusted unit value transfer, (3) value transfer function and (4) meta-analytic function transfer.

(1) Unit value transfer does not undertake any adjustment of the transferred value to the circumstances at the policy site. Thereby, it assumes identical ESS supply and demand at the study and policy site. Thus, a study site should be chosen, that is as similar as possible to the policy site. Alternatively, also the mean value estimate of a number of primary valuation studies are used. (2) Adjusted unit value transfer use a single adjustment to account for different circumstances at the policy site. Typically, the adjustment accounts for differences in the ESS demand only, such as the use of different income and price levels or the number of beneficiaries in the catchment of the sites. (3) Value transfer function uses a function including a number of spatial variables to adjust for the site specific circumstances, which may account for differences in ESS demand and supply. The value function is estimated by a primary valuation study, which uses certain valuation methods, that allows it to incorporate a number of variables in a value function, such as hedonic pricing, travel cost method or choice modelling. Value is transferred to the policy site by plugging in data for the variables in the value function, which describe the circumstances at the policy site. (4) Also meta-analytic function transfer uses a function including a number of spatial variables to adjust for the site specific circumstances at the policy site. However, the function is parameterised based on the results of statistical regression analysis of a number of primary valuation studies. Meta-analytic function transfer does not only account for site specific effects on demand and supply, but also for methodological effects resulting from the different methods used in the primary valuation studies (Brander *et al.* 2010). In the chapters 4, 5, 6 and 7 of this thesis we apply unit value transfer and meta-analytic value transfer for the valuation of recreational services.

Nevertheless, even if value transfer may be less costly and time consuming as compared to primary valuation, it introduces an additional source of uncertainty. Transfer errors may result in less accurate value estimates. A number of studies investigate the different value transfer methodologies and estimate transfer errors. However, no consensus exists on what method is best for a specific purpose. In general, transfer errors tend to be higher, if study sites and policy sites are more heterogeneous. Some authors argue that function transfers may result in lower transfer errors, even though evidence

[2] The methodology was introduced as benefit transfer. However, not only benefits are transferred but also costs and thus, the term value transfer seems more appropriate. Therefore, the term value transfer is used more and more in literature to describe the same methodology as benefit transfer (Navrud and Ready 2007). Both terms are used as a synonym here.

is mixed (Akter and Grafton 2010). However, due to the potential of value function approaches to make adjustments that reflect site-specific characteristics, these methods tend to be superior to (adjusted) unit values transfer, especially in cases where sites differ heavily (Eigenbrod *et al.* 2010b). Therefore, we conclude in chapter 2, that meta-analytic value transfer may be in particular preferable for ESS value mapping, when considering that values are displayed across a larger area, which makes it likely that sites differ heavily.

1.1.4 Mapping of ESS Values

In recent years, a new field of research has emerged within environmental economic valuation, which operates under the term of up-scaling or mapping ESS values. Mapping of ESS values means valuing ESS across a relatively large geographical area and assessing how values vary across space (chapter 2). The spatial variations of the ESS values may result from variations in ESS supply and demand.

The mapping of ESS values offers substantial advantages over traditional single-site valuation studies. By displaying ESS values across a larger area, new primary valuation or benefit transfer studies may not be necessary anymore. Policy makers may simply consult the ESS value map in order to derive a value estimate for the location of interest. Furthermore, displaying how values differ across space helps to quickly identify locations of high ESS values and thus, may help for restoration prioritisation and resource allocation. Payments schemes for ESS of environmental taxes can also be designed more efficiently by adjusting their amount based on location-specific ESS values. ESS maps are also used for land-use policy evaluation and the identification of synergies and trade-offs among different ESS and biodiversity. Furthermore, deriving ESS values from maps is helpful to estimate green accounts of natural capital at different scales. The advantages of ESS value mapping are investigated more in detail in chapter 2.

The need of spatially ESS assessments is explicitly recognised in a number of recent policy documents and initiatives such as the EU Biodiversity Strategy 2020, the Aichi Targets of the convention of biodiversity or the Millennium Ecosystem Assessment. Mainstreaming ESS into policy decisions requires spatially explicit information on ESS supply and values as well as their trends and how ESS values may respond to policy actions. Within national ecosystem assessments conducted in various countries, the mapping of ESS is an upcoming issue (UK NEA 2011a). The TEEB project ESS and their values are mapped at various scales (Costanza *et al.* 2014; de Groot *et al.* 2012). Spatial ESS assessments are conducted within the regional and global assessments of the Intergovernmental Platform on Biodiversity and Ecosystem Services. The European Commission has set up a working group to assist the Member States with the spatial ESS assessments and to develop a common framework of understanding and methods (EC 2011a) and spatial ESS assessments play significant role in the so-called *nature-based solutions* concept under the Horizon 2020 research and innovation program of the EU.

A wide variety of methodologies have been used for mapping ESS values and multiple reviews, frameworks and guidelines on ESS mapping evolved in recent years (Maes *et al.* 2012; Crossman *et al.* 2013; Paracchini *et al.* 2014; Troy and Wilson 2006; Egoh *et al.* 2008; Martínez-Harms and Balvanera 2012, chapter 2). Still no consensus is found on which methods to best used for a specific purpose. Several studies use participatory approaches to map ESS values, but using a non-monetary value metric. Typically, such studies conduct stakeholder or expert workshops, in which participants have assessed ESS values across relatively small case study areas based on local knowledge (Bailey *et al.* 2006; Sherrouse *et al.* 2011; Raymond and Brown 2006; Brown 2006; Tyrväinen *et al.* 2007). However, the thesis at hand focuses on the mapping of monetary ESS values as demonstrated in case studies in chapter 4, 5, 6 and 7. Therefore, we discuss methods for spatial monetary ESS accounts more in detail (see also chapter 2). Most studies mapping monetary ESS values focus either on assessing spatial variation in ESS supply, or spatial variations in ESS demand. Only some studies incorporate both dimensions of spatial ESS value variations. Theoretically, the same methods can be used for the mapping of ESS values that are known from primary ESS valuation studies. However, it is very time consuming to conduct multiple ESS valuation studies across a larger study area. As a result, ESS mapping is typically done by transferring the results of primary valuation studies and ESS supply measurements. However, contrary to traditional value transfer, now the values are not transferred from one or a number of study sites to one single policy site (see Figure 5 a), but to multiple policy sites across a continuous space. Brander *et al.* (2010) describe this process as an up scaling of ESS values (see Figure 5 b). However, the field of ESS value mapping works far more interdisciplinary than the traditional environmental economic value transfer, as the assessment of spatial ESS supply variations plays a more substantial role. Based on the findings of our literature review presented in chapter 2, we would rather describe the process as an inter- and extrapolation of ESS supply and demand observations across space to form a map (see Figure 5 c).

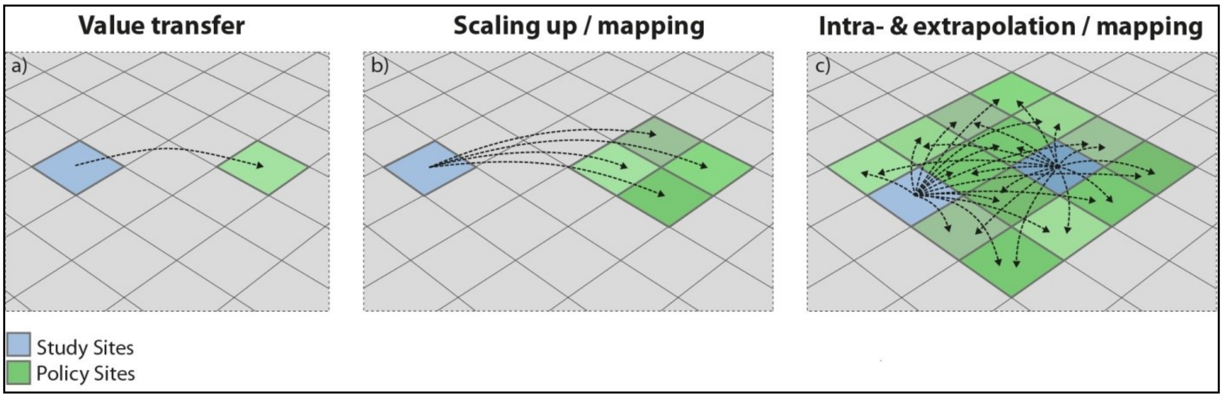

Figure 5: The concept of value transfer and ESS value mapping. (a and b modified from EEA 2010; c own illustration)

In most studies spatial variations in ESS supply are assessed by using one single indicator, such as land cover. Some studies use spatial ESS models that include a number of spatial variables. In cases, where simple unit value transfer is used, spatial variation of ESS demand is not accounted for. More advanced methods are used in recent studies, such as meta-analytic value transfer or value functions. Missing primary data on ESS supply and demand is one major obstacle for developing more accurate and precise spatial ESS value maps. Compiling necessary data bases is a time consuming procedure (de Groot *et al.* 2010). Evaluating the quality of data circulating in literature as well as the requirements for obtaining all the necessary information of data at hand is only one of the issues encountered. In chapter 3, we describe the process and requirements of primary data collection. A detailed review of the value mapping methods applied in literature and an analysis of their specific strengths are given in chapter 2.

For about two decades, the number of studies assessing ESS and their values spatially explicit has grown exponentially (see chapter 2). The chosen approaches, their spatial scales and the kind of used input data differ widely. There is still no consensus on what method is best for mapping ESS and their values given a specific purpose and application. A variety of approaches exist, each of them having its specific strengths and weaknesses and the method to be chosen depends on multiple aspects, such as the data availability, the ESS assessed, temporal and spatial scales as well as the specific purpose of the study. Often, the different disciplines involved are not sufficiently integrated (see also chapter 2). On the one side, economists value ESS and on the other side natural scientists quantify ESS by focusing on biophysical assessments described by the first three steps of the cascade model. Bridging the gap between the two disciplines is in process but there is still a long way ahead (Braat and de Groot 2012;

chapter 2). Natural scientists focus for example on the ESS modelling of carbon capture in peat land (Crow and Wieder 2005; Wilson *et al.* 2006), the assessment of natural landscapes' recreational potential (Paracchini *et al.* 2014), the impact of vegetation on erosion control (de Asis and Omasa 2007; Ludwig *et al.* 2005), the microclimatic effects of urban trees (Shashua-Bar *et al.* 2010) or the role of coastal wetlands for storm protection (Wamsley *et al.* 2010). On the other hand economists focus for example on discounting issues in EES valuation (Baumgärtner *et al.* 2014), on methodological issues within contingent valuation (Meyerhoff and Liebe 2006) or on discrete choice valuation techniques (Meyerhoff *et al.* 2014). In recent years multiple reviews, frameworks and guidelines on ESS mapping beyond the ones presented in chapter 2 have evolved (Maes et al. 2012; Crossman *et al.* 2013; Paracchini *et al.* 2014; Troy and Wilson 2006; Egoh *et al.* 2008; Martínez-Harms and Balvanera 2012; chapter 2).

1.1.5 Mapping Nature Recreation and its Economic Value

As part of this thesis, recreation is chosen as an ESS for exemplifying the methodological findings on the mapping of ESS values presented in chapter 2. Four case studies are presented in the subsequent chapters, which address the spatially explicit assessment of recreational visits, their economic value or both, visits and economic values, across different case study areas (see chapters 3, 4, 5, 6 and 7). In the following subsections, the ecosystem service recreation is introduced and related research is reviewed.

1.1.5.1 The Ecosystem Service Recreation

Recreation is an activity of enjoyment that people do to relax and to refresh their strength, mind and spirits after the demands of work and daily life. Nature recreation represents one subgroup of such activities, which takes place in nature. People enjoy nature by taking a walk, going for a hike, bird-watching, enjoying nature’s beauty or by just listening to the sound of the wind rustling the leaves of trees accompanied by singing birds and by appreciating the fresh air and smell of nature. The opportunity to undertake such activities within nature is considered a cultural ecosystem service that provides direct use-benefits (see also sections 1.1.1 and 1.1.2).

Evidence show that nature recreation plays a substantial role for human well-being (Matsuoka and Kaplan 2008). Epidemiological studies find that nature recreation and access to green space decreases morbidity (Dadvand *et al.* 2012; Maas *et al.* 2006; Nielsen and Hansen 2007; Richardson and Mitchell 2010; van den Berg *et al.* 2010) and mortality (Mitchell and Popham 2008; Richardson *et al.* 2012; Takano *et al.* 2002; Villeneuve *et al.* 2012). Nature recreation supports emotional stress-recovery (Hansmann *et al.* 2007; Tyrväinen *et al.* 2014; Ward Thompson *et al.* 2012) and self-reported emotional well-being (Korpela *et al.* 2014; Krekel *et al.* 2016; Zhang *et al.* 2014).

Within the large body of research on the economic value of ESS, recreation is the ESS that is addressed by the biggest share of primary valuation studies (Schmidt *et al.* 2016; Van der Ploeg and de Groot 2010). By synthesising the findings of this broad body of scientific literature, the annual recreational ESS value has been estimated at about US$ 21 trillion globally, which represents more than 16% of the TEV of global ESS supply (Costanza *et al.* 2014). Protected areas on its own provide recreational values of approximately US$ 250 billion by attracting approximately 8 billion visitors worldwide (Balmford *et al.* 2015a). European national parks attract 2 billion visits a year amounting to a total ESS value of € 14.5 billion (see chapter 4).

1.1.5.2 The Linkage between Nature Recreation and Biodiversity

Due to advancing urbanisation and ecosystem degradation, nature recreation opportunities and biodiversity are becoming increasingly scarce and their management is therefore even more important. Both, biodiversity and nature recreation, are strongly interacting. In many rural areas, land use is heavily affected by nature recreation as well as by nature conservation policies. The quality of recreational experiences within nature depends largely on the characteristics and quality of the visited ecosystem. While recreational visitors may prefer more natural, beautiful and (bio)-diverse landscapes, they may themselves have adverse effects on natural ecosystems such as by disturbing wildlife and by fostering ecosystem degradation due to the development of recreational infrastructure (Pickering and Hill 2007; van der Duim and Caalders 2002). On the contrary, nature recreation and tourism have a substantial economic value (see section 1.1.5.1) and may spur rural economic development by generating employment and income for local communities. If nature recreation supports the livelihoods of local communities it can increase the acceptance of nature conservation within these societies (Nyaupane and Poudel 2011). Therefore, if developed with caution, nature recreation may provide a win-win situation for nature conservation and rural economic development.

Making the recreational value of natural ecosystems explicit can be crucial in making a case for nature and biodiversity protection. Capturing possible synergies between nature recreation and biodiversity protection by establishing multifunctional landscapes may moderate possible trade-offs and reduce pressures on increasingly scarce natural ecosystems. Nowadays, nature recreation plays a significant role in nature conservation policies and protected area management (Gössling 1999; Hall 2010; Stenseke 2012).

However, the direct linkage between biodiversity and recreation is difficult to prove and evidence is mixed. While stated preference techniques indicate that recreational visitors favour biodiverse recreation areas (Biggs *et al.* 2016; Koo *et al.* 2013) and that they have a positive WTP for conserving and restoring biodiversity (Mladenov *et al.* 2007; Wang and Jia 2012), revealed preferences may not always support such findings (Beeco *et al.* 2014). Some studies indicate synergies between specific recreational activities and biodiversity. Siikamäki *et al.* (2015), for example, find that for Finnish national parks, recreational visitors prefer a higher presence of endangered species; Booth *et al.* (2011) and Collins-Kreiner *et al.* (2013) find that, among people who go birdwatching, rarity and number of birds' presence correlate with the number of bird watchers, Ruiz-Frau *et al.* (2013) find that divers prefer biodiverse habitats and Graves et al. (2017) find that forest visitors give broadly higher ranks to species rich wildflower forest communities. Nevertheless, other studies highlight that synergies between nature recreation and biodiversity are not clear-cut (Stenseke 2012). Human recreational preferences may not necessarily relate to high biodiversity values (Anderson *et al.* 2009; Qiu 2014) and recreational services may even have trade-offs with biodiversity conservation (Qiu *et al.* 2013; Vangansbeke *et al.* 2016). However, such trade-offs can be moderated by appropriate land-use planning (Cordingley *et al.* 2016; Larsen *et al.* 2008: 2; Qiu 2014).

1.1.5.3 The Spatial Assessment and Valuation of Nature Recreation

For the evaluation of land-use policies and for the identification of trade-offs and synergies among different ESS, mapping of ESS supply and its values became a widely applied approach in recent years (Maes *et al.* 2012c; chapter 2). Displaying the spatial variations of recreational ESS within a map does not only have communicative and illustrative advantages, but allows also to quickly identify areas of high demand and supply (hotspots), to recognize potential trade-offs and to evaluate the effects of land-use policies. To allocate resources for nature conservation more efficiently, it is important to know how and where recreational co-benefits of nature conservation are high and how they may alter as a consequence of land-use change. Multiple studies aim at supporting such policy decisions by assessing the spatial dimension of recreation supply and its value by using a wide variety of methodologies and focusing on a diverse spectrum of case study areas.

Mapping Visitor Numbers

The most important indicators of the recreational value of an ecosystem is the number of visitors, as it differs far more across space than the value per recreational visit (Bateman *et al.* 2006a; Jones *et al.* 2003; see also chapter 4, 6 and 7). Therefore, understanding the spatial drivers that determine the number of visitors is crucial for the designation and the management of recreational sites. In consequence, several studies focus on the mapping of recreational visitor numbers or closely related indicators.

A few studies use comprehensive data on visitor numbers or an indicator of recreational use across entire study areas to map recreational services. Such analyses exist typically only either at coarse resolution or for small case study areas (Eigenbrod *et al.* 2010b; chapter 2). Villamagna *et al.* (2014) use for example anglers license data in North Carolina, USA and Rees *et al.* (2010) use fishing boat trip data from operators within a bay of the UK. Some visitor monitoring studies map the spatial distribution of recreational use across local case study areas by comprehensive visitor counting (Arnberger 2006; Lehar *et al.* 2004) or GPS tracking (Beeco *et al.* 2014; Wolf *et al.* 2012). They monitor for example, where visitors enter a recreational area, which places they visit and how the visitors distribute across the site.

At regional to national scales, several studies map recreational visitor distribution by conducting regression analysis of survey data on recreational behaviours (Kienast *et al.* 2012). Several studies use general population survey data to estimate the total number of recreational trips. Then, choice models are applied that use either on-site or general population survey data on the origins and destinations of each trip. The choice models allow to distribute the total number of recreational trips across space (Sen *et al.* 2013; Termansen *et al.* 2008; Termansen *et al.* 2013; Vries *et al.* 2007). Some of these studies combine this kind of analysis with monetary value estimates (also see below in this section). However, due to the difficulty in obtaining required survey data for choice models, such applications are less frequent and are only applied at regional to national scale.

In consequence to primary data scarcity, some studies use primary data on related recreational indicators, such as tourist cabins (Van Berkel *et al.* 2014: 201), social media photo uploads (NCP 2015; Wood *et al.* 2013a) or camp sites (Weyland and Laterra 2014), which they extrapolate across space by combining statistical modelling with GIS data. Thereby, they map these recreational indicators across lager areas.

Mapping Recreational Supply

Several studies do not focus on visitor numbers or related indicators, but map the recreational supply at local to continental scale by combining various spatial data sets. However, in most of these studies, the term "*recreational supply*" is not clearly defined, but it is mainly considered to determine the recreational quality and/or opportunities of an area. Parameterisation of the models is either based on the researchers' judgements (also called "*non-validated model*"; see chapter 2; Koppen *et al.* 2014; Paracchini *et al.* 2014) or based on survey results that elicit the preferences for the different explanatory variables from probands such as citizens, stake holders or experts (also called "*validated model*"; see chapter 2; Plieninger *et al.* 2013; Sherrouse *et al.* 2011).

Paracchini *et al.* (2014) map for example the recreational potential of the landscape across Europe using a non-validated model. They use the best available knowledge to define the contribution of various geographic information system (GIS) data layers on recreational potential. Surveys to parameterise models are applied for example by (Peña *et al.* 2015), who map recreational supply and demand for a local case study area in Northern Spain. Supply is assessed by recreational potential and accessibility similar to Paracchini *et al.* (2014), but the recreational demand model is parameterised based on photo survey results. Similar approaches are applied to map the recreational opportunity spectrum (known as ROS) to highlight the recreational opportunities at different locations. Different GIS layers are combined based on the researchers judgements in order to classify different recreational opportunities (Casado-Arzuaga *et al.* 2013; Harshaw and Sheppard 2013; Joyce and Sutton 2009; Paracchini *et al.* 2014). The AIRES[3] model on recreation uses only one variable taken from a viewshed analysis for mapping recreational services (AIRES 2016).

Some survey based mapping techniques also employ statistical regression analysis in order to quantify the contribution of different landscape features to recreational supply. Within workshops or surveys, images or descriptions are presented to probands who are then asked to state their preferences for the different landscape features. The results of the analysis are then translated into GIS layers (Plieninger *et al.* 2013; Sherrouse *et al.* 2011). Nahuelhual *et al.* (2013) map for example recreation potential and opportunities in southern Chile. In this study, participants judged the importance of different spatial variables within two steps and a discussion inserted in between. Kliskey (2000) maps recreational suitability of a mountain range in the USA based on principal component analysis of results of a survey on the importance of different GIS features. Van Berkel and Verburg (2014) map recreational ESS based on a photo survey on the preferences for landscape features and structures. Caspersen and Olafsson (2010) synthesise the findings of past surveys on recreational preferences of the Danish population and translate these preferences into GIS data to map recreational opportunities.

Participatory Mapping of Recreational ESS

Other studies employ participatory approaches to map recreational ESS. Participatory mapping approaches elicit the preferences, perception and knowledge on the spatial distribution and relative importance of different ESS from stakeholders such as the general public, experts or decision makers. However, the term "*participatory mapping*" of ESS is not clearly defined in literature. While some authors define participatory mapping as the employment of surveys regarding the importance of different landscape features for the considered ESS (as described above), most commonly participatory mapping is described as asking probands to actually map ESS by indicating locations of ESS supply and/or values on a map. Raymond *et al.* (2009) ask for example 56 decision-makers to stick dots on a map indicating positive or negative values for certain ESS. De Valck *et al.* (2016) map recreation hot- and cold-spots by using an internet survey in which people have to locate past recreational visits in a digital maps. Pert *et al.* (2015) use a participatory approach consisting of a series of workshops and interviews to map the perceptions of a local community concerning cultural ecosystems in northeast Australia. Such participatory mapping approaches are typically applied to relatively small case study areas only.

Mapping Monetary Recreational ESS Values

Another body of literature maps recreational service values in monetary terms. One of the early studies that mapped global monetary recreational ESS values is the well-known Costanza *et al.* (1997) paper. Global mean value coefficients for each ESS are assigned to different land cover classes. This so-called Costanza approach may give a first indication of the magnitude of the ESS value, but presents little

[3] AIRES refers to ARtificial Intelligence for Ecosystem Services (http://aries.integratedmodelling.org/).

support for local policy decisions (Plummer 2009; chapter 2). Still, the approach has been replicated multiple times for local (Petrosillo *et al.* 2009; Petrosillo *et al.* 2010), regional/national (Liu *et al.* 2010; Troy and Wilson 2006) and global case study areas (Costanza *et al.* 2014; de Groot *et al.* 2012). Other studies use models in order to quantify the supply of recreational services and combine the model results with a monetary unit value transfer (Eade and Moran 1996; O'Farrell *et al.* 2011) or an adjusted unit value transfer (Baerenklau *et al.* 2010a). The models may either be calibrated based on the assumptions of the researchers and best available knowledge (so-called non-validated models) (Chan *et al.* 2011) or they are calibrated based on regression analysis of real world observations of recreational visitors (Bateman *et al.* 1995). Some studies use a value function that is typically estimated by a travel cost analysis and combine it with either validated (Brainard 1999; Moons *et al.* 2008; Termansen *et al.* 2008) or non-validated models (Baerenklau *et al.* 2010b) on spatial variations in visitor numbers. A general review and more details on studies mapping ESS values (including recreation) can be found in chapter 2.

Mapping of Recreational ESS Values in this Study

In the light of the EU Biodiversity Strategy 2020, the aim of this thesis is to contribute to the general understanding and methodological development of mapping ESS and their values on a large scale. Therefore, we analyse past ESS value mapping studies within the broad literature review that is presented in chapter 2. We identify a best-practice approach for mapping ESS values, which maps ESS values by two separate models, one biophysical model on ESS supply (in the case of recreational services the number of recreational visits) and one economic model on the value per unit of ESS supply (in the case of recreational services the value per recreational visit). Both models are to be calibrated by regression analysis of primary data. These methodological findings are exemplified by mapping recreational services and its values at continental scale in several case studies, which are presented in the chapters 3, 4, 5, 6 and 7.

However, contrary to other ESS, the supply of cultural ESS, such as recreation, is intangible and cannot be clearly distinguished from its demand, because the human dimensions of preferences and demand are inherent. Recreational supply cannot be defined as it is done for other ESS such as tons of carbons captured, flood risk reduced or m^3 of timber yields. We therefore, define recreational supply by all biophysical processes involved, which are expressed by the number of recreational visits. Visitor distribution is modelled as is done within species distribution modelling in ecology. The economic dimension is then expressed by the monetary value per recreational visit.

So far no study has mapped recreational services for a case study area larger than at national scale, with the exception of Paracchini *et al.* (2014) who use a non-validated model to map recreational potential by combining several GIS layers based on the researchers assumptions. One major difficulty in mapping ESS at an international scale by our proposed best-practice approach is the lack of consistent international primary data sets on ESS supply and its value. Therefore, we collect such data European-wide and construct an online data-sharing tool to share our database and future recreational ESS studies (see chapter 3). The online data-sharing tool may present a valuable tool for future research and the presented case studies in the chapters 3, 4, 5, 6 and 7 may shape future directions in recreational ESS value mapping.

1.2 Thesis Objective and Research Questions

1.2.1 Research Questions

This thesis is elaborated in the light of the EU Biodiversity Strategy 2020 as part of the PRESS project (PEER [Partnership for European Environmental Research] Research on EcoSystem Services) and the MAES working group (Mapping and Assessment of Ecosystems and their Services). Within this context, the main objective of this thesis is to contribute to the understanding of the spatial variations of ESS constituents to human well-being. Thereby, it aims at contributing to the development and implementation of the Biodiversity Strategy, which requires the EU member states to map and value their ESS spatially explicit. ESS and their values emanate from the interaction of natural, human, social, and built capital. Much of this interaction is governed by spatial relationships between these four kinds of capital. Capturing and modelling the nature of these interactions is one of the major challenges in the spatially explicit valuation of ESS. The thesis at hand aims at the methodological advancement in modelling these interactions in general as well as for the specific case of recreational services. Attempting to capture and model the processes that produce ESS and determine their values as a result of these interactions, mandates the acquisition of spatially referenced data for all four kinds of capital (natural, human, social and built). This raises several questions associated with suitable indicators for different ESS and their values, the spatial and temporal resolution of in- and output data as well as their associated uncertainties, the functional form of the relationships between different indicators and the parameterisation of models defining ESS supply and its values. Against this background and the overall objective of this thesis, the following research questions are deduced:

1. What makes the space-related perspective on ESS and on their values to be of particular interest and what advantages arise from spatially explicit ESS assessment as compared to traditional ESS valuation?
2. What methods are applied or may be applied for spatial assessments of ESS values and what is the state of the art in ESS value mapping?
3. What data is required to best map ESS and their values? What data gaps exist and how should available data be presented and organized?
4. What are the advantages and disadvantages of the different methods for mapping ESS values?
5. Which method appears to be the best for mapping values of a specific ESS under consideration of specific circumstances and study purposes? How to define a best-practice approach?
6. How to exemplify a best-practice approach for mapping ESS values through a case study?
7. How to map recreational services and its values and what are their main drivers?
8. How to integrate ESS maps into environmental policy and how to evaluate policy scenarios with ESS maps?
9. How can spatial ESS maps contribute to biodiversity protection?
10. What are future research prospects in mapping of ESS and their values?

1.2.2 Thesis Outline

This thesis consists of eight chapters, which all contribute to answering the above raised research questions. Chapter 2 to 7 represent each peer-reviewed articles published in a journal, a book (chapter 5) or manuscripts of an article submitted to be published. Chapter 7 presents a conference paper. The author of this thesis is the first author and main contributor of articles presented in chapter 2, 3, 4, 6 and 7. Statements by the other authors exposing the authors' contribution to the thesis' individual articles are added in the appendix. In the following, a short executive summary of the subsequent articles is given, highlighting the research questions addressed in the individual chapters. In the final chapter, chapter 8, the individual articles are brought together and the overall methodology connecting the different articles is discussed in detail. The thesis contribution to solving the above raised research questions are highlighted in its entirety.

Chapter 2: Mapping Ecosystem Services' Values: Current Practice and Future Prospects

Within Chapter 2, we analyse and review the literature on the mapping of ESS values. First we identify the purpose, policy applications and informative advantages of the studies as compared to traditional environmental valuation studies. We analyse different methodologies used to assess the biophysical and economic dimension of ESS mapping. Based on our findings we develop a methodology matrix classification scheme, which allows us to classify every study that has mapped ESS values by locating it in this matrix. Thereby, we deliver quantitative findings on the use and developments within ESS value mapping methodologies. We identify strengths and weaknesses in the applied approaches as well as data requirements and needs for ESS value. Finally, we give guidance for future ESS value mapping exercises. A best-practice approach is recommended.

Chapter 3: Monitoring Recreation Across European Nature Areas: A Geo-database of Visitor Counts, a Review of Literature and a Call for a Visitor Counting Reporting Standard

Lack of real world observations on ESS supply is one major challenge in ESS modelling. In this chapter, we present a database of geo-referenced annual recreational visitor estimates to non-urban ecosystems across Europe. The number of recreational visitors represents the recreational use of a certain site and is the most important proxy to define its recreational economic value. For advanced spatial ESS modelling, primary data is of great importance for model parameterisation and validation. This data base fills an important gap in data availability. At the same time, we review visitor monitoring activities all across Europe and give insights into the activities in different European countries. We identify shortcomings in data availability, data reporting and data sharing practice within the European countries. Based on our findings we propose reporting standards and data sharing for visitor monitoring studies in order to facilitate future secondary research.

Chapter 4: Mapping Recreational Visits and Values of European national parks by Combining Statistical Modelling and Unit Value Transfer

Recreation was identified as one major ecosystem service and an important co-benefit of nature conservation. In this chapter, we map recreational use and its economic value across all national parks in most European countries. Therefore, we develop geostatistical models on recreational visitor numbers based on a set of continuous spatial predictor variables. As a result of our statistical regression analysis, we identify spatial drivers of recreational use across Europe. We use the model to predict recreational visitor numbers across national parks in most European countries and combine them with mean value estimates of meta-analysis on recreational valuation studies. We also demonstrate the use of our model for policy evaluation by investigating the effect of a change in a national park area on visitor numbers and their values all across Europe.

Chapter 5: GIS-Based Mapping of Ecosystem Services: The Case of Coral Reefs.

This chapter combines a geostatistical visitor arrival function with a meta-analytic value transfer function to map the recreational value of coral reefs in Southeast Asia. By combining these two spatial models, it is not only accounted for the spatial variations in recreational visitor numbers, but also for spatial variations within the economic value per recreational visit. Thereby, the approach follows the methodological recommendations presented in chapter 2. The approach is used to evaluate the effect of expected future coral reef loss on the value of recreation.

Chapter 6: **Spatial Dimensions of Recreational Ecosystem Service Values: A Review of Meta-Analyses and a Combination of Meta-Analytic Value-Transfer and GIS**

In chapter 6, a similar approach is chosen, but it is focused on nature recreation across all types of non-urban ecosystems in Europe. More detailed and finer resolution GIS predictor variables are applied to gain additional insights into the spatial dimension of recreational ESS values. Therefore, a meta-analytic value transfer function is estimated by regression analysis on European primary valuation studies. It is applied to map the value per recreational visit across Europe. The results are critically evaluated against results of past meta-analysis of recreational valuation studies. Uncertainties involved with the geostatistical analysis are discussed. The model is combined with the visitor predictions presented in chapter 4 to map total recreational values per hectare for a potential national park anywhere across Europe. The results are compared to an alternative ESS value mapping approach in order to illustrate the effect of different methodologies.

Chapter 7: Mapping the Recreational Value of Non-Urban Ecosystems across Europe: Combining Meta-Analysis and GIS

In this chapter, we model recreational visitor numbers across all types of non-urban ecosystems throughout Europe, again based on statistical regression analysis. By combining the model with the meta-analytic value transfer function from chapter 6, we map recreational ecosystem service values by making spatially explicit predictions of the number of visits and the value per visit (VV) throughout all of rural Europe. Thereby, we are able to estimate the recreational value of any location across Europe's countryside at any given scale. In total we estimate 11 billion recreational visits a year which amounts to a value of € 57 billion for Europe's countryside. In addition, we deliver quantitative findings on the importance of spatial variations in recreational visitor numbers and the VV for determining overall variations in the recreational value per ha across space. This may shape future research priorities in recreational ESS valuation.

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2 Mapping Ecosystem Services' Values: Current Practice and Future Prospects

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Keywords:

Ecosystem service assessment
Ecosystem service mapping
Ecosystem service valuation
Ecosystem service modelling
Value transfer
Land use policy assessment

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 the 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.

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2.1 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. In particular, the mapping of monetary values for ESS value has become an active research topic in recent years (Troy and Wilson 2006; Maes *et al.* 2011a). In this paper we review studies that map monetary 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.[4] 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 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 6). The methodologies applied in these studies differ widely, in particular with respect to how spatial variation in ESS values is 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 monetary ESS values. Articles were obtained by searching the Scopus, 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 map ESS. We excluded all studies from the review that map only ESS supply (54) and that map non-monetary ESS values (20), because non-monetary valuation follows a different theoretical framework and applies a different set of valuation techniques. We analysed the remaining 69 articles and reviewed them according to the methodologies

[4] 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.

used for ESS quantification and valuation, the ESS assessed, study rationale and case study area characteristics. The purpose of this review is to identify current practices and developments in the mapping of monetary EES values with a view to providing recommendations for future applications and research.

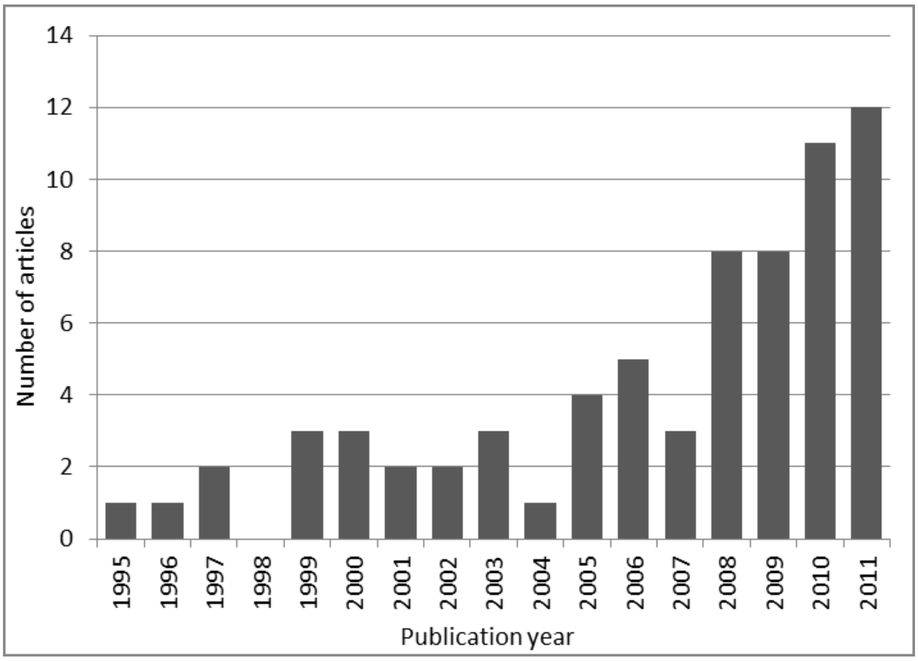

Figure 6: Published articles per year.

The paper is organised as follows: in Section 2 we give an overview of the rationale 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.

2.2 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 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 in 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 has emerged and become 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 spatial 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 7 presents the frequency with which specific policy applications are mentioned as the potential end-use of value data in the ESS mapping literature.

(1) Green accounting: mapping of ESS values allows for estimating a green Gross Domestic Product (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 not only supports decisions on whether or not to conduct a policy measure, it also indicates where to conduct a policy measure. It allows the identification of locations in order to minimise negative or maximise positive ecological side effects. For example, by identifying ESS hot spots for conservation it allows the assessment of ‘‘*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 to allow for more efficient incentives across providers of ESS.

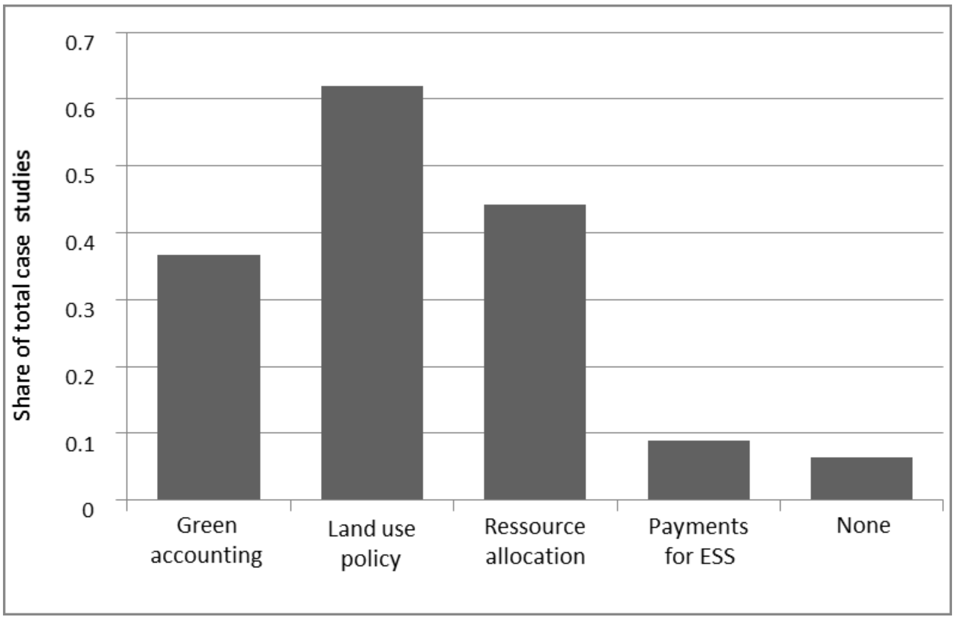

Figure 7: Citation of policy applications in ESS mapping literature.

2.3 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 ecosystems 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 8 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.

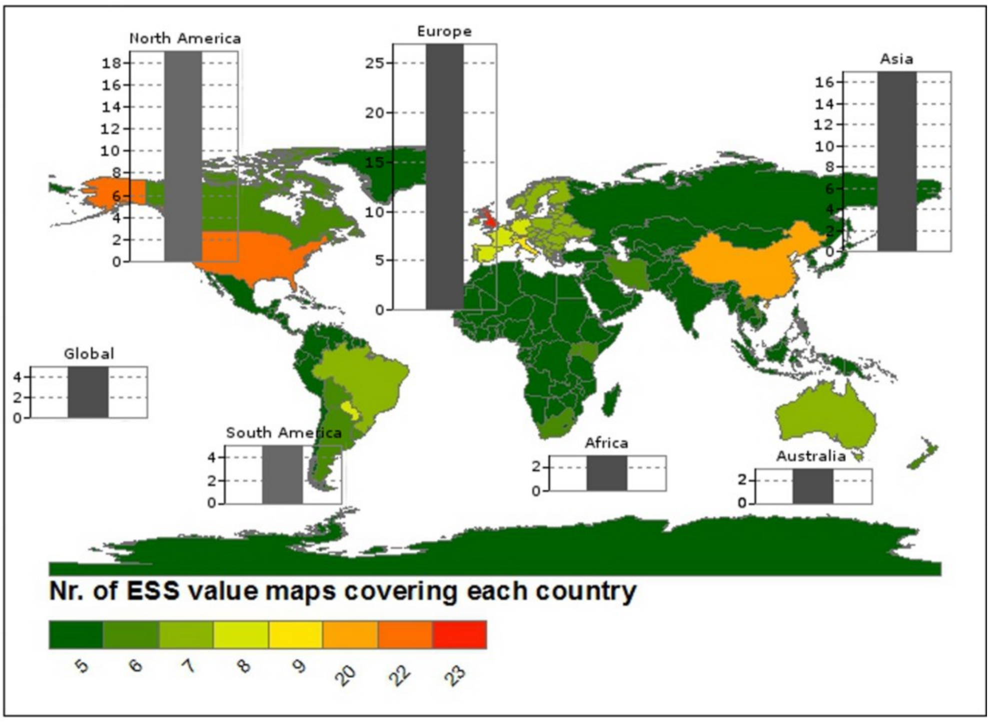

Figure 8: Spatial distribution of case study areas.

Study areas differ in size, ranging from global to local assessments (see Figure 9), with the smallest case study area comprising a 550 ha 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 1000 km². Typically, they focus on a single protected area, a single forest or an urban area. Approximately 23% focus on case study areas between 1000 km² and 10,000 km². Most of them are defined by the borders of an administrative region. Study site areas from 10,000 km² up to 100,000 km² comprise 24% of all studies. They contain mainly regional to national assessments. Approximately another 24% of all study areas are continental, supra national or global ESS value assessments with study areas above 100,000 km².

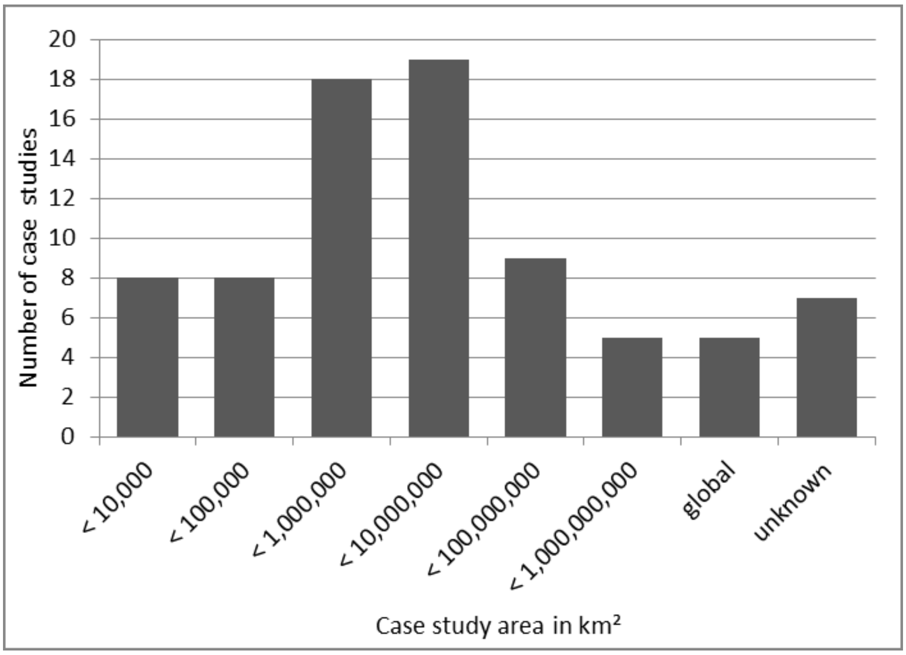

Figure 9: Study site area size.

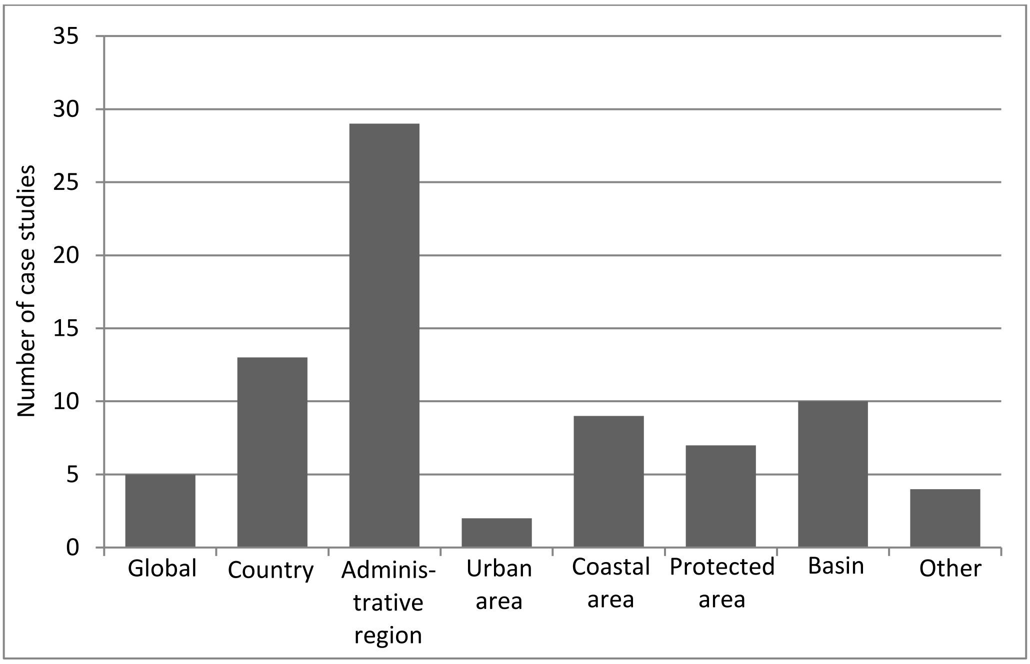

Figure 10: Types of study areas.

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 estuary (see Figure 10).

Most studies focus on more than four (multiple) land cover or land use classes (LCLU) (see Figure 11), 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 *et al.* 1999a).

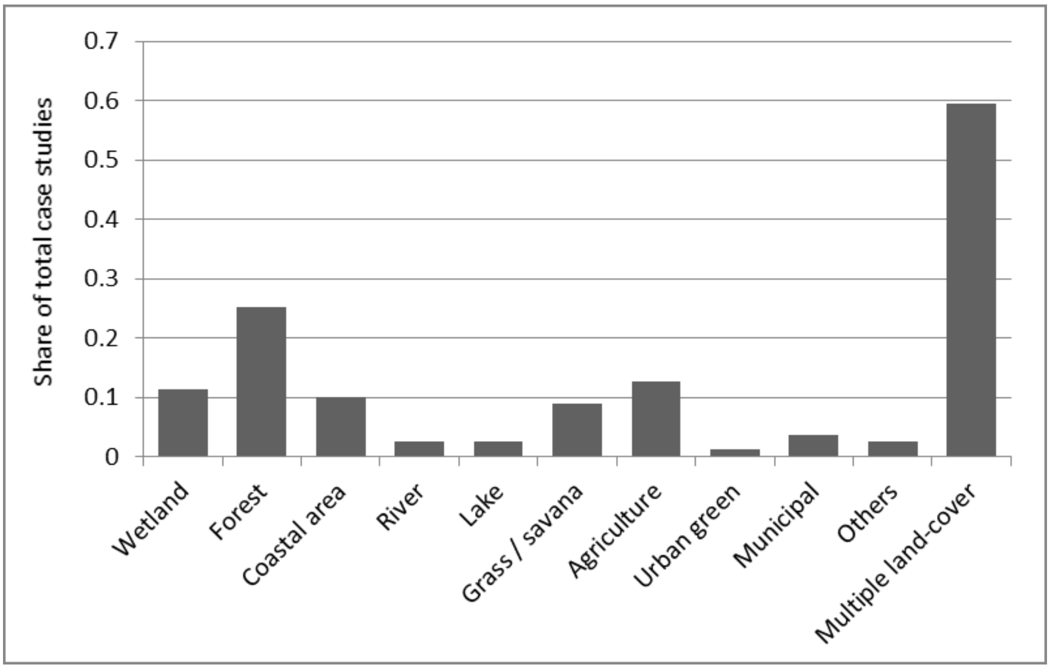

Figure 11: Ecosystems assessed.

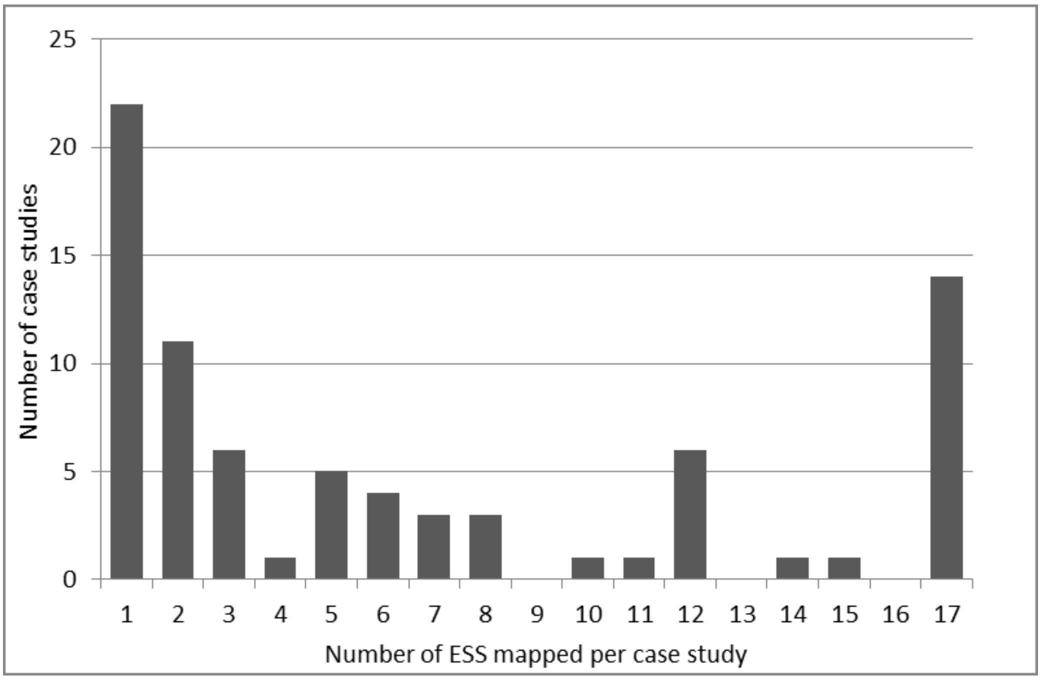

Figure 12: Number of ESS mapped per case study.

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 12).

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 13.

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 m to 10,000 m resolution (see Figure 14).

2.4 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 or by a combination of both dimensions.

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

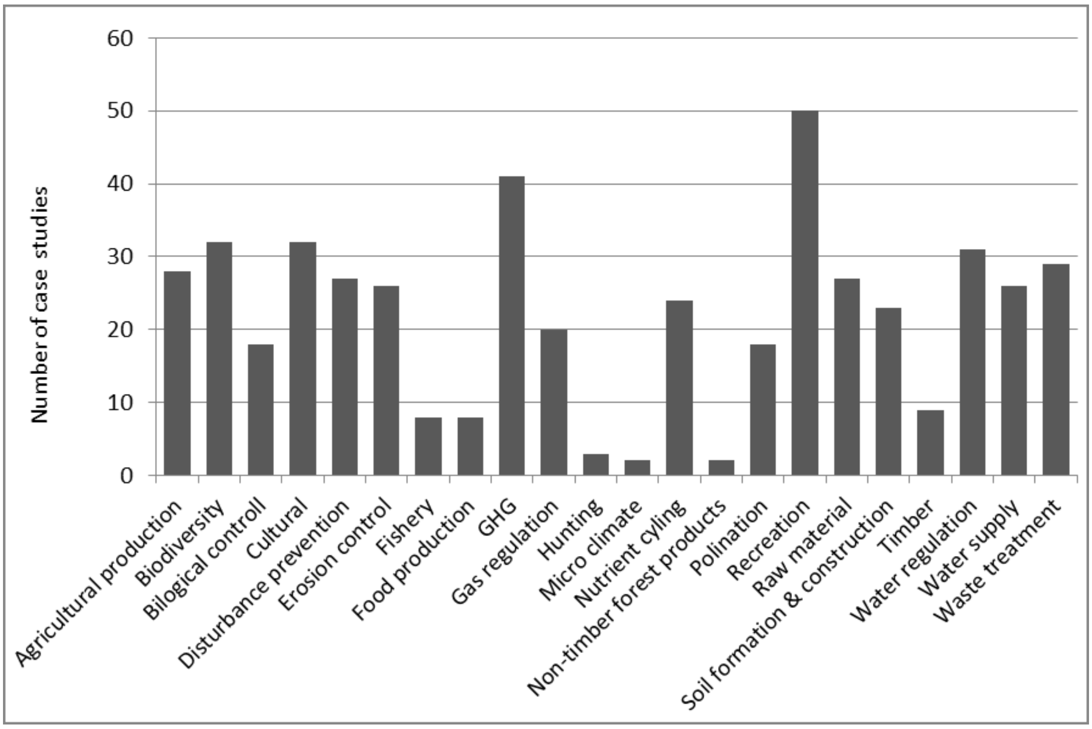

Figure 13: Frequency with which each ESS is mapped.

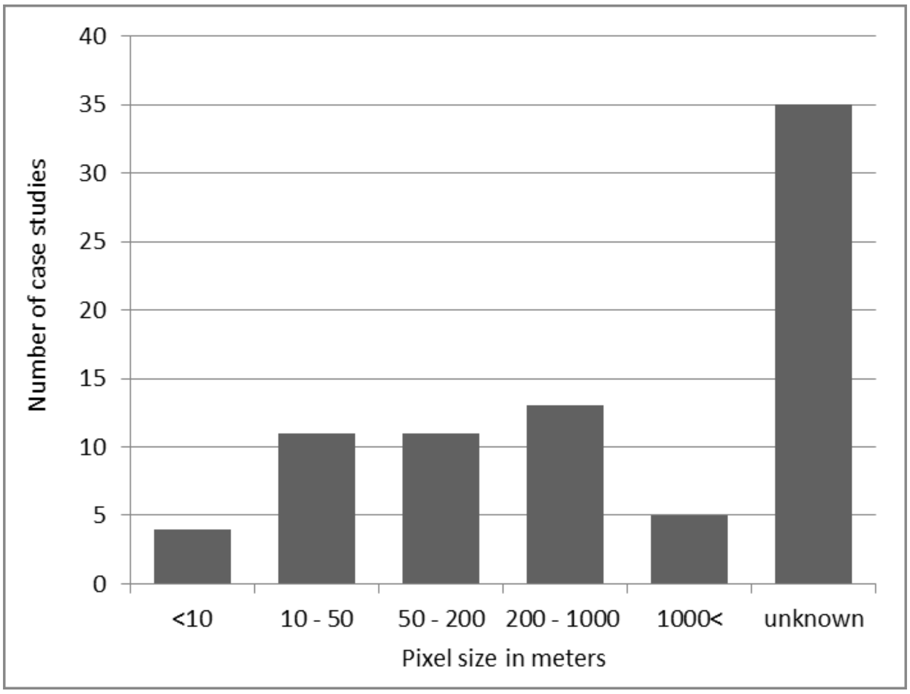

Figure 14: Resolution of ESS value maps.

2.4.1 Mapping of Ecosystem Service Supply

Methodologies used for mapping ESS supply can be divided into five main categories: (1) One-dimensional proxies for ESS, such as LCLU, (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 data of the study area: data on ESS supply that is collected for the specific study area, and (5) implicit modelling of ESS supply within a monetary value transfer function: the quantity of ESS supply is modelled within the valuation of the ESS. Figure 15 shows the share of studies using each of these methodologies for assessing ESS supply.

(1) Most common are ESS maps that are based on one easily available proxy. Such ESS maps use one biophysical variable to map variations of ESS supply across space, mainly LCLU data, but also others such as water depth or slope angle are used. Approximately 52% of all studies map ESS based on proxies.
ESS models (also called ecological production functions) have also been widely used for mapping ESS. Such models assess the supply of ESS based on a set of spatial explanatory variables.
(2) In the absence of any primary data on ESS supply for model calibration and validation, researchers tend to build non-validated models for mapping ESS supply (23% of all studies). These models are based on likely causal combinations of explanatory variables, but the causal combinations are grounded on researchers' or experts' assumptions or on information taken from the literature. No real world observations on ESS supply are used to calibrate the model or to test the model's validity.
(3) In contrast, validated models use primary or secondary data on ESS supply in order to calibrate the model parameters, for example by statistical regression analysis or by manual model optimisation. This approach is used by 34% of all studies. It is worth noting, however, that the distinctions between models that are calibrated based on primary or secondary data

(validated models) and those that are based on researchers' assumptions (non-validated models) are not clear cut. Almost every complex ESS model relies to some extent on researchers' assumptions. Moreover, in the absence of data on ESS supply for the study area, some studies use data for calibration, which were obtained for a different spatial context and for different purposes.

(4) ESS maps that are based on representative data use a minimum of one real world observation to quantify ESS supply within each patch of the ESS supply map. The application of this approach is limited and has been used mainly either for small study areas or at coarse resolutions (Eigenbrod *et al.* 2010b). Approximately 13% of all studies map at least one ESS based on representative data.

(5) A relatively small number of studies – typically with a strong environmental economics background – use implicit modelling to map ESS supply. Approximately 9% of the reviewed studies use this approach. Such studies use value functions that relate variation in unit ESS values to variation in the characteristics of the ecosystem, context and population of beneficiaries. Site-specific parameter values are plugged into the value function in order to derive a value estimate at every location of the study area. In applications in which the value function contains several biophysical variables that have a causal relationship with ESS supply, the model can be interpreted as providing an implicit modelling of ESS supply, although the ESS supply is not derived explicitly.

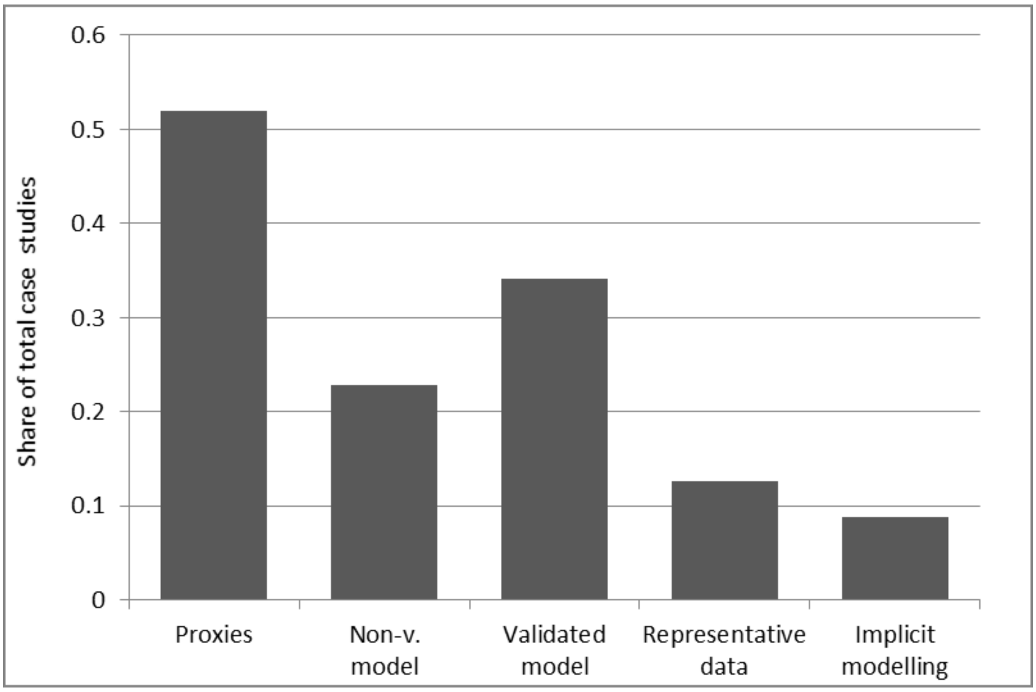

Figure 15: Share of studies using a specific methodology for mapping ESS supply.

2.4.2 Mapping of Ecosystem Services' Values

Mapping of ESS values requires that monetary values are assigned to mapped ESS provision. This can either be done by conducting a new primary valuation study for the case study area or by transferring values from existing studies for other similar study areas (known as value or benefit transfer). Primary valuation involves estimating the monetary value of the ESS supply of the case study area through the application of one or more market or non-market valuation methods.[5] Value transfer involves transferring values from one or multiple study sites, for which the ESS has been valued, to the current study site (often termed policy site). Typically, the reason for performing value transfer is to obtain information on ESS values without conducting time consuming and costly primary valuation studies. In total 42% of the reviewed studies conduct primary valuation, whereas 84% use value transfer for at least one ESS. In order to map variation in ESS values, value estimates are then distributed across the study area using the methods described below.

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.[6]

(1) In the unit value approach, a constant value per unit of ESS 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 per unit of ESS across the study area using simple variables in order to account for spatial variations in value. Typically, such variables are population density, income levels or consumer price index. Thereby, such adjustments account for the number of beneficiaries of an ESS, the effect of income levels on willingness to pay, and differences in price levels. About 5% of all studies use adjusted unit values for ESS value mapping.
(3) Value functions are used to 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 use value functions.
(4) The meta-analytic value function transfer approach also transfers values to the entire study area by plugging in site-specific characteristics into a value function. In this case, however, the function is estimated through statistical regression analysis of the results of a number of primary valuation studies. Only 4% of the reviewed case studies use this methodology (see Figure 16).

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[5] Detailed information on the underlying theory and practical implementation of non-market valuation techniques can be found in a number of texts including Hanley and Spash (1993), Pearce et al. (1994) and Freeman (2003).

[6] For a general overview on the different value transfer methodologies see for example Navrud and Ready (2007) and Navrud and Bergland (2001).

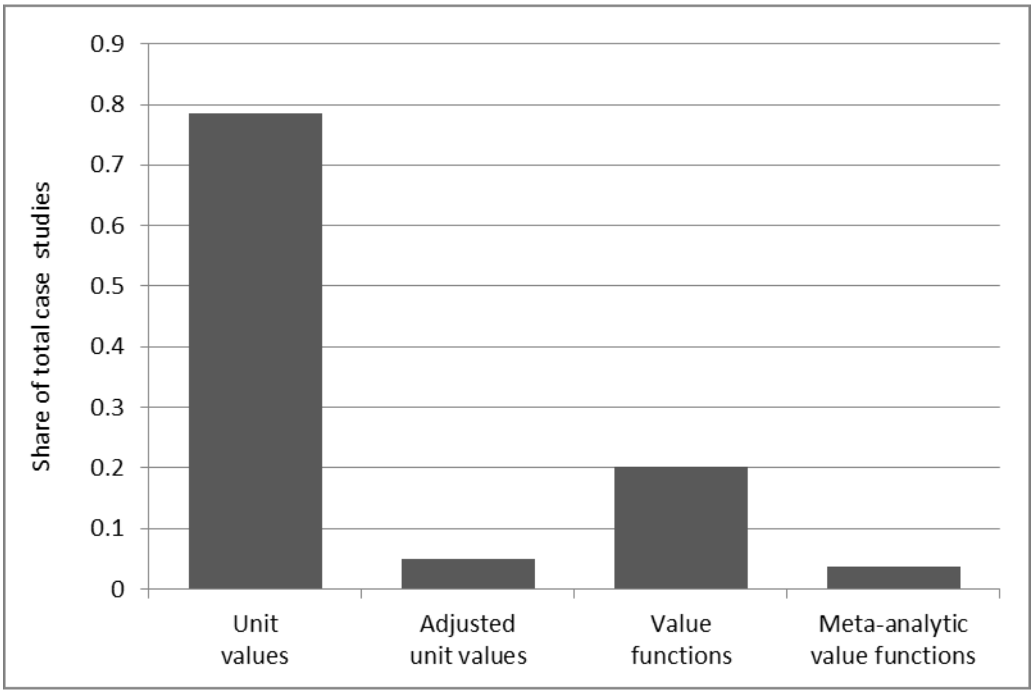

Figure 16: Share of studies using a specific methodology for valuing ESS.

2.4.3 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 according to the methodologies used for mapping ESS supply values (see Table 1)[7]. In each cell of the matrix, we include abbreviations for the different ESS. Each abbreviation is followed by numbers, which refer to studies that map the specific ESS using the combination of methodologies indicated for that cell. The abbreviations and studies are listed in the lower part of the Table. Readers that are interested in a particular methodology or a particular ESS can find the references of the relevant studies listed in the Table.

[7] The classification of some studies was difficult (mainly the differentiation between validated and non-validated models) in cases for which not all relevant information is available in the published article. In such cases, we searched for further information within the mentioned references.

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

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:** Biodiversity, **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 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)

Almost half of the reviewed studies combine LCLU proxies with unit values (46%).[8] 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. The only biophysical variable used to describe differences in ESS supply across space is LCLU (proxy). The ESS values per unit of ESS do not differ across space (unit value). This approach has been replicated 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 a specific ESS, which they investigate more in depth. Typically, such studies focus on one or a small number of ESS using more detailed methods. Additional ESS values are then included by the rather simple combination of LCLU and unit values in order to provide a more comprehensive assessment of ESS values.

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 of each location in the study area. Thus, the value per unit of water retention does not differ across space (unit value). However, water flow regulation differs across space based on a model using vegetation, soil and slope angle as spatial explanatory variables. The model is calibrated based on ''*in-situ surveys and field experiments*'' (validated model). 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 (validated model). Carbon is valued using one uniform value per ton sequestered carbon (unit value). Simonit and Perrings (2011) model the impact of wetlands on the water quality in Lake Victoria. Data for model calibration is not taken from the study area itself, but from ''*closely allied systems*'' (validated model). A uniform value is estimated per unit of nutrient retention based on an estimated impact on fish catch in the downstream lake (unit value).

The combinations of non-validated models with unit values (19%) and representative data with unit values (10%) are also used relatively frequently. Eade and Moran (1996) map recreational values based on the assumptions that the recreation service is distributed across the study area based on ''*distance and visibility from tourist areas*''. However, no reference is given on whether this relationship is based on any real world observation (non-validated model). The total recreational value estimate for the entire study area is then distributed in accordance to the mapped ESS distribution (unit value). Crossman *et al.* (2010) map agricultural production values based on yield statistics for the study area (representative data). An ESS value is attached to the yield by combining it with constant farmer net returns for each LCLU type (unit value). O'Higgins *et al.* (2010) map values for recreational clamming in a 1800 ha bay in Oregon, USA. Recreational use is quantified in a spatially explicit manner based on a comprehensive survey of the study area (representative data). A constant willingness to pay (WTP) value is attributed to each recreational user (unit value).

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 in Willamette Basin in Oregon, USA. The models on agricultural yields

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[8] Note that a number of studies use different methodologies for mapping values of different ESS.

and timber production use land use, climate and soil data as explanatory variables. The models are calibrated based on yield data (validated model). The net revenues of each land use are modelled spatially explicitly using a function that includes spatial variables, such as parcel location, slope and land use (value function). 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 *et al.* 1999a; b). The recreational demand models use visitor survey data for model calibration. The value per visit (VV) is then modelled using a travel cost function, which results in different values per visit for different locations in the map (value function). Grêt-Regamey *et al.* (2008) model the impact of forest cover on avalanche protection based on avalanche probability, slope and land cover data. The model is calibrated based on avalanche records (validated model). Values are a function of avalanche risk reduction and property and human lives at risk (value function).

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. Thus, the ESS storm protection is not explicitly modelled within an ESS model but still the ESS supply is quantified implicitly based on the biophysical variables within the value function. Powe *et al.* (1997) use a value function based on a hedonic pricing model for mapping recreational and amenity values of forests. The model, however, does also include forest characteristics in form of an access index, which correlate with recreational use. Thus, the value function quantifies the ESS implicitly.

About 4% of the reviewed studies 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. However, this relationship is based on the researchers' assumptions and not on real world observations (non-validated model). Values are a function of visitor numbers, visibility and travel costs estimated for each access point (value function).

Meta-analytic value functions are still relatively rarely used within ESS value mapping, although 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) mapped 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, which we classify in our matrix as a proxy. The meta-analytic value function used by Ghermandi *et al.* (2011) to map global coastal recreational values includes multiple biophysical variables that correlate with recreational use (e.g. climate, biodiversity and accessibility). Therefore, an implicit modelling of the ESS recreation within the meta-analytic value function is conducted.

Proxies in combination with value functions are used by 3% of all studies. Guo *et al.* (2001) mapped recreational values by using a travel cost model for valuation (value function). However, the only biophysical feature affecting spatial value distribution is LCLU (proxy).

Only one study uses non-validated models in combination with adjusted unit value transfer. McPherson *et al.* (2011) mapped 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 (non-validated model). A value per large tree is taken from one hedonic pricing study. The value per tree is then adjusted by one variable, the number of beneficiaries in terms of residential housing density. Thus, we classified this approach as an adjusted unit value.

We identified some correlations between the methodology used and other study characteristics. However, due to the limited number of studies for some methodological combinations, it is difficult to conclude an overall trend. Typically, studies that use a combination of proxies and unit values map values of multiple ESS (mean 10), whereas more complex methodologies result in fewer ESS being addressed (an average 1–2 ESS per study).[9] Studies that attempt to cover all ESS values are commonly mapped using the combination of proxies and unit values.

We can only identify a few concentrations of certain combinations of methods 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 functions (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 exist (5), mainly in combination with value functions.

There are also some patterns with respect to the policy application that is addressed by a study and the methodology used. Green accounting is dominantly mentioned within studies using unit values, either in combination with proxies, non-validated models or representative data. Resource allocation and land use policy evaluation are mentioned frequently within studies using unit values or value functions.

The spatial extent of the study area tends to be smaller 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 published after 2007 use proxies or non-validated models combined with unit values or adjusted unit values. For the sum of all other combinations of methods, this share amounts to 75%.

2.4.4 Accuracy and Precision in ESS Values Mapping

An important and insufficiently assessed issue in mapping ESS values is the accuracy and 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 the 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 studies do not address the question of accuracy and precision of their mapped values at all. About 58% of all studies at least discuss potential value mapping errors qualitatively. However, only a minor share of the reviewed studies give quantitative information on error margins of their results either by displaying parameter estimates

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

from the statistical analysis, by estimating boundaries within which the actual values may most likely lie, by conducting sensitivity analyses or by comparing predictions with real world observations (see Figure 17). 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 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) regionalisation or extrapolation errors (Eigenbrod *et al.* 2010a; b).

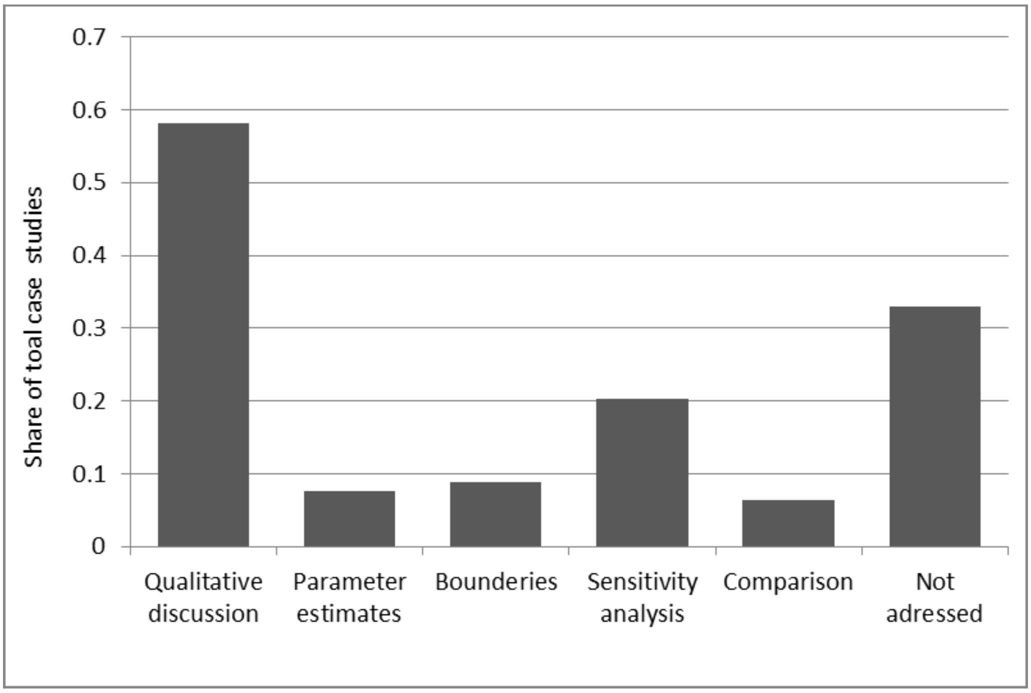

Figure 17: Assessment of results accuracy.

(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 vary due to multiple factors that 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 include the 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) Regionalisation or extrapolation errors may occur when values are transferred between different areas that are characterised 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.* 2010a; b; 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 data sets 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 a meta-analytic value function 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 that 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. Costanza *et al.* (2008) conducted a sensitivity analysis by setting a maximum marginal value for a wetland. As a result, the total value estimate differs by a factor of almost seven. 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.* (2010a; b) estimate errors associated with ESS mapping using land cover proxies. They make a comparison between ESS maps that assume a constant ESS supply per LULC class and maps that are based on real world observations of ESS supply. The correlations (Spearman's rho) between the predicted and observed provision of ESS are low (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, they find that including additional explanatory variables for population and accessibility increased the correlation between predicted and observed data 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.

Table 2: Evaluation of methodologies.

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

For conventional value transfer most studies find site specific transfer errors between 0 and 100% (Eigenbrod *et al.* 2010b), 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.* 2010b). 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).

2.4.5 Discussion of Methodologies

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

The different policy applications of ESS value mapping may demand different requirements in terms of accuracy and precision. If results are used for green accounting, an accurate overall value estimate of the entire study area's ESS may be desired. However, precision – meaning the accuracy of value estimates for each pixel of the map – may be of minor importance. Also land use policy evaluation may require accurate total value estimates of the different land use scenarios rather than precise value maps. In contrast, if results are used for resource allocation or for designing spatially explicit payments schemes for ESS accuracy but also precision are of greater 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 alter 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 2010a; b). The assumptions of uniform ESS supply and values across the same land covers, as used by Costanza *et al.* (1997) and repeated by many others, can be considered as a huge simplification (Plummer 2009; Eigenbrod *et al.* 2010a). It may hold for small and homogeneous case study areas and for some 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 their values, if the study area is characterised 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, if correct mean values per LCLU are applied. Thus, it may be appropriate for green accounting and land use policy evaluation at a broad scale, but offers little information for a specific location on the map. Nevertheless, if mean values are transferred that were derived within totally different spatial contexts without adjusting them to the case study area's characteristics, the information provided may be low; both in terms of precision and accuracy (Plummer 2009; Tallis and Polasky 2009).

Validated and non-validated models 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. For example, adjusting water retention services spatially based on slope, soil and LCLU (Guo *et al.* 2001) may allow for more precise ESS maps than if only mean retention capacities per LCLU class are considered. However, the application of ESS models may be limited due to the 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 the best available knowledge (De Groot *et al.* 2010). However, the quality of non-validated models remains unknown and depends heavily on the researchers' judgment regarding the causal relationships between variables. In contrast, validated models allow for validity testing by comparing the model predictions with real world observations. The share of studies that do not discuss the issue of mapping errors 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 potential errors of their ESS value map.

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. Therefore, its application is limited to small case study areas, coarse resolutions or to ESS for which such data is available in official statistics, such as timber and agricultural production.

Implicit modelling has the advantage that it allows research with a limited ecological background to include bio-physical indicators as explanatory variables in (meta-analytic) value functions. This approach can thereby account for variations in ESS supply and the value per ESS unit at the same time. This may result in more precise ESS value maps. For example, in a meta-analysis on forest values, Zandersen and Tol (2009) used not only strictly value- determining explanatory variables (such as GDP per capita and other socioeconomic characteristics) but also biophysical variables which correlate with ESS supply, such as fraction of open land, biodiversity and forest age diversity. However, modelling ESS supply and its value simultaneously introduces additional complexity, which may result in less accurate and spatially explicit ESS value maps, than if each would be estimated in separate models. The number of variables used within meta-analytic value functions is limited by the availability of primary value estimates used for the regression analysis. ESS that are not frequently valued, such as most regulating services, can therefore 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.

The specific strengths and weaknesses of different value transfer methodologies are discussed widely in value transfer literature (see for example Brouwer (2000), Navrud and Ready (2007), and Johnston and Rosenberger (2010)) and remain similar for ESS value mapping, but with some further specifications. The way in which these methods account for value-determining spatial characteristics is in particular relevance if values are mapped across large case study areas, as study areas tend to be more heterogeneous with size. Furthermore, accounting for spatial variations is in particular of importance for ESS value mapping, as one of its main purposes is to reveal how values differ across space. Whereas unit value applies one unique value per unit of ESS supply, adjusted unit values allow the adaptation of values across space by selected variables, such as income levels or the number of beneficiaries. The precision of an ESS value map produced using adjusted unit values may therefore be higher. For example, the value of flood control may not be constant across space but depends on property values at risk (De Kok and Grossmann 2010); or the amenity value of trees depends on the number of people benefiting from this service (McPherson *et al.* 2011).

ESS values may differ spatially due to further spatially variable characteristics such as the availability of substitutes and differences in human preferences across dissimilar sociocultural groups. (Meta-analytic) value functions allow such factors to be incorporated in the value mapping exercise and may thereby deliver more accurate and precise 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, this approach is generally more complex and time consuming to develop and requires comprehensive data sets of the explanatory variables across the entire study area, which may limit 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. For example, Moons *et al.* (2008) value forest recreation for a suburban region in Belgium using a travel cost model. The model is estimated based on local survey data, which may capture the local circumstances, but may be less accurate if applied within a very different spatial context.

An advantage of 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. Meta-analytic value functions are thereby able to capture the impacts of greater heterogeneity in site and context variables and methodologies in primary valuation studies (Bateman and Jones 2003; Brander *et al.* 2010). There is some evidence to suggest that meta-analytic value functions outperform other value transfer techniques, if sites differ strongly and if the number of primary valuation studies used for estimating the value function is large (Rosenberger and Phipps 2007). This suggests that meta-analytic value functions may be favourable for value mapping and that its potential may increase as the body of available primary valuation studies continues to grow. Furthermore, meta-analytic value functions allow the comparison of predictions with real world observations and thereby for quantification of prediction errors. However, meta-analyses require broad and quantitative databases of primary value estimates, which is a time consuming procedure and which may limit its application for ESS that are less widely valued.

2.5 Future Prospects in ESS Value Mapping

There are several issues within ESS value mapping that are of interest for future research. The challenge is to make ESS value maps more accurate, more precise and more comprehensive and to tailor them to support decision making. Finally, the role of biodiversity and ecosystems resilience in ESS provision remains insufficiently understood and has not been incorporated into ESS value maps.

The barriers to developing highly accurate ESS value maps are manifold. ESS values emanate from the spatial interaction of natural, human, social and built capital. Capturing these interactions is the principal challenge in ESS value mapping. Mapping of ESS and their values is dependent on quantitative, comprehensive and high resolution input data for all kinds of capital underlying the provision of ESS (social, human, built, and natural). Such data is required for both, as explanatory variables within ESS models and for model calibration. 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 harmonise available data and to construct online meta-databases to enhance access, such as the initiatives of the ''*The Ecosystem Services Partnership*'' (ESP) (http://www.es-partnership.org/esp) and ''*Earth Economics*'' (http://www.eartheconomics.org/). Quality and reporting standards for primary data collection have been repeatedly proposed in order to allow easier statistical assessments (Eigenbrod *et al.* 2010b; Rosenberger and Phipps 2007; Johnston and Rosenberger 2010). Furthermore, still little is known about many spatial determinants of ESS supply and its values. For example, how values differ across space due to differences in institutions and attitudes (Kotchen and Reiling 2000; Spash and Vant 2006; Pritchard *et al.* 2000), how different ESS are interlinked and how biodiversity contributes to ESS supply (Nicholson *et al.* 2009).

Accounting for the determinants of both ESS supply and its values requires a deeper integration of the disciplines involved (Bockstael *et al.* 2000). Still, many studies take rather mono-disciplinary approaches and only a limited number combines the strengths of multiple perspectives. Studies that are dominated by an ecological perspective tend to use sophisticated ESS models, but then apply rudimentary unit values methodologies. In absence of case specific valuation data, many studies use 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, studies that are dominated by an economic perspective 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. Comprehensive ESS value maps allow the identification of trade-offs and synergies between different ESS values. Thereby, land use policies can be identified, that maximise total ecosystem service values (Tallis and Polasky 2009; De Groot *et al.* 2010). However, due to the complexity and the interdisciplinary nature of such research, there tends to be a trade-off between comprehensive inclusion of ESS and the accuracy and precision of the analysis. Typically, studies mapping values of multiple ESS combine simple LCLU proxies with unit values. It is not only a challenge to combine multiple models of ESS, but also to link them by creating meta-ESS models that include the feedbacks and linkages between different ESS. Difficulties are faced in harmonising input and output variables of different models (Tallis and Polasky 2009; Nicholson *et al.* 2009).

Furthermore, the policy orientation of many studies is still poor. Only about 35% of the reviewed studies evaluate some kind of scenario that may allow for policy evaluation. 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 decisions 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 correlations between biodiversity and ESS supply are mixed (Cardinale *et al.* 2002; 2012; Maes *et al.* 2011b; 2012). However, the contribution of biodiversity to ecosystem resilience (its capacity to resist disturbances) and insurance values (the value of ensuring future ESS supply) are as 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. 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 scenario 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 (http://www.teebweb.org/) is mapping global ESS values based on LCLU proxies, but transferring values based on meta-analytic value function (TEEB 2010). Similarly, the AIRES project[10] (http://www.ariesonline.org/) develops value up-scaling methodologies in order to derive more accurate ESS value maps. The UK NEA (National Ecosystem Assessment) (http://uknea.unep-wcmc.org/Home/tabid/38/Default.aspx) maps ESS values of agricultural and timber products, carbon storage and recreation across the UK. It combines different methodologies for 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 (http://www.naturalcapi talproject.org/InVEST.html) aims at combining the capacities of researchers with different disciplinary backgrounds, in order to derive qualitative ESS supply and value maps for multiple ESS by combining different models and valuation methodologies (Tallis and Polasky 2009).

2.6 Conclusion

With the emergence of advanced GIS technology, spatial issues in environmental valuation have gained increasing attention and the importance of spatial relationships in ESS valuation has become widely recognised. The number of studies mapping ESS values by displaying how ESS values vary across space has grown exponentially in recent years. As compared to traditional site-specific valuation, ESS value mapping offers additional information by displaying trade-offs and synergies of alternative policy scenarios and enables the identification of preferable locations for policy measures.

Studies that map ESS values differ widely in terms of their spatial scope, purpose, disciplinary foundations and by the ESS assessed. A great variety exists in the methodologies used for revealing how ESS supply and values vary across space. Spatial variations in ESS values can be assessed by estimating spatial variations in ESS supply, the value per unit of ESS or through a combination of both of these determinants. In this paper, we developed a matrix for classifying studies with respect to the methodologies used for ESS value mapping. Methodologies for ESS supply mapping include one-dimensional proxies, validated and non-validated models, representative data and implicit modelling within (meta-analytic) value functions. Methodologies for the spatial distribution of ESS values include unit value, adjusted unit value, value function and meta-analytic value function. However, until now, no consensus exists on which methodology is best to use for what purpose.

[10] AIRES refers to ARtificial Intelligence for Ecosystem Services. For further information, consult http://www.ariesonline.org/.

Accuracy and precision are issues of great concern in ESS value mapping, which is yet insufficiently addressed in literature. Only a minor proportion of the reviewed studies assess this issue in a quantitative manner, even though evidence shows that error margins can be large. Due to coarse assessments and large uncertainty within mapped values, some studies may not reliably provide 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 limited information for site-specific assessments. The Costanza *et al.* (1997) study represents a significant step in the mapping of ESS values, but its limitations have been widely discussed in literature and are also largely recognised within the study itself. The current challenge is to develop spatially explicit models of ESS supply combined with spatially explicit (meta-analytic) value functions; both validated on real world observations in order to allow for accuracy assessment. Some promising initiatives exist, such as UK NEA, AIRES, INVEST or TEEB. However, most studies still focus either on the spatial distribution of ESS supply or on the spatial distribution of its value per unit of ESS. Only a few studies undertake efforts to incorporate both dimensions in a sophisticated manner. Mapping ESS values is a highly interdisciplinary exercise and requires the integration of ecological and economic research in order to utilise their specific strengths in assessing either the spatial biophysical or socioeconomic dimension of ESS values.

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3 Monitoring Recreation Across European Nature Areas: A Geo-database of Visitor Counts, a Review of Literature and a Call for a Visitor Counting Reporting Standard

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Key words:
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Meta-analysis
Database

Abstract:
Nature recreation and tourism is a substantial ecosystem service of Europe's countryside that has a substantial economic value and contributes considerably to income and employment of local communities. Highlighting the recreational value and economic contribution of nature areas can be used as a strong argument for the funding of protected and recreational areas. The total number of recreational visits of a nature area has been recognised as a major determinant of its economic recreational value and its contribution to local economies. This paper presents an international geo-database on recreational visitor numbers to non-urban ecosystems, containing 1,267 observations at 518 separate case study areas throughout Europe. The monitored sites are described by their centroid coordinates and shape files displaying the exact extension of the sites. Therefore, the database illustrates the spatial distribution of visitor counting throughout Europe and can be used for secondary research, such as for validation of spatially explicit recreational ecosystem service models and for identifying relevant drivers of recreational ecosystem services. To develop the database, we review visitor monitoring literature throughout Europe and give an overview of such activities with special attention to visitor counting. We identify one major shortcoming in available literature, which relates to the presentation, study area definition and methodological reporting of conducted visitor counting studies. Insufficient reporting hampers the identification of the study area, the comparability of different studies and the evaluation of the studies' quality. Based on our findings, we propose a standardised reporting template for visitor counting studies and advanced data sharing for recreational visitor data. Researchers and institutions are invited to report on their visitor counting studies via our web interface at rris.biopama.org/visitor-reporting to contribute to a global visitor database that will be shared via the ESP Visualisation tool (http://esp-mapping.net).

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3.1 Introduction

Recreation is a major ecosystem service provided by non-urban ecosystems that is of substantial economic importance. All across Europe, national parks are estimated to receive more than 2 billion recreational visits per year, which accounts for an economic recreational value of about € 14.5 billion (Schägner *et al.* 2016a). Globally, protected areas are considered to provide an economic recreational value of $US 250 billion annually through receiving 8 billion recreational visitors, who spend $US 600 billion within the destination country (Balmford *et al.* 2015a). The economic value of nature recreation and its contribution to local economies can be used as a major argument for funding nature conservation and recreational facilities (Eagles 2014).

The number of visits is the most important indicator of the economic value of recreational ecosystem services (Bateman *et al.* 2006b; Jones *et al.* 2003). Therefore, generating accurate and fine-resolution estimates of total annual recreational visits is of major importance in order to highlight the relevance and economic value of different ecosystems and landscape features for recreation as well as for the improvement of an efficient management of environmental capital. However, no aggregated data on visitor numbers to various nature areas exist on the international level. Eagles (2014) names visitor use and economic impact monitoring as two of the ten most important research priorities for recreational nature areas. By supplying site-specific visitor estimates, the importance and value of different ecosystems at different locations can be identified. As a result, resources can be allocated more efficiently and recreation sites can be defended against competing use. Site specific visitor estimates also have crucial relevance for designing the supply of recreational facilities, the protection of nature against overuse, avoiding visitor crowding and for the evaluation of site management strategies (Hadwen *et al.* 2007). Highlighting the importance of protected areas and ecosystems for recreational services has multiple effects on local and national policies in many different countries (Sievänen *et al.* 2008) and it is also required by the EU Biodiversity Strategy 2020 (Maes *et al.* 2013) and the Convention on Biological Diversity's (CBD) Strategic Plan (CBD 2010).

Nevertheless, within outdoor recreation research, studies focusing on the economic valuation of recreation are far more common than studies on estimating accurate visitor numbers, even though the number of visits is the most important indicator for the economic value of recreational ecosystem services. Furthermore, visitor numbers vary far more across recreational sites than the value per visit (Jones *et al.* 2003; Bateman *et al.* 2006). Several studies on the recreational value of nature undertake extensive valuation exercises, but are based on relatively poor visitor estimates. Several papers review studies on the economic valuation of recreation by conducting meta-analysis in order to identify the determinants of the studies' results (Bateman and Jones 2007; Rosenberger and Loomis 2000; Shresta *et al.* 2007; Zandersen and Tol 2009) or they present databases on the vast amount of studies, their results and methodologies used (McComb *et al.* 2006). For studies estimating the total recreational visitor numbers of certain sites, such information is relatively rare and less professionally organised. Bateman *et al.* (2006b) describe this disparity with "*The Tale of Horse and Rabbit Stew*", in which the cook spends most of his time preparing the rabbit for his king, even though it is the horse that makes the stew delicious. Schägner *et al.* (2016a) find that the spatial standard deviations of recreational visitor numbers are about 360 times larger than those of the economic value per visit. Cole (2006) states that visitor monitoring is "*lost in the gulf between science and management*". In recent years, the importance of accurate visitor estimates has become more and more recognised within the scientific community. The Tourism and Protected Areas Specialist *(*TAPAS) Group[11], *a joint initiative* by the International Union for Conservation of Nature (IUCN) and its World Commission on Protected Areas (WCPA), is currently acquiring funding for developing a global database on visitor numbers to IUCN Category II Protected Areas (national parks) (Spenceley 2016). A single conference session is dedicated to *"Visitors count! - Count visitation! Tourism in protected areas …"* at IUCN World Conservation Congress 2016 (Engels 2016).

The importance of nature-based recreation is recognised by the EU Biodiversity Strategy to 2020. The physical and monetary mapping and assessment of ecosystem services including cultural services such as nature-based recreation is an essential part of this strategy under Action 5. Maes *et al.* (2016) describe an indicator framework that can be used to ensure that coherent assessment approaches are used throughout the European Union. The number of visitors is retained as the most important indicator to quantify nature-based recreation but they observe that no harmonised, spatially-explicit data for this indicator are available at EU level.

Data on long-term trends in recreational use for various sites is critical for the economic valuation of different recreational sites, in order to identify determinants of recreational use and to evaluate the effects of various management strategies. It is crucial to make the acquired data available to the international research community, such as by other data sharing tools in other disciplines (DEIMS 2015; Drakou *et al.* 2015; JRC 2015).

So far, only some publications review visitor monitoring studies. For example, Kajala *et al.* (2006) review trends of visitor monitoring in Scandinavian and Baltic countries. They highlight the importance of standardised approaches and methodologies across countries. In the follow-up report, Kajala *et al.* (2007) propose some standards for monitoring visitors in the Nordic and Baltic countries, but with a more general focus. Whereas Hornback and Eagles (1999) propose visitor monitoring standards for protected areas in an international context and focus more on the results of the conducted studies than on detailed reporting, Sievänen *et al.* (2008) and Sievänen *et al.* (2009) review recreational monitoring programs across Europe as well as recreational supply indicators, but with a focus on forests only. They also propose a harmonisation of visitor monitoring and counting programs.

We instead promote the application of a variety of approaches and methodologies in recreational visitor monitoring and counting in order to let the methods evolve and develop, but call for detailed and standardised reporting of results and applied methodologies. A wide variety of methods can be used to estimate the number of recreational visits including the evaluation of trail use, samples of personal counting, and automated remote controlled counting devices. Counting samples can be scaled up over time and space by different means of accounting for counting times, days, season and weather as well as counting locations. The emerging use of GPS tracking and social media may allow for new and more efficient ways of estimating visitor numbers for recreational sites (Brandenburg *et al.* 2008; Wood *et al.* 2013b). Each visitor counting method may have its specific advantages and disadvantages and the methodological choice may have a strong and systematic effect on the estimated visitor numbers and on the accuracy of the estimate. By comprehensive reporting of the methodological choice, statistical regression analysis by means of meta-analysis can identify these effects and thereby help to improve visitor counting methods and give insights into the drivers of recreational use. Thereby, detailed reporting allows comparing results of different methods, but also

[11] For more information, *see* http://www.iucn.org/protected-areas/world-commission-protected-areas/wcpa/what-we-do/tourism-tapas.

for visitor monitoring and counting methods to evolve and progress. A harmonisation of visitor monitoring and counting approaches would increase the comparability of different studies even more, but may require the application of methods that do not fit the site-specific circumstances and the purpose of the study. In addition, it may hamper methodological developments and innovations in visitor monitoring and counting. Quality and reporting standards for primary data collection have been repeatedly proposed in other disciplines in order to ease statistical assessments such as in environmental economic valuation (Eigenbrod *et al.* 2010a; Johnston and Rosenberger 2010; Loomis and Rosenberger 2006; Rosenberger and Phipps 2007; Rosenthal and DiMatteo 2001; Stanley *et al.* 2013) or species distribution sampling (EU BON and GBIF 2015; Walls *et al.* 2014).

Based on a broad review of visitor monitoring studies with special focus on visitor counting, we propose that recreational visitor counting should (1) receive far more attention in scientific literature and funding schemes and (2) apply a more scientific and professional approach towards presentation of the gathered results and knowledge as well as reporting of the used methodologies. Multiple visitor counting studies are characterised by rudimentary reporting that does neither allow identifying the study area without local knowledge nor the study's quality. Officially published visitor numbers that are based on rough guesses may overstate real numbers by up to 26 fold (Job *et al.* 2014; Mehnen 2005; Ruschkowski 2010).

Within this paper we contribute to the field of visitor monitoring and counting by: (1) presenting a harmonised, spatially-explicit geo-database at EU level containing 1,267 total annual visitor observations at 518 separate nature areas including their exact locations and extension, (2) giving a review on visitor monitoring activities throughout Europe with a specific focus on visitor counting, (3) proposing a methodological reporting standard template for visitor counting studies based on the findings of our literature review (see appendix of this chapter) and (4) inviting the community to submit their visitor counting data via an web interface to contribute to a global database at rris.biopama.org/visitor-reporting.

The visitor number database allows for identifying visitor counting studies across Europe and can be used to estimate the importance of different drivers of recreational use. Thereby it may help to design and manage attractive recreational areas. The review provides insights into the trends of visitor monitoring across Europe and gives guidance on future prospects in visitor monitoring and counting practice. The reporting standards may support the quality and transparency of future visitor counting studies by allowing for assessments of the quality of single visitor estimates and for drawing conclusions on future visitor counting practice. It may also support the use of study results for secondary research, such as reviewing methodological evolvements and to allow for conducting meta-analysis as done in other disciplines such as in recreational economic valuation (Rosenthal and DiMatteo 2001; Zandersen and Tol 2009).

This paper is organised as follows: Section 2 gives a brief description on why and how the data was collected. Section 3.1 gives some summary statistics on the database. Then, in section 3.2, we describe general trends in visitor counting across Europe, and in section 3.3 we identify shortcomings in recent methodological reporting in visitor monitoring and counting studies. Therefore, we propose a reporting standard for visitor counting studies. Section 4 discusses our main findings before we conclude in the final section.

3.2 Methodology and Data

The aim of the study was to build up a database of total annual recreational visitor numbers of non-urban ecosystems all across Europe, in order to highlight the importance and value of different ecosystems for nature recreation. The database serves as a basis for statistical regression analysis of the drivers of recreational use and the effects of different methodologies in order to identify what ecosystem characteristics and landscape features attract and deter recreational visitors. The modelling results are published in Schägner *et al.* (2016a) and Schägner *et al.* (2016b). Therefore, we collected recreational visitor estimates that relates to a clearly defined nature area within Europe and that represent the number of visitors for an entire year and also appear to be a reliable estimate. To collect data we conducted a vast review of visitor monitoring literature. Visitor monitoring consists of a variety of survey and counting exercises that are implemented in order to obtain systematic information about recreational visitors. Total annual visitor estimates are often produced as part of a visitor monitoring study (Kajala *et al.* 2007). To search for visitor data we consulted online search tools, using general search engines, such as Google scholar, web of science, science direct and Scopus. Furthermore, we contacted relevant stakeholders from governmental and non-governmental agencies as well as researchers and managers of national park administrations across Europe. Finally, relevant conference proceedings were scanned, particularly the International Conferences on Management and Monitoring Visitors in Recreational Areas (MMV). The primary search for data was conducted in English, searching for data published within international scientific publications. However, a large amount of data is published in grey literature, which is solely published in national languages. Therefore, we also conducted an extensive online search for data in German and more rudimentary searches in Italian, Spanish, French and Portuguese, the languages accessible to the authors of this study. All total annual visitor estimates were entered into an ArcGIS geo-database and combined with referenced bibliographic information, all available methodological information and a GIS-shape file that indicates the exact location and extension of the case study area. We obtained shape files for each case study area by extracting them from an existing database on protected areas (EEA 2013; IUCN and UNEP 2015), by contacting study authors or stake-holders or by manually drawing them from map images presented in the publication or on the internet. For further analysis, all area covered by water (either inland or ocean water) was erased from the shape files in order to derive shapes of the terrestrial area only. This was done in order to derive comparable estimates of visitors per hectare. Some case study areas, such as a lake or a marine protected area, consist of more than 90% of water cover and since visitors spend most of their time on land, water covered areas would be a distortion. The database allows extracting site-specific information of the different case study areas by using available GIS data, such as ecosystem characteristics, socio-demographic and climate data, without consulting single publications, stake-holders or collecting data on-site. While hunting for visitor data, we reviewed relevant visitor monitoring studies and activities in different European countries.

3.3 Results

3.3.1 A Geo-database of Visitor Counts

In total we found 1,267 total annual visitor observations of about 518 separate case study areas all across Europe, which estimate a total of about 400 million visits a year. By far, the most case study areas are located in the UK (170), but also in Italy (57) and the relatively small countries Denmark (57) and the Netherlands (50) show a large amount of case study areas. Surprisingly, only very few estimates for the large countries Germany (13) and France (5) were found. For the following EU countries we could not obtain any observation: Portugal, Bulgaria, Estonia, Lithonia, Greece, and

Iceland and the small countries Luxembourg, Cypress and Malta, even though visitor monitoring activities take place in most countries.

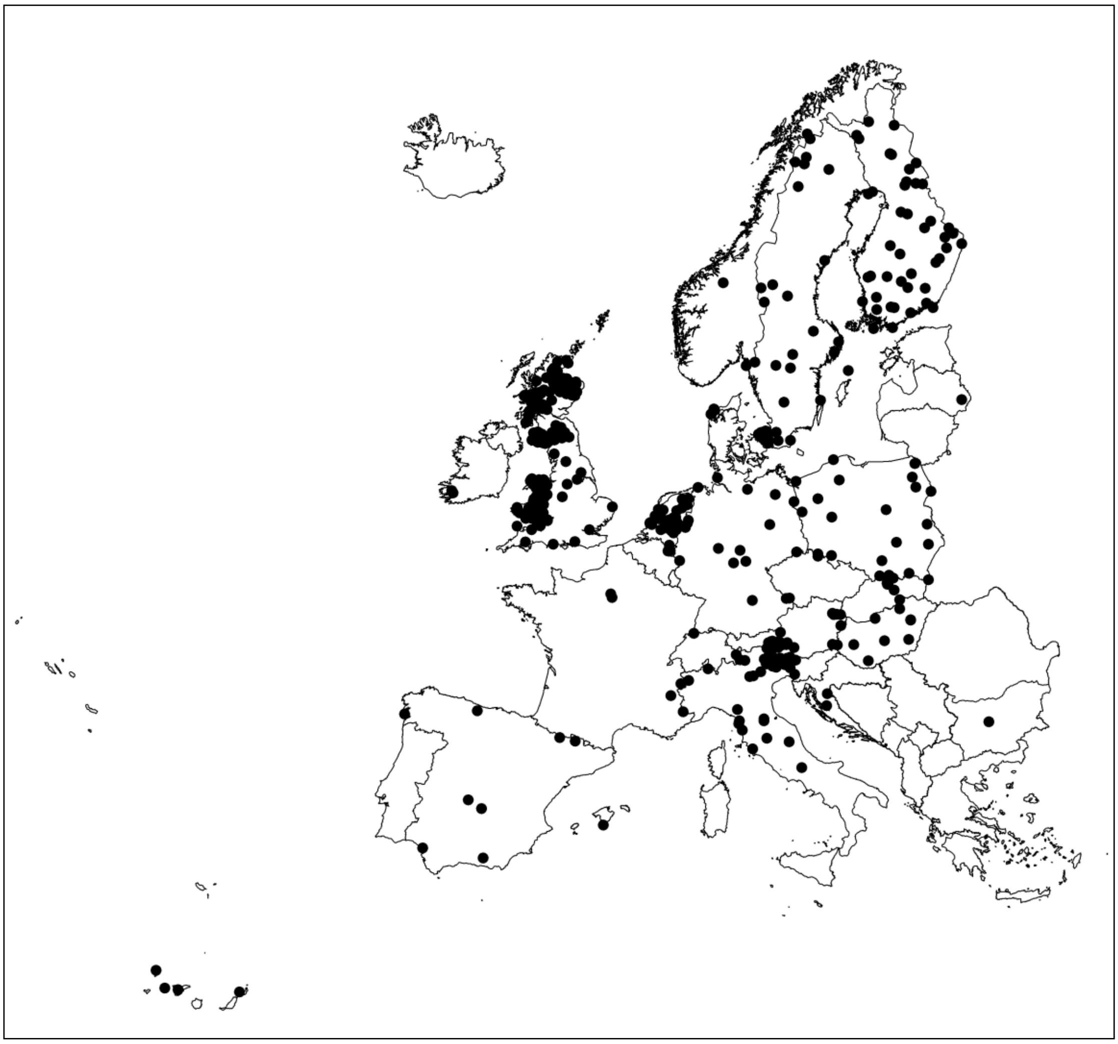

Figure 18: Location of total annual visitor observations across Europe.

About 40% of all the observations represent visitor estimates of national parks or parts of national parks. Another 15% of case study areas are other types of protected areas[12]. We found a considerable amount of monitored sites not being protected at all only in four countries; 140 in the UK, 41 in Denmark, 27 in the Netherlands and 17 in Italy.

On average, each case study area receives about 760,000 visits a year. However, annual visits differ widely with regard to visitation rates per km² and the case study area size. The overall average of the case study areas size is about 194 km² big (excluding water cover), but ranges from only 1 hectare up to almost 9,000 km². Country averages range from 13 km² and 20 km² in Denmark and the Netherlands up to 2,200 km² in France. The average annual visits per terrestrial km² are about 4,163, but differ widely. It ranges from three visits per km² in large remote sites to up to 15.7 million in small visitor hot spot areas. There are stark differences between countries. For the Netherlands, the average is 36,600 visitors per km² and for Finland only 213. Detailed statistical analysis of drivers explaining the differing visitation rates can be found in Schägner *et al.* (2016a) and Schägner *et al.* (2016b) . Summary statistics on the gathered data are presented in Table 3. The entire database is presented in the SOM.

We were not able to obtain information on the methodology of the visitor counting studies for most of the case study areas in our database due to incomplete methodological reporting. As a result, it is impossible to apply a statistical assessment of the impacts of different visitor counting methodologies on the total annual visitor estimates. Some studies and databases list visitor numbers from multiple sites within tables without giving any reference to how this data was collected and how total annual visitor estimates are obtained. In many cases the information might be available in a language not accessible to the authors (BR 2008; BR 2012; GOBT 2007; GOBT 2009; GOBT 2010). A positive example of detailed visitor counting methodology reporting are the visitor monitoring studies of the UK Forestry Commission (TNS and FCS 2006b; TNS and FCS 2006a; TNS and FCS 2008; TNS and FCW 2005). On request, we could obtain detailed information including shapes of the study areas, precise counting locations, counting length and used devices as well as methodologies used to upscale the counted visits to the entire area and year. Many studies on single sites do have a different focus than finding total annual visitor estimates such as economic valuation of recreation (Cullinan *et al.* 2008a), evaluating the effect of crowding (Arnberger and Brandenburg 2007; Kalisch 2012) or the effects of dog walking (Jaarsma and Kooij 2010), but do include a total annual visitor estimate as a by-product. Other studies do collect all data required, but do not come up with a total annual visitor estimate, due to a different study focus (Andersen *et al.* 2014; Fredman *et al.* 2009).

[12] We classified each site as a protected area, if at least 50% of its area is classified as protected. For estimating the share of the total area classified as protected we used the intersect tool of ArcGIS 10.2, the World Database of Protected Areas and the Common Database of Designated Areas (EEA 2013; IUCN and UNEP 2015).

Table 3: Summary of the database of annual visitor counts to sampled nature areas.

Country	Observations	National parks	Other protected areas	Share national parks	Share protected areas	Total visitors of sampled sites	Mean visitors per observation	Mean visitors per km²
Austria	30	28	0	93%	0%	8,021,604	267,387	1,370
Belgium	2	0	1	0%	50%	136,835	68,418	289
Bulgaria	1	1	0	100%	0%	15,000	15,000	21
Croatia	2	2	0	100%	0%	1,056,726	528,363	2,674
Czech Republic	3	3	0	100%	0%	7,460,771	2,486,924	5,648
Denmark	57	5	11	9%	19%	16,097,268	282,408	22,257
Finland	46	39	6	85%	13%	2,337,254	50,810	213
France	5	3	0	60%	0%	30,566,216	6,113,243	2,750
Germany	13	12	1	92%	8%	28,001,142	2,153,934	3,269
Hungary	11	11	0	100%	0%	6,110,000	555,455	1,310
Ireland	2	1	0	50%	0%	78,504	39,252	672
Italy	55	11	28	20%	51%	22,493,267	408,968	1,769
Latvia	1	1	0	100%	0%	55,667	55,667	93
Netherlands	50	7	16	14%	32%	53,666,782	1,073,336	36,609
Norway	1	1	0	100%	0%	30,000	30,000	19
Poland	24	24	0	100%	0%	13,296,300	554,013	3,983
Slovakia	2	2	0	100%	0%	4,900,000	2,450,000	3,758
Slovenia	1	1	0	100%	0%	2,000,000	2,000,000	2,386
Spain	14	14	0	100%	0%	9,321,895	665,850	2,742
Sweden	27	26	1	96%	4%	2,256,369	83,569	369
Switzerland	1	1	0	100%	0%	165,000	165,000	1,001
UK	170	16	14	9%	8%	184,132,506	1,083,132	7,638
sum	**518**	**209**	**78**			**392,200,000**	**800,000**	
mean				**40%**	**15%**			**3,899**

3.3.2 Visitor monitoring and Counting Activities in Europe

The number of total annual visitor observations reported in this study represent an indicator of the visitor counting and monitoring activities in different European countries. For collecting the data, we reviewed visitor monitoring literature broadly, but with a special focus on studies estimating total annual visitor numbers. The results of this review are presented in the following section. However, we do not claim that the review is exhaustive and fully representative for visitor counting and monitoring in Europe for several reasons. First, we encountered difficulties as a lot of the primarily grey literature is published in national languages and is not accessible to the authors. Second, publication policies of visitor monitoring programs differ across countries and institutions. Asking stakeholders to supply data was characterised by varying success. Policies and helpfulness in supplying data differed across institutions and individuals and sometimes it was just a matter of luck to contact the right person at the right time, willing to help and having access the desired data. Finally, the primary purpose of this study was to construct a database on visitor counts and therefore we did not search and analyse visitor monitoring studies in depth that do not provide the desired data.

The visitor monitoring and counting activities differs not only in scope but also in focus across European countries. In many countries household surveys on the recreational activities are conducted. Such surveys offer valuable information, such as the number of trips, destinations, activities and recreational needs and attitudes, but are only rarely used for estimating site specific total visitor numbers. Often they allow for conclusions on the relative recreational use of different ecosystem types and/or regions, but not to estimate total numbers for a specific location. On-site surveys are also a common visitor monitoring practice, sometimes combined with visitor counting. However, in many cases such studies are not used to estimate total annual visitors, although the required information is collected. Some studies estimate visitor numbers only for some periods (peak days and seasons) or locations, but do not up-scale them to the entire area and year. A number of studies publish total annual visitor estimates for some sites, but because of incomplete reporting, the sites cannot be identified since either the extension or the locations of the sites are not distinct.

Applied methods for estimating total annual visitor numbers to recreational areas are manifold (see Table 4) and have diversified in recent years. Whereas most studies conduct on-site visitor counting to estimate total visitor numbers, some studies use on- and off-site surveys to estimate total visitor numbers. The application of survey data for recreational destination choice modelling has been applied several times in recent year (Sen *et al.* 2011a; Termansen *et al.* 2008; UK NEA 2011b). In the past, personal on-site counting, ticket sales or simple expert judgment based on indirect methods such as trail use etc. were most common, but new technical developments increase the options of visitor counting and estimation. Nowadays, automated remote controlled counting devices are widely used and offer great opportunities for extensive counting at relatively low costs. The application of drones, aerial images and high resolution satellite images are used to monitor species such as whales (Fretwell *et al.* 2014), elephants (McMahon *et al.* 2014) and penguins (Fretwell *et al.* 2012) as well as human crowds (Coghlan 2012). It may also be used for large scale visitor counting in recreational areas. Since the start of the digital and new media age, the vast amount of "*big data*" may open visitor estimation options that are currently hardly exploited and still to be explored. Mobile phone traffic and Wi-Fi tracking may be used to track visitors and their movements on sites, as what is done to estimate traffic jams (Stenovec 2015). On social media platforms, users share a vast amount of data that can also be used to estimate their recreational behaviour (Wood *et al.* 2013b). Search engine queries reveal the interest in certain locations. Online map surveys allow the researchers to generate surveys on recreational behaviour with increased spatial resolution at lower costs (Maptionnaire 2016). Smart phone app such as geocaching or sports activity trackers record movement patterns and the activity of recreational visitors (SDI4Apps 2016; Vítek 2012).

In the **UK**, where we found the most observations, but also in the **Netherlands, Italy and Denmark**, total annual visitor estimates of recreational areas are used as an indicator for the recreational importance and economic value of different recreational sites. This indicates a long term and widely accepted importance of expressing the value of recreational areas in economic terms that are often used to promote their conservation and compete for public funding. Also, many recreational valuation studies were found for these countries. In the **UK**, visitor monitoring is widely applied, not only for national parks and sites of recreational areas, but also for the general countryside (Cope *et al.* 2000; TNS and FCS 2006b). The Forestry Commission and as well *Natural England* provide a number of visitor monitoring studies, some publish only total annual visits, but others include general surveys on visitor needs, perception and behaviour (FC 2015; Kajala *et al.* 2007; NE 2014). Many of these studies focus on forest recreation. Visitor numbers are based on on-site counting (TNS and FCS 2006b; TNS and FCS 2006a; TNS and FCS 2008; TNS and FCW 2005) or on up-scaling of survey results (Jones *et al.* 2003; Morris and Doick 2009). Some data could be extracted from secondary studies, mainly environmental economic valuation studies (Bateman *et al.* 1998; Hill and Courtney 2006; Jones *et al.* 2003). Governmental databases present visitor numbers to a variety of visitor sites including indoor attractions such as museums and amusement parks, but also country parks and nature reserves. However, due to incomplete reporting, the quality of these estimates could not be assessed and in many cases it was difficult to define exact case study areas (VE 2014; VS 2013; VW 2014).

Most reports in **Denmark and the Netherlands** are published in national languages only and thus, we had difficulties in evaluating the visitor monitoring activities in detail. Multiple visitor estimates do show that they are vibrant. For Denmark we found indications of many total annual visitor estimates of mainly forest sites, resulting from a large scale of survey based methods and car traffic counts, some dating back to the 1970s (Jensen 1992; 2003; Jensen & Guldager 2005; Kajala *et al.* 2006; Koch 1978; 1980; 1984; Sievänen *et al.* 2008; de Vries & Veer 2007). However, we did succeed in accessing only some of these numbers. For the Netherlands, some publications list total visitor numbers for a variety of different sites (GOBT 2010; Goossen *et al.* 2011), but there is no information on the methodology and on the spatial location of the sites reported, making it a difficult task to include them in the geo-database. In addition, we found some isolated visitor estimates in separate visitor monitoring studies (Hein *et al.* 2006; Jaarsma and Kooij 2010; Ligtenberg *et al.* 2008; Nunes *et al.* 2005).

Although most of the **Southern European** countries show less experience with visitor monitoring and counting, many visitor estimates were found for sites in **Italy**, some of them in combination with a monetary valuation study (Tempesta 2010), but most focusing solely on visitor numbers as a value indicator of different sites (Sanesi *et al.* 2008; Tempesta *et al.* 2002). Nevertheless, the methodological reporting was limited in most publications except Lehar *et al.* (2004). In particular, the study area definitions are deficient in many cases and consequently, it was not possible to locate part of the case study areas.

The countries **Sweden and Finland** show strong activity in visitor monitoring, but seem to have a slightly different focus. Studies are mainly concerned with on-site visitor managing and the quality of the recreational experiences, and less on highlighting the recreational economic value and importance, by publishing total visitor numbers. General population surveys resulting in outdoor recreation demand inventories are applied widely, but do not offer site-specific numbers (Kajala *et al.* 2006; Kajala *et al.* 2007; Sievänen *et al.* 2008; Sievänen 2012). We were able to obtain visitor numbers based on on-site counting for all national parks and some other official recreational sites. Most of these estimates are based on institutionalised visitor monitoring programs including long time series of visitor counting. Metsähallitus, a Finnish state-owned enterprise, runs electronic counters continuously in national parks and recreational areas. Summary reports in English and study area maps are available online, but no detailed methodological reporting is included (Metsähallitus 2015). Swedish visitor numbers including basic information on the methodology, such as counting devices etc. were obtained on request via email (Nasstrom 2012). Further visitor estimates for urban forests and other sites are indicated in literature, but it was not possible to obtain them (Ankre and Fredman 2012a; Fredman *et al.* 2012).

Alternatively for **Norway**, it was possible to obtain only a single visitor estimate for one site. Even though visitor monitoring is not as widespread as in the other Scandinavian countries, more visitor estimates exist from on-site counting (Andersen *et al.* 2012; Andersen *et al.* 2014). More than 13 national recreation surveys were conducted in Norway, which might also include site-specific numbers (Aasetre 2008; Kajala *et al.* 2006).

In **Germany**, intensive visitor counting programs have evolved only in recent years. The recreational value of nature areas has been approached less in a quantitative manner by research and policy documents than in other countries (Mann 2007). Even though some economic valuation studies on recreation exist (Elsasser and Meyerhoff 2007), we could obtain visitor numbers only for national parks provided by studies that are mainly supervised by Hubert Job from the University of Würzburg (Job *et al.* 2003; 2005; 2010; Job and Stein 2010). Müritz national park is the only area for which we found time-series of total visitors (NPA 2010). Nevertheless, in 2011 the Federal Agency for Nature Conservation initiated a socioeconomic monitoring program, which resulted in visitor monitoring and counting activities in several protected areas.

In **France and Spain** the situations are similar. We obtained visitor numbers of national parks and some single additional sites only, but without reference to the applied methodologies. Nevertheless, visitor estimates have existed for all Spanish national parks for several years. Only very few publications on visitor monitoring and recreational valuation are published in English and language barriers made it difficult to derive further information on the visitor monitoring activities. We found some studies on the economic valuation of recreation, indicating that further visitor numbers exist in France, but could not obtain them (Bonnieux and Rainelli 2003; Scherrer 2003).

In **Austria and Switzerland** some isolated studies were found that provided visitor numbers to most national parks and some other sites. Studies result from individual initiatives of researchers and site managers. We could not identify an institutional setting for collecting such data across sites. In Austria, the team of Arne Arnberger from BOKU University is active in visitor monitoring, but focuses more on aspects such as evaluating device accuracy (Arnberger *et al.* 2005), crowding effects (Arnberger and Brandenburg 2007) or visitor structures (Arnberger and Brandenburg 2002), than on the recreational value of various recreation sites.

Language barriers particularly hindered the search for visitor numbers in eastern and southeastern Europe. Nevertheless, also thanks to the helpfulness of stakeholders, we could obtain visitor numbers and spatial information for all national parks in **Poland** and **Hungary**. An evaluation of general visitor monitoring activities in Eastern Europe beyond these activities was only possible in parts. We found some isolated studies offering visitor estimates for single sites in **Slovakia** (Taczanowska 2004) and in the **Czech Republic** (Cihar *et al.* 2008a; Cihar *et al.* 2008b). For the **Baltic countries**, only one estimate was discovered in **Latvia**, although some publications indicate growing activities in visitor monitoring (Kajala *et al.* 2006; Livina 2014). We obtained some visitor numbers from an extensive visitor counting in **Estonian** forest- and national parks, but they were not yet not scaled up to an annual basis (Karoles & Maran 2014; Roose & Sepp 2012; Vítek 2012).

In addition, we obtained some visitor estimates from isolated studies also for **Croatia** (Lukač 2002; Pettenella 2008)**, Slovenia** (Erhartic *et al.* 2012)**, Belgium** (Doidi *et al.* 2012; Gilissen and Van Den Bosch 2013), **Iceland** (Ólafsson 2012) and **Ireland**. Several of the Irish visitor estimates are part of economic valuation studies (Cronin *et al.* 2000; Cullinan *et al.* 2008a; Hynes and Hanley 2006). Even though some visitor monitoring take place, we could not obtain any total annual visitor estimate for a specific case study area in **Portugal** (de Oliveira and Mendes 2014; Mendes *et al.* 2012), in **Greece** (Xanthopoulou 2007) and in **Cyprus** (Kakouris 2007).

3.3.3 Proposed Reporting Standard for Visitor Counting

Surprisingly, visitor monitoring studies and in particular visitor counting are typically characterised by relatively rudimentary reporting on the applied methodologies and study areas. In many publications not even the case study area is sufficiently defined, although this information is crucial because recreational behaviour has a highly spatial dimension. The size of the study area is fundamental for defining the average visitors per hectare, which is the most important indicator to assess the recreational value of different landscapes and to compare different recreational sites. Geo-locating the case study area — by some centroid coordinates, or better by displaying clear borders of the site – is essential for assessing any characteristics of the site not reported in the study itself. Even if the study estimates visitor numbers of a national park, the definition of the case study area is not always as clear as someone may expect. National parks may consist of zones of different protection levels and its borders may change over time. For many studies, study area identification is impossible without contacting the authors. If the monitored site cannot be identified, then what is the use of the estimated visitor numbers? Researchers may want to use the data for future research, acquire further information on the site, compare it to other sites and may display it on larger scale GIS maps. For identifying ecosystem characteristics and landscape features that attract recreational visitors, accurate, spatially explicit and fine-resolution visitor estimates are required.

The methodologies used to estimate total recreational visitors are manifold and may have a substantial effect on the accuracy of the result and may introduce a systematic bias. Even though some publications call for standardised visitor monitoring programs (Kajala *et al.* 2006; Kajala *et al.* 2007), the used methodologies will never be the same across all studies. Detailed reporting standards allow for comparing different studies by controlling for the effects of different methodologies. Statistical analysis in terms of meta-analysis (a common procedure in many other disciplines) is a helpful tool to identify effects of different methods on study results as well as the effects of different ecosystem characteristics. Thereby, intra-area comparison can be done even though non-standardised approaches are used, and drivers of recreational use can be identified. In addition, methods for estimating total visitors numbers can evolve and new ways of recreational use estimation can develop. New data sources such as GPS tracking, remote sensing and social media data, may allow for new methods of visitor number estimation.

We therefore propose a reporting standard for recreational visitor counting studies in a language accessible to the international research community (Table 3). All the methodological aspects that may have an impact on the final visitor estimates should be reported. A spreadsheet template for visitor counting reporting can be found in the appendix of this chapter. This could be used as a minimum requirement for peer reviewed publications that contain visitor counting. The spreadsheet template contains a "*must have reporting standard*" sheet, which is considered to be the absolute minimum methodological and spatial reporting on visitor counting studies, a “*should have reporting standard*” sheet, which we strongly recommend in order to allow statistical analysis of different methodological variables and a “*nice to have reporting standard*” sheet, which contains more detailed reporting options on the spatial distribution of visitors and visitor counting within the study area. The template is flexible as it allows users to add new variables and questions in order to fit it to specific user needs and to be extended to more general visitor monitoring studies.

Table 4: Proposed reporting standard for visitor counting studies.

Methodology	Description
Study area	A clear definition of the study area including information on the size and location, preferably by a GIS shape file, otherwise by a map illustration in combination with reference coordinates; Further information on the type of ecosystem and the availability of recreational facilities such as trail length, activities offered, visitor centres, etc.
Year	Declaration of data collection periods and the year that are correlated with the final visitor estimates.
Counting methods	Clear description of the counting methods used (on-site vs. off-site methods): - On-site: direct vs. indirect methods; direct counting: personally, interviews, automated counting via turnstiles, photoelectric counters, pressure sensitive devices or video counters etc.; validation of automated counter against false counts; Indirect methods: analysis of car parks, trace use, garbage, ticket sales or deterioration of certain facilities; self-registration via guest books and boxes at summits or huts etc.; use of related statistic such as overnight stays in hotels etc. - Off-site: catchment population interviews via post, telephone or personally, expert judgment. Detailed description of visitor counting methods can be found in (Cessford and Muhar 2003; Muhar and Arnberger 2002).
Number of counts / interviews	The number of interviews taken and / or of counts made in order to estimate the total visitor numbers for the study area; the refusal rate of interviews and the targeted survey population
Type of visitors counted	The type of visitors, if only a certain type of visitors is assessed, such as defined by the type of activities (anglers, hikers or boaters), mode of transport or length of stay (day trip vs. overnight)
Spatial and temporal counting resolution	The counting resolution including information on the number and length of counting, number of counting samples and number of counting locations; the time of counting (day time, week days, months, seasons); the type of counting locations (entrance point, central hub, peripheral location etc.); coordinates of counting locations; selection of counting locations and temporal counting samples (random, systematic)
Up-scaling methodology	Methodology used to scale-up counting samples to entire area and entire year (temporal: all-year counting, visitor interview information, expert guessing, temporal trends, accounting for weather etc.; spatial: comprehensive all entrance points counting, statistical modelling, trend analysis, visitor interview information, expert guessing)

In collaboration with the TAPAS Group, IUCN and the WCPA, the visitor counting reporting template has also been translated into a web interface that allows users to report their visitor counting studies online and obtain a filled spread sheet. The web interface is meant to automate visitor data collection and to construct a global visitor database that will be shared via http://esp-mapping.net. Please visit the site and encourage everybody to share their data at http://rris.biopama.org/visitor-reporting.

3.4 Discussion

Detailed reporting of the visitor counting methodology is of great importance for two reasons. First, it enables readers to distinguish sound studies from rudimentary ones. Some visitor numbers circulating in the web may result from an unverified guess only, whereas others are based on long-term intensive visitor counting and monitoring programs and therefore are far more reliable. Visitor data quality has been given little consideration in secondary research (Hill and Courtney 2006), partly because the lack of given information makes it difficult to judge the quality of the visitor estimates. Empirical findings in Schägner *et al.* (2016a) indicate that rough guesses have the tendency of over-estimating visitor numbers. Managers and stakeholders may tend to exaggerate the recreational importance of their sites, as for example, in the case of Harz national park in Germany. Initial visitor numbers circulated by the national park administration amounted to about 45 million visitors a year (Mehnen 2005; Ruschkowski 2010), but this estimate was reduced later, first to 10 million (Lehar *et al.* 2004), then to about 4 million ('Nationalpark Harz' 2015), and finally, after a solid visitor monitoring to 1.7 million (Job *et al.* 2014). The counting method and the spatial and temporal counting resolution may be a good indicator of the uncertainties involved with visitor estimates. Visitor counting programs that are based on a few and short counting periods at a few counting locations across a large study area, require more assumptions to be made in order to generate the total visitor estimate. These assumptions should be made transparent. Presently, relatively cheap visitor counting devices are available allowing for remote access and thus comprehensive visitor counting within recreational areas is on a rise. Reporting on the used methodology becomes therefore even more important in order to distinguish reliable results from the vast amount of unverified numbers published on the web.

Second, methodological reporting allows for conducting meta-analysis of multiple visitor counting studies and thereby estimates how different recreational sites characteristics and counting methods may affect estimated visitors. This may help to improve visitor counting methods and give insights into the drivers of recreational use. Meta-analysis of multiple studies is common in various disciplines to synthesise research findings and identify patterns among study results, the effects of methodological choices and the effects of study object characteristics that may be observed by analysing multiple studies. It has a long history, mainly in epidemiology (Deeks *et al.* 2001) and clinical trials (DerSimonian and Laird 1986), but also in environment economic valuation (Rosenthal and DiMatteo 2001) psychology (Lipsey and Wilson 2000) or ecology (Claudet *et al.* 2008). Quality and methodological reporting standards are prerequisite for a successful application.

The collection of high quality, standardized and spatially explicit statistics of the number of visitors is also relevant for recreational service mapping and spatial modelling, which is one major input for natural capital accounting. Real world observation of recreational use is required to calibrate and validate geo-statistical models for ecosystem service mapping (Schägner *et al.* 2013). The EU 7th Environment Action Programme (EAP) and the EU Biodiversity Strategy include objectives to develop natural capital accounting (NCA) in the EU, with a focus on ecosystems and their services. In 2015, the European Commission has launched a dedicated initiative called INCA (Integrated system of Natural Capital and ecosystem services Accounting). The data collected in this study and our call for a reporting standard constitute a first valuable input to developing accounts which track the recreational use of nature in the EU over time (EKC 2015).

Ideally, for a spatial ecosystem service model calibration, study sites of primary data collection would be randomly selected, as done for example in ecology for estimating species distributions (Keirle 2002). Random sampling is of great importance to obtain unbiased estimators in regression analysis. Nevertheless, the visitor data presented in this paper is strongly biased towards sites being prone of receiving high recreational visitor numbers, such as national parks or other protected areas. However, the aim of many visitor counting exercises as well as of spatial recreational service modelling is to highlight the recreational value or importance of certain ecosystems as compared to others. Therefore, it is not only important to know how many people visit a specific national park or recreational area, but also how few people visit an ordinary landscape. We therefore encourage the collection of visitor data for the general countryside and not only for specific recreational areas.

Finally, data sharing offers great benefits to science in general by allowing researchers to access multiple data sets at low costs and to combine them into valuable findings. Information technologies, metadata tools and repositories offer great opportunities for data sharing and many online data sharing tools have evolved (Drakou *et al.* 2015; JRC 2015). The Digital Observatory for Protected Areas (DOPA, http://dopa.jrc.ec.europa.eu/), for example, provides a set of web services and applications that can be used primarily to assess, monitor, report and possibly forecast the state of and the pressure on protected areas at multiple scales (Dubois *et al.* 2013; 2015). The data, indicators, maps and tools provided by the DOPA can be used to support spatial planning, resource allocation, protected area development and management as well as national and international reporting by a number of end-users including policy makers, funding agencies, protected area agencies and managers, researchers and the Convention on Biological Diversity (CBD). Although currently following a top-down approach that provides local data derived from global data sets, it is the objective of the forthcoming Open DOPA to capture information from the ground by allowing end-users to submit local information on the presence of key species, threats and pressures, projects, infrastructure and recreational visitors. Sharing visitor numbers through our web interface at rris.biopama.org/visitor-reporting presents a first contribution to the Open DOPA and will allow researchers to easily access, visualise and further analyse such data to better understand recreational patterns and stimulate the exchange of ideas and knowledge.

3.5 Conclusion

We reviewed visitor monitoring activities across Europe with a special focus on visitor counting and composed a geo-database on annual recreational visitor numbers to non-urban ecosystems across Europe, including 1,267 observations of 518 separate case study areas. The database gives insights into visitor monitoring and counting activities and recreation trends across Europe and it highlights the importance of recreation as an ecosystem service of non-urban ecosystems. Based on the review, we identify shortcomings and fields of improvements for future visitor monitoring and counting activities. In particular, we find that the presentation of results and methodologies is relatively unsatisfactory compared to other disciplines. Therefore, we propose a general reporting standard template for visitor counting studies with a special focus on: (1) case study area definition, (2) methodology documentation and (3) data sharing. It is meant to increase visitor monitoring professionalism and its scientific perception, and to facilitate the use of data for further research as well as the exchange of knowledge.

Visitor monitoring has moved on from sole visitor counts towards a manifold research topic, focusing on a variety of aspects such as visitor experiences, needs, attitudes and perceptions as well as activities, movement patterns, crowding effects, conflicts and wildlife disturbance (Aoki *et al.* 2014; Loomis

2000). However, it is necessary to note that simple visitor numbers are still a crucial piece of information and missing accurate visitor estimates are still a major obstacle in site management and secondary research (Booth 2006; Eagles 2014; Hill and Courtney 2006; Loomis 2000). Information on total recreational use is essential for assessing the value and importance of different nature areas for recreation and for identifying the determining factors of different sites' recreational values, but also for estimating visitors' impacts on resources, recreational facility management, budget allocation, for assessing the economic contribution of tourism and finally to defend recreational areas against competing uses. Advancements in automated visitor counting technologies, but also new data sources such as GPS tracking, drones, high resolution satellite imaginary, social media data, mobile phone traffic and smart phone apps may allow for more accurate and precise visitor estimates at lower costs.

By sharing data across the scientific community via online data sharing tools, the data provides a valuable asset for secondary research activities. The importance of reliable, comparable and accessible recreational visitor statistics has been recognised within the scientific community (Engels 2016; Spenceley 2016). Therefore, we aim at facilitating the reporting on visitor counting studies as well as the sharing of visitor data by providing a new web interface that allows users to insert their data. Please visit and promote our web interface and contribute to a global database on recreational visitor numbers in protected and nature areas at: rris.biopama.org/visitor-reporting.

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3.8 Appendix

Table A3.1: Database of annual visitor counts to sampled nature areas

Site Name	km²	1985	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	mean	Reference
Austria																														
Donaube Floodplain NP Forest (Lobbau)	24																					600,000							600,000	(Arnberger 2006)
Lower Austria Donaube Floodplain NP	69																400,000												400,000	(Arnberger & Brandenburg 2007)
Nationalpark Hohe Tauern	1,751																		1,750,000										1,750,000	(Lehar et al. 2004)
Nationalpark Hohe Tauern, Kärnten	299																		165,180										165,180	(Lehar et al. 2004)
Nationalpark Hohe Tauern, Kärnten2	314															102,200													102,200	(Wiederwald et al. 2000)
Nationalpark Hohe Tauern, Salzburg	805																		917,488										917,488	(Lehar et al. 2004; Wiederwald et al. 2000)
Nationalpark Hohe Tauern, Tirol	611																		446,720										446,720	(Lehar et al. 2004)
Naturpark Raab	147																					30,796							30,796	(Weixlbaumer et al. 2007)
Nockberge	171															320,000													320,000	Wiederwald et al. 2000)
NP Hohe Tauern, Kärnten, Ankogelgruppe	144																		109,130										109,130	(Lehar et al. 2004)
NP Hohe Tauern, Kärnten, Mölltal	169																		56,050										56,050	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Felbertal	6																		39,180										39,180	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Fuschertal	91																		23,020										23,020	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Gasteinertal	95																		87,790										87,790	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Groáarltal	44																		93,260										93,260	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Habachtal	43																		19,110										19,110	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Hollersbachtal	63																		30,360										30,360	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Kaprunertal	28																		133,120										133,120	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Obersulzbachtal	114																		29,080										29,080	(Lehar et al. 2004)
NP Hohe Tauern, Salzburg, Stubachtal	25																		44,070										44,070	(Lehar et al. 2004)
NP Hohe Tauern, Tirol, Debanttal	42																		50,850										50,850	(Lehar et al. 2004)
NP Hohe Tauern, Tirol, Defreggental	148																		50,190										50,190	(Lehar et al. 2004)
NP Hohe Tauern, Tirol, Kalsertal	116																		91,170										91,170	(Lehar et al. 2004)

NP Hohe Tauern, Tirol, Matrei Umgebung	90																		25,200										25,200	(Lehar et al. 2004)
NP Hohe Tauern, Tirol, Tauerntal	60																		44,350										44,350	(Lehar et al. 2004)
NP Hohe Tauern, Tirol, Virgental	127																		93,790										93,790	(Lehar et al. 2004)
NP HT Salzburg, Zillertaler Alpen, Wilde Gerlos	131																		458,620										458,620	(Lehar et al. 2004)
NP HT, Salzburg, Rauris-Seidlwinkeltal	126																		170,580										170,580	(Lehar et al. 2004)
Viena Ottakringer Forest	2																		400,300										400,300	(Arnberger & Eder 2007)
Wienerberg	0.53																	1,240,000											1,240,000	(Arnberger 2006)
Belgium																														
Kempen Broek	266																											105,117	105,117	(Gilissen & Van Den Bosch 2013)
Maas Valley River Park	208																										30,538	32,898	31,718	(Gilissen & Van Den Bosch 2013)
Bulgaria																														
Central Balkan National Park (CBNP)	727																		15,000										15,000	(Taylor 2004)
Croatia																														
Paklenica National Park	99												40,000				90,000												65,000	(Lukač 2002)
Plitvice Lakes National Park	296																						900,000				1,083,451		991,726	(Čulinović 2012; Pettenella 2008)
Czech Republic																														
Krkonoše Mountains NP	550								6,000,000							5,700,000													5,850,000	(Stursa 2002; Stastna 2006)
Sumava National Park	685												1,200,000																1,200,000	(Trebicky & Cihar 2006)
Sumava National Park (core area)	86												406,582	425,780	403,997	395,967	406,858	301,101	454,370	445,567	428,687	438,797							410,771	(Cihar et al. 2008)
Denmark																														
Aggebo og Græsted Hegn	2																											33,710	33,710	(Mette 2011)
Avderod Sk.	0.71																											4,295	4,295	(Mette 2011)
Bistrup Hegn	0.48																											6,878	6,878	(Mette 2011)
Brødemose Sk.	0.70																											6,752	6,752	(Mette 2011)
Brødeskov	2																											20,600	20,600	(Mette 2011)
Charlottenlund Sk.	0.75																											956,625	956,625	(Mette 2011)
Danstrup og Krogenberg Hegn	6																											145,879	145,879	(Mette 2011)
Egebæks Vang	1																											68,923	68,923	(Mette 2011)

Folehave	3																											584,622	584,622	(Mette 2011)
Freerslev Hegn	3																											40,194	40,194	(Mette 2011)
Ganlose Eget, Krogelund og Klokkekilde	2																											92,466	92,466	(Mette 2011)
Ganlose Ore, Farum Lillevang m.v.	8																											246,745	246,745	(Mette 2011)
Geelskov	2																											344,168	344,168	(Mette 2011)
Gribskov Stenholtvang	56																											656,897	656,897	(Mette 2011)
Gronholt Hegn	3																											88,809	88,809	(Mette 2011)
Gronholt Vang m.v.	2																											46,985	46,985	(Mette 2011)
Gronnæsse Sk.	1.00																											10,417	10,417	(Mette 2011)
Gurrevang	2																											101,153	101,153	(Mette 2011)
Hareskovene, Jonstrup Vang og Bondernes Hegn	8																											1,115,033	1,115,033	(Mette 2011)
Hobjerg Hegn	2																											29,351	29,351	(Mette 2011)
Hornbæk Pl.	2																											154,114	154,114	(Mette 2011)
Horneby Sand	0.77																											54,052	54,052	(Mette 2011)
Jugersborg Dyrehave	11																											3,310,682	3,310,682	(Mette 2011)
Jugersborg Hegn	4																											928,342	928,342	(Mette 2011)
Klosterris og Horserod Hegn m.v.	10																											93,299	93,299	(Mette 2011)
Knorrenborg Vang	1																											7,481	7,481	(Mette 2011)
Kohaven m.v.	0.34																											20,958	20,958	(Mette 2011)
Kongelunden	3																											566,053	566,053	(Mette 2011)
Krogerup, Babylone og Hejre Sk.	0.68																											63,314	63,314	(Mette 2011)
Lave Sk.	0.76																											109,693	109,693	(Mette 2011)
Lyngby Skov	0.66																											18,843	18,843	(Mette 2011)
Lystrup Skov	2																											69,096	69,096	(Mette 2011)
Nejede Vesterskov	1																											9,093	9,093	(Mette 2011)
Norresk.	2																											251,546	251,546	(Mette 2011)
Nyrup Hegn	2																											116,697	116,697	(Mette 2011)
Ravnsholt Sonderskov	4																											109,120	109,120	(Mette 2011)
Rude Skov	6																											598,398	598,398	(Mette 2011)

Sjælso Lund	0.82																											65,907	65,907	(Mette 2011)
Slagslunde	2																											129,908	129,908	(Mette 2011)
Snevret Skov	0.69																											4,392	4,392	(Mette 2011)
Sollerod Kirkesk. og Rygsrd	2																											334,498	334,498	(Mette 2011)
Sonnerup Sk. m.v.	0.61																											12,680	12,680	(Mette 2011)
St Dyrehave	11																											267,218	267,218	(Mette 2011)
Stasevang	0.72																											35,153	35,153	(Mette 2011)
Teglstrup Hegn og Hellebæk Sk.	9																											256,270	256,270	(Mette 2011)
Thy (northern beaches)	2																		51,000										51,000	(NIRAS 2005)
Thy (northern part, line between Stenbjerg Tvorup)	132																		300,000										300,000	(NIRAS 2005)
Thy National Park	244																		777,000										777,000	(NIRAS 2005)
Thy National Park (southern beaches)	4																		175,000										175,000	(NIRAS 2005)
Thy(southern part, line between Stenbjerg Tvorup)	111																		300,000										300,000	(NIRAS 2005)
Tisvilde Hegn m.v.	20																											523,000	523,000	(Mette 2011)
Tokkekob Hegn	6																											170,275	170,275	(Mette 2011)
Trorod Hegn	0.56																											104,669	104,669	(Mette 2011)
Uggellose Skov	2																											57,917	57,917	(Mette 2011)
Ullerup Sk.	0.95																											8,915	8,915	(Mette 2011)
Valby Hegn	4																											37,814	37,814	(Mette 2011)
Vestskoven Vest	13																											1,404,369	1,404,369	(Mette 2011)
Finland																														
Bothnian Sea	918																										67,000		67,000	(MNHS 2002-2011)
Evo	91																	50,000	50,000	50,000	50,000	50,000	50,000	50,000	50,000	84,500	61,000		54,550	(MNHS 2002-2011)
Helvetinjärvi	53																	32,000	32,000	32,000	32,000	32,000	33,000	33,000	33,000	23,000	20,500		30,250	(MNHS 2002-2011)
Hiidenportti	45																	8,000	7,500	7,700	10,000	10,000	6,500	9,000	12,000	11,000	10,000		9,170	(MNHS 2002-2011)
Hossa	91																	44,500	42,000	42,000	48,100	49,000	53,000	53,000	54,500	48,000	52,200		48,630	(MNHS 2002-2011)
Isojaervi	22																	8,000	8,000	9,000	8,000	7,000	8,000	11,000	10,500	10,500	10,500		9,050	(MNHS 2002-2011)
Iso-Syoete	11																	20,000	22,000	24,000	25,000	23,500	23,000	25,500	25,000	22,000	19,500		22,950	(MNHS 2002-2011)
Itäinen Suomenlahti (Eastern Gulf of Finland)	7																	15,000	15,000	15,000	16,000	17,000	17,000	17,000	19,000	19,000	16,500		16,650	(MNHS 2002-2011)

Kauhaneva-Pohjankangas	61																	6,000	6,000	6,000	6,000	6,000	6,000	3,500	4,500	5,500	5,000		5,450	(MNHS 2002-2011)
Koli	30																		120,000					110,000	127,500	138,500	134,500		126,100	(MNHS 2002-2011)
Kolovesi	48																	6,000	6,000	6,000	6,500	7,000	7,000	6,500	7,500	7,500	8,000		6,800	(MNHS 2002-2011)
Kurjenrahka	31																20,000		20,000	20,000	25,000	25,000	32,500	31,500	28,500	26,500	25,500		25,450	(MNHS 2002-2011)
Kylmaeluoma	73																	35,000	34,000	34,000	35,000	35,000	37,000	31,000	28,500	25,500	26,000		32,100	(MNHS 2002-2011)
Lauhanvuori	50																	30,000	25,000	27,000	27,000	27,000	27,500	10,000	10,000	9,500	10,000		20,300	(MNHS 2002-2011)
Leivonmäki	30																		4,500	7,000	10,000	11,000	12,000	14,500	12,500	12,500	15,000		11,000	(MNHS 2002-2011)
Lemmenjoki	2,859																	10,000	10,000	10,000	10,000	10,000	10,000	10,000	10,000	10,000	15,000		10,500	(MNHS 2002-2011)
Liesjärvi	21																25,000		15,000	16,000	25,000	25,000	22,000	29,500	30,500	31,000	22,000		24,100	(MNHS 2002-2011)
Linnansaari	265																	27,500	28,000	28,000	28,000	29,000	29,000	29,000	31,000	31,000	31,000		29,150	(MNHS 2002-2011)
Nuuksio	56																100,000		100,000	100,000	110,000	142,000	170,000	175,500	179,500	178,000	197,000		145,200	(MNHS 2002-2011)
Oulanka	294																	162,000	165,000	173,000	173,500	183,500	185,500	163,000	165,500	169,000	171,500		171,227	(MNHS 2002-2011)
Oulujaervi	15																	27,000	27,000	25,500	25,000	25,000	24,000	25,000	21,000	24,000	38,500		26,200	(MNHS 2002-2011)
Päijänne	16																	8,000	8,000	10,000	12,000	12,000	12,000	14,500	15,000	13,500	14,000		11,273	(MNHS 2002-2011)
Pallas-Ounastunturi	594																	98,000	125,000	125,000									116,000	(MNHS 2002-2011)
Pallas-Yllästunturi	1,022																		217,000		300,000	310,000	312,000	329,500	419,000	436,000	435,500		344,875	(MNHS 2002-2011)
Patvinsuo	105																15,000		15,000	20,000	14,000	15,000	14,000	12,000	12,000	12,000	12,500		14,150	(MNHS 2002-2011)
Perämeri	159																	6,500	7,200	7,200	2,500	5,500	6,000	5,000	9,000	9,500	10,000		6,840	(MNHS 2002-2011)
Petkeljärvi	7																15,000		17,000	17,000	17,500	18,500	23,000	20,000	19,500	20,500	19,000		18,700	(MNHS 2002-2011)
Puurijärvi-Isosuo	27																22,000		15,000	15,000	17,000	12,500	10,000	11,000	11,500	7,000	8,500		12,950	(MNHS 2002-2011)
Pyhä-Häkki	13																	11,000	11,000	11,000	9,000	15,500	14,500	13,500	17,000	16,500	15,500		13,450	(MNHS 2002-2011)
Pyhä-Luosto (Holy Luosto)	144																		95,000		95,000	103,500	109,500	114,000	128,000	119,000	118,500		110,313	(MNHS 2002-2011)
Pyhätunturi	43																	35,000	25,000	25,000									28,333	(MNHS 2002-2011)
Repovesi	16																		65,000	65,000	65,000	69,000	70,000	75,500	74,500	76,500	78,500		71,000	(MNHS 2002-2011)
Riisitunturi	76																	6,000	7,000	7,000	7,000	7,000	8,000	8,000	15,000	23,500	22,000		11,050	(MNHS 2002-2011)
Rokua	9																	24,000	24,000	20,000	20,000	18,000	23,500	23,500	23,500	23,500	17,000		21,700	(MNHS 2002-2011)
Ruunaa	31																	110,000	118,000	115,000	117,000	94,000	82,500	87,500	89,000	88,000	84,000		98,500	(MNHS 2002-2011)
Saaristomeri (archipelago)	495																	40,000	80,000	80,000	60,000	60,000	60,000	51,000	53,500	59,000	56,000		61,773	(MNHS 2002-2011)
Salamajärvi	62																	7,000	7,000	9,000	10,000	12,000	11,000	9,000	10,500	12,500	13,000		10,100	(MNHS 2002-2011)

Seitseminen	45																	37,000	40,000	40,000	40,000	42,000	44,000	51,000	45,500	40,500	37,500		41,750	(MNHS 2002-2011)
Sipoonkorpi	18																										75,500		75,500	(MNHS 2002-2011)
Syöte	300																25,000		24,000	34,000	33,500	33,000	36,000	34,500	40,000	31,000	33,500		32,450	(MNHS 2002-2011)
Tammisaaren saaristo (Ekenäs Archipelago)	55																	22,000	20,000	20,000	23,000	25,000	47,000	49,000	44,500	54,000	51,000		35,550	(MNHS 2002-2011)
Teijo	35																50,000		60,000	60,000	60,000	60,000	80,500	75,000	75,000	72,000	74,500		66,700	(MNHS 2002-2011)
Tiilikkajärvi	34																	6,000	6,000	7,000	6,500	7,000	7,000	6,500	7,500	8,500	7,500		6,950	(MNHS 2002-2011)
Torronsuo	30																15,000		20,000	20,000	20,000	20,000	27,000	22,500	20,500	17,000	17,000		19,900	(MNHS 2002-2011)
Urho Kekkosen kansallispuisto	2,548																	150,000	160,000	160,000	165,000	170,000	180,000	252,000	289,000	287,500	277,000		209,050	(MNHS 2002-2011)
Valkmusa	17																6,000		5,000	5,000	6,000	6,500	6,200	7,000	7,000	8,500	8,500		6,570	(MNHS 2002-2011)
France																														
Le troncon	28																			3,174,603									3,174,603	(Chegrani 2007)
Lorraine forest	8,954												25,620,000																25,620,000	(Després 1998)
Mercantour, central zone	680															800,000	427,226												613,613	(Wiederwald et al. 2000)
Parc national de la Vanoise, central zone	534															400,000	366,000												383,000	(Wiederwald et al. 2000)
Parc nationale des Ecrins, central zone	919															800,000	750,000												775,000	(Wiederwald et al. 2000)
Germany																														
Altmuehltal	2,966																			910,000									910,000	(Job et al. 2005)
Bayrischer Wald	242																						760,000						760,000	(Job et al. 2010)
Beach of B•üsum (Wadden Sea NP)	0.09															500,000													500,000	(Gätje et al. 2002)
Berchtesgaden NP	214																	1,129,538											1,300,000	(Job et al. 2003)
Eifel	109																						450,000						450,000	(Job et al. 2010)
Hainich	75																						290,000						290,000	Job et al. 2010)
Kellerwald-Edersee	57																						200,000						200,000	Job et al. 2010)
Mueritz Nationalpark	322														475,000	635,000	495,000	536,500	584,500	660,000	528,000	520,000	478,000	502,000	517,000				515,077	(Job et al. 2005; LFGMV n.d.; NPA 2006- 2010)
Naturpark Hoher Flaeming	827																			300,000									300,000	Job et al. 2010)
Niedersaechsisches Wattenmeer	2,777																						20,630,455						20,630,455	Job et al. 2010)
Sächsische Schweiz NP	94																								1,712,000				1,712,000	(Job & Stein 2010)
Schaalsee	778																						227,610						227,610	(HP 2008)

Unteres Odertal	106																						206,000						206,000	(Rein et al. 2008)
Hungary																														
Aggtelek National Park (zone b,c)	154																		200,000										200,000	(Benkhard 2004)
Balaton-felvidéki Nemzeti Park	480																		2,000,000										2,000,000	(Benkhard 2004)
Bükk (Bukki) NP Zone C	21																		1,350,000										1,350,000	(Benkhard 2004)
Duna-Dráva Nemzeti Park	499																		130,000										130,000	(Benkhard 2004)
Duna-Ipoly National Park	618																		1,500,000										1,500,000	(Benkhard 2004)
Ferto Hanság National Park (zone C)	50																		300,000										300,000	(Benkhard 2004)
Hortobagy Nemzeti Park	748																200,000												200,000	(Karácsonyi et al. 2002)
Hortobagy Nemzeti Park no zoneA	643																		200,000										200,000	(Benkhard 2004)
Kiskunsag Nemzeti Park	502																		130,000										130,000	(Benkhard 2004)
Körös-Maros Nemzeti Park	539																		80,000										80,000	(Benkhard 2004)
Orség Nemzeti Park (zone C)	410																		20,000										20,000	(Benkhard 2004)
Ireland																														
Killarney-Nationalpark	103								62,500	62,500	65,000	75,000	75,000	80,000	80,000														71,429	(Luddy n.d.)
River Roughty	13																		7,075										7,075	(Hynes & Hanley 2006)
Italy																														
Agner- Pale S	141																75,748												75,748	(Tempesta et al. 2002)
Antelao-Marmarole	210																163,150												163,150	(Tempesta et al. 2002)
Appennino Tosco-Emiliano	263																				23,000	30,000							26,500	(Sanesi et al. 2008)
Asiago - Monte Grappa	803																2,179,223												2,179,223	(Tempesta et al. 2002)
Baldo-Lessini	566																617,641												617,641	(Tempesta et al. 2002)
Bosco di Pianura	0.05																								30,839				30,839	(Tempesta et al. 2002)
Bosconero	48																52,441												52,441	(Tempesta et al. 2002)
Canal del Ferro	296																131,165												131,165	(Tempesta et al. 2002)
Cansiglio Alpago	248																355,435												355,435	(Tempesta et al. 2002)
Carnia	1,190																847,890												847,890	(Tempesta et al. 2002
Carso	373																719,067												719,067	(Tempesta et al. 2002
Castelfranco	0.10																								7,500				7,500	(Tempesta 2010)

Civetta Moiazza	126																355,435													(Tempesta et al. 2002)
Colline Moreniche	233																74,952													(Tempesta et al. 2002
Collio Colli Orientali del F	166																163,957													(Tempesta et al. 2002
Dolomiti Bellunesi NP	314														285,000			120,000											202,500	(Tempesta et al. 2002; Wiederwald et al. 2000)
Dolomiti Friulane	392																220,267													(Wiederwald et al. 2000)
Duranno-Cima Preti	117																64,095													(Tempesta et al. 2002
Foce dell Isonzo	25														31,000														31,000	(Tempesta et al. 2002
Foreste Casentinesi M. Falterona e C. (Emilia-R.)	190																				30,134	22,919							26,527	(Sanesi et al. 2008)
Foreste Casentinesi M. Falterona e C. (Tuscany)	181																				55,683	45,060							50,372	(Sanesi et al. 2008)
Gran Paradiso	712													1,700,000															1,700,000	(Wiederwald et al. 2000)
Griffon vulture project	5														8,000														8,000	(Tempesta et al. 2002
Historical garden Villa Varda	0.20														69,500														69,500	(Tempesta et al. 2002
Iris	0.06																								109,100				109,100	Tempesta 2010)
Manin	0.04																								43,800				43,800	Tempesta 2010)
Maremma	93																				64,810	64,058							64,434	(Sanesi et al. 2008)
Marmolada	31																308,820												308,820	(Tempesta et al. 2002
Monti Sibillini	697																		500,000										500,000	(Taylor 2004)
Natural Park of the Dolomiti	103														540,000														540,000	(Tempesta et al. 2002
Naturpark Rieserferner - Ahrn	216																		569,670										569,670	(Lehar et al. 2004)
Nuvolau-Averau-Croda da L	98																145,670												145,670	(Tempesta et al. 2002
Oasi Mulino di Cervara	0.27																										14,900		14,900	(Tempesta 2013)
P R Ampezzane Cortina	168														340,000														340,000	(Tempesta et al. 2002
Parco Buzzacarini Monselice	0.03																								5,803				5,803	(Tempesta 2013)
Parco Nazionale d'Abruzzo, Lazio e Molise	507																5,368,202												5,368,202	(Taylor 2004)
Parco Scan Martino delle Vaneze	0.02																								2,814				2,814	(Tempesta 2013
Pelmo	130																180,631												180,631	(Tempesta et al. 2002
Piccole Dolomiti - Pasubio	147																891,500												891,500	(Tempesta et al. 2002
Prealpi Carniche e P C Merid	668																374,758												374,758	(Tempesta et al. 2002

Prealpi Giulie	345																430,972												430,972	(Tempesta et al. 2002
Prealpi Giulie Meridionali	398																765,912												765,912	(Tempesta et al. 2002
Prealpi Venete	360																306,833												306,833	(Tempesta et al. 2002
Quadris nature area	0.21														9,000														9,000	(Tempesta et al. 2002
Sorapiss Cadini	80																46,614												46,614	(Tempesta et al. 2002
Stilfser Joch (Stelvio) Lombardy	595																27,609												27,609	(Wiederwald et al. 2000)
Stilfser Joch (Stelvio) Trient	177																200,000												200,000	(Wiederwald et al. 2000)
Tofane Cristallo	143																279,686												279,686	(Tempesta et al. 2002
Tre Cime-Croda dei Toni-Popera	73																297,167												297,167	(Tempesta et al. 2002
Val Grande	119																15,000												15,000	Wiederwald et al. 2000)
Valcanale	428																2,140,805												2,140,805	(Tempesta et al. 2002
Valle Canal Novo2	0.57														12,850														12,850	(Tempesta et al. 2002
Vette Feltrine- Monte del Sole	542																												(Tempesta et al. 2002	(Tempesta et al. 2002
Vincheto Celarda	1														8,000														8,000	(Tempesta et al. 2002
Waterfall of Molina	2														34,000														34,000	(Tempesta et al. 2002
Latvia																														
Razna NP	596																					42,000	62,000	63,000					55,667	(Muskare 2012)
Netherlands																														
Alkmaarder en Uitgeestermeer	16																				643,725	534,868	527,677	525,022	450,433				536,345	(Goossen et al. 2011)
Amstelland De Hoge Dijk Gaasperzoom	0.74																					140,429	128,931	122,032	120,325				127,929	Goossen et al. 2011)
Amstelland Elsenhove	0.25																					257,490	270,527	257,661	258,887				261,141	Goossen et al. 2011)
Amstelland Ouderkerkerplas	1																					533,694	464,549	456,749	459,653				478,661	Goossen et al. 2011)
Bijland	3																				55,000	80,000	30,000	40,000	45,000				50,000	Goossen et al. 2011)
Bosjes van Poot	0.28																						439,000						439,000	(Jaarsma & Kooij 2010)
Bosjes van Poot 1	0.08																						375,606						375,606	(Jaarsma & Kooij 2010)
Bosjes van Poot 2	0.02																						226,632						226,632	(Jaarsma & Kooij 2010)
Bosjes van Poot 3	0.01																						215,538						215,538	(Jaarsma & Kooij 2010)
Bosjes van Poot 4	0.02																						294,780						294,780	(Jaarsma & Kooij 2010)
Bosjes van Poot 5	0.06																						218,708						218,708	(Jaarsma & Kooij 2010)

Bosjes van Poot 6	0.08																					358,173						358,173	(Jaarsma & Kooij 2010)
De Wieden wetland	52																	172,456										172,456	(Hein et al. 2006)
Dobbeplas	0.64																						1,400,000					1,400,000	Goossen et al. 2011)
Drentsche Aa National Landscape	106																1,000,000											1,000,000	Goossen et al. 2011)
Dwingelderveld National Park	38																							1,800,000				1,800,000	(Ligtenberg et al. 2008)
Eiland van Maurik Maurik	3																			165,000	250,000	110,000	100,000	185,000				162,000	Goossen et al. 2011)
Gaasperzoom Gaasperplas	2																				1,740,775	1,703,817	2,131,410	2,231,874				1,951,969	Goossen et al. 2011)
Gaasperzoom Overdiemen	0.20																					51,434	53,851	44,290				49,858	Goossen et al. 2011)
Geestmerambacht	2																			894,879	847,388	725,864	735,995	768,713				794,568	Goossen et al. 2011)
Ginkelse Heide Ede	8																						30,000	33,000				31,500	Goossen et al. 2011)
Greater Veluwe area	988																29,700,000											29,700,000	(Nunes et al. 2005)
Het Boomkroonpad	0.01																			91,000	103,000	114,000	104,000	116,500				105,700	Goossen et al. 2011)
Het Hulsbeek	3																			386,000	276,900	226,000	234,400	395,147				303,689	Goossen et al. 2011)
Het Nulde	0.22																			371,000	435,000	302,500	424,000	518,000				410,100	Goossen et al. 2011)
Het Rutbeek	1																			259,000	236,800	190,000	229,600	234,814				230,043	Goossen et al. 2011)
Het Twiske	6																			1,550,911	1,389,352	1,307,968	1,096,732	1,117,487				1,292,490	Goossen et al. 2011)
Het Zandenbos	3																			130,000	118,000	125,000	125,000	130,000				125,600	Goossen et al. 2011)
Hoge Veluwe National Park	51																564,000	680,000	547,000	525,000	527,205	527,394	501,055	518,580				537,651	(GOBT 2007-2010)
Kievitsveld	0.81																						99,000	110,000				104,500	(GOBT 2010)
Kievitsveld Emst	0.40																						99,000	110,000				104,500	(GOBT 2010)
Landgoed Fraeylemaborg	0.28																			88,429	92,552	75,784	76,037	82,527				83,066	Goossen et al. 2011)
Lingebos	0.89																			95,000	80,000	92,000	50,000	16,500				66,700	Goossen et al. 2011)
Meijendel	1																			913,000	839,000	875,000	859,704					871,676	Goossen et al. 2011)
Mookerplas Plasmolen	2																			223,000	295,000	165,000	244,750	253,500				236,250	(GOBT 2010)
Park van Luna	3																					338,335	319,229	278,805				312,123	Goossen et al. 2011)
Recreatiegebied Hilgelo W	0.74																			135,000	175,000	80,000	110,000	125,000				125,000	(GOBT 2010)
Recreatiegebied Beldert Z	0.78																			205,000	290,000	120,000	110,000	150,000				175,000	(GOBT 2010)
Recreatiegebied Berendonck	2																			270,000	375,000	209,000	270,000	293,250				283,450	(GOBT 2010)
Recreatiegebied Bussloo V	3																			885,000	1,028,000	759,000	1,045,000	1,172,000				977,800	(GOBT 2010)

Recreatiegebied Heerderstr	0.67																				221,000	230,000	115,500	181,450	183,500				186,290	(GOBT 2010)
Recreatiegebied Rhederlaag	7																				352,000	466,000	220,000	371,900	432,750				368,530	(GOBT 2010)
Recreatiegebied Zeumeren	0.72																				230,000	325,000	187,000	317,800	405,250				293,010	(GOBT 2010)
Recreatieschap West-Friesland	3																						215,150						215,150	Goossen et al. 2011)
Rijkerswoerdse Plassen Elst	0.79																				190,000	220,000	100,000	105,000	155,000				154,000	(GOBT 2010)
Stichting Recreatie Nienoord	0.70																				292,000	296,000	306,000	304,000	295,000				298,600	Goossen et al. 2011)
Strandpark Slijk Ewijk	0.94																				210,000	255,000	115,000	115,000	150,000				169,000	(GOBT 2010)
National Park	64																								970,000				970,000	Goossen et al. 2011)
Veluwezoom National Park	50																				2,000,000	2,010,000	2,025,000	2,035,000	2,040,000				2,022,000	(GOBT 2010)
Zuid-Kennemerland National Park	36																							2,000,000					2,000,000	Goossen et al. 2011)
Norway																														
Dovrefjell Sunndalsfjella management area	1,596																								30,000				30,000	(Gundersen & Andersen 2010)
Poland																														
Babiog¢rski	34																										75,000		75,000	(Skarbek 2012)
Bialowieza (Bialowieski) National Park	104															100,000											133,800		116,900	(Kun 2002; Skarbek 2012)
Biebrzanski	597																										27,200		27,200	(Skarbek 2012)
Bieszczady NP (Bieszczadzki)	292																										290,000		290,000	(Skarbek 2012)
Bory Tucholskie Park Narodowy	46																										60,000		60,000	(Skarbek 2012)
Drawienski Park Narodowy	113																										48,000		48,000	(Skarbek 2012)
G¢r Stolowych Park Narodowy	67																										335,000		335,000	(Skarbek 2012)
Gorczanski Park Narodowy	70																										65,000		65,000	(Skarbek 2012)
Kampinoski Park Narodowy	377																										1,000,000		1,000,000	(Skarbek 2012)
Karkonoski Park Narodowy	56								2,500,000																		2,250,000		2,250,000	(Skarbek 2012; Stursa 2002)
Magurski Park Narodowy	197																										45,000		45,000	(Skarbek 2012)
Narwianski Park Narodowy	68																										10,000		10,000	(Skarbek 2012)
Ojcowski Park Narodowy	22																										400,000		400,000	(Skarbek 2012)
Pieninski Park Narodowy	24																										710,000		710,000	(Skarbek 2012)
Poleski Park Narodowy	98																										23,700		23,700	(Skarbek 2012)

Roztoczanski Park Narodowy	83																										100,000		100,000	(Skarbek 2012)
Slowinski Park Narodowy	215																										317,100		317,100	(Skarbek 2012)
Swietokrzyski Park Narodowy	78																										193,400		193,400	(Skarbek 2012)
Tatra NP (Polish part, mountains only, Tatry)	176																			1,900,000									1,900,000	(Blazejczyk 2002)
Tatry NP (Polish part, Tatra, Tatrzanski)	212										2,500,000																		2,500,000	(Taczanowska 2004)
Ujscie Warty Park Narodowy	76																										20,000		20,000	(Skarbek 2012)
Wielkopolski Park Narodowy	76																										1,200,000		1,200,000	(Skarbek 2012)
Wigierski Park Narodowy	151																										110,000		110,000	(Skarbek 2012)
Wolinski Park Narodowy	108																										1,500,000		1,500,000	(Skarbek 2012)
Slovakia																														
Tatra National Parks (Slovak part)	743																		4,000,000										4,000,000	(Taczanowska 2004)
Tatra National Parks (Slovak part, mountains only)	561																		900,000										900,000	Taczanowska 2004)
Slovenia																														
Triglav	838															2,000,000													2,000,000	(Wiederwald et al. 2000)
Spain																														
Aig•estortes i Estany de Sant Maurici	139											333,734	345,545	349,021	369,223	382,264	410,427	362,822	356,411	341,759	337,484	355,633	322,555	304,606	329,227	294,547			346,351	(INE 2012)
Archipiélago de Cabrera	100						21,891	28,729	35,934	32,226	45,000	39,265	43,215	52,796	47,302	44,983	64,078	66,302	66,535	73,540	71,987	74,532	76,541	60,804	60,662	64,688			53,551	(INE 2012)
Caballeros	409									6,400		22,984	30,145	51,000	52,921	59,015	51,822	63,277	63,739	70,782	66,935	73,926	72,688	78,767	90,001	92,578			59,186	(INE 2012)
Caldera de Taburiente	44				100,000	120,000	142,167	157,520	324,705	212,179	200,000	250,000	210,141	265,961	288,032	347,619	377,726	375,753	395,264	367,938	380,399	371,558	389,024	408,088	337,649	387,805			291,342	(INE 2012)
Doñana	554				250,000	250,000	225,818	249,526	269,331	202,954	250,000	366,287	417,287	385,393	384,276	385,563	394,401	407,693	361,984	391,536	381,964	376,287	384,638	350,005	380,155	341,961			336,685	(INE 2012)
Garajonay	38				125,000	150,000	245,386	221,581	300,000	450,000	500,000	450,000	550,000	525,000	569,000	615,000	520,000	507,000	641,754	859,860	854,824	842,467	884,858	860,000	625,801	610,254			541,263	(INE 2012)
Islas Atlánticas de Galicia	11																		171,999	182,394	213,897	220,240	238,939	254,000	274,716	292,374			231,070	(INE 2012)
Monfrag•e	184																						351,885	331,788	306,041	297,976			321,923	(INE 2012)
Ordesa y Monte Perdido	157				450,000	500,000	585,000	600,000	650,000	650,000	702,700	624,503	601,500	603,004	624,263	635,876	657,045	622,014	619,700	582,800	598,950	616,700	617,950	616,600	617,500	614,059			608,644	(INE 2012)
Picos de Europa	641				700,000	800,000	860,234	836,511	950,825	941,080	1,100,000	1,676,392	1,535,376	1,451,697	1,619,588	1,869,063	1,669,973	1,596,825	1,990,255	2,221,761	1,939,803	1,863,847	1,774,955	1,712,668	1,818,671	1,610,341			1,479,085	(INE 2012)
Sierra Nevada	863														250,000	275,000	292,128	302,520	315,000	558,489	645,738	728,137	737,183	684,573	673,302	667,319			510,782	(INE 2012)
Tablas de Daimiel	19				101,602	96,183	94,916	82,690	73,952	83,746	60,190	130,774	285,371	146,652	112,195	115,503	109,753	100,099	107,437	128,640	123,413	100,666	122,955	94,687	105,957	398,742			126,187	(INE 2012)
Teide	190				1,000,000	1,100,000	2,227,000	2,250,000	2,250,000	2,434,152	2,500,000	3,000,000	3,237,000	3,554,782	3,868,839	3,722,913	3,589,164	3,488,622	3,364,873	3,540,195	3,349,204	3,567,701	3,142,418	2,866,057	3,052,830	2,407,480			2,886,965	(INE 2012)
Timanfaya	52				800,000	700,000	1,000,000	1,200,000	1,300,000	1,757,513	1,450,000	1,575,135	1,606,638	1,691,347	1,742,087	1,800,000	1,866,000	1,768,566	1,841,431	1,815,186	1,778,882	1,787,776	1,748,149	1,600,175	1,371,349	1,434,705			1,528,861	(INE 2012)

Sweden																														
Abisko	77																											50,000	50,000	(Nasstrom 2012)
Ängsö	0.83																											8,000	8,000	(Nasstrom 2012)
Blå Jungfrun	0.70																					3,215	4,010	4,180	3 440	2,684	3,400	3,400	(Nasstrom 2012)
Dalby söderskog	0.36																											100,000	100,000	(Nasstrom 2012)
Djurö	3																											2,000	2,000	(Nasstrom 2012)
Färnebofjärden	70																											48,000	48,000	(Nasstrom 2012)
Fulufjället National Park	408																38,000	53,000	35,000									50,000	44,000	(Nasstrom 2012; Taylor 2004; Fredman et al. 2007)
Garphyttan	1																											30,000	30,000	(Nasstrom 2012)
Gotska Sandön	45																					4,404	4,490	5,279	4,813	4,461	4,469	4,469	(Nasstrom 2012)
Hamra	14																											11,000	11,000	(Nasstrom 2012)
Haparanda Skärgård	7																											5,000	5,000	(Nasstrom 2012)
Kosterhavet	8																											500,000	500,000	(Nasstrom 2012)
Muddus	511																											5,000	5,000	(Nasstrom 2012)
Pieljekaise	155																											1,000	1,000	(Nasstrom 2012)
Sarek	1,986																											5,000	5,000	(Nasstrom 2012)
Skuleskogen	30																											36,000	36,000	(Nasstrom 2012)
Söderåsen	16																											300,000	300,000	(Nasstrom 2012)
Sonfjallet	104																											10,000	10,000	(Nasstrom 2012)
Stenshuvud	4																											500,000	500,000	(Nasstrom 2012)
Stora Sjöfallet	1,280																											10,000	10,000	(Nasstrom 2012)
Store Mosse	76																											100,000	100,000	(Nasstrom 2012)
Tiveden	14																						120,340	116,750	125,268	120,886	120,800	120,800	(Nasstrom 2012)
Töfsingdalen	17																											200	200	(Nasstrom 2012)
Tresticklan	29																											4,000	4,000	(Nasstrom 2012)
Tyresta	20																											320,000	320,000	(Nasstrom 2012)
Vadvetjåkka	27																											500	500	(Nasstrom 2012)
Valadalen, a natural reserve	1,217	38,000																											38,000	(Bojö & Hultkrantz 1985)

Switzerland																														
Parc naziunal Svizzer	165						250,000		150,000												110,000								165,000	(Wiederwald et al. 2000; Rupf-Haller et al. 2006; Lozza 1996)
UK																														
Aberbeg	3																			3,000									3,000	(TNS & FCW 2005)
Aberdulais Falls	0.05																								20,267	25,705	25,962		23,978	(BR 2008)
Aberglasney Gardens	0.08																								44,625		32,683		38,654	(BR 2008)
Achany	8																					4,000							4,000	(TNS & FCS 2006b)
Achduchil	4																					23,322							23,322	(TNS & FCS 2008)
Alwen/Clocaenog	55																			59,000									59,000	(TNS & FCW 2005)
Ardmore	6																					19,000							19,000	(TNS & FCS 2006b)
Aros Park	2																					20,000							20,000	(TNS & FCS 2006b)
Avon Forest Park in Dorset	1					41,600																							41,600	(Hanley & Spash 1993)
Balcardine East	7																					18,000							18,000	(TNS & FCS 2006b)
Balchers	0.82																			500									500	(TNS & FCS 2006b)
Ballater	2																					3,148							3,148	(TNS & FCS 2008)
Banchory	11																					57,807							57,807	(TNS & FCS 2008)
Barrhill	0.16																			35,000									35,000	(TNS & FCS 2006a)
Beddgelert	8																			37,000									37,000	(TNS & FCW 2005)
Bennachie	23																			224,000									224,000	(TNS & FCS 2006a)
Bentley Community Woodland	0.93																							395,373		989,160			692,266	(Morris & Doick 2009a)
Birches Valley Forest Centre	4																							14,893,208		14,244,270			14,568,739	(Morris & Doick 2010)
Bodnant Garden	0.05																								149,036	190,913	170,929		170,293	(BR 2012)
Braehour	16																					500							500	(TNS & FCS 2006b)
Braelangwell	2																					500							500	(TNS & FCS 2006b)
Brechfa	56																			23,000									23,000	(TNS & FCW 2005)
BRECON BEACONS NATIONAL PARK	1,347																				4,789,000	4,836,000	4,963,000	5,246,000	5,021,300	4,955,300			4,968,433	(STEAM 2010)
Breidden	3																			50,000									50,000	(TNS & FCW 2005)
Brickfield Pond	0.06																										80,000		80,000	(BR 2012

Brownmoor	1																				1,740							1,740	(TNS & FCS 2008)
Bwlch Nant yr Arian	8																		45,000									45,000	(TNS & FCW 2005)
Caerphilly Woods	2																		9,000									9,000	(TNS & FCW 2005)
CAIRNGORMS NATIONAL PARK	3,816																	1,419,650	1,408,280				3,053,000					1,960,310	(STEAM 2009)
Callendar Crags	2																				47,353							47,353	(TNS & FCS 2006b)
Callop	7																		15,000									15,000	(TNS & FCS 2006a)
Carradale Walks	9																		10,000									10,000	(TNS & FCS 2006a)
Carrick	38																		27,000									27,000	(TNS & FCS 2006a)
Carrick Forest Drive	87																		58,000									58,000	(TNS & FCS 2006a)
Castlemaddy Dundeugh	25																		10,000									10,000	(TNS & FCS 2006a)
Chepstow Park	10																		40,000									40,000	(TNS & FCW 2005)
Clashindarroch	57																		14,000									14,000	(TNS & FCS 2006a)
Coed Creigiau	5																		4,000									4,000	(TNS & FCW 2005)
Coed Taf	15																		200,000									200,000	(TNS & FCW 2005)
Colby Woodland Garden	0.02																							28,377	33,785	33,885		32,016	(BR 2012)
Cologin	3																				2,000							2,000	(TNS & FCS 2006b)
Conwy Nature Reserve	0.36																							96,193	89,876	88,996		91,688	(BR 2012)
Corsemalzie	5																		500									500	(TNS & FCS 2006a)
Corwen	7																		6,000									6,000	(TNS & FCW 2005)
Cosmeston Lakes Country Park	1																							210,000	245,000	230,000		228,333	(BR 2012)
Cosmeston Medieval Village	0.02																							18,980	17,828	10,357		15,722	(BR 2012)
Craig Fawr	2																		3,000									3,000	(TNS & FCW 2005)
Craig Phadraig	0.91																				26,513							26,513	(TNS & FCS 2006b)
Criffel	2																				3,932							3,932	(TNS & FCS 2006b)
Culloden Muir	4																				23,619							23,619	(TNS & FCS 2006b)
Culloden Wood	1																				4,092							4,092	(TNS & FCS 2006b)
Cwm Idwal in the Snowdonia National Park	0.83																77,190											77,190	(Christie et al. n.d.)
Cwm Giedd	3																		2,000									2,000	(TNS & FCW 2005)
Cwmcarn	17																		95,000									95,000	(TNS & FCW 2005

DARTMOOR NATIONAL PARK	955																3,800,000							3,013,000					3,406,500	(STEAM 2009)
Delgaty	2																			18,000									18,000	(TNS & FCS 2006a)
Devils Bridge Waterfalls	0.01																								39,155	37,773	35,124		37,351	(BR 2012)
Dinas Mawddwy	5																			17,000									17,000	(TNS & FCW 2005
Dingle Local Nature Reserve	0.11																									54,950	51,238		53,094	(BR 2012)
Dorset Heathland	91																				5,000,000								5,000,000	(Liley et al. 2006)
Dunbennan	1																			5,000									5,000	(TNS & FCS 2006a)
Dunnyduff	0.37																			4,000									4,000	(TNS & FCS 2006a)
Duror	18																				9,000								9,000	(TNS & FCS 2006b)
Durris	17																					18,366							18,366	(TNS & FCS 2008)
Dyffryn Gardens	0.21																								49,398	56,753	36,032		47,394	(BR 2012)
EXMOOR NATIONAL PARK	690																			1,970,000	2,000,000	2,130,000	2,080,000	1,990,800					2,032,467	(ENPA 2009)
Flat Holm	0.26																										1,835		1,835	(BR 2012)
Garlty	13																			21,000									21,000	(TNS & FCS 2006a)
Garw	10																			2,000									2,000	(TNS & FCW 2005)
Gelli Aur Country Park	0.20																									130,284	148,029		139,157	(BR 2012)
Gelliwion/Gellieion	2																			67,000									67,000	(TNS & FCW 2005)
Glansevern Hall Gardens	0.25																								3,487		4,344		3,916	(BR 2012)
Glen Creran	27																				1,000								1,000	(TNS & FCS 2006b)
Glen Lochy South	3																				20,000								20,000	(TNS & FCS 2006b)
Glen Loy	9																			26,000									26,000	(TNS & FCS 2006a)
Glenmore Forest Park	26																						68,380						68,380	(Hill & Courtney 2006)
Glentress	11																			385,000	190,000	55,372							210,124	(Hill & Courtney 2006; Christie et al. n.d.; TNS & FCS n.d.)
Gnoll Estate	0.15																								170,051		163,195		166,623	(BR 2012)
Golitcarg Rumster	14																				3,000								3,000	(TNS & FCS 2006b)
Gunns Wood	0.69																				2,000								2,000	(TNS & FCS 2006b)
Gwern Ddu Cefn Mabli	4																			17,000									17,000	(TNS & FCW 2005)

Gwydyr North Central	28																		27,000									27,000	(TNS & FCW 2005)
Gwydyr South	32																		4,000									4,000	(TNS & FCW 2005)
Hafren	35																		19,000									19,000	(TNS & FCW 2005)
Harriets	3																				500							500	(TNS & FCS 2006b)
Heathall	1																				17,993							17,993	(TNS & FCS 2008)
Hensol	3																		14,000									14,000	(TNS & FCW 2005)
Ingrebourne Hill Community Woodland	0.47																						1,792,226		2,394,675			2,093,451	(Morris & Doick 2009b)
Inverarden and Benmore	14																				4,000							4,000	(TNS & FCS 2006b)
Inverawe	4																				1,000							1,000	(TNS & FCS 2006b)
Inverliever Collaig	3																		1,000									1,000	(TNS & FCS 2006a)
Kilmichael	89																		43,000									43,000	(TNS & FCS 2006a)
Kilsture	2																		20,000									20,000	(TNS & FCS 2006a)
Kirkhill	5																				86,123							86,123	(TNS & FCS 2008)
Knapdale Crinan	66																		4,000									4,000	(Hill & Courtney 2006)
Knockbain	1																		500									500	(TNS & FCS 2006a)
Ladyurd	3																		13,000									13,000	(TNS & FCS 2006a)
Laiken	2																				2,098							2,098	(TNS & FCS 2008)
LAKE DISTRICT NATIONAL PARK	2,284									12,000,000							22,000,000					23,100,000							19,033,333	(STEAM 2009; Taylor 2004)
Lamington Lamington CCF	9																				12,000							12,000	(TNS & FCS 2006b)
Leanachan	29																		143,000									143,000	(TNS & FCS 2006a)
Learnie	3																				15,981							15,981	(TNS & FCS 2008)
Lethem	42																		3,000									3,000	(TNS & FCS 2006a)
Littleburn Cloutie Well	2																				10,958							10,958	(TNS & FCS 2008)
Littlemill	1																				3,501							3,501	(TNS & FCS 2008)
Llangwyfan	0.42																		46,000									46,000	(TNS & FCW 2005)
LOCH LOMOND & THE TROSSACHS NATIONAL PARK	1,863																								7,000,000				(MC et al. 2011)
Loch Sunart East	2																		17,000									17,000	(TNS & FCS 2006a)
Loch Sunart West	5																		7,000									7,000	(TNS & FCS 2006a)
Lochaline	22																				9,000							9,000	(TNS & FCS 2006b)

Lossie	8																			10,000								10,000	(TNS & FCS 2006b)
Lundy Island	5																									17,000		17,000	(Chae et al. 2012)
Lussa Forest	81																		1,000									1,000	(TNS & FCS 2006a)
Lynford Stag	5											42,010																42,010	(Bateman et al. 1998)
Mabie	11																199,267				123,793							161,530	(TNS & FCS 2008; Bateman et al. 1998)
Meadshaw	44																		8,000									8,000	(TNS & FCS 2006a)
Mill Wood	1																		4,000									4,000	(TNS & FCW 2005)
Moel Famau	6																		134,000	32,695						140,000		102,232	(Hill & Courtney 2006)
Monaughty	3																			36,000								36,000	(TNS & FCS 2006b)
Morangie	16																			19,000								19,000	(TNS & FCS 2006b)
Mynydd Dinas	1																			2,000								2,000	(TNS & FCW 2005)
Mynydd Du	11																			21,000								21,000	(TNS & FCW 2005)
Nercwys	1																			50,000								50,000	(TNS & FCW 2005)
NEW FOREST NATIONAL PARK	568						7,000,000												13,345,400		7,150,000					3,000,000		11,193,636	(Gallagher et al. 2007; Sharp et al. 2008; STEAM 2009)
Newborough	10																		296,000									296,000	(Hill & Courtney 2006)
Newcastleton	28																		26,000									26,000	(TNS & FCS 2006a)
NORFOLK AND SUFFOLK BROADS	305															5,400,000							11,300,000					8,350,000	(STEAM 2009; Sharp et al. 2008)
NORTH YORKSHIRE MOORS NATIONAL PARK	1,435															8,000,000							10,700,000					9,350,000	(STEAM 2009; Sharp et al. 2008)
NORTHUMBERLAND NATIONAL PARK	1,054															1,500,000							1,700,000					1,600,000	(STEAM 2009; Sharp et al. 2008)
Ordiequish	7																			8,000								8,000	(TNS & FCS 2006b)
PEAK DISTRICT NATIONAL PARK	1,435															19,000,000							10,389,000					14,694,500	(STEAM 2009; Sharp et al. 2008)
Pembrey	10																		25,000									25,000	(TNS & FCW 2005)
PEMBROKESHIRE COAST NATIONAL PARK	610																						12,868,000					12,868,000	(STEAM 2009)
Queen Elizabeth Forest Park	173		94,359																	50,383								79,700	(Hanley 1989)
Queen Elizabeth Forest Park Forest Drive	4		6,978																									6,978	Hanley 1989)
Quinish	7																			1,000								1,000	(TNS & FCS 2006b)

Radnor	13																		4,000									4,000	(TNS & FCW 2005)
Rascarrel	2																				26,393								26,393	(TNS & FCS 2008)
Rivox	31																				1,318								1,318	(TNS & FCS 2008)
Rogart	3																			500									500	(TNS & FCS 2006b)
Rosal	32																			500									500	(TNS & FCS 2006b)
Roseisle	0.07																			35,000									35,000	(TNS & FCS 2006b)
Rothiemurchus	26																			250,000									250,000	(Christie et al. n.d.)
Scootmore	5																			500									500	(TNS & FCS 2006b)
Shankend	43																		6,000										6,000	(TNS & FCS 2006a)
Shira Achnatra	29																		500										500	(TNS & FCS 2006a)
Sillyearn	3																		500										500	(TNS & FCS 2006a)
Skelbo	1																			11,000									11,000	(TNS & FCS 2006b)
Skipwith Common	3			15,264																									15,264	(Willis & Benson 1988)
SNOWDONIA NATIONAL PARK	2,143																						10,390,000						10,390,000	(STEAM 2009)
Soflen Soiflen	2																		500										500	(TNS & FCW 2005)
South Downs	1,653																	39,000,000											39,000,000	(STEAM 2009)
South Laggan	17																		18,000										18,000	(TNS & FCS 2006a)
St Pierre Coppice Barnets	3																		42,000										42,000	(TNS & FCW 2005)
Strathconon	5																				1,786								1,786	(TNS & FCS 2008)
Strome	16																				258								258	(TNS & FCS 2008)
Swinnie	2																		5,000										5,000	(TNS & FCS 2006a)
Tarenig Myherin	23																		4,000										4,000	(TNS & FCW 2005)
Teindland	2																			3,000									3,000	(TNS & FCS 2006b)
Theford	120						168,997																						168,997	(Jones et al. 2003)
Thornielee	1																		24,000										24,000	(TNS & FCS 2006a)
Torinturk	15																		2,000										2,000	(TNS & FCS 2006a)
Torrs Warren	2																		37,000										37,000	(TNS & FCS 2006a)
Trecastle	3																		7,000										7,000	(TNS & FCW 2005)
Trecwn	3																		500										500	(TNS & FCW 2005)

Upper Teesdale	145	9,094																											9,094	(Willis & Benson 1988)
Woodend Ord Hill	2																					25,917							25,917	(TNS & FCS 2008)
YORKSHIRE DALES NATIONAL PARK	1,768															9,000,000								12,600,000					10,800,000	(STEAM 2009)
Yr Allt Battle Hill	1																		1,000										1,000	(TNS & FCS 2008)
Ystwyth	9																		5,000										5,000	(TNS & FCW 2005)

4 Mapping Recreational Visits and Values of European National Parks by Combining Statistical Modelling and Unit Value Transfer

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Keywords:

Ecosystem service modelling
Recreational demand modelling
Ecosystem service mapping
Ecosystem service valuation
Recreational visitor numbers
Protected area

Abstract:

Recreation is a major ecosystem service and an important co-benefit of nature conservation. The recreational value of national parks (NPs) can be a strong argument in favour of allocating resources for preserving and creating NPs worldwide. Managing NPs to optimize recreational services can therefore indirectly contribute to nature conservation and biodiversity protection. Understanding the drivers of recreational use of national parks is crucial.

In this study we use a combination of primary data on annual visitor counts for 205 European NPs, GIS and statistical regression techniques to analyse how characteristics of NPs and their surroundings influence total annual recreational visitor numbers. The statistical model can be used for land-use planning by assessing the impact of alternative conservation scenarios on recreational use in NPs. The recreational use of new NPs can be estimated ex-ante, thereby aiding the optimisation of their location and design.

We apply the model to: (1) map recreational visits to potential new NPs across Europe in order to identify best NP location; (2) map recreational visits to a proposed new NP in the west of Germany in order to estimate monetary values and to show how visits are distributed across the site; and (3) predict annual visits to all NPs of 26 European countries. Total annual visits amount to more than 2 billion annually. Assuming a mean value per visit derived from 244 primary value estimates indicates that these visits result in a consumer surplus of approximately € 14.5 billion annually.

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4.1 Introduction

NPs are protected areas (PA) for the conservation of extraordinary landscape and wildlife for posterity and as a symbol of national pride. NPs contribute to stop the loss of biodiversity, maintain the naturalness and beauty of our landscape and the supply of ecosystem services. Thereby, NPs contribute to achieve the targets defined in EU Biodiversity Strategy 2020, such as "*halting the loss of biodiversity and the degradation of ecosystem services*" (EC 2011b), and the Aichi targets, such as "*to improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity*" (CBD 2013).

However, financial resources and political support for nature conservation are limited and halting ecosystem degradation remains a great challenge. In the past, major policy goals on biodiversity protection have typically not been met, such as those set by Convention on Biological Diversity, ratified after the global summit in Rio de Janeiro (1992) (Barbault 2011; Leadley *et al.* 2010). And still, the future outlook reveals that biodiversity remains under threat and substantial action needs to be undertaken (SCBD 2014).

One major co-benefit of nature conservation is the supply of recreational opportunities. NPs provide opportunities of visiting, experiencing, enjoying and learning about nature and biodiversity, and thus contribute to human well-being and environmental awareness. Nature recreation and tourism present a great economic value and an opportunity for rural economic development by generating income and employment through visitors' expenditures. The value of nature recreation and its economic opportunities can be used as a strong argument in favour of allocating financial resources towards nature conservation at different spatial scales (Balmford *et al.* 2015b).

Nature conservation should not only focus on biodiversity and habitat protection, but should also take recreational co-benefits into account. Efficient land-use planning needs to consider all ecosystem services supplied. For allocating resources for nature conservation, it can be important to know how recreational co-benefits of nature conservation can be optimized. The most important indicator of the contribution of recreation to the local economy is the number of visitors (Bateman *et al.* 2006a; Jones *et al.* 2003). Therefore, understanding the drivers that determine the number of visitors to PAs is crucial for PA management and for PA designation.

The aim of this study is to analyse the effects of NP characteristics and their spatial context on total annual visits that are considered the main determinant of recreational economic value (Bateman *et al.* 2006). To this end, we develop regression models of visitor numbers using primary data for European NPs combined with additional spatial variables derived from GIS data. The estimated models give insights into the drivers of recreational use within European NPs and thus allow the prediction of visitor numbers for designated new NPs and alternative management scenarios. Similar to the study of Balmford *et al.* (2015), we combine our predicted visitor numbers with a mean value estimate per recreational visit, but derived from a much larger set of primary valuation studies. Thereby, the relative importance of recreational services is highlighted as compared to other ecosystem services and man-made goods.

Several studies have modelled visitor numbers of PAs or nature areas based on spatial variables. One widely applied approach is to use choice models to predict recreational behaviour at the individual level. Typically, such studies use survey data containing information on the origin and destination of an individual recreational trip. However, such data sets are time-consuming to develop and are usually only available for relatively small areas (Pouta and Ovaskainen 2006; Bateman *et al.* 2011; Hausman *et al.* 1995; Jones *et al.* 2010; Loomis 1995; Feather *et al.* 1995; Parsons and Hauber 1998; Sen *et al.* 2013; Shaw and Ozog 1999; Termansen *et al.* 2008). The purpose of the present study is to investigate the determinants of recreational use of NPs at a European scale and therefore we use data from visitor monitoring studies for NPs across Europe. Some existing studies have used similar approaches in order to investigate drivers of recreational park visits. For example, Neuvonen *et al.* (2010) analyse effects of park characteristics on visitation rates to 35 Finnish NP. Mills and Westover (1987) model the visitation rates for 121 Californian State Parks using four predictors representing park characteristics and the distance to the nearest population agglomeration. Hanink and White (1999) model recreational demand for 36 US NPs using age and size as variables for describing the park, its distance and the population of the closest metropolitan area, as well as substitute availability as context characteristics. Hanink and Stutts (2002) model the demand for 19 recreational battlefields in the US. They use a substitute availability indicator weighted by individual substitute's characteristics. Loomis *et al.* (1999a) find a significant effect of GDP per capita and of availability of wilderness on the number of recreational trips to wilderness areas per capita in the US. Ejstrud (2006) use a number of GIS indicators for modelling visitor frequency to 10 Danish open-air museums using six predictor variables, but do not report whether they show significant effects. The only study using international visitor data is from Balmford *et al.* (2015), which uses visitor data of PAs worldwide. Their study uses only a limited number of relatively simple predictor variables and finds few significant effects. Their model may be appropriate to assess overall trends in PA visitation rates, but may have few site specific implications. Loomis (2004) uses regression techniques to estimate the effect of elk and bison populations on visitation rates in Grand Teton NP, US, using explanatory variables on how the park changes over time, but does not compare effects of alternative sites' characteristics.

All expect one of the above mentioned studies use national data only for their statistical analysis. Thereby, the number of primary observations is in general relatively low. The purpose of the present study is to investigate drivers of recreational use for NPs Europe-wide and therefore, use visitor data from NPs in 21 European countries comprising 205 case study areas in total. Consequently, we can include more predictors in our initial model and try to estimate a more robust model. For example, national study areas are relatively small and therefore climatic conditions are often too similar to be considered as a predictor in a recreational demand model. Furthermore, we use more refined site and context characteristics as predictors in our model, which are computed and extracted from Europe-wide GIS data layers. As all our predictors are derived from large scale GIS data layers, the final model can easily be used to make predictions of visitors' frequency for any potential NP in Europe. Thus, recreational use can be mapped for any location in Europe without the need for an additional collection of information on the predictor values. Our spatial assessment can thereby be used for ecosystem service mapping as required by the EU Biodiversity Strategy 2020, improving resource allocation and calculating a green GDP (Maes *et al.* 2012a; UN 2014). Finally, we use a number of different statistical regression techniques to deal with spatial autocorrelation (SAC) for a more in-depth identification of the spatial dimension of recreational use.

This paper is organized as follows: in section two we describe the data we use, first the primary data of visitor monitoring studies and second the predictors used in our models. In section three we explain the statistical regression techniques applied and present the estimated visitor models. The results are discussed in section five. Finally, we conclude.

4.2 Data

4.2.1 Primary Data

Our primary data are 205 total annual visitor estimates to European NPs and 245 estimates of monetary values per recreational visit for 147 separate nature areas in Europe. We collected the data through internet searches, review of relevant literature and by contacting researchers involved in this field, NP administrations and relevant governmental bodies in all EU countries. The data is described more in detail in Schägner *et al.* (submitted).

For the visitor data to be included, we required as a minimum quality criteria that the total annual visitor estimates are based on some form of on-site visitor monitoring, which is then scaled up to the entire area and the entire year. In order to check whether the quality criteria is met, we analysed the relevant publications on the visitor monitoring programs. In cases in which the information was not available or not accessible due to language barriers, we contacted the authors and relevant institutions. In total we could obtain annual visitor observations for 205 separate case study areas within Europe, which are either an entire NP or a subsection of a NP (see Figure 19). All collected data were attached as attributes to a spatial layer in vector format, containing the boundaries of NPs or of their surveyed part. We obtained NP polygons from World Data Base of Protected Areas (WDPA) and the Common Database on Designated Areas (CDDA) (IUCN and UNEP 2015; EEA 2013) and from national agencies. If case study areas differed from the available polygons, we tried to obtain polygons from the authors of the studies, the park management or other stakeholders. In some cases we manually draw polygons with ArcGIS, based on information available in the case study publications or supplied by the authors. If multiple observations of visitor numbers are available for the same study area, we used the average.

NP and case study area characteristics differ widely in terms of size, location, visitation rate and ecosystem characteristics. The smallest case study area is a nine hectare beach within the Wadden Sea NP in Germany, whereas the largest case study area is the Cairngorms NP in Scotland comprising 3,816 km². Most of the case study areas in our database are located in Northern Europe. For the Southern Europe we could obtain visitor numbers for all Spanish, most Italian and French NP. For our statistical analysis we divided the total annual visitor numbers by the total terrestrial area of the single study areas[13] and thereby obtained total annual visitor densities per ha as our dependent variable in our models. Visitor numbers range from 0.03 visitors/ha/year in the large Sarek NP in northern Sweden up to 56,680 visitors/ha/year on a small beach within the Wadden Sea NP. The total median and mean is 13 and 368 with standard deviation of 3,962 visitors/ha/year, indicating a skewed distribution with a tail of very high visitation rates. The mean relative deviation is about 167%. For more information on the primary data, it can be accessed via the ESP Visualisation Tool (Drakou *et al.* 2015).

For our statistical analysis we divided the total annual visitor numbers by the total terrestrial ha size of the single study areas and thereby obtained total annual visitor densities per ha as our dependent variable in our models, which is common within species distribution modelling.

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[13] We used the terrestrial area not including area covered with water because some NP — in particular marine NP — comprise mainly of water. Including the area of water would bias our analysis since this area is hardly visited.

The valuation studies use either Travel Cost Method (TCM) (57%) or Contingent Valuation Method (CVM) (43%). For the valuation studies, we transfer all value estimates to Euro 2013 price level using purchasing power parity and country specific inflation data. We exclude one outlier with an extreme deviation of 60 times the mean value. The remaining value estimates range from € 0.16 to € 64.7 per visit with a mean of € 7.17, a median of € 2.8, a standard deviation of 11 and a mean relative deviation of 95%. Most study sites are located in Western Europe (51%). The UK has the highest number of observations (81), followed by Italy (32), Ireland (28), Finland (27) and Germany (22).

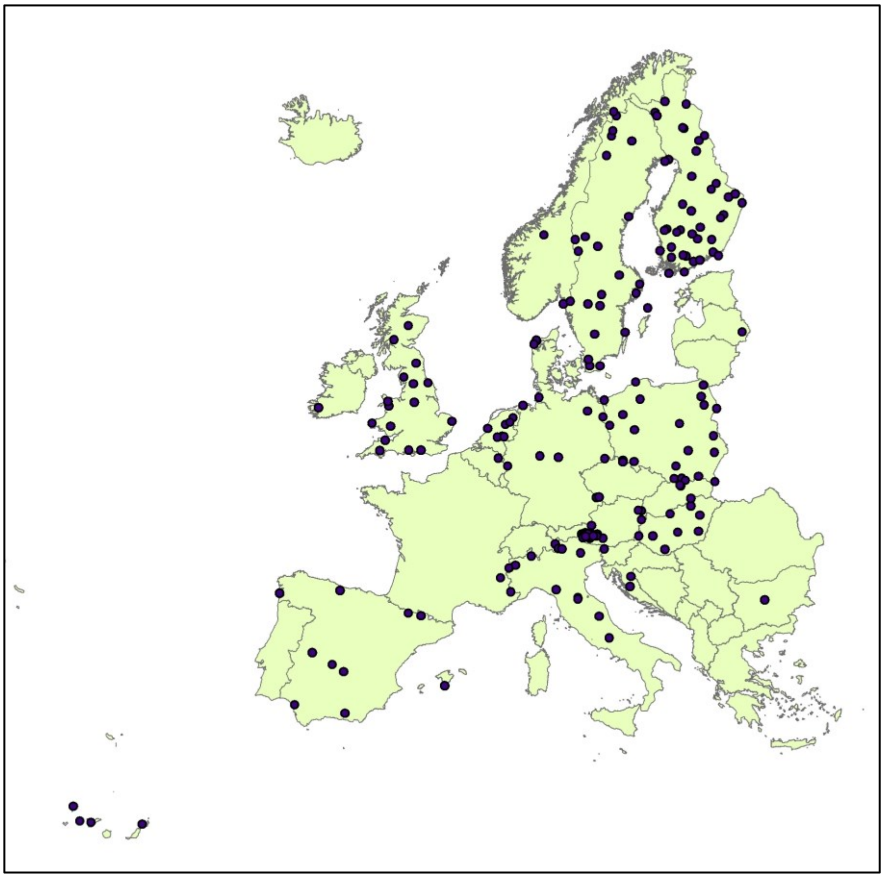

Figure 19: Location of visitor counts across Europe.

4.2.2 Explanatory Variables

Explanatory variables used to model visitation rates can be divided into three categories: (1) site characteristics, which describe the NP itself, (2) context characteristics, which describe the spatial context of the NP and (3) study characteristics, which describe the methodology of primary data collection. The selection of variables was based on a review of the literature on recreational demand modelling and environmental recreational value transfer studies. However, limitations in the availability of comprehensive and consistent Europe-wide data sets and in the information provided in visitor monitoring publications restricted our choice of predictors. A complete list of all predictors used in our analysis is presented in Table 5 in this chapter. Detailed description is presented in the following sections. Each variable is available in geospatial raster format, therefore site and context characteristics for each site could be easily calculated in a GIS environment. We extracted mean values of all predictor variable for each case study area using an automated model built in ArcGIS, including the use of the zonal statistics tool (ArcGIS 10.1). The raster layers of the predictors were either taken from available GIS data sets or we computed them by reprocessing or combining existing data sets using ArcGIS (ArcGIS 10.1). Then we conducted an exploration of our data following the recommendations of Zuur *et al.* (2010) in order to gain initial insights into distributions and dependencies. For some predictors we used logarithmic or square root transformations either because they showed a relatively skewed distribution or because we wanted to approximately linearize an expected non-linear relationship. We tested all our predictors for multicollinearity, but could not identify anything of concern.

4.2.2.1 Site Characteristics

The following **site characteristics** are used to model visitation rates: **(1) Share of land cover/use:** We used the CORINE[14] land cover/use data set (EEA 2006) to determine the shares of different land cover/use classes and aggregates of single land cover/use classes for each NP. In particular we focused on natural vegetation cover. We do not, however, have strong prior expectations regarding the signs of these land cover predictors. In general, one may assume that natural vegetation supports nature recreation. However, NPs typically offer plentiful natural vegetation and therefore additional natural vegetation of any kind may not necessarily attract additional visitors. Our analysis of the different land covers has an exploratory character and does not aim to test specific hypotheses. The separate classes and aggregated areas are presented in Table 5 within this chapter. **(2) Water bodies:** We computed a 300 m resolution grid of the share of surface area covered with rivers, lakes or ocean using the Euro Regional Map as input data set (EG 2010). Then we applied a kernel density function tool (ArcGIS 10.1) to compute the amount of surface covered with water within a 3 km radius of each pixel. The density function allows water area that is further away to be weighted less than water nearby and thereby incorporates a distance decay effect. The presence of water bodies in a NP are expected to have a positive impact on recreational use (Termansen *et al.* 2008).

We expect that more diverse landscapes are perceived as more beautiful (Dramstad and Sundli Tveit 2006) and thereby attract more visitors. Based on the basic economic principle of decreasing marginal utility and rates of substitution, diversity tends to be rated higher than uniformity (Mankiw 2001). In order to account for landscape diversity we computed three different indicators. **(3) Three dimensionality:** We computed the area visible from each pixel within a 30 km radius using the view shed tool (ArcGIS 10.1) and a 1000 m resolution digital elevation map from the European Environmental Agency (EEA 2015a). We believe that visitors prefer three-dimensional landscapes offering great views. **(4) Land use/cover diversity:** Based on the CORINE land use/cover data set we computed the Simpson Diversity Index (Magurran 1988) of land use/cover within a 3 km radius for each pixel of the CORINE map. In their study Neuvonen *et al.* (2010a) use the number of biotopes as a diversity indicator and find a significant positive effect on visitation frequency in Finnish NPs. However, the number of biotopes may be positively correlated with the study area size. Therefore, this predictor may pick up part of the size effect. Furthermore, larger NPs may have more biotopes even if their landscape is not more diverse. **(5) Forest edges:** Using the Joint Research Centre forest cover map (EC 2006), we computed the number of forest pixels (25m resolution) that are not classified as forest core. We consider these forest pixels as the transition area between forest and other land use/cover and therefore, as a major visible change in the ecosystem type (EC 2006). **(6) Temperature:** We applied a data set from Biavetti *et al.* (2014) indicating the number of days with maximum temperature above five degrees Celsius. Due to the predominance of southbound tourism fluxes in Europe, we expect temperature to have a positive effect on visitation rates. **(7) Regions:** Sites were further classified according to their membership of bio-geographical and geographical regions. We do

[14] CORINE refers to COoRdination of INformation on the Environment.

not have expectations regarding the signs of these factor variables, but might discover some cultural effects. **(8) Trail density:** We used trail density as proxy for overall recreational facilities, which may attract visitors. From the OSM (Open Street Map) data set (OSM 2012), we extracted all vector elements that can be classified as non-motorized traffic infrastructure. We used five OSM classes: trails, foot paths, bike paths, bridle paths and steps. On a 100 m resolution we applied the line density tool (ArcGIS 10.1) to compute an indicator for trail availability. Again, trails that are further away from a pixel were weighted less than trails close by. Other studies found significant positive impacts of trails (Neuvonen *et al.* 2010a) or recreational facilities in general (Mills and Westover 1987), but they used individual park data and no comprehensive large scale GIS data sets. **(9) Street density:** Similar to trail density we computed an indicator for street availability for all minor roads (Tele Road Atlas road classes 4-6) based on the Tele Road Atlas data set (TS 2006). Roads are an important infrastructure for accessing remote locations and thereby are expected to increase visitor numbers. However, if roads are too abundant, they may negatively affect the quality perception of nature recreation in a NP and thus, deter visitors. **(10) Study area size:** We expect that area size has a negative impact on the mean number of visitors per ha because of two reasons: First, larger study areas act as a substitute in itself, because visitors can be distributed across a larger area. Second, visitor counting tends to result in lower mean visitor numbers for lager areas. If a visitor hikes through a large study area, he is counted once. If the same study area is split into separate study areas, the same visitor may eventually be counted several times. Most existing studies of NP visits use total visitor numbers as the dependent variable and therefore find a positive influence of study area size on visitor numbers (Hanink and Stutts 2002; Hanink and White 1999; Mills and Westover 1987). However, by working with linear models they potentially miss out that visitor numbers do not increase in direct unitary proportion to the size of the study area. **(11) Age of NP:** Finally, we characterized each NP by its age (number of years since foundation until 2015). Existing studies have found a positive correlation between park age and visitor numbers (Neuvonen et al. 2010; Mills and Westover 1987; Hanink and Stutts 2002; Hanink and White 1999). This may be caused by the general tendency that the most attractive locations were designated as PAs earlier or that older NPs have had more time to establish recreational facilities. The designation of a NP may create an advertisement effect and establish a good reputation increasing the parks popularity over time. **(12) Biodiversity:** In this case we used the total number of red list species encountered in a study area as an indicator for biodiversity (IUCN 2013).

4.2.2.2 Context Characteristics

As **context characteristics** we used the following variables: **(1) Accessibility:** We expect that the number of people that can access a certain location within a certain time is likely to have a positive effect the visitation rate. We define this variable as the total population living within a 50 km radius around the site, using population data from Batista e Silva *et al.* (2013). In order to account for distance decay, we applied a Gaussian weight function, which causes that the population that is further away from the NP is weighted less than the population nearby. The weight function was calculated so that 95% of its integral was located within the 50 km radius. Other studies find significant positive effects of accessibility on visitor numbers. They use for example distance to nearest towns (Mills and Westover 1987) or consider the population of metropolitan areas (Hanink and Stutts 2002; Hanink and White 1999) and do not include distance decay effects (Neuvonen *et al.* 2010a). **(2) NP substitutes:** We computed a raster in which each pixel is the sum of areas classified as NP within 130 km radius. The Europe-wide NP data set was a combination of sites from the WDPA and CDDA data base. In order to account for distance decay, we used the same methodology as for population. As a result, large NPs and NPs with small distance from each other have a relatively high availability of substitutes. Other studies have found negative influences of substitute availability on visitor numbers.

They use for example distances to competing recreational sites (Hanink and White 1999; Hanink and Stutts 2002) or the number of parks within a certain distance (Neuvonen *et al.* 2010a). They do not, however, account for the size of substitute areas. **(3)** Finally, we introduce **GDP per capita** as a proxy of visitor income, which we extracted from the Eurostat database (EC 2013). We took the mean values of the last ten years (as far as available) and the highest data resolution available, which is either NUTS2 or NUTS3 level[15]. We expect that visitation rates are likely to be higher in locations with higher per capita GDP. Existing studies have observed that people engaging in nature recreation have above average incomes (Loomis *et al.* 1999a).

4.2.2.3 Study Characteristics

Initially, we considered collecting detailed information on study characteristics describing the methodology of the visitor monitoring procedure for each case study area. In that way, we hoped to identify the influence of different visitor monitoring techniques on the final total annual visitor estimate. Similar attempts have been successfully implemented in meta-analysis studies of environmental economic valuation studies (Brouwer *et al.* 1999; Zandersen and Tol 2009). However, we encountered difficulties in coding such methodological study characteristics due to the language and incomplete reporting in the underlying case study publications. Therefore, we only introduce two study characteristics as predictors in our analysis: **(1)** the **year of the visitor monitoring survey** for which we used the mean values of the years in which visitor monitoring took place. **(2)** Furthermore, we classified all visitor monitoring studies according to different levels of primary **data collection quality** from one for the lowest and ten for the highest quality. The quality judgment represents a composite indicator of different quality dimensions: the type of publication (scientific vs. grey literature), the visitor monitoring study purpose (scientific vs. political), the institution conducting the study (academic, NP management, others), the methodological documentation of study (full, incomplete, none). If the documentation of the study was available, we assessed the quality of methodologies based on details such as the temporal and spatial counting resolution, manual or electronic counting devices and the temporal and spatial up-scaling methodology. Finally, a very important aspect for the visitor monitoring studies quality is the description of the study area. Some publications do not supply maps and only rough descriptions of the study area. If the area of the study area is uncertain, then the number of visitors per hectare is uncertain as well.

[15] NUTS refers to Nomenclature of Territorial Units for Statistics.

Table 5: List of Predictors used in the Models.

Type	Variables	Explanation*	Mean / Standard Deviation
Site Characteristics:	Sqrt (grassland)	Share of grasslands cover of the study area (100 m resolution raster)	0.2 / 0.24
	Sqrt (wetland)	Share of wetlands cover of the study area (100 m resolution raster)	0.14 / 0.23
	Sqrt (water)	Share of water bodies of the study area (300 m resolution raster)	0.23 / 0.26
	Log (broadleaf)	Share of broadleaf forest of the study area (100 m resolution raster)	0.73 / 0.86
	Conifer	Share of conifer forest of the study area (100 m resolution raster)	4.44 / 4.63
	Log (forest edge)	Transition area between forest and other land use/cover (25 m resolution raster)	0.83 / 0.4
	Sqrt (land cover diversity)	Simpson Diversity Index of Corine land use/cover within a 3 km radius (100 m resolution raster)	1.61 / 0.22
	Log (viewshed)	Area visible from each location within in a 30 km radius (1 km resolution raster)	5.43 / 0.69
	Log (red list species)	Total number of red list species found in study area	2.65 / 0.84
	Temperature	Total number of days with maximum temperature above 5 degrees Celsius (10 km resolution raster)	256 / 57.5
	NP age	Years since NP foundation until 2015	40.6 / 26.94
	Log (trails)	Trail density using density function in order to account for distance decay effect	5.69 / 1.87
	Log (roads)	Density of minor roads using density function in order to account for distance decay effect (100 m resolution raster)	0.9 / 0.83
	Study area km²	Size of the study area in km²	352 / 621
Context Characteristics:	Log (NP substitutes)	Area of NP within 130 km radius of the study area using a Gaussian weight function in order to account for distance decay (1 km resolution raster)	11.35 / 1.5
	Log (Population 50 km²)	Population living within 50 km radius of the study area using a Gaussian weight function in order to account for distance decay (100 m resolution raster)	12.88 / 1.75
	GDP/ capita	GDP/ capita in the NUTS 2 or 3 region in which the study area is located	21,856 / 7,713
Study Characteristics:	Survey year	Year of visitor monitoring survey	2005.6 / 4.16
	Survey quality	Quality of the visitor monitoring survey methodology and study area definition	7.17 / 1.53

* For all predictors mean values per study area were computed.

4.3 Methodology

We applied a number or regression techniques in order to model the total annual visits per ha to European NPs using the above described predictors. All models were estimated using the open source statistical software R. We started our analysis with a simple linear regression, but it showed a strong spread of the residuals for larger fitted values and therefore a violation of the homogeneity assumption. We tried to control this effect by introducing a number of different variance structures, but were not successful in eliminating the heterogeneity to an acceptable degree.

As our dependent variable is a count, we continued our analysis with generalized linear models using a Poisson and a negative binomial distribution (using R-package glmmADMB, MASS, lme4, nlme and gamlss (Bates *et al.* 2015; Bolker *et al.* 2012; Pinheiro *et al.* 2015; Ripley *et al.* 2015; Stasinopoulos *et al.* 2015), which are typical distributions of count data (O'Hara and Kotze 2010). However, model results show spatial residual patterns similar to the one displayed in Figure 20. The negative (grey bubbles) and positive residuals (black bubbles) are clustered, which is a violation of the independence assumption of general linear regression analysis. In order to overcome this problem we added a spatial residual structure, either by a spatial random effect or a SAC, but we ran into numerical conversion problems of the optimization algorithm trying to solve the complex statistical model. We therefore abandoned this approach and do not present the interim results of these attempts in the following.

Because our count data shows relatively large values (mean value 367), log transformation is an alternative approach, which should have a negligible effect on the parameter estimates but decreases the model processing complexity substantially (O'Hara and Kotze 2010). We therefore continued our analysis with linear log transformed model of the following form:

$$\log(V_i) = \alpha + \beta * X_i + \mu_i \qquad \text{where} \qquad \mu_i \sim N(0, \sigma^2)$$

V stands for the dependent variable (in our case the total annual visits per ha), α is a constant, β represents a vector of parameters, X is a vector of explanatory variables and μ is the residual, which is normally distributed with mean of zero and variance σ. Again, we had to deal with spatial residual patterns, which we tried to control for using a spatial random effect in a mixed model[16] and by a residual SAC structure. We tried a number of different random intercepts and random slopes in the mixed model and also a number of SAC structures[17]. We investigated all estimated models on how successful they are in controlling for the spatial residual patterns and on their AIC and BIC scores[18] (as criteria for model selection). The best model contained a spatial spherical correlation structure, which models the residuals' correlation across space as a spherical function of distance. The model formula remains the same as before, but this time we assume that the residuals μ_i of different locations are correlated based on the function f and their distance.

$$\operatorname{cor}(\mu_a, \mu_b) = \begin{cases} 1 & if\ a = b \\ f(\mu_a, \mu_b, \rho) & else \end{cases}$$

We used this model as a starting point and conducted stepwise model selection by dropping the least significant predictor until every predictor was significant. We determined starting values for the range (maximum distance of spatial correlation) and the nugget (one minus the correlation of two arbitrarily close observations) of spatial correlation structure based on interpretation of variogram and spatial residual plots in order to improve consistency across the different models. In the following section on results, we present detailed results on our initial log transformed model, the starting model including

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[16] In other disciplines, mixed modelling is also referred as to multilevel analysis, nested data models, hierarchical linear models, and repeated measurements.

[17] For an introduction into mixed modelling we would like refer the reader to (Zuur *et al.* 2009) and for in introduction into spatial autocorrelation to (Bivand *et al.* 2013a).

[18] AIC and BIC refer to the Akaike Information Criterion and the Bayesian Information Criterion.

the spatial spherical correlation structure and on the final model after stepwise model selection. We validated our final model against the assumptions of linear regression analysis. Therefore, we plotted our residual against fitted values and against each predictor. We could not identify any linear or non-linear patterns of concern. To present a comparable measure of the goodness of fit of all models we compute the root mean square deviation (RMSE) and the coefficient of variation of the RMSD (CV RMSE).

We use our final model (1) to make predictions of the total annual visits to all European NPs within the countries covered by our explanatory variable layers, (2) to map the total annual visits to a fictive new 80 km² NP, located anywhere in the European countries covered by our explanatory variable layers and (3) to map the distribution of the predicted total annual visits to a proposed new NP (Teutoburger forest and Senne heathland) in the western part of Germany.

In order to predict the number of visits to NPs of most European countries, we extracted all shape files from the WDPA and the CDDA (UNEP 2015; EEA 2013), which fall into the International Union for Conservation of Nature (IUCN) category II (NP). Furthermore, we accessed national databases to obtain shapes of NPs, which were missing in those two databases. In total we included 449 separate NPs areas. It is to be noted that not all of these sites fall into IUCN category II. No uniform definition of the term NPs exists and it was used long before the IUCN categories system was created. Many existing NPs all over the world are differently managed than demanded by the requirements of category II (IUCN 2008), but are still called NP based on the decision of governments and other local stakeholders. We used vector layer of all NPs boundaries and zonal statistics (ArcGIS 10.2) to drive mean values for the explanatory variables. Predictions were made using the rms R-package (Harrell Jr 2015). In order to improve our predictions and account for unobserved effects on visitation rates, we kriged the residuals of our model across the entire study area using the gstat, GeoR and raster R-packages (Diggle *et al.* 2015; Hijmans *et al.* 2015; Pebesma and Graeler 2015). We then added the result to the prediction of each NP.

For predicting the number of visits of a marginal increase of NP area, we assume a fictively created medium size NP of 80 km². We then created explanatory variable raster layers accounting for the average substitute effect of the new NP and the size of the new NP. The quality of the visitor monitoring methodology, which is one explanatory variable in our model, was set to the highest quality available in our primary data base (9.5). The NP age was set to zero. We then used the model to map the annual number of visits for each 100 m resolution grid cell across Europe, as though it is part of the newly created NP. The mapping was conducted using the raster, gstat and geoR R-packages. Again, we added the kriged residuals to our predictions.

In order to test our visitor mapping procedure in a realistic policy setting, we applied it to a proposed new NP in the western part of Germany (Teutoburger forest and Senne heathland). The area of this proposed NP is approximately 20,000 ha and comprises a forested mountain range and a heathland, which had been used as an army base in the past. It is already largely protected and has been proposed for NP designation (NABU 2015). We again made predictions on 1 ha resolution in order to estimate total visits to the area and show how visitors distribute across the area.

Finally, we combine the predicted number of visits with a monetary value estimate, derived by taking the overall mean value per visit (€ 7.17) from the 244 value estimates described above, which is almost the same value estimate applied in a similar study by Balmford *et al.* (2015) (US$ 7) , but based on much larger primary valuation data base. This approach, so-called *unit value transfer* or *average value transfer,* is a common approach used for value transfer and ecosystem service value mapping

(Balmford *et al.* 2015b; Rosenberger and Loomis 2001; Schägner *et al.* 2013) and a method considered for aggregating ecosystem service values to develop a System of Environmental-Economic Accounting (SEEA) (Costanza *et al.* 1998; UN 2014). It assumes a constant value per recreational visit across space, which is indeed a simplification. However, as the value per recreational visit varies by far less across space than the number of recreational visits (Bateman *et al.* 2006a; Jones *et al.* 2003), its effect on the overall recreational value of an area is relatively small. Given the fact that we focus only on NPs and on an area of relatively similar socioeconomic and cultural characteristics, we consider unit value transfer as a good approximation for the case study presented (see section discussion for further details).

4.4 Results

The results of the NP visitor model using a log-transformed dependent variable are presented in Table 6. 14 of the 19 predictors show statically significant coefficients and the multiple R^2 of 0.68 indicates relatively high explained variance. Most coefficients have the expected sign. However, the residual plots of the model show some spatial patterns, which are to be controlled for. The residual bubble plot in Figure 20 of the spatial distribution of the full model's residuals without spatial correlation structure shows clustering of positive and negative residuals across Europe. We applied a number of different techniques to control for these patterns.

Table 6: National park visitor model. Dependent variable is the log of annual number of visitors per hectare. Spatial patterns in the residuals are not controlled for.

Variable	Coefficient	p-value	
(Intercept)	15.64	0.79	
Sqrt (grassland)	-0.75	0.19	
Sqrt (wetland)	-1.05	3.49E-02	*
Sqrt (water)	1.32	4.50E-03	**
Log (broadleaf)	-0.51	3.70E-03	**
Conifer	-0.04	0.18	
Log (forest edge)	-0.48	0.13	
Sqrt (land cover diversity)	1.47	3.60E-03	**
Log (viewshed)	0.34	3.28E-02	*
Log (red list species)	-0.39	3.71E-02	*
Days > 5 Degrees	6.70E-03	9.40E-03	**
NP age	8.08E-03	3.44E-02	*
Log (trails)	0.47	0.00E+00	***
Log (roads)	0.38	5.00E-03	**
Study area km²	-4.91E-04	7.00E-03	**
Log (NP substitutes)	-0.25	1.13E-02	*
Log (population 50 km)	0.48	0.00E+00	***
GDP/ capita	-3.50E-05	3.02E-02	*
Survey year	-1.12E-02	0.70	
Survey quality	-2.93E-02	0.68	

Multiple R^2: 0.68	RMSE: 1.21	AIC: 796.9
Adjusted R^2: 0.65	CV(RMSD): 0.45	BIC: 864.5

*** $p \leq 0.001$, ** $p \leq 0.01$, * $p \leq 0.05$, . $p \leq 0.1$

First we added different regional factor variables to the model, in order to explain the spatial patterns. We tried bio-geographical regions, geographical regions and countries [19] as factor variables. However, adding one of these variables reduced the degrees of freedom and increased the complexity of the model to such an extent that we ended up with models having a lot of non-significant variables. Also most of the different levels of the regional factor variables did not show any significant effect. In addition, AIC and BIC values did not show any favourable scores for the models.

[19] For the country variable we combined some countries to one region in order to reduce the levels of the factor variable, such as Benelux countries, Alpine countries and Baltic countries.

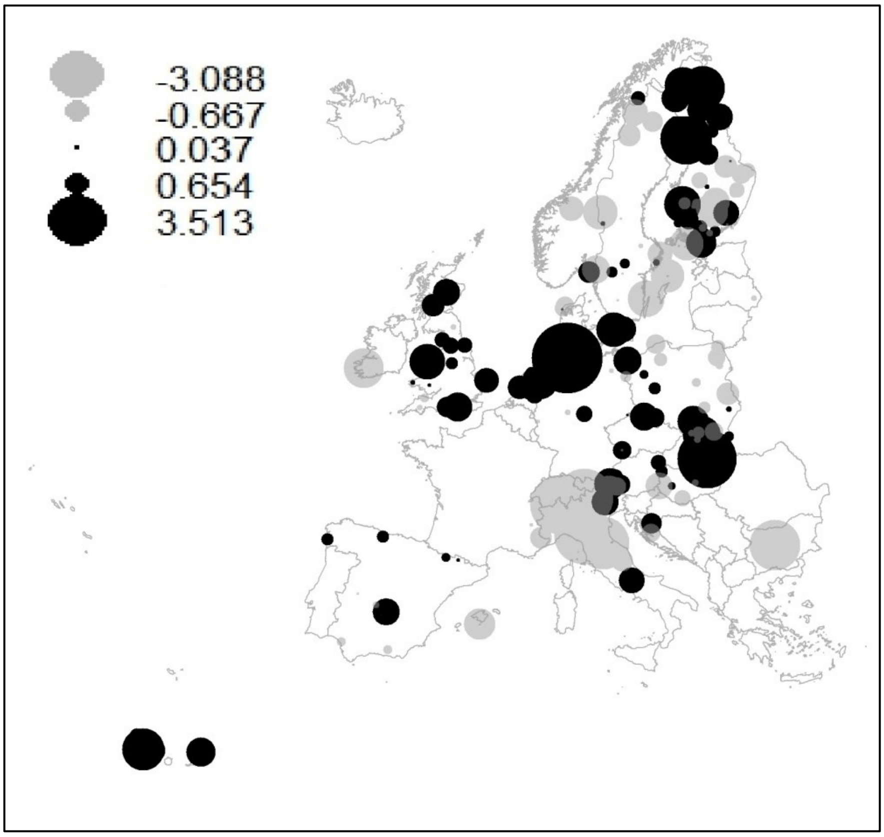

Figure 20: Bubble plot of the spatial distribution of the full model's residual without spatial correlation structure.

Then, we tried to implement a mixed model by adding the regional variables as a random part in order to control for the spatial patterns in the residuals. We tried various combinations of random intercept and random slope models, which significantly improved the model in terms of AIC and BIC values, but a considerable spatial residual pattern still remained. Finally, we tried different SAC structures, which improved the model's AIC and BIC values substantially, beyond all the models we tried before. The best model in terms of AIC and BIC values as well as in controlling for the spatial residual patterns applied is a spherical spatial correlation structure. The result of the full model including the SAC structure is shown Table 7. In total, 13 predictors of the full model show a significant correlation with total annual numbers of visits per ha. After stepwise elimination of the least significant variable until only significant predictors remained (at least at the 0.1 level), we ended up with the same 13 significant predictors as before and substantially low AIC and BIC values (see Table 8).

Our final models show a SAC between single observations up to a range of 530 km for the full model and up to a range of 580 km for the final model. The nugget refers to differences between observations, which can neither be explained by the model nor by the SAC due to measurement errors or micro variability.

Table 7: Full model including spherical spatial correlation structure.

Variable	Coefficient	p-value		Beta coefficient
(Intercept)	-9.30	0.88		2.02%
Sqrt (water)	1.61	3.00E-04	***	7.26%
Sqrt (grassland)	-0.61	0.27		2.53%
Sqrt (wetland)	-0.98	3.77E-02	*	3.98%
Log (broadleaf)	-0.39	2.72E-02	*	5.74%
Conifer	-0.03	0.37		2.31%
Log (forest edge)	-0.55	6.91E-02	.	3.84%
Sqrt (land cover diversity)	1.34	5.50E-03	**	5.02%
Log (viewshed)	0.13	0.40		1.54%
Log (red list species)	-0.24	0.28		3.46%
Days > 5 degrees	7.43E-03	3.59E-02	*	7.40%
NP age	1.07E-02	5.10E-03	**	4.98%
Study area km²	-5.69E-04	3.40E-03	**	6.12%
Log (trails)	0.44	0.00E+00	***	14.24%
Log (roads)	0.50	1.70E-03	**	7.16%
Log (NP substitutes)	-0.30	1.57E-02	*	7.82%
Log (population with 50 km)	0.37	6.00E-04	***	11.19%
GDP/capita	-1.00E-06	0.96		0.13%
Survey year	2.40E-03	0.94		0.17%
Survey quality	-0.12	0.10	.	3.11%

Spherical spatial correlation structure	RMSE: 1.26	AIC: 768.7
Range: 530 km , nugget: 0.40	CV(RMSD): 0.48	BIC: 842.7

*** $p \leq 0.001$, ** $p \leq 0.01$, * $p \leq 0.05$, . $p \leq 0.1$

A strong positive and highly significant influence is shown for the presence of water bodies, both in the full and in the final model. The beta coefficients indicate that it is the fourth most important predictor for explaining recreational use in our models. Interestingly, even though we did not have strong prior expectations regarding the signs of predictors representing the type natural vegetation, all of them— broadleaf and coniferous forest, grassland and wetlands — show negative signs in the full model. Only broadleaf forest and wetlands show a significant effect in the full model as well as in the final model. Also the variable forest edges, contrary to our expectations, shows a negative and significant sign. However, forest edges are strongly correlated with total forest (the sum of broadleaf and coniferous forest). Therefore, forest edges may pick up some of the negative impacts of forest cover on recreational use in our model. Both, broadleaf and coniferous forests have negative signs, even if only broadleaf forest shows a significant effect. We initially thought that we could separate the effect of forests on the numbers of visits from the effect of forest edges by including single predictors for broadleaf and conifer forest. One explanation of the negative signs of the vegetation cover predictors could be that NPs do have natural vegetation to such an extent, that it becomes abundant and thereby, more of it deters visitors. The transformation of the predictors indicates that their negative effect on the number of visits decreases with their increasing share of land cover. Nevertheless, the beta coefficients of the single vegetation-cover predictors indicate that they only have a relative small effect on the total visitation rate. Also the predictor measuring land cover diversity shows a significant positive effect. On the contrary, the predictor view shed and red list species abundance do not prove to have a significant effect. Red list species abundance does even have a negative sign, which is contrary to our expectations. Nevertheless, both variables drop out of the model during the variable selection procedure. We also find a positive effect of the numbers of days with a maximum temperature above five degrees. Another predictor, which shows a significant positive but relatively small effect on the number of visits is the age of the national park. The most important and highly significant predictor is the availability of trails. In the final model, it explains almost 17% of the number of visits. However, the question of correlation and causality is in particular relevant for this predictor. To what extend trails attract visitors and to what extend trails are put in place due to high visitor numbers cannot be answered by this analysis. The same may apply to the availability of minor roads, which also show a significant positive effect but being less important for explaining the observed visitor numbers. A significant negative impact can be found for the size of the study area of the visitor monitoring study, but a low beta coefficient indicates a relatively low importance. A stronger and significant, but negative impact shows the availability of other national park areas within 130 km surrounding. It is the third most important variable in our models. The second most important variable in explaining the observed number of visits is the population living in the surrounding of the study area, which shows a significant positive effect. A minor negative but not significant effect is found for the GDP/capita and the year of the visitor monitoring survey. This is contrary to our initial expectations. It could be that other cultural aspects interfere with this effect. It may also be that Southern European countries with lower GDP/capita (e.g. Italy and Spain) receive more visitors in NPs because of high tourist visits, whereas richer northern European countries (e.g. Scandinavian countries) receive fewer visitors because of lower tourist numbers. At the edge of the 0.1 significance level, the predictor measuring the quality of the visitor monitoring study shows a relatively small and negative effect. Initially, this variable was considered for explaining residual patterns. We expected that visitor monitoring studies with a lower quality judgment would result in less precise visitor estimates and therefore in higher residuals. However, in our pre-analysis we could not find a significant effect of the visitor monitoring quality on the residuals. Moreover, we find that visitor monitoring studies of lower quality tend to overestimate visitor numbers. This could be caused by the incentive of NP managers to highlight the importance of their NP and thereby use assumptions made within the visitor monitoring study in favour of higher visitor numbers. Visitor monitoring studies of higher quality may allow for less of these assumptions to be made (by more complete counting and less up-scaling). Furthermore, complete reporting of the assumptions made may stimulate more realistic judgments.

Table 8: Final model after stepwise model selection including spherical spatial correlation structure.

Variable	Coefficient	p-value		Beta coefficient
(Intercept)	-3.35	0.11		2.26%
Sqrt (water)	1.8	0.00E+00	***	9.29%
Sqrt (wetland)	-0.83	4.81E-02	*	3.84%
Log (broadleaf)	-0.31	3.41E-02	*	5.18%
Log (forest edge)	-0.57	3.32E-02	*	4.53%
Sqrt (land cover diversity)	1.32	4.70E-03	**	5.65%
Days > 5 degrees	6.89E-03	3.72E-02	*	7.83%
NP age	1.07E-02	4.30E-03	**	5.72%
Study area km²	-5.14E-04	5.20E-03	**	6.31%
Log (trails)	0.46	0.00E+00	***	16.95%
Log (roads)	0.44	2.90E-03	**	7.26%
Log (np substitutes)	-0.36	2.50E-03	**	10.81%
Log (population with 50 km)	0.32	1.20E-03	**	10.98%
Survey quality	-0.11	0.1	.	3.38%

Spherical spatial correlation structure	RMSE: 1.29	AIC: 727.5
Range: 580 km , nugget: 0.38	CV(RMSD): 0.48	BIC: 782.8

*** $p \leq 0.001$, ** $p \leq 0.01$, * $p \leq 0.05$, . $p \leq 0.1$

We used our final model to make predictions for all NPs sites in our primary visitor database and also for all NPs in most of the EU[20] as well as in Norway and Switzerland. Comparing our predictions with our primary data, we estimate an average relative prediction error of about 185% (the full model 174%), which seems reasonably good. Interestingly, the four observations contributing most to our relative prediction error are all located in Italy.

[20] We could not make predictions for some EU countries for which we are missing raster layers of the explanatory variables in our model. These countries are Bulgaria, Croatia, Cypress, Island and Malta.

Table 9: Estimates of total annual visits to national parks in European countries and their estimated monetary value.

Country	Km² of NP	Predicted Visits	95% Confidence Interval (lower / upper)	Monetary Value
Austria	3,098	24,098,000	14,001,000 / 41,660,000	172,684,000 €
Belgium	3,200	63,569,000	32,294,000 / 125,388,000	455,527,000 €
Switzerland	170	135,000	72,000 / 256,000	969,000 €
Czech Republic	3,543	32,835,000	17,148,000 / 63,127,000	235,290,000 €
Germany	2,363	534,188,000	309,773,000 / 921,987,000	3,827,911,000 €
Denmark	846	77,623,000	55,797,000 / 108,203,000	556,236,000 €
Spain	10,450	121,666,000	89,810,000 / 170,467,000	871,840,000 €
Estonia	1,618	2,182,000	1,561,000 / 3,078,000	15,635,000 €
Finland	8,196	6,427,000	4,564,000 / 9,456,000	46,054,000 €
France	13,565	71,408,000	36,506,000 / 140,680,000	511,700,000 €
United Kingdom	21,754	700,862,000	429,126,000 / 1,162,686,000	5,022,270,000 €
Greece	4,677	14,713,000	10,287,000 / 21,934,000	105,432,000 €
Hungary	6,234	18,543,000	11,457,000 / 30,336,000	132,878,000 €
Ireland	2,221	3,510,000	2,447,000 / 5,070,000	25,152,000 €
Italy	17,419	145,719,000	93,198,000 / 231,777,000	1,044,203,000 €
Lithuania	1,345	2,398,000	1,482,000 / 3,909,000	17,186,000 €
Luxembourg	465	2,912,000	1,560,000 / 5,441,000	20,866,000 €
Latvia	3,201	3,711,000	2,508,000 / 5,538,000	26,592,000 €
Netherlands	1,889	93,133,000	48,749,000 / 182,005,000	667,375,000 €
Norway	30,696	2,150,000	1,821,000 / 2,602,000	15,404,000 €
Poland	10,168	46,227,000	25,125,000 / 85,506,000	331,254,000 €
Portugal	930	15,245,000	10,006,000 / 23,227,000	109,244,000 €
Romania	5,670	2,662,000	1,565,000 / 4,546,000	19,077,000 €
Slovakia	7,679	18,218,000	9,079,000 / 37,180,000	130,544,000 €
Slovenia	1,157	4,121,000	2,425,000 / 7,004,000	29,531,000 €
Sweden	8,370	7,773,000	5,457,000 / 11,191,000	55,700,000 €

Using our model to predict the number of visits to all 449 NPs across our study area, we estimate a total annual number of visits of more than 2 billion (2,016,028,000; lower and upper 95% confidence interval: 1,217,818,000; 3,404,254,000)[21]. Combining this estimate with the average monetary value per visit (€ 7.17, prices 2013), which we extracted from a meta-analysis of recreation valuation studies, the total recreational value of the 449 NPs amounts to € 14.5 billion annually. The result compares well to the estimates of Balmford *et al.* (2015), who estimate 3.8 billion visits annually and a value of $US 26,943 billion for all PAs within Europe, not only NPs. Our aggregated estimates per country are shown in Table 9.

Most visits are received by British NPs, which results from the large total area of NPs, high population density and intensive recreational facilities in terms of trail densities. Also other densely populated countries such as Denmark, Belgium and the Netherlands show relatively high visitor numbers. On the contrary, countries such as Sweden, Finland and Norway show relatively low visitor numbers for their large and mainly forested NPs in the low populated north. Germany shows exceptionally high visitor numbers considering the relatively small NP area. However, these numbers are dominated by one large NP, for which our model may over-predict the total number of visits. The Wadden Sea NP — an UNESCO natural heritage — is by far the largest NP of Germany and stretches almost all along the North Sea shore of Germany. The area lies in the catchment of large cities such as Hamburg and Bremen. It is a touristic hot spot receiving by far the highest number of day and overnight visits of German NPs (Job *et al.* 2010). All variables used in our model, except size, show values in favour of high visitor numbers for the Wadden Sea NP. This combination of such variable values is exceptional in our data and may cause an unreasonable over prediction.

Our predictions of visits per ha for a marginal increase of NP supply in Europe are shown below in Figure 21. We assume a hypothetical newly created NP of about 80 km anywhere throughout Europe an estimate the number of visits it would receive. All urban areas are excluded from this prediction (EEA 2015c), as it seems unrealistic that such areas would be converted into a NP and because urban areas are typically characterized by explanatory variable values that lie beyond the range of the explanatory variable values of our primary data. The map shows values from almost zero up to the maximum of about 147,000 annual visits per ha. Low numbers of visits are predicted for remote areas, which are characterized by low population and little access infrastructure. The maximum predicted visits of 147,000 per ha seems high, but 34 visitors for an average daylight hour may not be unreasonable for a popular visitor hot spot in a NP. However, it should be considered that the predicted visitor numbers are strongly skewed with a mean and median values of about 87 and 4.8. More than 90% of the pixels receive visitors on less than 100 visitors a year and anything above 2,000 is to be expressed in per mile. A map presenting the spatially explicit economic values can be found in the appendix.

[21] We used the rms R-package for estimating confidence intervals.

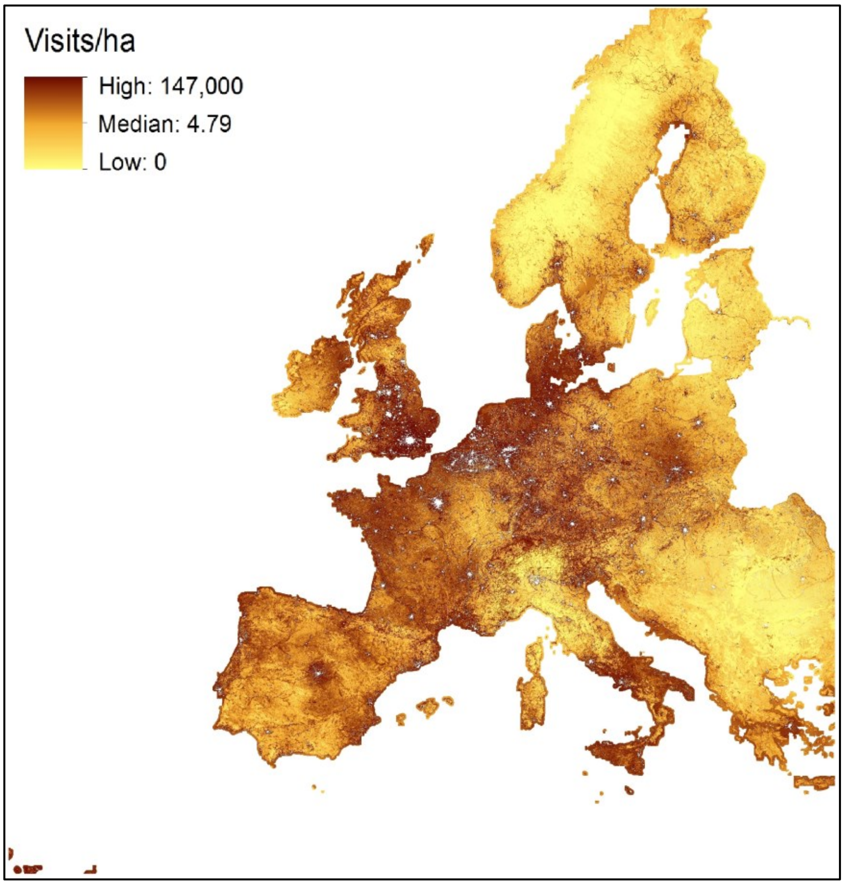

Figure 21: Predicted visits per ha and year for a potential new national park of about 80 km².[22]

To exemplify our model for a realistic setting we chose the Teutoburger forest and the Senne heathland in the west of Germany, which is proposed for NP designation. Figure 22 shows how the predicted annual visits per ha distribute across the area. On average, we expect about 283 annual visits per ha for the area. The highest visitation rate is predicted in the peripheral areas, close to the population centres of cities of Detmold and Paderborn receiving up to 24,000 visits per ha and year. In contrast, the centre of the proposed NP, which is hardly accessible, is predicted to receive less than one visit/ha/year. In total we predict about 5.8 million annual visits for the entire area (95% confidence interval lower bound 3.38 and upper 9.91 million)[23], which accounts for an annual monetary value of approximately € 41.5 million. A map presenting the spatially explicit economic values can be found in the appendix. Negative impacts on the number of visits include the relatively low presence of water bodies, high forest cover, low trail availability and the low age of the potential new NP. Positive impacts include the small size of the NP, the high population pressure, low substitute availability and the high land cover diversity. The number of visits is expected to increase with the age of the NP and if recreational facilities are established.

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[22] Note that for illustrative purpose the color scheme is set to display the same amount of pixels per color shade.

[23] Note that the predicted total visitor number of the entire area is less than the sum of the predicted visitors for each ha because of two reasons: visitors may cross more than one ha during a visit and it is not possible to take the linear mean of a model containing non-linear variables.

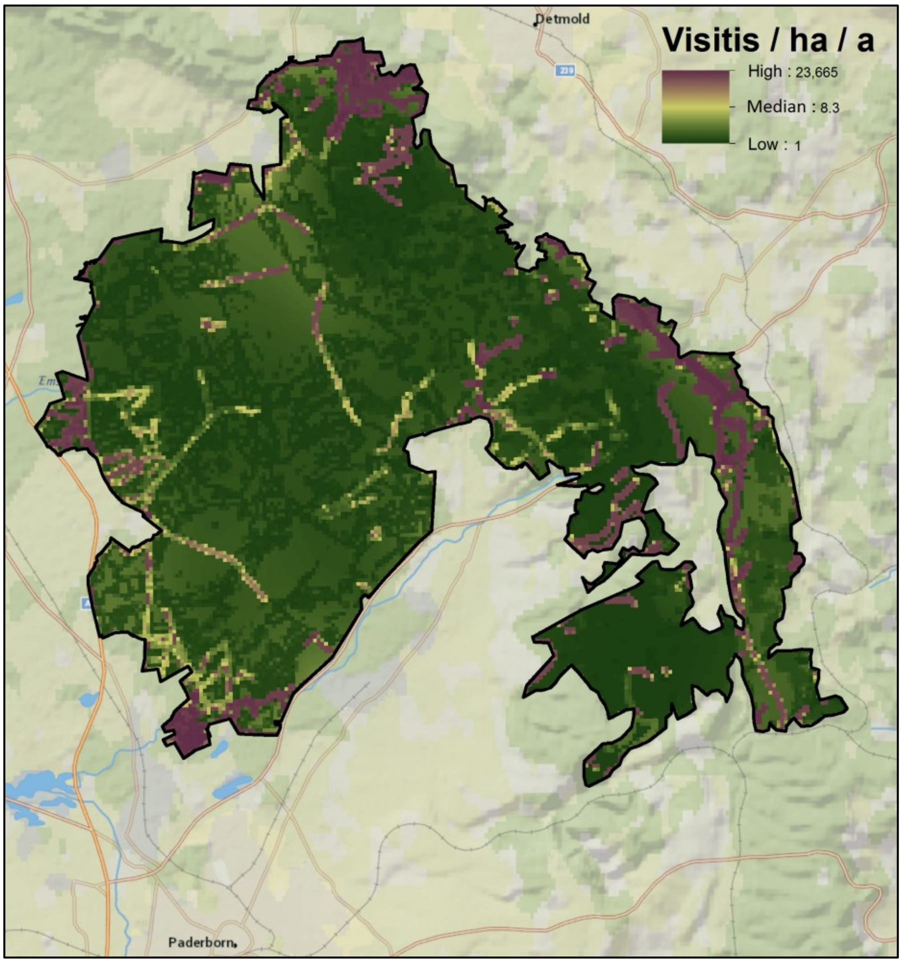

Figure 22: Predicted visits per ha and year for a potential national park in the Teutoburger forest and the Senne heathland (west of Germany).

4.5 Discussion

4.5.1 Spatial Effects and Modelling

Our estimated model fits the data reasonably well and therefore offers valuable information on the main drivers of recreational use within European NPs. All predictors with statistically significant effects on the number of recreational visits have signs that are in line with our interpretations and theoretical expectations.

Nevertheless, there are some uncertainties in the model and prediction accuracy which may be improved by further research. The question remains, what may be the source of the SAC. In an optimal statistical textbook world, introducing SAC in a model would not influence parameter estimates, but only reduce the degrees of freedom of the model. However, looking at real world spatial data, this is hardly ever the case. If parameter estimates are affected as in our case, this may indicate some common spatial econometric problems, such as missing predictors, which are picked up by the spatial error term, a spatial weight matrix or a non-linear relationship (Diggle *et al.* 2000; Fingleton and Gallo 2010; Smith and Lee 2011). A likely explanation could be that unobserved determinants of recreational visits exists, which are spatially related. Such determinants could be manifold and include everything from site, context and methodological study characteristics as well as their interactions. One important aspect could be related to the social-cultural context and path dependencies, which may result in specific recreational patterns in certain countries and regions. Also differing property rights could play an important role. Investigating human recreational behaviour across a study area as big as Europe is such a complex issue that all of these econometric problems may arise. There may hardly be any model that can incorporate all relevant drivers of recreational use, their interactions and non-linear effects.

Encountering such problems is common for modelling spatial data and therefore, we have to be cautious in interpreting p-values and parameter estimates. An option to gain further insights and confidence in model result interpretations is to try different spatial modelling approaches and compare their results. In particular, compare the confidence intervals of the parameter estimates. There is number of model setups, which would qualify for evaluating such spatial data sets. Since in this study we are analysing count data, one option would be to use a negative binomial or a quasi-Poisson distribution, even though it should not change the model results too much (O'Hara and Kotze 2010). However, there are only a very limited number of statistical R-packages, which allow for combining these distributions with SACs and as we stated above we had problems in solving the maximization algorithms for these models. An option would be to try alternative models incorporating the SAC either within the fixed or the random part of the model, such as a spatial lag model, a Durbin model, spatial autoregressive models with autoregressive disturbances, geographically weighted regressions or even by using Bayesian approaches. However, there is no consensus on which model to use best for this specific purpose. Fitting all or at least some of these models and comparing their results may be subject to further research (Bivand 2011; Elhorst 2010; Gerkman 2011; Brunsdon *et al.* 1996).

Nevertheless, considering the complexity of the spatial processes driving human recreational behaviour, we can confidently say, that we model recreational use reasonably well. None of the predictors' signs differ across the different estimated models, neither for models without autocorrelation nor for the mixed models, which indicates the robustness of our analysis. Anyhow, other publications conducting spatial modelling of recreational use do not at all engage to such a depth in the spatial dimensions nor do they take into account such considerations on uncertainties, potential alternative regression techniques and model setups (Hanink and Stutts 2002; Hanink and White 1999; Mills and Westover 1987; Neuvonen *et al.* 2010a).

Future research on this issue may benefit from greater and more reliable primary data availability. Errors in primary data collection impose huge difficulties for identifying relevant predictors. In recent years visitor monitoring studies encountered a huge dynamic in terms of interest and technical advancement. Recent remote controlled electronic visitor counters allow far more accurate visitor estimates at lower costs as compared to conventional personal counting. More refined GIS data sets may allow for more accurate, detailed and comprehensive predictors for modelling recreational demand.

4.5.2 Valuation of Recreational Services

Another aspect of improvement may be to account for spatial variations in the value per recreational visit by applying a value function transfer (such as meta-analytic value transfer). Using a unit value transfer for mapping ecosystem service values across a larger area is associated with transfer errors, in particular with so-called generalisation errors. Nevertheless, the value of a recreational visit varies across space due to differences in ecosystem characteristics and the local population's preferences, differences that are not accounted for in a unit value transfer (Rosenberger and Loomis 2001). Value function transfer allows adjusting transferred values to site specific circumstances and may therefore be more accurate for ecosystem service value mapping. However, even though, value function transfer is considered to produce lower transfer errors, there is no consensus on which value transfer method is best for specific circumstances. Evidence on transfer errors show mixed results and unit value transfer may be superior to other value transfer techniques for some applications (Brouwer 2000; Johnston and Rosenberger 2010; Lindhjem and Navrud 2008; Navrud and Ready 2007; Rosenberger and Phipps 2007; Rosenberger and Stanley 2006). In ecosystem service value mapping, the unit value transfer method is most common (Schägner *et al.* 2013). It is also proposed for the aggregation of values to set up, for example, a SEEA, even though aggregation over a large area is controversial and should be interpreted with caution (Costanza *et al.* 1998; UN 2014).

In the case of recreational services, meta-analysis of recreational valuation studies show that most of the variations in the value per visit result from different valuation methodologies and not from site specific circumstances, indicating large measurement errors. Moreover, it remains difficult to identify robust relationships between spatial explanatory variables and the final value estimate. Meta-analysis on recreational valuation studies identify only few significant and typically weak effects of biophysical, socio-economic and regional or national dummy variables (Shrestha *et al.* 2007; Zandersen and Tol 2009; Brander *et al.* 2015; Sen *et al.* 2013; Sen *et al.* 2011; Rosenberger and Loomis 2001; Londoño and Johnston 2012). By using the mean value of a large number of primary valuation studies, we aim at averaging out measurement errors within our value transfer (Johnston *et al.* 2006), which may result in lower transfer errors as compared to the usage of single studies or regional subsets, even though cultural differences across countries may affect value per recreational visit (Hynes *et al.* 2012; Kaul *et al.* 2013; Lindhjem and Navrud 2008; Ready and Navrud 2006; Shrestha and Loomis 2001).

Finally, the overall recreational value of a site is predominantly determined by spatial variations in the number of recreational visits. Spatial variations in value per recreational visit play only a minor role (Bateman *et al.* 2006a; Jones *et al.* 2003). This insight is also supported by mean relative deviation of our primary data, which is considerably higher for the visitor numbers as compared the value per visit estimates. In consequence, accurate visitor estimates are by far more important for defining the overall recreational value of a certain location than accurate estimates of the recreational value per visit. As compared to meta-analysis of recreational valuation studies, the explanatory power of our spatial variables explaining visitor numbers is high. We are therefore confident that we capture the main spatial variations in the overall recreational value NP recreation and that the value estimates give a good indication of the relative importance of European NP recreation as compared to other ecosystem services and man-made goods.

4.5.3 Policy Implications

The model can be used for a number of policy applications: (1) The model may contribute to the fulfilments of the EU Biodiversity Strategy 2020, which require of EU members sates to "*map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020*" (EC 2011b) and the achievement of the Aichi Targets, which aim at *"reflecting the values of biodiversity in spatial planning and resource management exercises including through the mapping of biodiversity and related ecosystem services"* (CBD 2013). (2) The mapped recreational visitor numbers and the related economic value of recreational ESS can act as a spatial value data base that can be used for value transfers. Policy makers can quickly derive a value estimate of the recreational services of any NP across Europe by consulting the map. (3) The maps may contribute to an efficient resource allocation by allowing policy makers to prioritize areas for conservation due to their high recreational value. In addition, recreational infrastructure may be designed to match the needs of the expected visitor numbers within a given NP. Furthermore, it may be valuable to compare the model's predictions with real world observations on recreational use and values (if available) and, for example, investigate why some NP might remain below their recreational potential and how the recreational use and its value could be increased. However, it should be noted that the model allows only for assessments of NPs. Even if predictions can be made for a new hypothetical NP, no conclusion can be made on whether NPs designation results in an in- or decrease of recreational use and its values. (4) The model allows to evaluate the effect of land use policies within European NP on recreational services and values. (5) Finally, the estimated recreational service values may contribute to set up a green GDP or a SEEA as proposed by the UN (2014), which may act as a counterpart to traditional GDP accounts and represent an additional measure for the impacts of human action on human well-being.

4.6 Conclusion

We model recreational use of European NPs using a large number of spatially variable predictors. Our model fits the data reasonably well and we identify the main determinants of variation in recreational use in European NPs. Among analysed variables trails density, population density, presence of substitutes, presence of water bodies and number of days with temperature above 5 degrees are those that show a higher explanatory value. The model allows the estimation and valuation of total recreational use of existing and planned NPs. For our study area covering most of Europe and in total 449 NPs, we estimate more than 2 billion recreational visits a year, with an economic value of approximately € 14.5 billion. The latter information is particularly relevant to support the task that EU countries should fulfil by 2020, according to EC (2011) of assessing the economic value of ecosystem services and integrate such values into accounting and reporting systems by 2020.

Since all our predictors are obtained from GIS raster layers, which cover entire Europe, the model can be applied for ex-ante evaluation of alternative policy scenarios of change for existing NPs and on the creation of new NPs at a European scale. This information may be useful in planning the supply of recreational facilities such as parking and accommodation. Furthermore, NP locations and design features optimizing recreational use can be identified. Thereby, the model has implications for NP policy of European countries. Based on our findings, we can conclude that to ensure high numbers of recreational visits, potential new NPs should be located in close proximity to populated areas but not close to other NPs. The total conservation area should be used for a larger number of small parks rather than for a smaller number of large ones. The availability of water bodies and the diversity of the land cover contribute to higher visitation rates, whereas extensive forest cover tends to deter visitors. However, it should be kept in mind that the main purposes of NPs are not to supply recreational services but preserve a beautiful and natural landscape as well as biodiversity for posterity. Recreational opportunities are a co-benefit of NPs, which can be used as an argument for allocating resources towards NP creation and conservation.

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4.9 Appendix

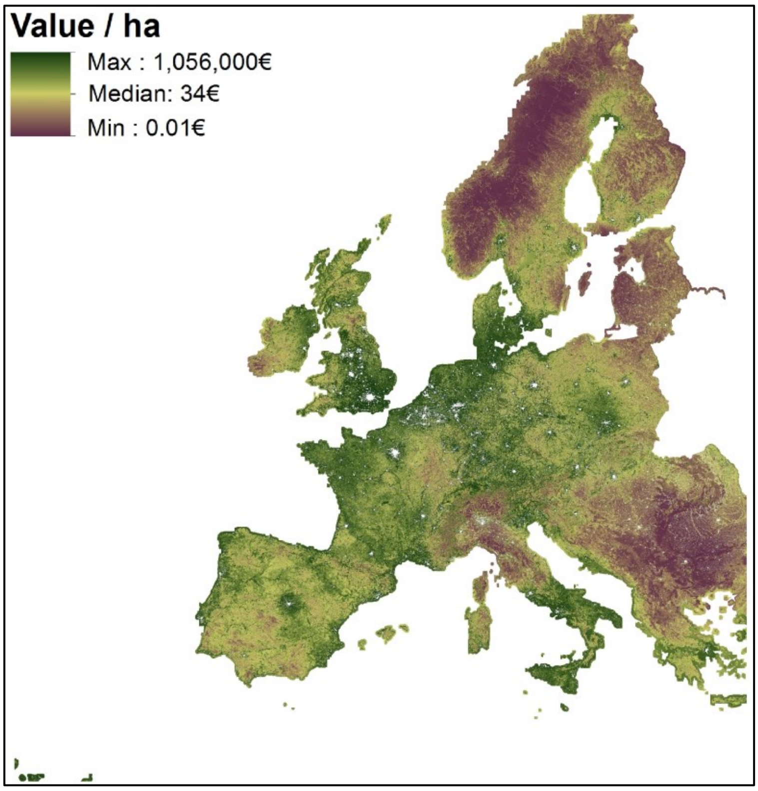

Figure A 4.1: Predicted recreational value per ha and year for a potential new national park of about 80 km².

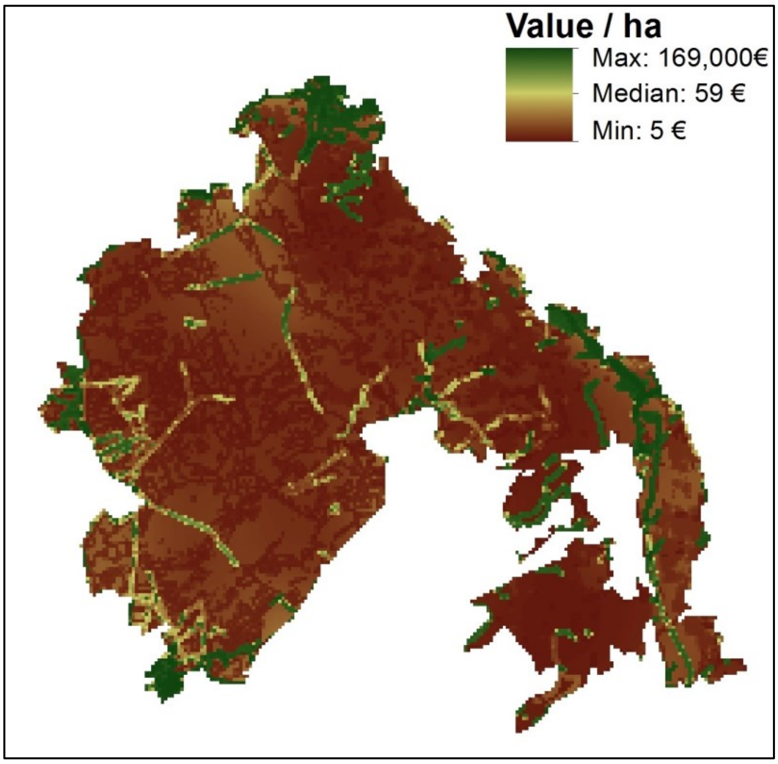

Figure A 4.2: Predicted values per ha and year for a potential national park in the Teutoburger forest and the Senne heathland (west of Germany).

5 GIS-Based Mapping of Ecosystem Services: The Case of Coral Reefs

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Recreation

Abstract

This chapter illustrates the process of mapping ecosystem service values with an application to coral reef recreational values in Southeast Asia. The case study provides an estimate of the value of reef-related recreation foregone, due to the decline in coral reef area in Southeast Asia, under a baseline scenario for the period 2000 – 2050. This value is estimated by combining a visitor model, meta-analytic value function and spatial data on individual coral reef ecosystems to produce site-specific values. Values are mapped in order to communicate the spatial variability in the value of coral reef degradation. Although the aggregated change in the value of reef-related recreation due to ecosystem degradation is not high, there is substantial spatial variation in welfare losses, which is potentially useful information for targeting conservation efforts.

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5.1 Introduction

The framework of ecosystem services (ESS) is widely used for understanding and communicating the links between ecosystems and human well-being (MA 2005). Many 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 of an ESS is largely determined by the characteristics of human beneficiaries of the ESS (population, preferences, distance to the resource, etc.) and modelling hereof has largely been taken up by economists. It has been recognized that the determinants of both the supply and demand of ESS are spatially variable, which makes the assessment of ESS values inherently a spatial analysis. In recent years, a growing body of literature has assessed ESS spatially by producing digital maps either of ESS supply or its value. With the development of advanced GIS technology, mapping of ESS values has emerged and become an important research issue, in particular the mapping of monetary values for ESS value (Bateman *et al.* 1999; Brainard 1999; Maes *et al.* 2013; Schägner *et al.* 2013; Troy and Wilson 2006). This literature therefore includes studies that produce graphical value maps as well as analyses that explicitly address spatial variability in values.

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. Thereby, mapping of ESS values reveals additional information as compared to traditional site-specific ESS valuation, which is beneficial for designing spatially efficient policies and institutions for maintaining ESS supply. Most often, this mapping involves some type of benefit transfer, in which values from one set of locations are used to project or approximate values in other areas.

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 1990s (Bockstael 1996). The first studies to map ESS values examined recreational values for Welsh forests (Bateman *et al.* 1995) and multiple ESS across a protected area in Belize (Eade and Moran 1996). Since then, the number of publications mapping ESS values has grown exponentially. Schägner *et al.* (2013) provide a review of the literature on mapping ESS values and show that almost 60% of such studies have been published after 2007. The methodologies applied in these studies differ widely, particularly with respect to how spatial variation in ESS values is estimated. The precision and accuracy of mapped ESS values have 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).

The purpose of this chapter is to develop and apply a method for mapping the value of the recreational use of ecosystems, based on a meta-analytic benefit function transfer. The chapter is organized as follows: Sect. 5.2 describes the methods that have been applied in the literature so far. Section 5.3 describes an application of value mapping to assess the welfare loss associated with coral reef degradation in Southeast Asia under a business-as-usual scenario for the period 2000 – 2050. This section contains details on the case study region, methodology, visitor model, meta-analytic value function, scenario for coral reef degradation and value maps. Section 5.4 provides conclusions on the results, methods and avenues for future research.

5.2 Methodologies for Mapping Ecosystem Service Values

The estimation of accurate ESS values requires that models account for spatial heterogeneity in biophysical and socioeconomic conditions. The spatial perspective of variation in ESS values is relatively new and has not been extensively researched (Schaafsma *et al.* 2012). Insufficient knowledge exists about how ESS values differ across space and the spatial determinants of these values (Bateman *et al.* 2002; Bockstael 1996; de Groot *et al.* 2010; Plummer 2009; Schaafsma *et al.* 2013). Spatial factors that affect the supply of ecosystem services include, among others: ecosystem area (possibly characterized by a non-linear relationship and/or with thresholds), networks, fragmentation, and biodiversity. Spatial factors that affect demand for ecosystem services include: the number of beneficiaries, distance to the ecosystem, availability of substitutes, complements, and accessibility. See Bateman *et al.* (2002) and Hein *et al.* (2006) for more detailed discussions of spatial determinants of ecosystem service demand and supply.

Besides communication and visualization, value mapping makes site-specific ecosystem service values available on a large spatial scale. It allows decision makers to extract estimated values from a map or database for the locations or areas of policy interest in order to evaluate potential policy measures. New time-consuming primary valuation studies may therefore not be necessary.

Spatially explicit ESS value maps have specific advantages for several types of policy applications including green accounting, land use policy evaluation, resource allocation and payments for ES. Green accounting includes information on environmental goods and services and/or natural capital in national accounts. Mapping of ESS values allows the estimation of values at different spatial scales, and the aggregation of total ESS values across the region of interest for inclusion in green accounts (TEEB 2010). For land use policy evaluation, the mapping of ESS values allows for the evaluation of broad land use policies at a regional or even supranational level. Typically, land uses are multi-functional and therefore provide multiple services. ESS value mapping displays the trade-offs and synergies in ESS values that may result from land use change. For improving resource allocation, the mapping of ESS values not only supports decisions on whether or not to implement a policy measure, it also informs where to implement a policy measure. It allows the identification of locations in order to minimize negative or maximize positive impacts on the provision of ecosystem service (Naidoo *et al.* 2008; Polasky *et al.* 2008). Regarding payments for ES, by making ESS values spatially explicit, schemes can be designed to allow for more efficient and cost-effective incentives across providers. The levels of payments can then be more closely related to the value of services provided by different locations.

Methodologies used for mapping ecosystem service supply can be divided into five main categories (Eigenbrod *et al.* 2010; Schägner *et al.* 2013): (1) one-dimensional proxies for ecosystem services, such as land cover or land use (e.g., Costanza *et al.* 1997; Helian *et al.* 2011; Simonit and Perrings 2011); (2) non-validated models: ecological production functions based on likely causal combinations of explanatory variables, which are grounded in researcher or expert assumptions (e.g., Holzkämper and Seppelt 2007; Naidoo and Adamowicz 2005; Zhang *et al.* 2011); (3) validated models: ecological production functions, which are calibrated based on primary or secondary data on ecosystem service supply (e.g., Coiner *et al.* 2001; Mashayekhi *et al.* 2010); (4) representative samples of the study area: data on ecosystem service supply that is collected for the specific study area (e.g., Chen *et al.* 2009; Crossman *et al.* 2010); and (5) implicit modelling of ecosystem service supply within a value transfer function, i.e., the quantity of ecosystem service supply is modelled within the valuation of the ecosystem service using variables that capture supply-side factors (e.g., Brander *et al.* 2012; Costanza *et al.* 2008).

5.3 Application: Mapping Coral Reef Values in Southeast Asia

This section provides an illustration of the process of mapping ecosystem services values in an application to value changes in coral reef recreational values in Southeast Asia. The purpose of this case study is to illustrate the data, methods and results of a value mapping exercise.

5.3.1 Coral Reef Recreation, Threats and Values in Southeast Asia

Southeast Asia has the most extensive and diverse coral reefs in the world. They cover approximately 70,000 km², which is 28% of the global total area of coral reef (Burke *et al.* 2011). Within the region, the Coral Triangle, which includes the reefs of Indonesia, the Philippines and Malaysia, contains 76% of all known coral species and hosts 37% of all known coral reef fish species. The coral reefs of Southeast Asia are highly productive ecosystems that provide a variety of valuable ecosystem services to local populations (Burke *et al.* 2011; UNEP 2006). These ecosystem services include coastal protection, habitat and nursery functions for commercial and subsistence fisheries, recreational and tourism opportunities, and the existence of diverse natural ecosystems. In this case study we focus on the recreational and tourism uses of coral reefs.

Tourism is one of the largest and fastest growing industries in the world. In Southeast Asia, tourism accounted for 11.1% of the region's GDP in 2012 and is forecast to grow at 5.8% per annum over the coming decade (WTTC 2013). Reef-related tourism is expected to increase even more rapidly (Musa and Dimmock 2012). Recreational activities associated with coral reefs include diving, snorkelling, viewing from boats, and fishing. In addition, many beaches are protected by reefs or formed from coral material. Cesar *et al.* (2003) estimate the total global annual value of coral reef-based recreation and tourism at US$ 9.6 billion.

Given the range and serious nature of threats to the ecological integrity of coral reefs, there is a need for more information on the value of welfare losses associated with a decline in the provision of ecosystem services (MA 2005). Information on the value of coral reef ecosystem services can be used in a number of different policy-making contexts, including the justification for establishing marine protected areas, determination of compensation payments for damage to coral reefs, setting of user fees for access to protected areas, cost-benefit analysis of conservation and restoration measures, and advocacy regarding the economic importance of properly functioning marine ecosystems (Van Beukering *et al.* 2007).

5.3.2 Outline of the Case Study Methodology

The aim of this case study is to provide an estimate of the loss in value of coral reef-related recreation resulting from the decline in coral reef area under a business-as-usual scenario for the period 2000 – 2050. In other words, it estimates one component of the cost of policy inaction from not adequately addressing the multiple threats facing coral reefs in the region. The changes in coral reef-related recreation values are mapped in order to account for spatial variation in the determinants of value and present the results in a spatially explicit way, allowing for the identification of high impact locations. Following Sen *et al.* (2014), the selected methodology uses a combination of a validated model for visits to coral reefs and a meta-analytic value function to estimate the value per visit. An alternative approach would be to use a meta-analysis to estimate recreational values on a per hectare basis and implicitly model the number of visits to each hectare of an ecosystem within the value function. This is the approach used, for example, by Ghermandi and Nunes (2013) for estimating the recreational value of the world's coasts. Due to data limitations on recreational visit flows at a global scale with which to estimate a model of visits, they transfer values on a per hectare basis rather than per recreational visit.

The methodology involves the following steps:

1. Estimate a model of recreational visits to individual coral reef sites. The visitor model relates the number of visits per day to the site and context characteristics of each coral reef ecosystem such as degree of siltation or fishing damage.
2. Estimate a value function for coral reef recreation through a meta-analysis of existing monetary estimates. The value function relates the value per visitor day to the characteristics of the ecosystem and its surroundings.
3. Develop a database of coral reef ecosystems in Southeast Asia containing information on the variables included in the visitor model and value function estimated in steps 1 and 2.
4. Develop a baseline scenario for the change in the quality and spatial extent of coral reef ecosystems in Southeast Asia for the period 2000 – 2050. This baseline scenario is spatially variable to reflect variation in location-specific pressures on coral reef ecosystems.
5. Combine the models and data generated in steps 1 through 4 to produce estimates of the value of the loss in coral reef-related recreation under the baseline scenario. This approach allows the estimation of spatially variable, site-specific values that reflect the characteristics and context (e.g., pressure or threat) of each coral reef.

5.3.3 Visitor Model

In the first step of the analysis, we estimate a visitor model which explains variation in the number of visits by individual visitors to a given coral reef site per day. This is modelled as a function of several explanatory variables describing the characteristics of the ecosystem and its surroundings. We estimate the visitor model using a large sample survey for coral reef sites in Southeast Asia.[24] These data have a panel structure in that multiple observations of visitor numbers are taken for the same coral reef site at different points in time. Using a GIS, the visitor data are combined with additional information on spatially referenced variables obtained from multiple sources (including area of other ecosystems, population and economic activity in the vicinity of each coral reef site).

[24] Reef Check is a volunteer survey program that has collected biophysical and visitor data at reef sites for more than 3000 survey sites in 80 countries globally since 1997 (see: www.reefcheck.org).

Table 10: Variables included in the visitor model for Southeast Asia.

Variable	Variable definition	Mean	Standard deviation
Visitors	Number of visitors per day	16.216	15.396
Siltation	Dummy: 1 = siltation; 0 = none	0.717	0.451
Fishing damage	Dummy: 1 = fishing damage; 0 = none	0.290	0.454
Air temperature	Average air temperature (oC)	30.795	1.751
Area of coral cover	Area of coral cover (km²)	11.351	38.553
Area of mangroves	Area of mangroves within 50 km (km²)	32.298	79.124
Population	Population within 50 km	739,273	920,681
GCP	Gross cell product within 50 km (US$)	6732	4533

The dependent variable in the estimated regression model (γ) is the number of visitors per day to a specific reef location. The explanatory variables are grouped in two matrices that include the site characteristics in Xs and context characteristics in Xc. Table 10 presents the list of variables included in the analysis with the mean and standard deviation of each.

The model fit was considerably improved, and heteroskedasticity mitigated, by using the natural logarithms of the area and context variables. Following Bateman and Jones (2003), Brander *et al.* (2007), and Brouwer *et al.* (1999), we use a multilevel modelling (MLM) approach to estimate the meta-regression.[25] MLM allows a relaxation of the common assumption of independent observations, and enables us to examine hierarchies within the data, such as similarity of observations for the same reef. The use of MLM provides an indication of where the assumption of independence may be invalid, and also improves the estimation of standard errors on parameter coefficients. The estimated model is given in following equation:

$$\gamma_{ij} = \alpha + \beta s X s_{ij} + \beta c X c_{ij} + \mu_j + e_{ij}$$

where the subscript i takes values from 1 to the number of observations of visits and subscript j takes values from 1 to the number of reefs. α is the constant term, μ_j is a vector of residuals at the second (reef) level, e_{ij} is a vector of residuals at the first (observation) level, and the vectors β contain the estimated coefficients on the respective explanatory variables. In this equation, both μ_j and e_{ij} are random quantities with means equal to zero. We assume that these variables are uncorrelated and also that they follow a Normal distribution so that it is sufficient to estimate their variances, σ_μ^2 and σ_e^2 respectively (Rasbash *et al.* 2003). This type of model is also known as an error variance components model, given that the residual variance is partitioned into components corresponding to each level in the hierarchy. In our model, the level 2 residuals represent each reef's departure from the population mean, represented by the constant term, and the level 1 residuals reflect the conventional error variance at observation level. The estimated regression model is presented in Table 11.

[25] The software used is MLwiN version 2.0 (see Rasbash et al. 2003).

Table 11: Estimated visitor model for Southeast Asia

Variable	Variable definition	Coefficient	Standard error
Constant	–	−37.301**	16.073
Siltation	Dummy: 1 = siltation; 0 = none	−5.866***	0.932
Dynamite fishing damage	Dummy: 1 = fishing damage; 0 = none	−7.036***	1.212
Air temperature	Air temperature (°C)	−0.569***	0.162
Area of coral cover	Natural log of area of coral cover (km²)	1.027	0.638
Area of mangroves	Natural log of area of mangroves within 50 km (km²)	0.685*	0.373
Population	Natural log of population within 50 km	−0.886*	0.467
GCP	Natural log of Gross Cell Product within 50 km (US$)	9.672***	1.373
Level 1 (observation) variance		145.509***	12.697
Level 2 (reef) variance		12.569***	0.927
−2*log likelihood		4447.873	
N		658	

*$p \leq 0.10$, **$p \leq 0.05$, ***$p \leq 0.01$

As expected, the presence of siltation and damage due to dynamite fishing at a coral reef site reduces the number of visitors to that site. Air temperature is also found to have a statistically significant negative effect on the number of visitors at a coral reef site. This indicates that additional increases in temperature reduce the attractiveness of recreation locations. An optimal temperature or possible non-linear effects with temperature were examined by including a quadratic term in the regression model, but no statistically significant effects were found. The estimated coefficient on the area of coral cover at the site is positive but not quite statistically significant at the 10% level. The area of mangroves within a 50 km radius of the coral reef site is found to have a positive and statistically significant effect on the number of visits. This suggests that there may be positive effects from the extent of other coastal ecosystems on the attractiveness of coral reef sites to visitors. This apparent complementarity between ecosystems possibly indicates the degree of naturalness of the site location. The size of the population living within a 50 km radius of a coral reef site is found to have a negative and statistically significant effect on the number of visitors. On one hand this result is somewhat surprising, since the population in the vicinity of a coral reef represents potential visitors.

On the other hand, visitors to coral reefs are often not local residents. This may particularly be the case in developing countries for which a large proportion of coral reef visitors are international tourists. In this respect, visitor models for coral reefs may differ substantially from visitor models for other ecosystems, for which the size and proximity of the local population are important explanatory factors (Sen *et al.* 2014).

The negative effect of population in the vicinity of a coral reef site is interpreted here as the pressure and impact of urbanization and other types of development on the attractiveness of a coral reef to visitors. The estimated coefficient on gross cell product (GCP), which is a spatially disaggregated measure of economic activity equivalent to gross domestic product (GDP),[26] indicates that visitor rates are higher in regions with higher income levels. This variable does not necessarily represent the income of visitors themselves, given that visitors are often international tourists, but may reflect the availability and quality of infrastructure in a region. The estimated level 2 (reef-specific) variance indicates that there remains unexplained reef-specific variation in visitor numbers. Calculating the variance partition coefficient [12.569 / (12.569 + 145.509) = 0.08] shows that approximately 8% of residual variance in visitor numbers can be attributed to unobserved differences between reefs.[27]

5.3.4 Meta-Analytic Value Function for Reef Recreation

Following Brander *et al.* (2007) and Londoño and Johnston (2012), a meta-analysis of the coral reef valuation literature is used to estimate a value function for coral reef-related recreation. The coral reef value data set used to estimate value functions for coral reef ecosystem services is an extension of the data described in Brander *et al.* (2007). These data have been expanded to include a number of recent coral reef valuation studies. We restrict this data set, however, to select only estimates obtained using contingent valuation or travel cost methods in order to ensure the theoretical validity of the welfare estimates (e.g., we excluded estimates that measure gross revenues). The restricted sample size is 74, of which 47 are contingent valuation estimates and 27 are travel cost estimates.

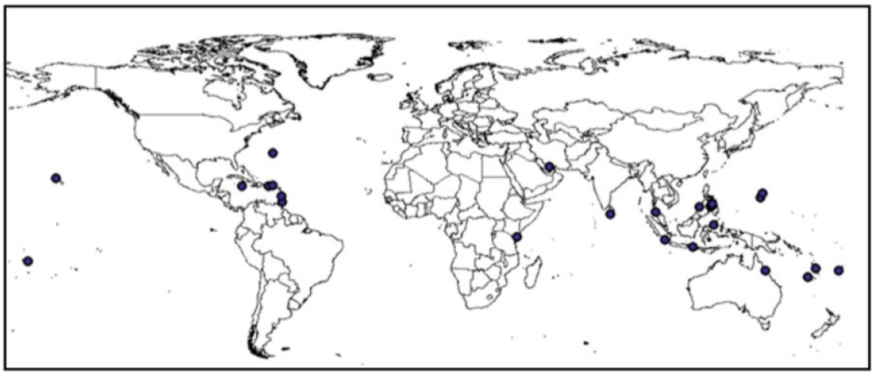

Figure 23: Location of coral reef recreation valuation study sites

The studies included in our analysis were published between the years 1992 and 2012. The geographic distribution of study sites is presented in Figure 23. Southeast Asia is reasonably well represented in

[26] The conceptual basis of GCP is the same as GDP as developed in national income accounts. The basic measure of output is gross value added in a specific geographical region. Gross value added is defined as total production of market goods and services less purchases from other businesses. Under the principles of national economic accounting, GCP will aggregate up across all cells within a country to GDP (Nordhaus et al. 2006). This variable is correlated with population, but not perfectly.

[27] We test the influence of unobserved reef specific effects using a likelihood ratio test, for which the null hypothesis is that σ_{μ}^{2} = 0. We compare the estimated model with a model where σ_{μ}^{2} is constrained to equal zero, i.e., a single level model. The value of the likelihood ratio statistic is 5157.32 − 4447.87 = 709.442. Comparing this to a chi-squared distribution on 1 degree of freedom, we conclude that there are significant unobserved differences between reef sites.

the data with 13 valuation estimates (17% of the sample). The locations of the remaining estimates are the Caribbean (16%), the United States (51%),[28] Indian Ocean (13%), and Australasia (3%).

The data on the value of reef-related recreation are standardized to a common currency, year of value and units using Purchasing Power Parity (PPP) adjusted exchange rates and GDP deflators from the World Bank World Development Indicators.[29] The standardized values are expressed in US$ per visitor day in 2007 prices. This is the dependent variable in the meta-analytic regression model. The model is given in the following equation:

$$\ln(y_i) = \alpha + b_S X_{Si} + b_R X_{Ri} + b_M X_{Mi} + \mu_i$$

The subscript i assumes values from 1 to 74 (number of observations), α is the constant term, b_S, b_R and b_M are the coefficients of the explanatory variables and μ is a vector of residuals. The explanatory variables consist of three categories, giving characteristics of: (i) the study site X_S , (ii) the recreational activities valued X_R , and (iii) the valuation method used X_M. Table 12 presents the full list of variables included in the analysis, with the mean and standard deviation of each.

The meta-regression results are presented in Table 13. Following best practice, heteroskedasticity-consistent standard errors are estimated. However, the null hypothesis of homogenous variance of the residuals cannot be rejected by White's test for heteroskedasticity (White's statistic = 21.589). The adjusted R^2 statistic indicates that approximately 41% of the variation in the dependent variable is explained by the explanatory variables, which is comparable with similar meta-analyses of the ecosystem service valuation literature (e.g., Brander *et al.* 2007; Ghermandi *et al.* 2010).

Table 12: Variables included in the meta-analytic value function

Variable	Variable definition	Mean	Standard deviation
Value per visit	US$ per visitor day	73.86	171.66
Visits per day	Visits per day	196.83	388.23
Area of coral cover	Area of coral cover (km²)	16.29	26.83
Caribbean	Dummy: 1 = Caribbean; 0 = other	0.16	0.37
Indian Ocean	Dummy: 1 = Indian Ocean; 0 = other	0.13	0.34
Southeast Asia	Dummy: 1 = SE Asia; 0 = other	0.17	0.38
Australia	Dummy: 1 = Australia; 0 = other	0.03	0.16
Diving	Dummy: 1 = diving; 0 = other	0.77	0.42
Snorkelling	Dummy: 1 = snorkelling; 0 = other	0.64	0.48
Fishing	Dummy: 1 = fishing; 0 = other	0.07	0.25
CVM	Dummy: 1 = CVM; 0 = other (travel cost method)	0.61	0.49

[28] Including Hawaii.

[29] http://data.worldbank.org/data-catalog/world-development-indicators.

Table 13: Estimated meta-analytic value function

Variable	Variable definition	Coefficient	Standard error
Constant		3.871***	1.087
Visits per day	Natural log of visits per day	−0.434**	0.174
Area of coral cover	Natural log of area of coral cover (km²)	0.451*	0.278
Caribbean	Dummy: 1 = Caribbean; 0 = other	1.482**	0.736
Indian Ocean	Dummy: 1 = Indian Ocean; 0 = other	2.932***	0.943
Southeast Asia	Dummy: 1 = Southeast Asia; 0 = other	1.456*	0.822
Australia	Dummy: 1 = Australia; 0 = other	0.065	1.087
Diving	Dummy: 1 = diving; 0 = other	−0.276	0.476
Snorkelling	Dummy: 1 = snorkelling; 0 = other	−0.980**	0.446
Fishing	Dummy: 1 = recreational fishing; 0 = other	0.131	0.491
CVM	Dummy: 1 = contingent valuation; 0 = other	−1.949***	0.449
Adjusted R2	0.41		
N	74		

*$p \leq 0.10$, **$p \leq 0.05$, ***$p \leq 0.01$

The estimated model broadly fits prior expectations. The estimated coefficient on the number of visitors to a reef has a negative sign and is statistically significant, suggesting that visitors prefer less crowded coral reefs. The area of coral cover has a positive effect on the welfare derived from a recreational visit. Visitors have a preference for coral reefs with larger areas. Regarding the results on the regional indicators, reefs in the Indian Ocean, Caribbean and Southeast Asia are all found to provide significantly higher recreational values than reefs in the U.S. (the omitted category in the set of regional dummy variables). The values of recreational visits to Australian reefs are not statistically significantly different from visits to U.S. reefs. Regarding the dummy variables indicating the principal recreational activity that is valued, only the estimated coefficient for snorkelling is statistically significant and indicates that the value of this activity is lower than for others.[30]

Regarding valuation methods, we find that contingent valuation (CVM) estimates are statistically significantly lower than estimates obtained using the travel cost (TCM) method. From a theoretical perspective we might expect CVM estimates to exceed TCM estimates, given that the former may include some element of nonuse value in addition to the direct use value of a recreational visit. On the other hand, TCM estimates for recreational visits that are part of a more complex multi-purpose trip, such as a vacation to a tropical island, may over-estimate the value of individual constituent activities (Armbrecht 2014). Empirical evidence with regard to the extent that these two methods produce similar results is somewhat ambiguous. Carson *et al.* (1996) review 83 valuation studies for

[30] The omitted category of reef-related recreation is a general category of "other" activities, including the viewing of coral reefs from boats. Our prior expectation is that the value of diving would be higher than other reef-related recreational activities. We do not, however, find evidence that the value of diving is different from recreational fishing or reef viewing. These activities can evidently also be of high recreational value.

quasi-public goods from which 616 comparisons of CVM and revealed preference (RP) estimates are made. The sample mean CVM/RP ratio is 0.89, with a 95% confidence interval of 0.81 – 0.96 and a median of 0.75. Although the results from this study show that RP methods produce higher value estimates than CVM, they also show that estimates from these two methods are within the same range. Mayor *et al.* (2007) compare TCM and CVM estimates specifically for recreational visits and find that the former tend to exceed the latter. Previous meta-analyses of the coral reef valuation literature have found similar results to those of the present study (Brander *et al.* 2007; Londoño and Johnston 2012).

5.3.5 Data and Scenario for Coral Reef Loss, 2000 – 2050

The next step in assessing the welfare change associated with the loss of coral reef area over the period 2000 – 2050 is to develop a database of coral reef ecosystems in Southeast Asia that contains information on the variables included in the visitor model and the meta-analytic value function. We then develop a baseline scenario for the change in the spatial extent of coral reef ecosystems in Southeast Asia for the period 2000 – 2050.

Individual ecosystem or patch-level data on coral reefs in Southeast Asia were obtained from the UNEP World Conservation Monitoring Centre (WCMC, described in Giri *et al.* 2011). For each of the 5290 coral reef patches in Southeast Asia that are included in the UNEP-WCMC database, we used a GIS to obtain information on the area of each coral reef and area of mangroves, population and gross cell product within 50 km.

We make use of the results of the Reefs at Risk Revisited assessment by the World Resources Institute (Burke *et al.* 2011) to define a baseline scenario for coral reef change for the period 2000 – 2050. This assessment provides a spatially explicit projection of the degree to which coral reefs are threatened. The threats included in the Reefs at Risk Revisited assessment are coastal development, watershed-based pollution, marine-based pollution and damage, over fishing and destructive fishing, thermal stress and ocean acidification. These local and global threats are combined into an integrated index representing the degree to which coral reefs are threatened. Threat levels are classified as low, medium, high, very high, or critical. The proportion of coral reefs in the low-or medium-threat categories declines over time, whereas the proportion of coral reefs that are highly, very highly or critically threatened increases dramatically. We used spatially differentiated change factors derived from the Reefs at Risk Revisited integrated threat data, combined with the patch-level data on coral reefs from the UNEP-WCMC, to calculate the change in area of each patch of coral reef for the period 2000 – 2050. The baseline loss of coral cover is presented in Figure 24.

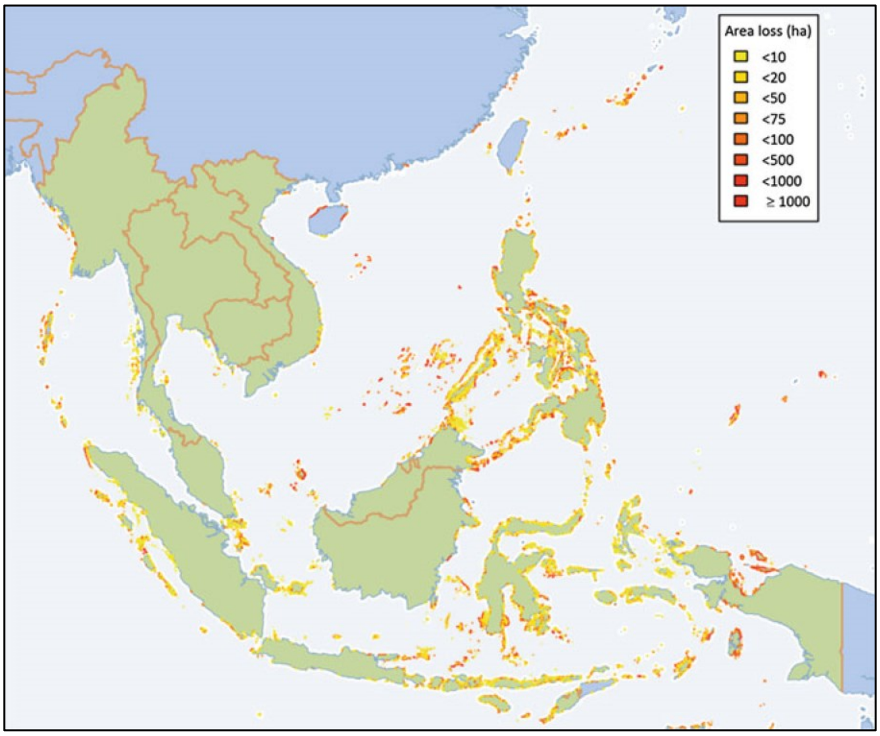

Figure 24: Change in area of coral cover 2000 – 2050 in Southeast Asia

5.3.6 Results and Value Maps

The final step in the assessment is to combine the models and data generated in the previous steps to produce estimates of the value of the loss in coral reef-related recreation under the baseline scenario.

At the level of individual patches of coral reef, patch-specific parameter values are substituted into the visitor model to estimate the number of visitors to each site. Visitor numbers are estimated for the year 2050 by using the areas of coral cover and mangroves existing in 2000 (i.e., under a conservation scenario) and the projected areas in 2050 (i.e., the baseline scenario). The difference between these two scenarios gives the estimated site-specific change in visitor numbers due to ecosystem degradation. The change in visitor numbers is represented in Figure 25 and is shown to be relatively insensitive to loss in coral cover. The average decrease in the annual visitation rate per site is only approximately 190 visitors. Nevertheless, there is substantial spatial variability across sites, due to both the underlying popularity of a site and the extent of change in the area of coral cover at that location. For example, the decrease in visitor numbers is shown to be higher for coral reefs on the east coast of Vietnam than for the west coast of Myanmar and Thailand.

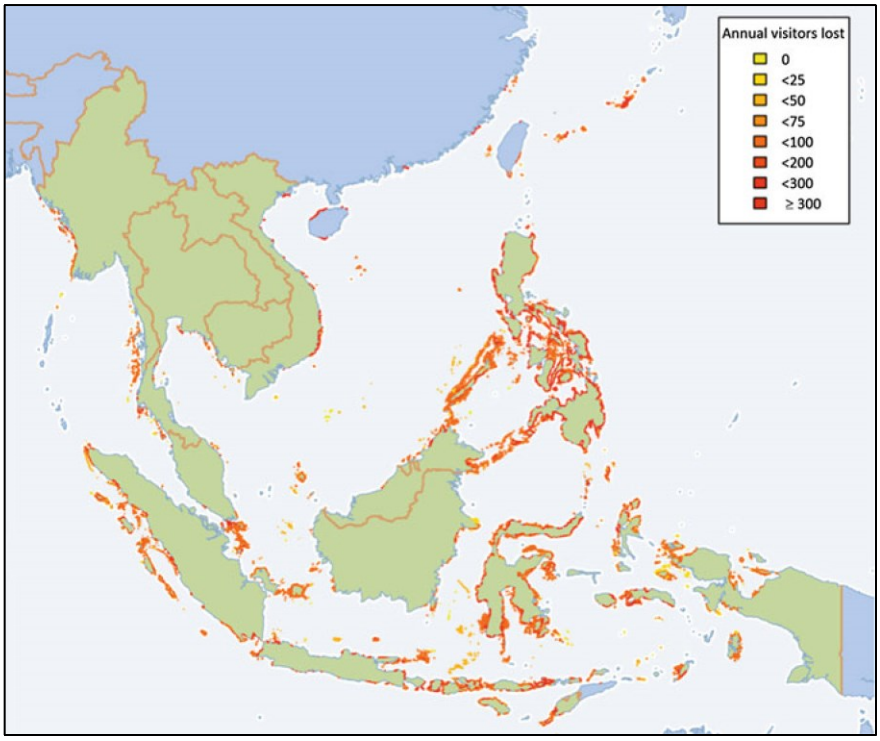

Figure 25: Change in coral reef-related recreation visits per day in Southeast Asia

The value per visit to each site is computed by substituting patch-specific parameter values into the meta-analytic value function. This is done using pre-and post-change areas of coral cover and visitor numbers in order to estimate the value of a visit to each site before and after ecosystem service degradation.

Two components of the change in welfare due to ecosystem degradation are then computed. The first component is the loss in consumer surplus associated with the decrease in the number of visitors. This is computed as the decrease in visitors at each site multiplied by the pre-change value per visitor (i.e., the loss in value to those that no longer visit). The second component is the loss in consumer surplus associated with the decrease in value of visits that still take place (i.e., visitors may continue to visit a site but derive lower utility per visit from doing so). This is computed as the decrease in value per visit at each site multiplied by the number of visitors under the degradation scenario. Lower-and upper-bound values are calculated using the 95% prediction intervals for each coral reef, which are computed using the method proposed by Osborne (2000). The prediction intervals provide an indication of the precision with which the estimated value function can predict out-of-sample values. The results are presented in Figure 26 and in Table 14, aggregated to the country level. For Southeast Asia as a whole, the annual loss in consumer surplus from reef-related recreation in 2050 due to coral reef degradation is approximately US$ 120 million (with a 95% prediction interval of US$ 3 million – 1.4 billion). The 95% prediction interval is very large and reflects the high uncertainty in estimating site-specific values per visitor day. The countries expected to suffer the highest losses are Indonesia and the Philippines, which have the largest areas of coral reef and numbers of reef-related recreational visits. There is considerable spatial variation in the change in value of reef-related recreation across sites reflecting differences in rates of coral cover loss, visitor numbers and values per visitor.

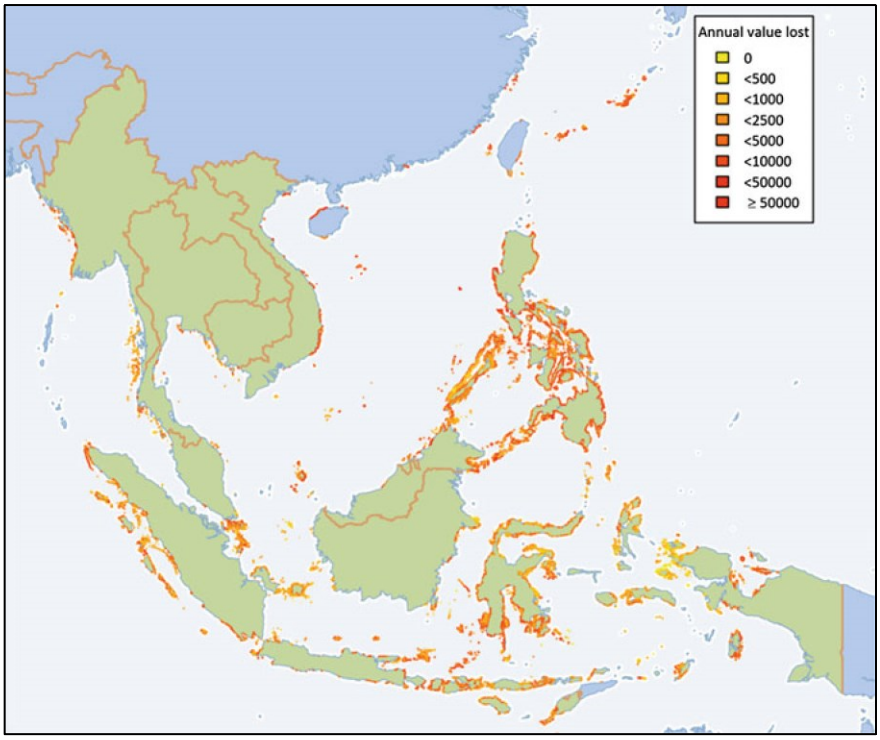

Figure 26: Loss in the annual value of coral reef-related recreation in 2050 due to policy inaction

Table 14: Change in consumer surplus of reef-related recreation in Southeast Asia caused by Ecosystem Degradation, 2050 (2007 US$)

Country	Value per visitor day	Total change in consumer surplus (000s)	Lower bound 95% prediction interval (000s)	Upper bound 95% prediction interval (000s)
Cambodia	11.20	−124	0	−1392
Indonesia	8.90	−59,468	−1099	−665,880
Malaysia	10.80	−3140	−280	−35,161
Myanmar	4.60	−2836	−253	−31,754
Philippines	6.50	−56,749	−5068	−635,440
Singapore	2.60	−176	−16	−1972
Thailand	5.80	−1936	−30	−21,680
Vietnam	4.00	−3577	−319	−40,058
Southeast Asia	6.80	−128,007	−2848	−1,433,337

It is important to note that the estimated welfare loss is only for the impact of coral reef degradation on the consumer surplus derived from reef-related recreation. The estimated values do not include producer surplus associated with reef-related recreation or impacts on other reef-related ecosystem services. The impacts on other ecosystem services provided by coral reefs, such as coastal protection and fisheries, are likely also to be substantial and possibly more sensitive to changes in coral cover.

5.4 Conclusion

This chapter illustrates the process of mapping ecosystem service values with an application to value changes in coral reef recreational values in Southeast Asia. This case study provides an estimate of the value of reef-related recreation foregone, caused by the decline in coral reef area in Southeast Asia under a baseline scenario of ecosystem degradation for the period 2000 – 2050. This value is estimated by combining a visitor model, meta-analytic value function and spatial data on individual coral reef ecosystems to produce site-specific values. The case study illustrates the data, methods and results of a value mapping exercise and allows several general conclusions to be drawn.

The estimated changes in visitors and values of reef-related recreation across Southeast Asia are not particularly high relative to their absolute values. Both visitation rates to coral reefs and values per visit are found to be relatively unresponsive to changes in the area of coral cover.[31] The aggregated loss of consumer surplus derived from reef-related recreation due to ecosystem degradation under the baseline scenario is therefore limited. The central estimate of annual loss in 2050 of US$ 128 million is not high, considering the size of the ecosystem providing the recreational services. The case study results do show, however, substantial spatial variation in the value of coral cover loss. This information can potentially be used in economic analyses for targeting conservation efforts to specific locations. With additional information on the spatial variability of conservation costs, a spatially explicit cost-benefit analysis could be conducted to identify the location of conservation efforts in the region that would generate the highest returns. Such an analysis could be useful in locating new protected areas or planning new tourism developments.

There are several important limitations to the case study that are worth noting. There is a substantial challenge in obtaining reliable spatially disaggregated data on visitor numbers and characteristics with which to estimate a visitor model. The Reef Check data that we use in the case study application are focused primarily on the status of the reefs themselves, rather than on visitor numbers or visitor characteristics. We are therefore unable to include potentially important variables describing visitor characteristics in the model, such as recreational activity, income, origin and travel time. Future research should aim to collect such visitor-level data and include it in the estimation of visitor models. The lack of visitor-level data also restricts the options for including visitor characteristics in the meta-analytic value function, since it is necessary to have policy site data on each explanatory variable included in value function. Information on the income of visitors as a determinant of recreational value is again notably absent.

A second important limitation of the case study application is the restricted extent to which the supply of the ecosystem service is modelled. The supply side of reef-related recreation is essentially modelled implicitly in the visitor function, i.e., coral reefs supply recreational opportunities to the extent that people want to visit them. This approach may be defensible in the case of a cultural ecosystem service such as recreation, but still neglects other potentially important ecosystem characteristics that may determine the provision of the service, such as coral and fish diversity or water clarity. The method makes the analysis relatively simple but sidesteps the greater complexity involved in modelling the ecological functioning that underlies the supply of most ecosystem services. In general, accounting for spatial variability in ecosystem service values requires a closer integration of the bio-physical

[31] The regional mean proportional changes in visitor numbers and value per visit are −6 and −12.5% for a −27% change in the area of coral cover.

assessment of ecosystem services into the valuation of ecosystem services. The disconnection between these steps in the ecosystem service assessment process remains challenging; future applications should attempt to better combine ecological and economic modelling of the determinants of ecosystem service values.

Third, the analysis of visitor behavior and recreational value does not account for the potential impact of changes to substitute (or perhaps complement) sites. The current model treats each site as independent, and does not allow for the possibility that simultaneous changes in the quality of multiple coral reef sites will influence visits and value in a way not captured through the aggregation of single-site estimates. To the extent that these cross-site effects are relevant, estimates may depart from those reported here.

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6 Spatial Dimensions of Recreational Ecosystem Service Values: A Review of Meta-Analyses and a Combination of Meta-Analytic Value-Transfer and GIS

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Abstract:

This paper investigates spatial determinants of recreational ecosystem service values by combining Geographic Information System (GIS) and meta-analysis, and by presenting the first review on meta-analysis studies in this field. Using meta-analytic value transfer, we map the spatial distribution of recreational values across Europe.

By combining meta-analysis and GIS we identify spatial biophysical and socio-economic determinants of recreational ecosystem service values. Nevertheless, comparing the results of past meta-analyses reveals difficulties in establishing robust relationships between spatial variables and recreational values per visit, as existing meta-analyses show contradicting results and methodological variables show stronger effects. Based on our findings we give guidance on how to improve geostatistical analysis within future meta-analyses on ecosystem service valuation studies.

Furthermore, we find that spatial variations of recreational visitor numbers are by far greater than variations of the value per visit. Therefore, we conclude that accurate estimates of visitor numbers are of greater relevance than accurate estimates of the value per visit.

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6.1 Introduction

The spatial assessment of ecosystem services (ESS) has gained increased attention in recent research and policy activities (Maes et al., 2012; Schägner et al., 2013; Crossman et al., 2013). The Strategic Plan for Biodiversity 2011–2020 of the Convention on Biological Diversity, adopted in 2010, calls for spatial ESS assessments and valuation to ''*be integrated into development plans to ensure that these ecosystems receive the necessary protection and investments*" (UNEP, 2013). The European Union (EU) has implemented this commitment within its Biodiversity Strategy, which requires member states to ''map and assess the state of ecosystems and their services in their national territory" and to ''*assess the economic value of such services*" (EC, 2011). Sustainable resource management strategies for nature areas require a comprehensive and spatially explicit assessment of their ESS values. Thereby, trade-offs and synergies from alternative land-use strategies can be identified, resources can be allocated more efficiently across space, and restoration prioritisation can be supported (Schägner et al., 2013).

Nature recreation represents a valuable ESS supplied by protected areas and the wider rural landscape. Humans enjoy nature areas for walking, hiking, biking, relaxing, experiencing and learning about nature and biodiversity. All this contributes to human well-being and environmental awareness. Nature recreation and tourism present an opportunity for rural economic development, by generating income and employment through visitors' expenditures. The value of nature recreation and its economic opportunities can be used as a strong argument for allocating financial resources towards nature conservation at different spatial scales (Balmford et al., 2015; Schägner et al., 2016; Fleischer and Tsur 2000; Jones et al., 2010).

The value of recreational ESS differs across space due to variations in the number of recreational visits and the value per visit (Brander et al., 2015; Schägner et al., 2013; Sen et al., 2014). In this paper we apply meta-analytic value transfer to explore spatial variations of the value per recreational visit and the importance of such variations in determining the total recreational value per hectare for different nature recreation sites.

Meta-analyses have become a widespread tool for investigating the effects of different valuation techniques, ecosystem features and socioeconomic characteristic on the value estimate, but also for meta-analytic value transfer. Several studies apply meta-analysis to studies on the value per recreational visit. In the first study we found, Smith and Kaoru (1990) analyse 186 value estimates from 77 studies in the U.S. As in most meta-analyses, their focus lies on identifying the effects of different valuation methodologies on the final value estimate. However, the valued good, the study area, is typically described only by some dummy variables indicating for example whether a forest, a lake or a national park has been valued or whether the site is located in a certain region. Similar meta-analyses have been conducted by Rosenberger and Loomis (2001) and Shresta et al. (2007) on valuation studies from the U.S., De Salvo and Signorello (2015) for Italy, Wang et al. (2013) for China and the U.S., Sen et al. (2011, 2012, 2014) on a global valuation dataset and Brander et al. (2007, 2015); Londoño and Johnston (2012) and Fitzpatrick et al. (2017) on global coral reef valuation studies.

Past studies have used different statistical model specifications, different valuation study datasets and different explanatory variables; however, the description of the valued good has remained rather rudimentary. Continuous explanatory variables on the characteristics of the study area and its context, which allow for a more detailed and smooth characterisation of the study areas, are less common. Only study area size is used in several studies (Brander et al., 2007; Zandersen and Tol 2009; Londoño and Johnston 2012; Brander et al., 2015; Fitzpatrick et al., 2017). Three studies (Zandersen and Tol

2009; Sen et al., 2011; Hysková, 2013) also used population density as continuous explanatory variable. In their meta-analysis on coral reef recreation valuation, Brander et al. (2007, 2015) describe the valued coral reef sites by their number of annual recreational visitors, and Londoño and Johnston (2012) and Fitzpatrick et al. (2017) by the share of live reef area. Hysková (2013) describes valued forest sites by the share of forest, and Zandersen and Tol (2009) by the share of open land. More than two continuous predictor variables on the study area and context characteristics are only used in the meta-analysis by Zandersen and Tol (2009). Besides the above mentioned, they describe valued forest sites also by the regional GDP/ capita, the latitude, a species diversity index and the tree age diversity.

Even though several meta-analyses have been conducted, still, ‘‘the absence of variables on site user socioeconomic characteristics and on supplementary site features (being un-reported in most primary studies), poses serious limitations to the use of this meta-analysis for benefit transfer exercises” (De Salvo and Signorello, 2015) and little is known about how biophysical site characteristics, socio-economic context characteristics and availability of substitutes affect estimated values. Simple dummy variables, which indicate whether the study site is for example a forest or not, are only a rough approximation. Typically, study sites comprise multiple land covers, and differ gradually by their recreational facilities and their socio-economic context.

To overcome these limitations, in this study, we combine meta-analysis of primary valuation studies on recreational values per visit with Geographic Information System (GIS). The GIS allows us to assess the biophysical and socio-economic characteristics of the study sites ex-post in an automated manner. When study sites are spatially located within a GIS environment, additional information can be extracted without time consuming consultations of primary studies, their authors and/or secondary sources. In addition, comprehensive spatial biophysical and socio-economic predictor variables allow us to extrapolate value estimates across space by applying meta-analytic value transfer across a larger area. Thereby, we map the recreational value per visit across European nature areas using several continuous biophysical and socio-economic predictor variables and show how values differ across space. We refrain from using regional dummy variables in order to identify the underlying effects of biophysical and socio-economic characteristics on the value per recreational visit. We compare our results to past meta-analyses by conducting the first review of literature in this field. We give a special focus to the uncertainties involved with identifying robust relationships between spatial variables and the value per recreational visit, and how analyses could be improved in the future. To estimate the total recreational value per hectare, we combine the predicted values per visit with predictions of recreational visitor numbers, which are based on a geostatistical model presented in Schägner et al. (2016). To identify the contribution of our meta-analytic value function to the total recreational value per hectare, we compare the results with a unit value transfer approach that is based on the mean value estimates of all primary valuations used in our meta-analysis. Results are illustrated using maps for Europe and for a case study area in Germany.

In the following section, we first describe the primary valuation data and then the additional spatial data used as predictors in our models. In section three we explain the statistical regression techniques applied and present our valuation models. The results are presented in section four and then discussed against the results of past meta-analyses in section five, followed by our conclusions.

6.2 Data

Our primary data are 245 estimates of monetary values per recreational visit for 147 separate nature areas in Europe. We obtained the data from 75 valuation studies using either Travel Cost Method

(TCM) or Contingent Valuation Method (CVM). These studies were identified through internet searches, a review of relevant literature and by contacting researchers involved in this field.[32]

We transfer all value estimates to € values and to the 2013 price level using purchasing power parity and country specific inflation data. From the total dataset we exclude one outlier (Lagoon of Venice, Italy), showing an extreme deviation of 60 times the mean value. The remaining 244 value estimates range from € 0.16 to € 64.7 per visit with a mean of € 7.17 and a median of € 2.8 and a mean relative deviation of 95% (see Table 15).

Table 15: Descriptive statistics of value per visit estimates (n = 244) in €, 2013.

Min.	1st Qu.	Median	Mean	3rd Qu.	Max.	St. dev.	mean relative deviation
0.16	1.54	2.8	7.17	7.76	64.7	10.98	95%

For each surveyed nature area we obtain or create a spatial layer in vector format with the area's boundaries. Some polygons were obtained from official sources (IUCN and UNEP-WCMC, 2015; EEA, 2013) or from the study authors; however, most polygons were drawn manually using ArcGIS, based on information supplied in the original valuation studies, the study authors or based on information from internet inquiries. In several cases, we were not able to get any approximation of the location and shape of the study area and thus could not include those studies in our data base. The study areas differ widely in terms of size, location, the estimated value per visit and ecosystem characteristics. The size of study areas ranges from 1.9 hectare for a small Nature Park east of Padova, Italy, up to 1.8 million hectare for the Jämtland mountain area in North Sweden. Most study sites are located in Western Europe (51%). The UK has the highest number of observations (81), followed by Italy (32), Ireland (28), Finland (27) and Germany (22). The distribution of the study sites is shown in Figure 27.

For the statistical regression analysis, we compile a number of predictor variables, divided into three categories: (1) study characteristics, describing the methodology of primary value estimation, (2) site characteristics, describing the study area itself and (3) context characteristics, describing the spatial and socioeconomic context of the study area. We select the variables based on a review of past meta-analyses on recreational valuation studies. A complete list of all predictors used in our analysis is presented in Table 16.

We consider three methodological characteristics, which we assume to have a strong influence on the valuation results (see also Table 16). First, we distinguish studies by their valuation method, which is relatively equally distributed with 140 studies using CVM and 104 using TCM. Second, we distinguish whether studies consider use values only (226) or if they consider use and option values (19). Finally, we use a factor variable (value-measure v/visit) to consider whether studies estimate values per visit, which is the case in the majority of studies (197), or if studies estimate the value per day visit, value per party visit or the value per month or year of access (48).

[32] Full bibliographic information of all studies used in the meta-analysis can be found in the SOM.

The main focus of our analysis, however, is to identify the effects of spatial determinants of recreational values in order to produce spatially distributed predictions. Therefore, we prepare several EU wide geospatial layers of site and context characteristics in raster format. Limitations in the availability, accuracy, comprehensiveness and consistency of Europe-wide datasets restrict our choice of predictors and may hamper the statistical analysis. We use available GIS datasets and, if necessary, process these layers to derive our predictor variable raster layers. GIS processing is done with ArcGIS 10.2.

Figure 27: Location of nature areas represented in the primary valuation data.

We use the following site characteristics in our regression analysis:

(1) Land cover: Based on the analysis of past meta-analyses of recreational valuation studies, we choose the following land cover/use classes as predictor variables for our analysis: share of all natural vegetation, agricultural area, grassland and forest. We do not have strong hypotheses regarding the effects of different land cover classes but expect more natural land covers may have positive effects on the value per visit. For forest we use the Joint Research Centre's forest cover map (EC, 2006) and compute the mean number of forest pixels (25 m resolution) per hectare that are classified as either coniferous or broadleaved forest within each study site. For other land cover types we used the CORINE dataset (EEA, 2006) to determine the percentage of land cover classes in the study sites. We

can only use a limited number of land cover combinations in our analysis because several CORINE land cover classes occur only rarely, which would result in many zero values in our spread sheet and, thus, the detection of significant effects would become more difficult and vulnerable to outliers. In addition, aggregates of land cover classes are often correlated with each other, and thereby can cause problems of collinearity.

(2) Land cover diversity: From the CORINE land cover dataset we compute the Simpson Diversity Index (Magurran, 1988) of land cover types within a 3 km radius for each pixel of a 100 m resolution raster map covering Europe. We expect that more diverse landscapes are perceived as more beautiful and may therefore positively affect the value per visit.

(3) Water bodies: We compute two 300 m resolution grids of the share of surface area covered with rivers and lakes or ocean using the Euro Regional Map (EG, 2010). Then we apply a kernel density function tool to compute the amount of surface covered with water within a 3 km radius of each pixel. The density function allows a water area, which is more afar, to be weighted less than a water nearby, thereby incorporating a distance decay effect. We expect that water bodies attract visitors and generate higher values per visit.

(4) Biodiversity: We use the total number of red list species in a study area as an indicator for biodiversity (IUCN, 2013). We assume that the presence of threatened species may attract visitors from distant locations and result in higher values per visit. In addition, we use a dummy variable to indicate whether at least 50% of the study site is designated as national park, here defined as protected areas that are either classified as IUCN Protected Area Management Category II (IUCN and UNEP-WCMC, 2015) or as national parks by national authorities.

(5) Climate: We use three climatic variables in our model, under the assumption that better climate in terms of higher temperature, less precipitation and more sunshine attracts visitors from distant locations for longer recreational trips. We use a dataset from Biavetti et al. (2014) indicating the mean number of days per year with maximum temperature above five degrees Celsius, and a similar dataset from Burek (unpublished) for the mean number of days per year with at least some precipitation and the mean hours of sunshine per day.

(6) Topography: We use the digital elevation map from the European Environment Agency (EEA, 2015a) for two topography indicators: (1) the slope value of the 100 m digital elevation map, and (2) the area visible from each pixel within a 30 km radius, computed using the viewshed tool. To accelerate the viewshed processing we aggregated the digital elevation map to a 1000 m resolution raster grid. We expect that mountain regions and regions offering large viewsheds have special attraction for recreation and generate higher values per visit.

(7) Trail density: We use trail density within 1 km radius as a proxy for overall recreational facilities, which may enhance the recreational experience. From the Open Street Map dataset (OSM, 2012), we extract all vector elements that can be classified as non-motorized traffic infrastructure. On a 100 m resolution we apply the line density tool to compute trail density using five OSM classes: trails, foot paths, bike paths, bridle paths and steps. The trails are weighted less with increasing distance from the pixel under analysis. (8) Street density: Similar to trail density we compute an indicator for street density for all minor roads (Tele Road Atlas road classes 4–6) based on the Tele Road Atlas dataset (TS, 2006). Roads are an important infrastructure for accessing remote locations and are thereby expected

to increase visitor numbers. However, since the relationship between road density, nature recreation and recreational values is unclear, our analysis has exploratory character.

(9) Accessibility: The number of people that can access a specific location within a certain time is likely to have an effect on the visitation rate (Schägner et al., 2016), which may in turn affect the quality of nature recreation negatively due to crowding effects (Kalisch, 2012). We use the weighted sum of the total population living within a 130 km radius around each pixel, using population data from Batista e Silva et al. (2013). In order to account for distance decay, we applied a Gaussian weight function, so that the population is weighted less with increasing distance from the pixel under analysis. The weight function was calculated so that 95% of its integral is located within the 130 km radius.

(10) Socio-economic effects: We use GDP per capita, the unemployment rate and the share of population with upper secondary or tertiary education as rough proxies for visitors' income and their recreational preferences. For these variables, we extract mean values for the last ten years (as far as available) and the highest data resolution available, which is either NUTS2[33] or NUTS3 level from the Eurostat database (EC, 2013).

[33] NUTS is referred to as Nomenclature of Units for Territorial Statistics, which is a hierarchical system defined by Eurostat for dividing up the EU territory in order to produce regional statistics at the resolution of different administrative levels.

Table 16: Predictor variables used in the regression analysis. See text for further explanations.

Type	Variables	Explanation*	Mean / Standard Deviation
Study Characteristics:	TCM	1 if TCM; 0 if CVM	0.43 / 0.5
	Use & option	1 if use value; 0 if use & option value	0.93 / 0.26
	V/visit	1 if Value/visit; 0 otherwise	0.81 / 0.4
Site Characteristics:	Ln (ha)	Natural log of the study site area in hectare	7.83 / 2.84
	Ln (sri)	Simpson Diversity Index of Corine land use/cover within a 3 km radius (100 m resolution raster)	1.1 / 0.31
	Ln (forest)	Natural log of the share of forest cover of the study area (100 m resolution raster)	1.76 / 0.81
	Ln (natural LC)	Natural log of natural land cover of the study area (100 m resolution raster)	1.98 / 1.62
	Ln (agriculture)	Natural log of agricultural land cover of the study area (100 m resolution raster)	2.1 / 1.55
	Ln (grassland)	Natural log of grassland land cover of the study area (100 m resolution raster)	1.44 / 1.35
	Ln (inland water)	Natural log of inland water body area within 3 km distance weighted by a kernel function (300 m resolution raster)	0.96 / 1.16
	Ln (ocean)	Natural log of ocean area within 3 km distance weighted by a kernel function (300 m resolution raster)	0.5 / 1.11
	Red list species	Total number of red list species found in study area (1 km resolution raster)	8,991 / 3,144
	National Park	1 if site is a national park; otherwise 0	0.19 / 0.39
	Rain days	Mean number of days with rain per year (1 km resolution raster)	144 / 34
	H sun/day	Mean hours of sunshine per day (1 km resolution raster)	4.19 / 1.12
	Days 50C	Mean numbers of days with an average temperature of above 5 degrees (1 km resolution raster)	304 / 53
	Viewshed	Area visible from each location within in a 30 km radius (1 km resolution raster)	276 / 214
	Slope	Slope (100 m resolution raster)	2.04 / 0.97
	Ln (trail)	Natural log of trail density within 1 km radius using density function in order to account for distance decay effect (100 m resolution raster)	1.37 / 0.97
	Ln (small roads)	Natural log of small roads density within 1 km radius using density function in order to account for distance decay effect (100 m resolution raster)	2.09 / 1.11
Context Characteristics:	Ln (population)	Population living within 130 km radius of the study area using a Gaussian weight function in order to account for distance decay (100 m resolution raster)	16.2 / 1.08
	GDP/capita	GDP/ capita in the NUTS 2 or 3 region in which the study area is located	25,768 / 6,593
	High education	Share of population with higher education in the NUTS 2 or 3 region in which the study area is located	70.4 / 11.5
	Unemployment	Unemployment rate in the NUTS 2 or 3 region in which the study area is located	6.24 / 3.29

* For all spatial predictors mean values per study site area are computed.

6.3 Methodology

Before the regression analyses we explore our data following the recommendations of Zuur et al. (2010) in order to gain initial insights into distributions and dependencies. For some predictors we use logarithmic transformations either because they show a skewed distribution or with the aim to approximately linearize an expected non-linear relationship. We test all our predictors for multicollinearity, but do not identify any concerns. One may expect high correlation between the variables of mean hours of sunshine per day and mean rainy days per year as well between GDP/capita, unemployment rate and share of population with higher education. However, our pre-analysis reveals only moderate correlations with a maximum of 0.5 between GDP/capita and the unemployment rate (see Figure A 6.1).

We apply a number of regression techniques to identify a model fitting the assumptions of linear regression best by evaluating residual plots and comparing the Akaike Information Criteria (AIC) and Bayesian Information Criteria (BIC). All models are estimated using the open source statistical software R. We start our analysis with a general linear regression (fixed effects model), but it shows a wider spread of the residuals for large fitted values, and therefore a violation of the homogeneity assumption. We control for this effect by using a linear log transformed model of the following form:

$$\mathrm{Ln}(VV_i) = \alpha + \beta * X_i + \mu_i \qquad \text{where} \qquad \mu_i \sim N(0, \sigma^2)$$

VV stands for the dependent variable (the monetary value per recreational visit), α is a constant, β represents a vector of parameters, X is a vector of explanatory variables and μ is the residual, which is normally distributed with mean of zero and variance σ. The results (AIC: 792.3, BIC: 878.9) are shown on the left (columns 2–4) in Table 17.

We validate our final model against the assumptions of linear regression analysis. Therefore, we plot our residual against fitted values and against each predictor used in the model as well as predictors not used in the model. One concern is the potential for spatially correlated residuals. As several valuation studies use different valuation methodologies to value recreation at the same site, it cannot be assumed that these observations are independent. Therefore, we use a mixed model[34] introducing the study site as random intercept[35] . However, as the introduction of the study site ID as random intercept has almost no effect on the model results and did not improve the AIC and BIC (789.4 AIC and 879.4 BIC), we abandon this approach. Author effects is another concern because several authors contribute multiple valuations to our data base and their specific approaches may influence the studies result. We therefore introduced the most common authors as random intercepts in a mixed model of the following form:

[34] In other disciplines, mixed models are also referred as to multilevel analysis, nested data models, hierarchical linear models, and repeated measurements.

[35] As almost no study investigated several study sites, study ID and study site ID can be considered to be equivalent.

$$\text{Ln}\left(VV_{ij}\right) = \alpha + \beta * X_{ij} + \gamma_j + \mu_{ij} \qquad \text{where} \qquad \mu_{ij} \sim N(0,\ \sigma_{\mu}^2) \qquad \text{and} \qquad \gamma_j \sim N(0,\ \sigma_{\gamma}^2)$$

The random effect is specified by γ_j, representing the correlation of observations from the same sites which is normally distributed with mean of zero and variance. The mixed effects model improves the model considerably the AIC and BIC (AIC: 760.3, BIC: 850.3). We then added the site ID as nested random effect within the author effect (multilevel approach), but again it had only a very limited effect on the model result (AIC: 759.7, BIC: 853.0) while adding complexity. Thus, we abandon this approach. The results of the mixed effects model with first author as random effect and the fixed effects model are shown in the right of Table 17.

For both models we conduct stepwise variable selection using maximum likelihood and restricted maximum likelihood estimators to compare AIC and BIC and the likelihood ratio test until all remaining variables are significant at the 0.1 level (see Table 18). We validate our final model against the assumptions of linear regression analysis. Therefore, we plot the residuals against fitted values and against each predictor. We do not identify any linear or non-linear patterns of concern. In addition, to investigate the validity of our models, we estimate our final model for subsets of our primary data containing either only observations from specific countries or estimated by specific valuation methods (TCM and CVM). The final model residuals are plotted against each predictor separated by country to investigate regional differences in the estimated effects (see Appendix 6.9). Model and validation plots are created using statistical software R and lme (Bates et al., 2015), lattice (Sarkar 2015, 2), sp (Pebesma et al., 2015), visreg (Breheny and Burchett 2017) and gstat package (Pebesma and Graeler 2015).

We use the model characterised by the lowest AIC and BIC values — the mixed effects model after variable selection — for predictions and map values per recreational visit across rural Europe on 1 km² resolution. The maps indicate how the value per visit differs across space. The mean value of an area can be used as indicator for the value per visit for certain recreational sites. We use a dataset of urban morphological zones (EEA, 2015b) to cut out all urban areas because our primary data covers only non-urban ecosystems. To analyse a realistic policy scenario in more detail we zoom into a proposed new national park (NP) in the western part of Germany (Teutoburger Forest). The area of this proposed NP is approximately 200 km² and comprises a forested mountain range and a heathland, which has been used as an army base in the past. It is already largely protected and has been proposed for NP designation (NABU, 2015).[36]

In a second step, we combine the predicted values per visit with a prediction of total annual recreational visits per hectare, based on a geostatistical model presented in Schägner et al. (2016). By multiplying the values per visit with the number of visits per hectare we obtain the total annual recreational value per hectare and the relative spatial variation. In the study of Schägner et al. (2016), the predicted visitor numbers per hectare are combined with the mean value estimate per visit of all primary valuation studies used for the meta-analysis in this paper. Whereas the recreational value map presented in Schägner et al. (2016) allows only for spatial variations of the number of visits per

[36] Additional statistical analyses in the SOM indicate that there are no significant country specific differences in the effect of the predictors on the value per visit in Germany. If there were more primary valuation studies available per country, it may prospect for future research to estimate country specific value transfer functions.

hectare (the value per visit is constant), the approach presented in this study also allows for spatial variations of the value per visit. We then compare the mapped values to investigate how the spatial variation of visits per hectare and the spatial variation of the value per visit contribute to the overall variation of the recreational value per hectare.

6.4 Results

In the following sections, for purposes of cross validation, the signs and levels of statistical significance of all spatial variables used in our models are compared to the results of other meta-analyses of recreational values per visit. A summary of the results for selected spatial predictor variables for which several estimates exists is presented in Table 19. A comprehensive summary of the reviewed meta-analyses is presented in Table A6.3 in the appendix of this chapter.[37] Thereafter, we use the model evaluated best to predict recreational values across Europe.

6.4.1 Model Results and Comparison with other Meta-Analyses

For the full fixed effects model, we find eight of the predictor variables to be statistically significant at the 0.1 level and an adjusted R^2 of 0.44, which is comparable to other meta-analyses (Brander et al., 2015; Brander, Van Beukering, and Cesar 2007; Ghermandi and Nunes, 2010). Some comparable meta-analyses have found a higher R^2 of up to 0.85 (Zandersen and Tol, 2009) or 0.91 (Hysková, 2013)[38] . The use of closely similar underlying valuation studies, such as those focussing on forest recreation only, and the inclusion of detailed methodological variables tend to improve the models' explained variance. However, the focus of this study is to identify spatial predictor variables of the value per recreational visit across a variety of sites. The results of our model are shown in Table 17 and 18. Figure A 6.2 shows box plots of the model's residuals for the studies conducted by each author. It indicates that the residuals of our fixed effects model depend on the primary valuation studies' authors. This may result in underestimated standard errors and inflated p-values. Some predictors may only show significant effects due to correlations across the results of the same author. By introducing a random effect accounting for correlations across studies from the same author (shown in the right of Table 18) some variables (grassland) become non-significant and others become significant (value-measure v/visit). The AIC and BIC values decrease considerably indicating superiority of the model. The residual variance (RV) is 0.81 and variance of the random intercept (VRI) is estimated to 0.61, which indicates how the intercepts for the studies of the different first authors are normally distributed around the intercept of all observations.

[37] Further meta-analyses on recreational values (Ghermandi and Nunes, 2013; Ghermandi and Nunes, 2010; Giergiczny et al., 2014) are not considered in this study because the dependent variable is not the value per visit, but the value per ha. Therefore, the results are not comparable as they may reflect more the spatial variation in the number of visits than the variation in of the value per visit.

[38] None of these studies specifies whether they report a multiple R^2 or an adjusted R^2.

Table 17: Mixed and random effect model with the ln of the value per visit as dependent variable (€, 2013) and first author as random intercept in the mixed effects model.

	Linear fixed effect model			Linear mixed effect model		
Variable	**Coefficient**	**p-value**		**Coefficient**	**p-value**	
Intercept	4.24	3.7e-02	*	4.28	4.8e-02	*
TCM	0.73	2.8e-06	***	0.69	<1e-16	***
Use & option	-0.16	0.53		0.10	0.67	
V/visit	-0.20	0.32		-0.50	2.8e-02	*
Ln (ha)	6.1e-02	0.10	•	8.7e-02	2.6e-02	*
Ln (sri)	-0.10	0.65		2.2e-02	0.92	
Ln (forest)	-0.25	5e-02	*	-0.20	0.10	•
Ln (natural LC)	1.5e-02	0.82		-3.6e-02	0.59	
Ln (agriculture)	-6.3e-03	0.92		-5.6e-03	0.92	
Ln (grassland)	-0.19	1e-02	*	-0.10	0.16	
Ln (inland water)	9.8e-02	0.14		1.6e-02	0.80	
Ln (ocean)	5.5e-02	0.41		1.4e-02	0.83	
Red list species	-4.1e-05	0.23		0.0e+00	0.99	
NP	0.21	0.29		-1.3e-02	0.94	
Rain days	-1.1e-02	3.1e-03	**	-5.7e-03	9.1e-02	•
H sun/day	8.7e-02	0.37		0.10	0.32	
Days 5	2.9e-03	0.19		3e-03	0.25	
Viewshed	-4.6e-04	0.15		-3e-04	0.31	
Slope	0.19	5.4e-02	•	0.18	5.2e-02	•
Ln (trail)	6.7e-03	0.81		-9.3e-03	0.72	
Ln (small roads)	6.9e-02	0.31		1.9e-02	0.76	
Ln (population)	-0.30	4.7e-03	**	-0.29	4.9e-03	**
GDP/capita	6.7e-06	0.66		7e-06	0.61	
High education	1.4e-02	0.21		7e-03	0.55	
Unemployment	0.55	1.6e-02	*	0.22	0.38	
	AIC: 792.3			AIC: 760.3	VRI = 0.61	
	BIC: 878.9			BIC: 850.3	RV = 0.81	

We report significance levels by indicating p-values of up to 0.001, 0.01, 0.05 and 0.1 by "*", "**", "*" and "•". Study site as random or nested random intercept was excluded from the model due to pre-analyses.**

Study characteristics:

The intercept shows the strongest effect on the final value estimate in all models (indicated by the beta-coefficient). The most important predictor is the valuation method TCM, which shows a significant positive effect in all models before and after variable selection. This confirms results from Brander et al. (2015); Brander et al. (2007); Hysková (2013); Sen et al. (2011, 2014) and Shresta et al. (2007) and may reject the concern about the hypothetical nature of CVM leading to over valuation as compared to revealed preference valuation methods (Carson et al., 1996). Whether the study assesses ''*use values*'' or ''*use and option values*'' does not show a significant effect in our analysis. However, only a small share of all studies in our database considers option value in the valuation approach, and it may therefore be difficult to identify a significant effect. Whether the study estimates the value of a single visit or another valuation measure (value per party visit, month of access etc.) does show significant negative effect in the mixed effects model, an outcome confirming expectations. It is however not significant in the fixed effects model.

Study area and context characteristics:

A positive significant effect is identified for the study area size in all our models. This confirms the results of Brander et al. (2015) and Brander et al. (2007) (significance level of *, ***)[39] for coral reef sites, but contradicts results of Zandersen and Tol (2009), who find a significant negative effect (***) for the study area size on forest recreation values in one of their models, but positive non-significant in two of their models. For coral reef recreation, Fitzpatrick et al. (2017) find positive effects in ten of their 18 models, of which five are significant (*, **, **, **, ***). Two of the remaining eight models with negative signs are significant (*, **). Londoño and Johnston (2012) find negative non-significant effects of the coral reef study area size. Summing up the results of our and reviewed meta-analyses, 18 parameters show positive signs of which 10 are significant at the 0.1 level and 14 show negative signs of which six are significant.

The share of forest cover shows significant negative effects in all our models. Contrary to our findings, Hysková (2013) reports a significant positive effect (**) for the share of forest cover on the estimated value of forest recreation, but the effect is only significant in one of her two models. Zandersen and Tol (2009) find a positive effect (***) of the fraction of open land (which is negatively correlated with forest cover in our database, -0.7) in their meta-analysis on forest recreation values. This result seems more reasonable than the one of Hysková (2013) given that their focus is on forest recreation and thus, open land is likely to be scarce and forest cover abundant in the valued forest areas. Other studies use a dummy variable to define whether the valued site is a forest or woodland. Smith and Kaoru (1990) find negative effects of sites being a forest in 10 of their 12 model specifications of which six are significant (*, •, **, *, *, *) and positive non-significant effects in the remaining two. Shresta et al. (2007) find a significant positive (*) effect of forest sites in their model for the Southeast of the U.S. (an area with relatively high forest cover; NASA, 1999), whereas their model on the Intermountain West (an area with relatively low forest cover) shows a significant negative effect (*). The signs of these two effects seem contrary to the theoretical expectation that scarce land uses would have relatively higher values. Sen et al. (2014) and Wang et al. (2013) find significant positive (*, **) effects for a factor variable ''woodlands and forests'' analysing a global (Sen) and a Chinese and U.S. (Wang) valuation dataset. Rosenberger and Loomis (2001) find a negative non-significant effect for North

[39] We report significance levels of other studies' findings in brackets by indicating p-values of up to 0.001, 0.01, 0.05 and 0.1 by ''***'', ''**'', ''*'' and ''•''.

American forest in general and a negative significant effect (*) for forest managed by the U.S. Forest Service. Analysing Italian valuation studies, De Salvo and Signorello (2015) find four times a positive effect and four times a negative effect for woodland sites, one of each being significant at the 0.01 level (**). Besides, Sen et al. (2011; 2012) find positive significant effects for two dummy variables summing up farmland, woods and grassland, as well as farmland and woods (**, *) for a global dataset, but these combinations of land covers are not very specific. The negative sign we and most of the other studies find for forests may be surprising as forests are considered to be of particular recreational value and a lot or recreation research focusses on forests (Zandersen and Tol, 2009; Ankre and Fredman, 2012; Bateman and Jones, 2003). The analysis of the residuals of our models does not indicate non-linear effects and the study sites in our primary valuation database cover a broad range of share of forest cover ranging from zero to above 90. Nevertheless, again the contradicting results of different studies indicate difficulties in identifying an overall effect. Of all meta-analyses, 13 parameters have a positive signs with six being significant, but 20 have a negative sign with 12 being significant.

The availability of water cover (either inland or ocean) shows positive effects in our models, but they are not significant. Other meta-analyses use only dummy variables on the availability of water cover. For a global valuation dataset, Sen et al. (2012, 2014) find significant positive effects (**, *) for a factor variable on coastal marine and a non-significant positive effect for a variable on freshwater and flood plains (Sen et al., 2014) and for freshwater, marine and coastal (Sen et al., 2011). Shresta et al. (2007) and Rosenberger and Loomis (2001) find negative effects for a factor variable on lakes for modelling recreational values across the entire U.S. (*) and the Northeast of the U.S. (*) and North America (*). In addition, Shresta et al. (2007) find a positive effect (*) of rivers for the entire U.S. and the Intermountain West (*), a region with relatively low mean temperatures, but a negative effect (*) for the Southeast (a region with high mean temperatures). However, water recreation may be more attractive for warmer regions. Rosenberger and Loomis (2001) find a positive significant effect for the factor river (*) and a negative significant effect for lakes (*). In China and the U.S., Wang et al. (2013) finds a negative effect for sites being a lake or a wetland (•), a significant positive effect for rivers (**) and costal marine sites (**), but a negative effect for beaches (*). Smith and Kaoru (1990) find a negative effect for lake sites in the U.S., which is significant in 11 of their 12 model specifications (**, *, ***, *, **, *, ***, *, *, ***, *). For rivers they find significantly negative effects in 10 models (**, •, *, •, *, •, •, *, •) and positive in the remaining two, but none of the positive parameters is significant. Of all analyses considering water 15 show positive signs (7 significant) and 27 negative signs (23 significant). Interestingly, all negative parameter signs come from dummy variables.

Grassland shows a negative effect in our analysis, which is significant in the fixed effects model, but not in the mixed effects model before variable selection. Grassland is not analysed separately in any of the meta-analyses that we review. Sen et al. (2011) find a significant positive effect (*) for a factor variable on whether the site is of grasslands, farms or woods. This may contradict our findings of negative signs for forest and grassland, but is only partly comparable. The share of agricultural area shows a negative sign in our model, but is far from being significant. Zandersen and Tol (2009) find a positive effect (***) for the fraction of open land, which is more or less the equivalent to the sum of the variables on grassland and agricultural area used in our models. They focus however on forest recreation only. Sen et al. (2012) finds a positive significant effect for sites being farmland or woods (***). As wetlands are rare among the study sites in our database, we did not consider this land cover as a predictor in our model, but some other meta-analyses do. Globally, Sen et al. (2014) also find a significant positive (**) effect for study sites that are wetland and Sen et al. (2012) for sites being either a wetland or a freshwater. De Salvo and Signorello (2015) find a positive (**) as well as a negative (**) effect in four of their different model specifications for Italy. One of each is significant at the 0.001 level.

The number of days with precipitation has a significant negative effect on the value per visit in all our models. Unsurprisingly, recreational visitors prefer not to get wet in the rain. However, the variable could also pick up some regional or other unobserved effects. Typically, climatic patterns do not change much across small areas. In Europe, rainy regions are located in particular along the Atlantic and North Sea coasts. A lot of study sites in our valuation database that show a lot of rainy days are located in Ireland, the West of the UK and in Brittany, France. We investigate the models' residuals against the number of rainy days in depth but can identify neither an overall nor a country specific significant trend (see Figure A 6.6). To our knowledge, there is no other meta-analysis on the recreational value per visit accounting for climatic variables.

The mean slope value of the study sites — an indicator for mountainous areas — shows a significant positive effect in three of our models, but not in the fixed effects model after variable selection. This indicates that people attach greater value to visiting mountains relative to other types of landscape. Our findings partly confirm the results of Sen et al. (2011, 2012, 2014), who report significant positive effects (*) for sites in mountains and heathlands, even though heathland may not share many characteristics with mountains.

On the contrary, viewshed shows a negative but not significant effect in our model. Recall that viewshed is not correlated with mean slope. High viewshed values are found for example in wide valleys offering broad panoramas, whereas areas with high slope values tend to have low viewshed values as other mountains tend to be in the line of sight.

The availability of trails — an indicator of recreational facilities — proves to be a strong significant predictor of recreational visitor numbers to European National Parks in Schägner et al. (2016). However, it does not show significant effects in any models of this study, and signs differ between the fixed and mixed effects model. Shresta et al. (2007) do find a significant negative effect for study sites having developed facilities (*). It is however not clear how sites are classified as having either developed facilities or not. The availability of small roads, which also has a strong effect in attracting visitor numbers (Schägner et al., 2016), does not show a significant effect on the value per visit.

Population pressure shows a strong and significant negative effect in all our models. This could indicate that people prefer nature recreation in areas with lower population density. High population pressure may decrease the quality of the recreational experience, for example diminishing the natural character of the landscape. Nevertheless, findings reported in the literature are again conflicting. For Europe, Hysková (2013) finds a negative effect (***) of population density, whereas Zandersen and Tol (2009) report significant positive effects for two models (**, ***). Globally, Sen et al. (2014) find a positive effect (***) on the value per visit estimated for sites located in urban fringes, which partly contradicts with our findings because sites located in proximity to cities would show high population pressure and probably also high population density values.

GDP/capita and share of the population with higher education show positive effects in our models, but the effect is not statistically significant. Zandersen and Tol (2009) report a non-significant positive effect of GDP/capita in one model, but significant negative effect (***) in another model. Unemployment shows a significant positive effect in all our models, which contradicts our expectations. However, a possible explanation could be that rural areas attract visitors from further away and that these visits incur higher values, although these rural areas tend to have higher unemployment rates (Copus et al., 2006). Nevertheless, it has to be considered that data on unemployment is only available at the NUTS2 or 3 level, and not spatially explicit. The average unemployment rate for all years available in EUROSTAT shows high values in our database for sites located in Finland, France, Germany, Spain and Sweden and low values for Italy. Again, the variable could also pick up some regional or other unobserved effects, but we could not identify any systematic pattern allowing for an explanation.

Table 18: Linear fixed and mixed effects model after stepwise variable selection with the ln of the value per visit as dependent variable (€, 2013) and first author as random intercept in the mixed effects model.

	Linear fixed effect model			**Linear mixed effect model**		
Variable	**Coefficient**	**p-value**		**Coefficient**	**p-value**	
Intercept	6.10	3.5e-06	***	5.13	3e-04	***
TCM	0.77	9.4e-09	***	0.68	<1e-16	***
V/visit	—	—	—	-0.35	5.1e-02	•
Ln (ha)	8.9e-02	4e-04	***	7.3e-02	8.1e-03	**
Ln (forest)	-0.17	4.8e-02	*	-0.18	4.7e-02	*
Ln (grassland)	-0.19	5e-04	***	-0.11	6.4e-02	•
Rain days	-6.3e-03	2.1e-03	**	-5.9e-03	7.4e-03	**
Slope	—	—	—	0.15	3.4e-02	*
Ln (population)	-0.32	2.4e-06	***	-0.23	9e-04	***
Unemployment	0.42	3.9e-03	**	0.27	9.2e-02	•
	AIC: 697.9			AIC: 661.1	VRI = 0.57	
	BIC: 729			BIC: 702.5	RV = 0.79	

We report significance levels by indicating p-values of up to 0.001, 0.01, 0.05 and 0.1 by "*", "**", "*" and "•".**

We could not identify significant effects for diversity indicators in our models, neither for land cover diversity, nor for red list species. Similarly, Zandersen and Tol (2009) find a non-significant negative effect of species diversity. They do find, however, a significant negative effect (***) of tree age diversity in one of their models. Whether the site is a national park does also not show a significant effect in our models, and signs in the mixed and fixed effects model differ. Sen et al. (2011) also report a non-significant positive effect for designated areas. It is however not clear how they define designated areas.

In addition to the variables used in this study, past meta-analyses typically use spatial dummy variables in their models, for example whether the specific study site is located in a specific region or dummy (Zandersen and Tol 2009; Brander et al., 2015; Brander et al., 2007; Sen et al., 2014, 2011; Hysková, 2013). We, however, focus on identification of significant spatial biophysical and socio-economic explanatory variables. Including spatial dummies in our model would diminish the part of the variance that can be explained by such variables, even though it might improve the overall fit of our model. Our residuals do not show significant trends with respect to the different countries.[40] However, we do find evidence of a slight spatial autocorrelation in our residual.[41] There is a spatial correlation, which decreases by distance up to about 650 km and a nugget of about 0.4, which indicates micro variability or measurement errors. The effect may bias standard errors, p-values and parameter estimates and it can increase type I errors (Bivand et al., 2013; Dormann et al., 2007; Legendre 1993). In consequence, and due to the uncertainties in the results of meta-analyses on recreational values per visit, they are to be interpreted with caution (see also Table 19).

Table 19: Summary of meta-analyses results for selected spatial predictor variables (total / dummies / continuous).

Predictors *	Number of parameter estimates	Number of data sets**	Negative signs	Positive signs	Significant negative signs (0.1)	Significant positive signs (0.1)
Forest & woodland***	33 / 25 / 8	9 / 6 / 3	20 / 16 / 4	13 / 9 / 4	12 / 8 / 4	6 / 4 / 2
Water cover	42 / 38 / 4	6 / 5 / 1	27 / 27 / 0	15 / 11 / 4	23 / 23 / 0	7 / 7 / 0
Inland water	35 / 33 / 2	6 / 5 / 1	26 / 26 / 0	9 / 7 / 2	22 / 22 / 0	4 / 4 / 0
Marine coastal	7 / 5 / 2	3 / 2 / 1	1 / 1 / 0	6 / 4 / 2	1 / 1 / 0	4 / 4 / 0
Share of live reef area	20 / - / 20	1 / - / 1	1 / - / 1	19 / - / 19	1 / - / 1	18 / - / 18
Artificial or natural reef	18 / 18 / 0	1 / 1 / 0	0 / 0 / 0	18 / 18 / 0	0 / 0 / 0	14 / 13 / 0
Protected area	62 / 62 / 0	7 / 7 / 0	19 / 19 / 0	43 / 43 / 0	6 / 6 / 0	24 / 24 / 0
Study area size	32 / - / 32	4 / - / 4	14 / - / 14	18 / - / 18	6 / - / 6	10 / - / 10
Population density	11 / - / 11	4 / - / 4	5 / - / 5	6 / - / 6	5 / - / 5	3 / - / 3

*** We consider only predictors that can be clearly attributed to a land cover type. Combinations such as grassland, farm & wood are ignored.**

**** We consider Brander 2007 and 2015, Sen 2011, 2012 and 2013 as well as Londoño and Johnston 2012 and Fitzpatrick et al. 2017 to be the same data set as they do differ substantially.**

***** Open land within forest sites is considered to be the antonym of forest and woodland.**

6.4.2 Model Predictions

The map presented in Figure 28 was generated by using the best model as indicated by the lowest AIC and BIC values to map values per recreational visit across the entirety of rural Europe as well as the proposed national park Teutoburger Forest, Germany. Across Europe, the predicted value per visit ranges from € 1 to € 45 with a mean relative deviation of 39% (standard deviation 4.3). The mean relative deviation is relatively low as compared to the mean relative deviation of the predicted number of visits found in Schägner et al. (2016), which is more than 4 times higher. The distribution of the

[40] We do find however a significant effect for the Netherlands in our residuals. However, there is only one observation from the Netherlands in our data base. It could have probably been excluded from the model, as being an outlier.

[41] Figure A 6.3shows an experimental variogram of the residuals from the full mixed effects model.

predicted values is skewed with skew value of 1.4[42], and the mean value per visit is € 8.34 and the median is € 7.27. About 73% of the predicted values are smaller than € 10 and less than 2% are above € 20.[43] High values per visit are found in the mountainous and sparsely populated North of Scandinavia as well as in the dry southern part of Europe, which show little forest and grassland cover and high slope values. Low values per visit are found in the highly populated and flat areas of Western Europe such as the Netherlands, UK and Germany. Remote places may be characterised by higher values per visit as they require longer travel distances on average and, therefore, trips tend to be longer (to justify the effort to get there). In consequence, the travel cost valuation method results in higher values.

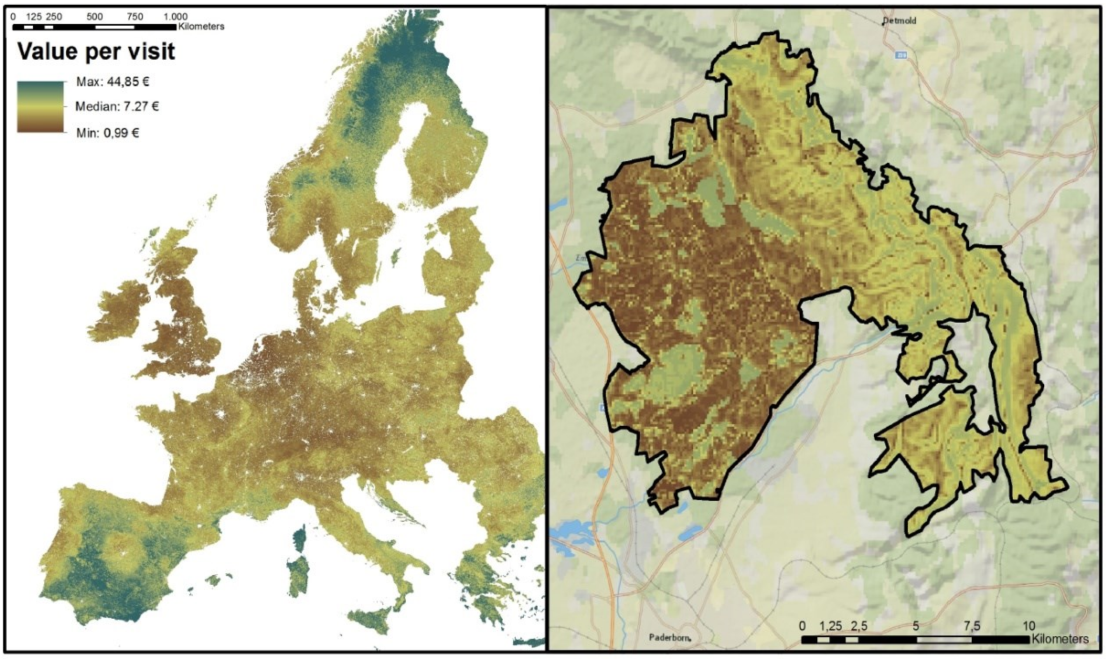


Figure 28: Predicted values per visit based on a meta-analytic value transfer function for Europe and a proposed national park in Germany.

[42] The skew value indicates how far a distribution is from being symmetrical. Negative values indicate a tail to the left and positive values indicate a tail to the right. Anything greater than 1 (or less than 1.0) indicates a distribution far from symmetrical. For further information please consult (Revelle, 2015).

[43] Note that for illustrative purpose, each map colour covers the same amount of map pixels, but not the same value range because skewed distribution maps using the same value range per colour would be dominated by few colours, which makes spatial differences difficult to distinguish.

The proposed national park Teutoburger Forest in the West of Germany shows a predicted mean value of € 3.37 per visit (median € 3.35) with a range of the predicted values per pixel from € 1.48 to € 6.69. Low values are found in the flat area in the West, whereas high values are found in the mountainous East. The predicted value is lower than the average prediction across all of Europe, because the Teutoburger Forest is characterised by an above average population pressure, number of rain days, grassland and forest cover as well as a below average slope value.

To obtain the annual recreational value per hectare (Figure 29b), we multiplied the spatially explicit prediction of the value per recreational visit with spatially explicit predictions of the annual number of visits per hectare (based on a geostatistical model presented in Schägner et al. (2016)). The results are compared to an estimation of the annual recreational value per hectare based on the same visitor predictions, but combined with the constant arithmetic mean value per visit of all observations of the primary valuation studies analysed in this study, which amounts to € 7.17 (Figure 29a). Whereas the spatial variations of the recreational value per hectare in Figure 29a) depend only on the spatial variation in the number of visits, the values in Figure 29b) also incorporate spatial variations of the value per visit. Due to the multiplication with a constant mean value per visit, the unit value approach results in a map strictly proportional to the prediction of the number of visits (Schägner et al., 2016). The predictors Schägner et al. (2016) used to map the number of recreational visits (including their signs indicated in brackets) ordered by their relative importance specified by the beta coefficients are: Trails density (+), population pressure (+), national park substitute availability (-), forest cover (-), water bodies (+), temperature (+), small road density (+), land cover diversity (+), and wetland cover (-).

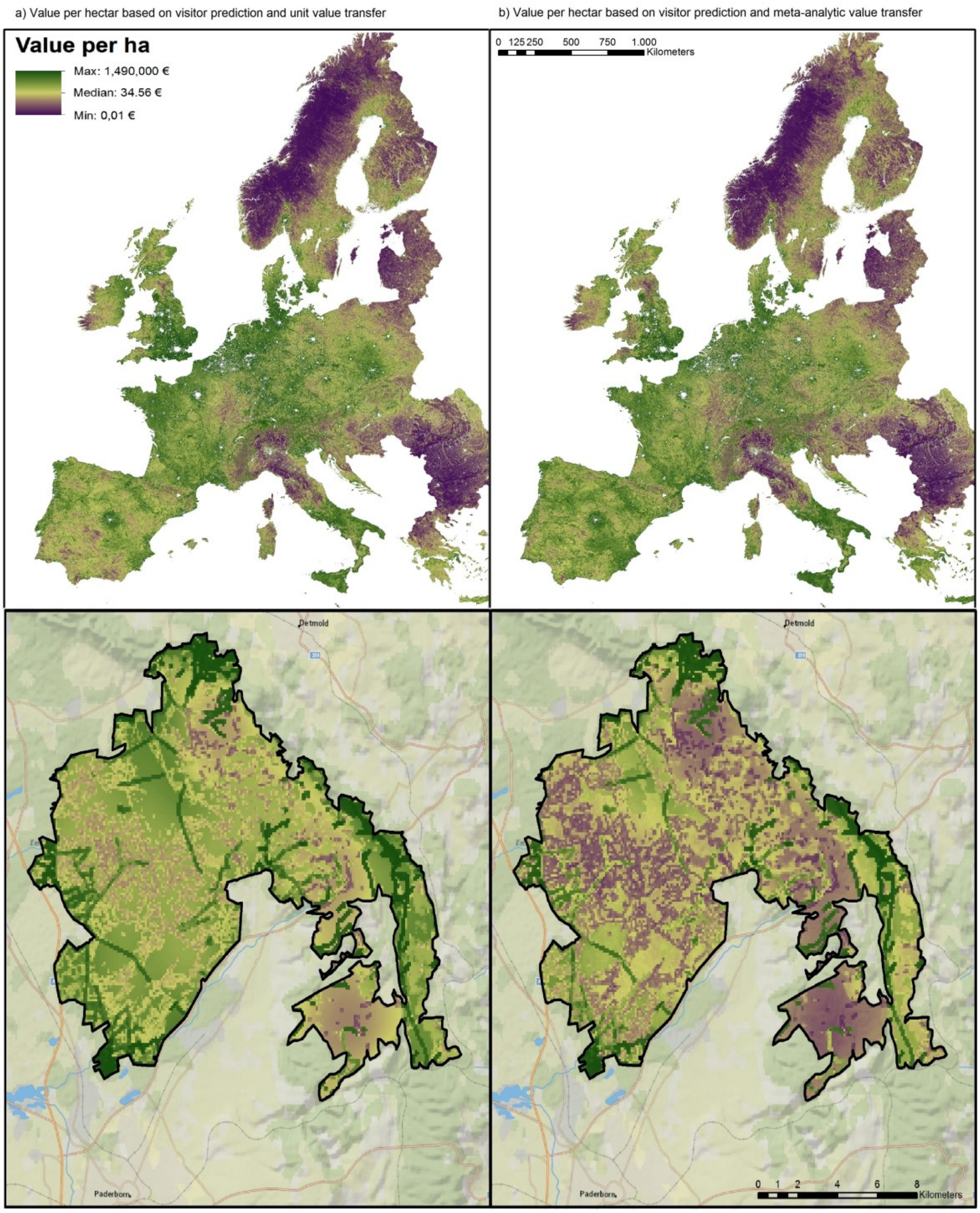
When comparing the two maps for Europe, differences can hardly be identified at a first glance. It is however visible that meta-analytic value transfer results in higher value estimates per hectare for Scandinavia and Spain but lower value estimates for Germany. The median value per hectare from the unit value transfer approach is € 34.07 compared to € 35.08 for the meta-analytic value transfer approach.

We also find that the total recreational value per hectare is highly dominated by spatial variations in the number of visits. The spatial variations in the value per visit generate only a secondary effect as the correlations between the values of the two rasters displaying the value per hectare is very high (0.92). The distributions of the predicted values are strongly skewed for both maps with a mean value and a skewness of € 676 and 37.6 for the unit value transfer and € 526 and 55.9 for the meta-analytic value transfer.

The mean relative deviation of the predicted values per hectare are 169% for the unit value transfer and 167% for the meta-analytic value transfer. If, by contrast, we combine the predictions of the meta-analytic value transfer with the mean number of visits per hectare of our primary data (reciprocal analysis), then the mean relative deviation of the predicted values per hectare is only 39%. Also this indicates a much higher influence of the spatial variation of the number of visits per hectare on the overall spatial variation of the value per hectare than of the spatial variation of the value per visit.

Even though an extreme value of € 1.5 billion per hectare is predicted for one pixel by the meta-analytic value transfer (€ 1.1 billion for the unit value transfer), both maps show less than 19% of the pixels with values above € 200 and anything beyond € 10,000 is to be expressed in per mille. Extreme values may be caused by over-predictions due to a combination of extreme values of explanatory variables beyond the range of the primary data used to estimate the models. In addition, the log transformed response variable of the model causes higher prediction errors for larger values.

Figs. 28 and 29 suggest a negative correlation between the value per visit and the overall value per hectare for both the meta-analytic value transfer (-0.03) and the unit value transfer (-0.08). On the contrary, the overall values per hectare for both approaches are strongly positive correlated (0.9)[44]. Recall that the latter is strictly proportional to the predicted visit rate and, thus, the correlation indicates that the overall value per hectare is almost solely explained by variations in the number of visits per hectare.



[44] We calculated the Pearson correlation coefficient between the raster layers using the R raster package.

Figure 29: Comparison of two approaches for mapping recreational values per hectare across Europe and a proposed national park in the West of Germany.

By zooming in on the Teutoburger Forest, differences between the two approaches become more visible. Again, the unit value transfer results in higher mean (€ 997 as compared to € 485) and median values (€ 59 as compared to € 27). Again, the spatial variations are dominated by the variations in visitors per hectare and not by variations in the value per visit. The correlation between the values of the two rasters is again very high with 0.98. Values range from € 4.71 to € 170,000 for the unit value transfer and from € 1.54 to € 68,000 for the meta-analytic value transfer. The distribution of the predicted values is strongly skewed for both maps with skewness of about 12. For the meta-analytic value transfer about 84% of the pixels show values below € 100.

6.5 Discussion

Our model results and the review of past meta-analyses help to identify predictors that have a significant effect on the value of recreational visits to natural areas, and the mapped values contribute to the efforts needed to fulfil the requirements of the EU Biodiversity Strategy (EC, 2011). The results help to identify relevant recreational sites and thereby may support efficient resource allocation and prioritisation for conservation. Furthermore, they contribute to the evaluation of land-use policies and to natural capital accounts (Haines-Young and Potschin, 2012).

The statistical analyses show that methodological variables are the most important predictors explaining differences across studies' results. In addition, we identify seven spatial variables with statistically significant effects on recreational values. However, due to uncertainties in our statistical analysis and due to inconsistencies among our and past meta-analyses, results are to be interpreted with caution.

The correlation between the two predicted values per hectare, which are either based on the meta-analytic value transfer or on the constant unit value transfer approaches, is very high. The mean relative deviation of the predicted visits per hectare is more than four times higher than the mean relative deviation of the predicted values per visit. The overall value per hectare is strongly positively correlated with the predicted visitor numbers, but negatively correlated with the predicted value per visit. This indicates that the spatial variation in the value per recreational visit is relatively low and that the main determinant of the overall recreational value of different nature areas is the number of visitors. This finding seems surprising, as the majority of the scientific literature focuses on the valuation, but not on the estimation of visitor number, even though the latter seems of greater relevance.

Based on the results of our meta-analysis and of our review of past meta-analyses on recreational valuation studies, one may question the criterion validity of monetary valuation for recreational ecosystem services. First, the spatial variation of ecosystems' recreational value depends mainly on the variation of the number of recreational visits, but only little on differences in the value per visit. Second, the main parameter of the models explaining the value per visit is the applied valuation methodology (CVM or TCM). Only few site and context characteristics that determine the demand and the quality of recreational experience show significant effects, and they have a much lower explanatory power. In addition, for several of these spatial predictors, results of different meta-analyses are contradictory.

However, modelling recreational values per visit by means of meta-analysis is not an easy task for several reasons. First, the statistical uncertainties involved in meta-analyses on value per visit estimates turn out to be substantial, because of common geostatistical problems, such as spatial autocorrelations, unobserved predictors, non-linear patterns and multiple interactions. Such problems may all affect the estimated standard errors and inflate p-values. Besides, several studies use different valuation methodologies to value the same study area, but most meta-analyses we review treat these observations as if independent. The impact of the implied violation of the independence assumption remains to be explored. We try to account for this by introducing random intercept for similar sites, but this did not improve the model as indicated by AIC and BIC values. One problem is the small group size (the number of observations per site). In addition, the valuation method has a strong impact on the value estimate. Both aspects complicate the identification of correlations across observations from the same site. The option to choose only one observation per site would overcome the violation of the independence assumption, but reduce the sample size considerably.

Second, most primary valuation data sets used in the different meta-analyses are not congruent. The value per recreational visit may depend on the spatial and temporal context and linear models may not detect general non-linear patterns that are to be expected for most ecosystem characteristics.

It may also be questioned whether recreational values per visit follow one overall pattern that can be explained by one model because the effect of different landscape characteristics on the values per visit may depend on the specific recreational activity pursued. Thus, it may be necessary to conduct regression analyses separately for different activities such as angling, camping, climbing, hiking etc.; however, primary valuation studies often do not discriminate by different activities.

Besides, there may also be substantial measurement errors in the explanatory variables. This results partly from the absence of consistent, high quality, fine resolution and comprehensive spatial data sets on relevant site and context characteristics. Consistent, detailed and comprehensive data sets, such as data on recreational facilities or activities, are hardly available across larger areas.

Finally, the strong explanatory power of methodological variables (in the meta-analyses presented and reviewed in this paper) indicate that measurement errors in monetary valuation of recreational services may be substantial, which hampers the identification of significant effects for spatial site and context characteristics. Like Brander et al. (2007) and Brander et al. (2015) we conclude that there are still insufficient primary valuation studies of high quality. They report not use more than 2/3 of all valuation studies in their meta-analyses because of insufficient methodology reporting. Insufficient and non-structured reporting within primary valuation studies, which hampers their use in secondary research, may have a big effect. Often it is difficult or impossible to identify the exact valuation methodology used and the precise location of the study area. However, as the different valuation methods have such a substantial effect on the value estimate, it is of major importance to consider them within statistical analysis. Our results presented in the appendix indicate that TCM studies in our data set are much more sensitive to site- and context characteristics than CVM studies. This may be caused by the fictive nature of CVM studies. It is however still to be investigated which valuation method results in superior results.

Most important, precise reporting on the location and extension of the study area is essential for identifying precise values of spatial explanatory variables. The spatial dimension of ecosystem services has received increasing attention and the advancement of GIS technology offers great potential for geostatistical assessments (Schägner et al., 2013; Maes et al., 2012; Maes et al., 2013). However, if the study area is not clearly defined, deriving exact information on the spatial characteristics of the study area becomes impossible. Therefore, exact spatial reporting on the study area is fundamental, including centroid coordinates, study area size and maps of its precise borders. If the only available information on the study site is, for example, a name of a forest in a certain country, it may not be possible to identify the location without additional information from the study authors. In consequence, authors of past meta-analyses describe study sites only by rough dummy variables and thereby treating very different sites as similar in their statistical analyses, which may be one reason for contradicting results. In contrast, if the study site can be identified clearly, researchers can assess site information ex-post, display sites in GIS and relate them to other spatial data. Several primary valuation studies could not be included in our analysis due to insufficient spatial information, and several other study sites are only approximated.

Quality and reporting standards for primary data collection have been repeatedly proposed in order to allow easier statistical assessments (Eigenbrod et al., 2010; Rosenberger and Phipps, 2007; Johnston and Rosenberger, 2010), but have rarely been put into practice. Schägner et al. (2017) propose a detailed reporting standard for recreational visitor counting studies including information on the methodology and the study area location. The reporting standard is available within a web interface that supports detailed reporting on all kind of nature recreation studies including studies on visitor counting, visitor monitoring, nature tourism's economic impact and recreational valuation. Researchers are encouraged to share their studies at http://rris.biopama.org/ visitor-reporting and thereby increase the outreach of their work and to contribute studies on the recreational importance of nature areas to this new global open access database. The database will be shared within a map-browser at http://esp-mapping.net/Home/.

6.6 Conclusion

In this study we combine meta-analysis and GIS to model spatial variations in the value of recreational visits to nature areas. Thereby, we identify statistically significant predictor variables, which can be used to map spatial variations of recreational values. The results of our analysis suggest that visits to larger recreational areas in remote mountainous locations are valued higher. Significant negative effects are shown for higher forest and grassland cover as well as for more rainy days. The maps of the predicted values per visit and value per hectare show how recreational values differ across space, and they can be used to assess the recreational importance of different nature areas. Thereby the results contribute to the assessment and valuation of cultural ecosystem services as specifically stipulated by Target 2, Action 5 of the EU Biodiversity Strategy and supported by MAES working group (Mapping and Assessment of Ecosystems and their Services). Whereas the value per visit is high in the North of Scandinavia, the total value per hectare for this area is low due to low visitor numbers. High recreational values per hectare are mainly found in highly populated areas, where in general there are higher visitor numbers but the value per visit tends to be lower, such as in Western Europe, but also in the South of Italy.

However, we conclude from our own analysis and other meta-analyses that meta-analytic value transfer of recreational values per visit encounters several important limitations and involves considerable uncertainties. Such analyses encounter common (geo)-statistical problems, such as violations of spatial independence, unobserved predictors, measurement errors in the dependent variable as well as in spatial predictor variables, spatial interaction etc. The overall explained variance of our meta-analysis is relatively low as compared to studies modelling recreational visitor numbers (Schägner et al., 2016). Whereas the modelling of recreational visitor numbers by choice models or meta-analysis has been successful in relating visitor numbers to the characteristics of recreational sites, the task proves to be more difficult for estimating values per visit. Meta-analyses have been successful in identifying the effects of study characteristics on the final value estimate. However, how site and context characteristics influence the recreational value per visit has only partly been resolved. Existing studies show mixed and partly conflicting results and it remains difficult to identify overall trends. The most important variables for explaining estimated values relate to the valuation methodology and not to site and context characteristics.

Statistical analyses are hampered by the lack of detailed, consistent and comprehensive data sets on relevant spatial predictor variables. Comprehensive and consistent data sets are required that describe the biophysical and socio-economic site and context characteristics that determine the quality of the recreational experience. However, most important for identifying the effects of these characteristics is a precise study area definition in primary valuation studies in order to accurately locate the site.

A lot of the unexplained statistical noise may also relate to insufficient reporting on the valuation methodologies. Detailed information on valuation methodology is necessary to account for their effects on the study results. Detailed reporting standards for primary data collection as proposed by Schägner et al. (2017) should become a requirement for the publication of primary valuation studies.

Nevertheless, our results clearly suggest that spatial variations in the overall recreational value per hectare depend far more on variations in the number of recreational visits than on variation in the recreation values per visit. Comparing the standard deviations of the predicted value per hectare, if either the number of visits or the value per visit is kept constant at the mean of the primary data, shows that in an European context, the spatial variations in the number of visits per hectare has a more than 13 times bigger effect on the overall spatial variations of the value per hectare than the spatial variation in the value per visit. Therefore, we conclude that, to map the overall recreational value of different ecosystems accurately and spatially explicit, it is far more important to obtain accurate and precise assessments of the number of recreational visits than estimates of the value per recreational visit. It is best not to value recreational ecosystem services by transferring a value per hectare, but to build the value from accurate use estimates combined with a value per visit estimate.

6.7 Acknowledgements

This research was funded by the European Commission as part of the MAES working group (Mapping and Assessment of Ecosystems and their Services). Within follow-up INCA (Integrated system for Natural Capital and ecosystem services Accounting) project, as well as the DOPA (Digital Observatory for Protected Areas) project and the BIOPAMA (Biodiversity and Protected Areas Management) program, we aim at extending this work in scope and scale by including further ecosystem services and covering the entire globe. We would like to thank our colleague Roxanne Leberger for her support in scripting with R.

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6.9 Appendix

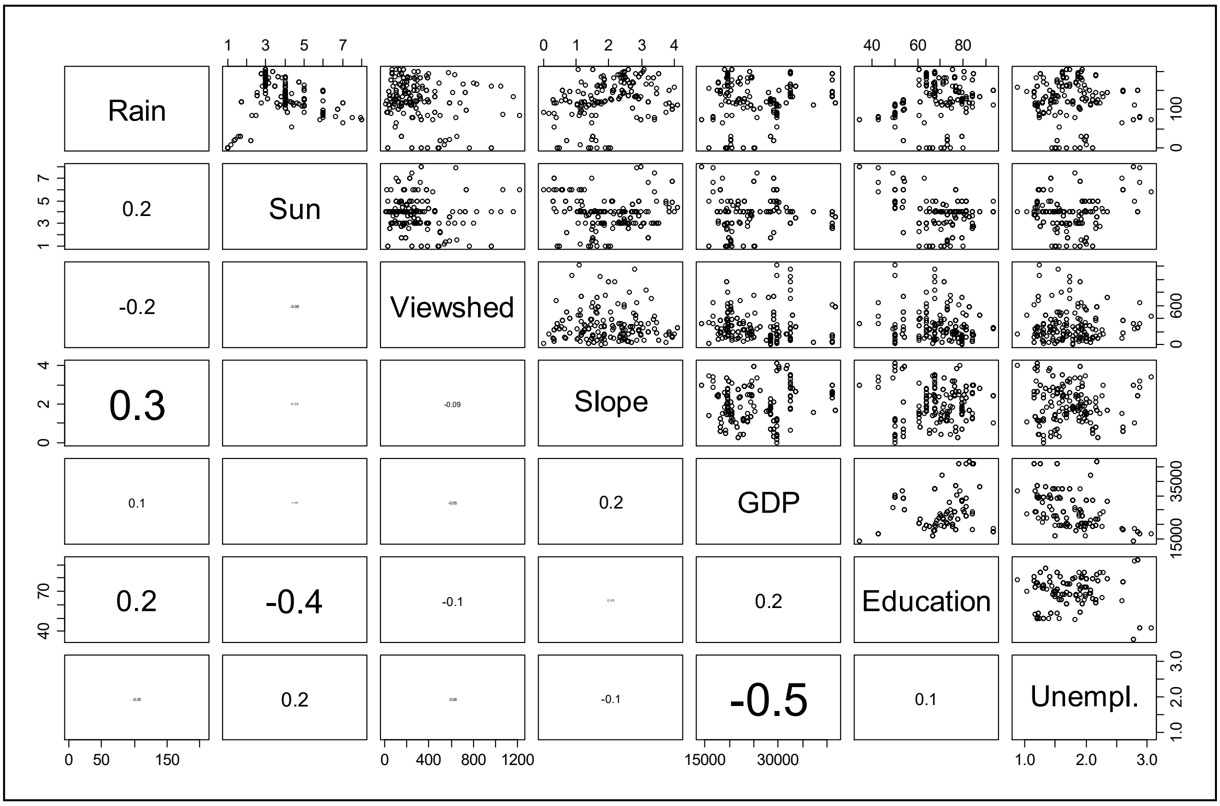

Figure A 6.1: Correlations between selected predictor variables.

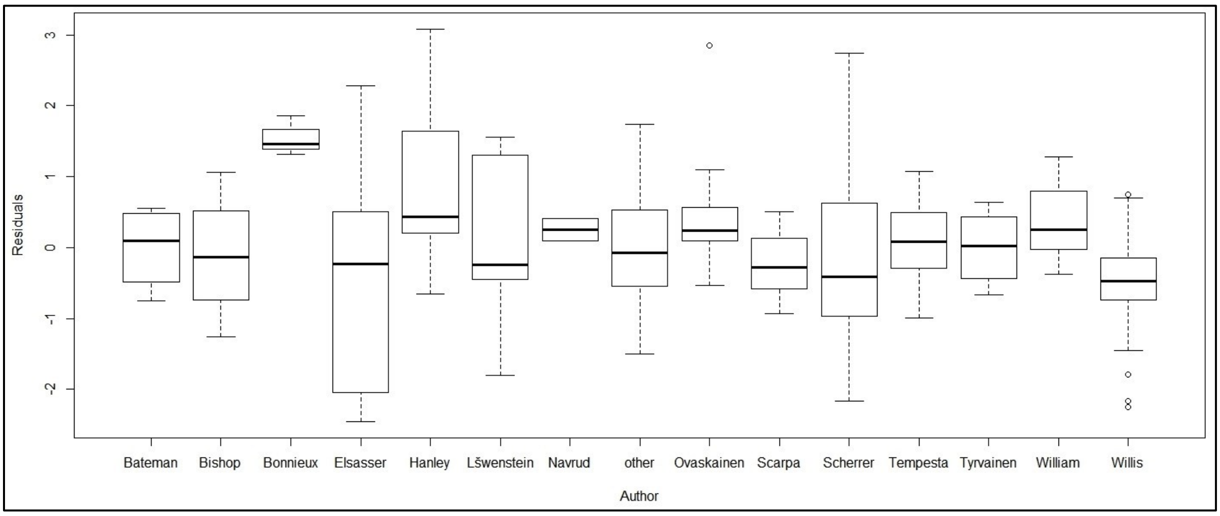

Figure A 6.2: Residuals of the linear fixed effects model by author.

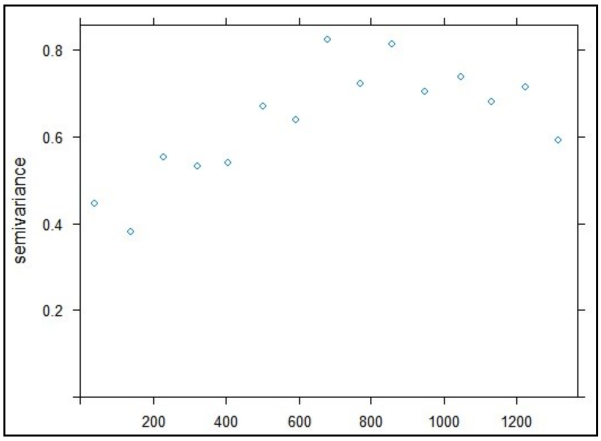

Figure A 6.3: Semivariance of the full mixed effect model

Recreational Values in different countries

Different countries might be characterised by different patterns in the recreational value per visit due to different cultures, preferences and other factors. Even though this may not be a major concern for EU countries – countries that are geographically and culturally close – we investigate this issue more in detail here.

Figure A6.4 shows the residuals of the final mixed model by country. Due to a small number of observations per country some variation is not surprising. Still, the zero-line is covered by the quantiles of the residuals for most countries and for all countries with a larger number of observations.

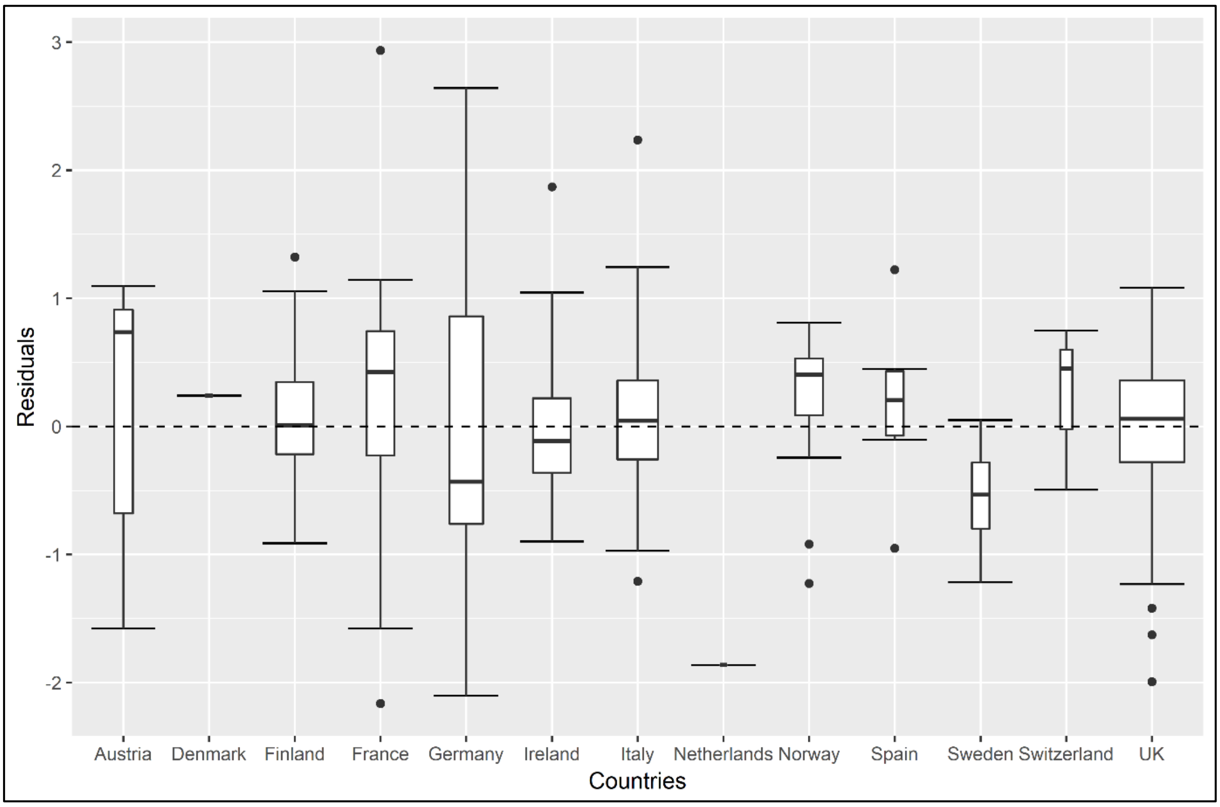

Figure A 6.4 Residuals of the final mixed models by country (boxplots' width represents the number of observations per country).

Due to the limited number of primary valuation studies per country it is not possible to estimate comparable models for each country. Figure A6.5 shows the number of observations per country. Only the UK shows a considerable number of observations and consequently we estimated the final mixed model presented in this paper also for a subset of observations with study sites located in the UK and for a subset of observations with study sites located outside the UK. For the UK subset we had to exclude the *value-measure* variable because there are not sufficient observations per level of this factor variable. The results are presented in Table A6.1.

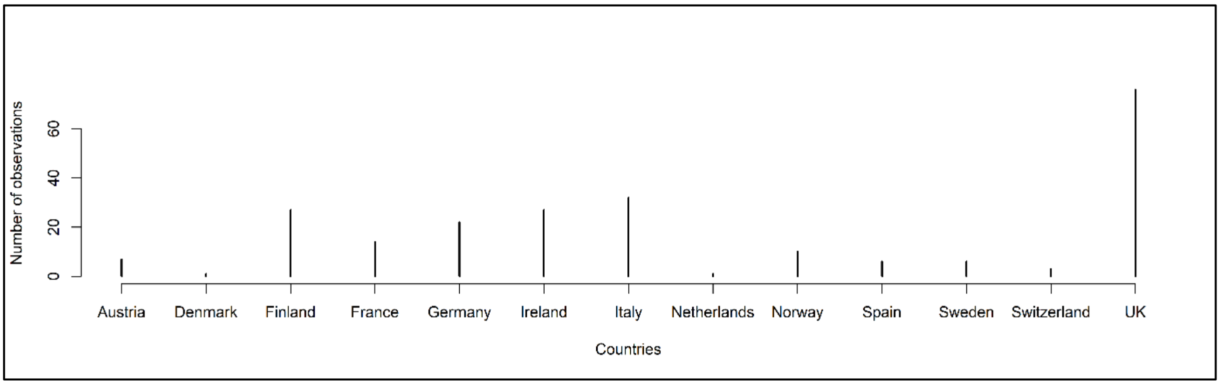

Figure A 6.5: Number of observations per country.

Table A6.1: Linear fixed and mixed effects model after stepwise variable selection with the ln of the value per visit as dependent variable (€, 2013) and first author as random intercept as the mixed effects model for the full data set, model with the same predictors for study sites in the UK only and for study sites outside the UK only.

	Full linear mixed effects model			Linear mixed effects model (UK only)			Linear mixed effects model (non-UK only)		
Variable	**Coefficient**	**p-value**		**Coefficient**	**p-value**		**Coefficient**	**p-value**	
Intercept	5.13	3e-04	***	2.35	0.42		5.45	6.40E-03	**
TCM	0.68	<1e-16	***	0.83	0.00E+00	***	0.60	7.00E-04	***
Value-measure v/visit	-0.35	5.1e-02	●	—	—		-0.48	4.09E-02	*
Ln(ha)	7.3e-02	8.1e-03	**	4.17E-02	0.32		7.53E-02	4.44E-02	*
Ln(forest)	-0.18	4.7e-02	*	0.13	0.44		-0.24	5.01E-02	●
Ln(grassland)	-0.11	6.4e-02	●	-1.56E-05	1.00		-7.83E-02	0.32	
Rain days	-5.9e-03	7.4e-03	**	-1.24E-02	3.57E-02	*	-7.17E-03	2.20E-02	*
Slope	0.15	3.4e-02	*	0.41	4.58E-02	*	0.13	0.16	
Ln(pop)	-0.23	9e-04	***	-0.16	2.77E-01		-0.20	3.97E-02	*
Unemployment	0.27	9.2e-02	●	0.72	7.19E-02	●	0.14	0.53	
N	244			81			163		

We report significance levels by indicating p-values of up to 0.001, 0.01, 0.05 and 0.1 by "*", "**", "*" and "•".**

As expected significance levels decrease on average due to the lower number of observations for the subsets. For the UK we find one predictor that changes its sign. Forest cover now shows a positive effect, but neither the effect nor the difference in the slope as compared to the full data set model is significant (see also Figure A6.9). Nevertheless, an explanation could be that there is a non-linear effect for forest cover as suggested by economic theory and as the UK has little forest cover, the variable turns positive. We did, however, not detect any significant non-linear effects for the residuals of the full data set model. For the non-UK countries subset, no changes in the predictors signs are found. For all countries with still some observations, we plot the country specific residuals of the final mixed model against each predictor that is either included or excluded in the final model. We added regression lines and 0.05 conditional confidence intervals bands (see Figure A6.6-29). If the dashed line is not covered by the grey confidence interval bands, it suggests that there is a significant difference in the effect of the predictor in the specific country compared to the overall model (Breheny & Burchett 2017). However, due to the large number of plots and the probabilities indicated by the 0.05 confidence intervals, the detection of significant differences is to be expected by some plots just by chance. For countries with few observations and/or wide spreads in the residuals (e.g. Finland), such significant differences may be caused just by a few or by one observation. In general, we could not detect anything of concern, but the interested reader may investigate these plots more in detail. Please note that for several countries there are not sufficient observations per level for some of the factor variables and consequently, those countries are excluded from the plots for these factor variables.

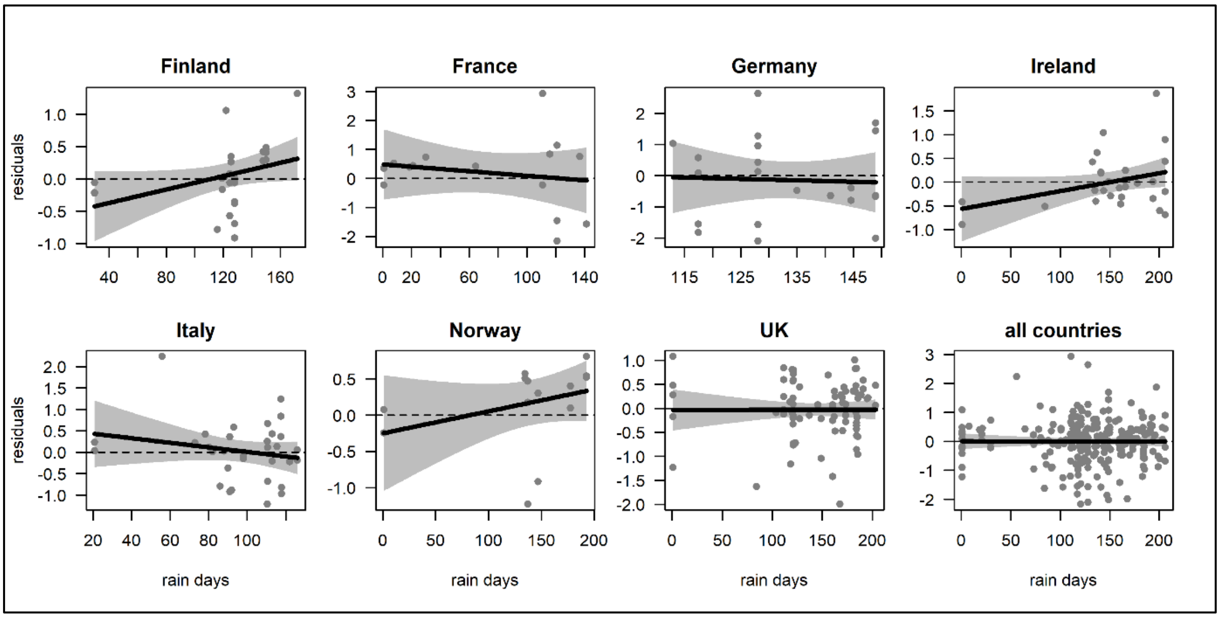

Figure A 6.6: Final mixed model residuals against predictor rain for all observations and for countries with multiple observations.

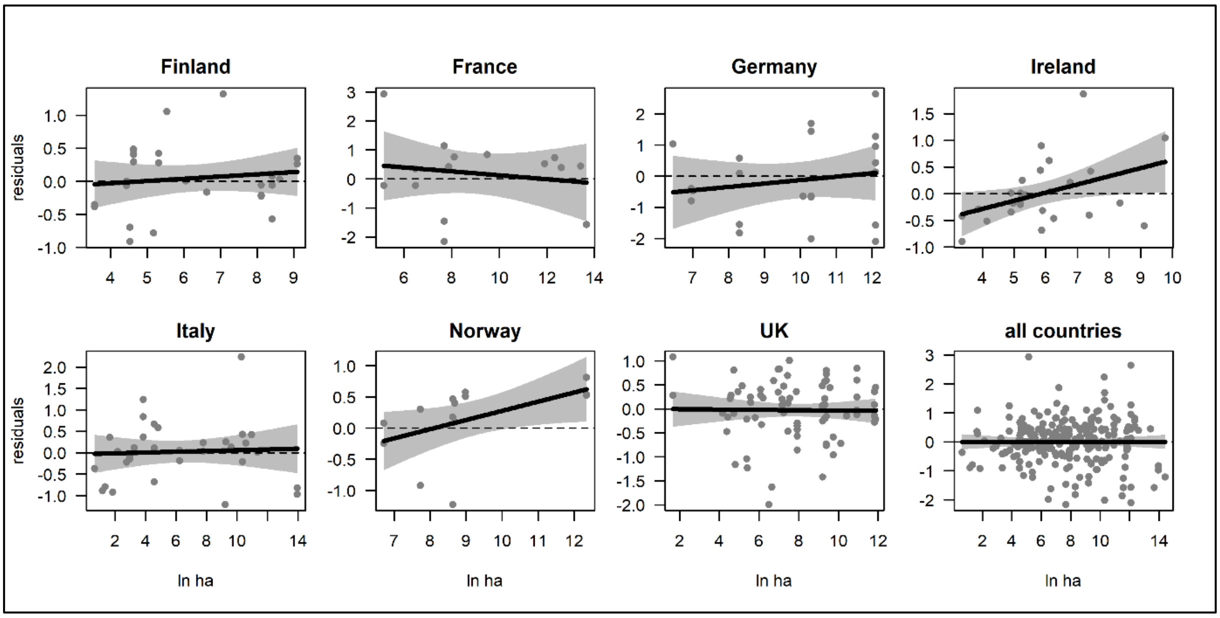

Figure A 6.7: Final mixed model residuals against predictor ln ha for all observations and for countries with multiple observations.

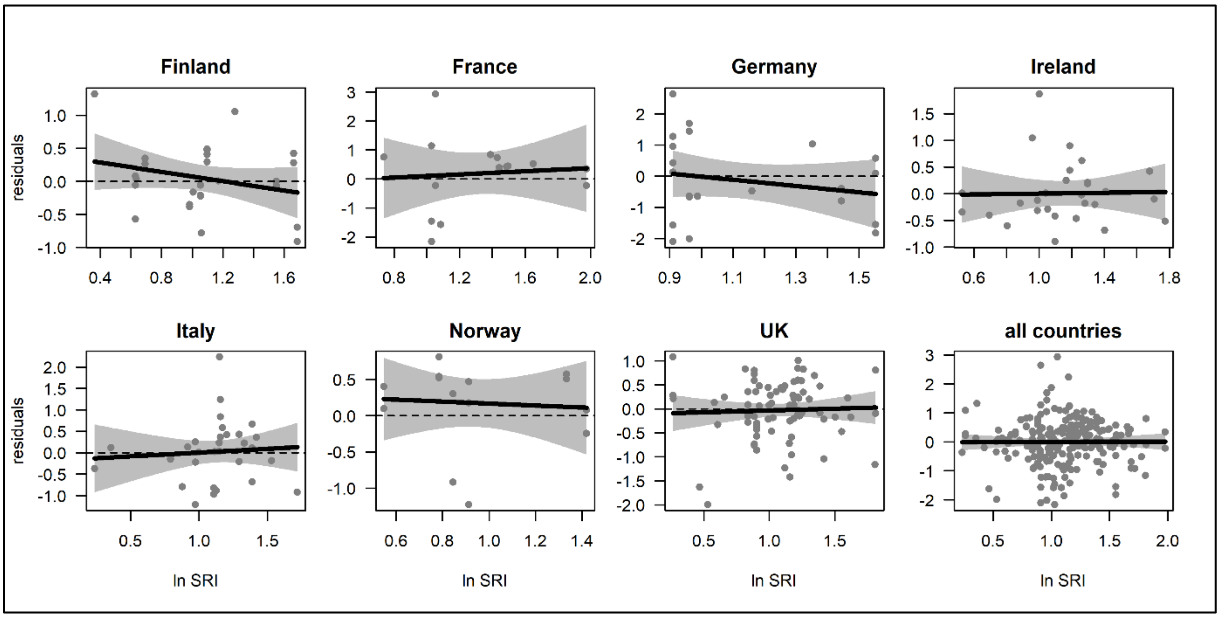

Figure A 6.8: Final mixed model residuals against predictor ln SRI for all observations and for countries with multiple observations.

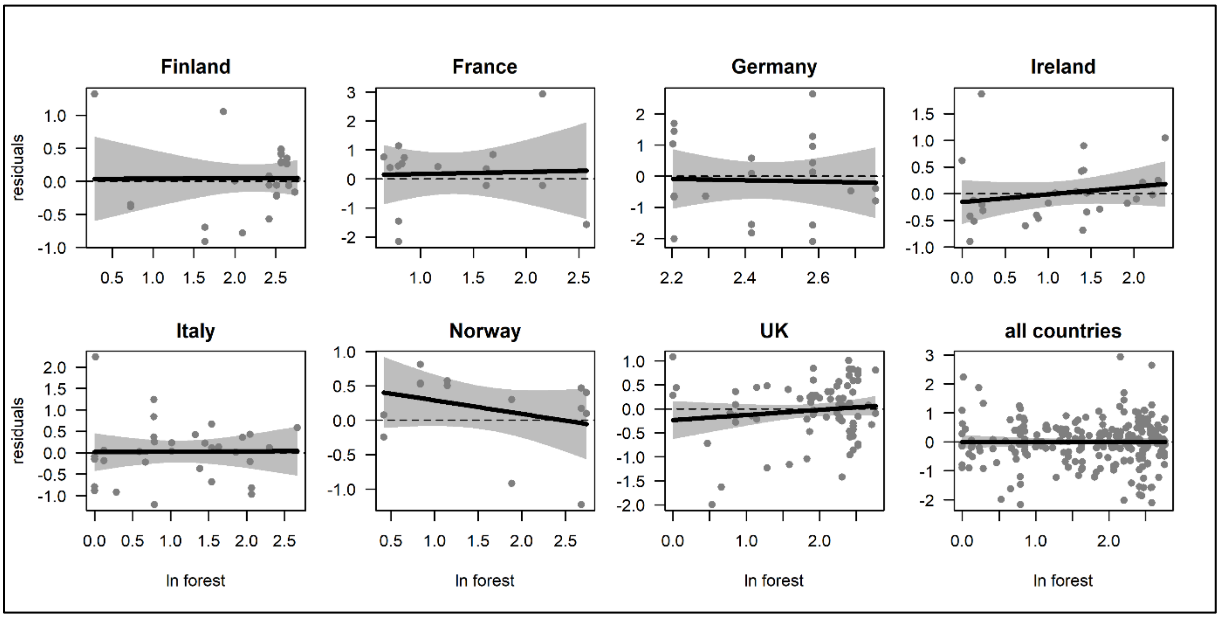

Figure A 6.9: Final mixed model residuals against predictor ln forest for all observations and for countries with multiple observations.

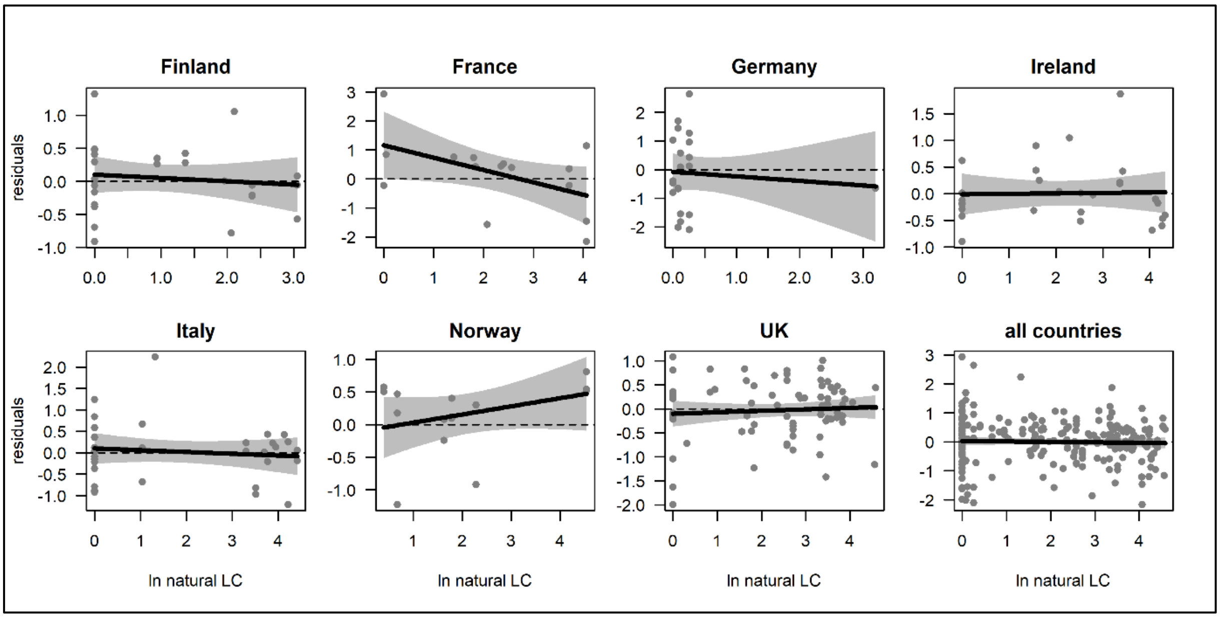

Figure A 6.10: Final mixed model residuals against predictor ln natural LC for all observations and for countries with multiple observations.

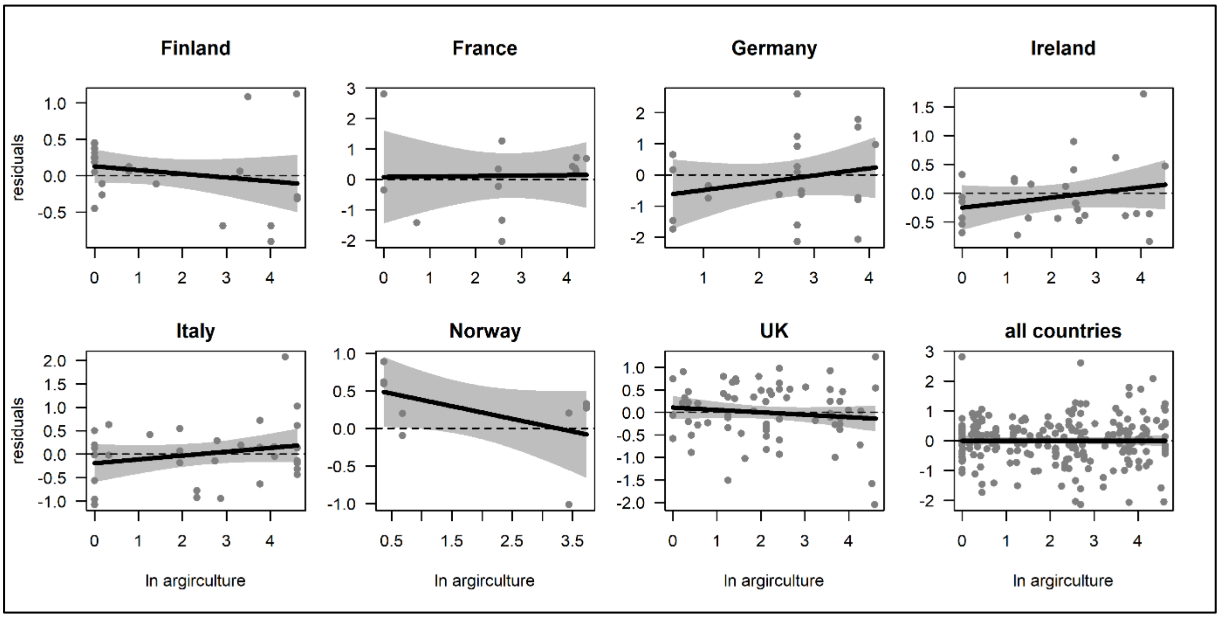

Figure A 6.11: Final mixed model residuals against predictor ln agriculture for all observations and for countries with multiple observations.

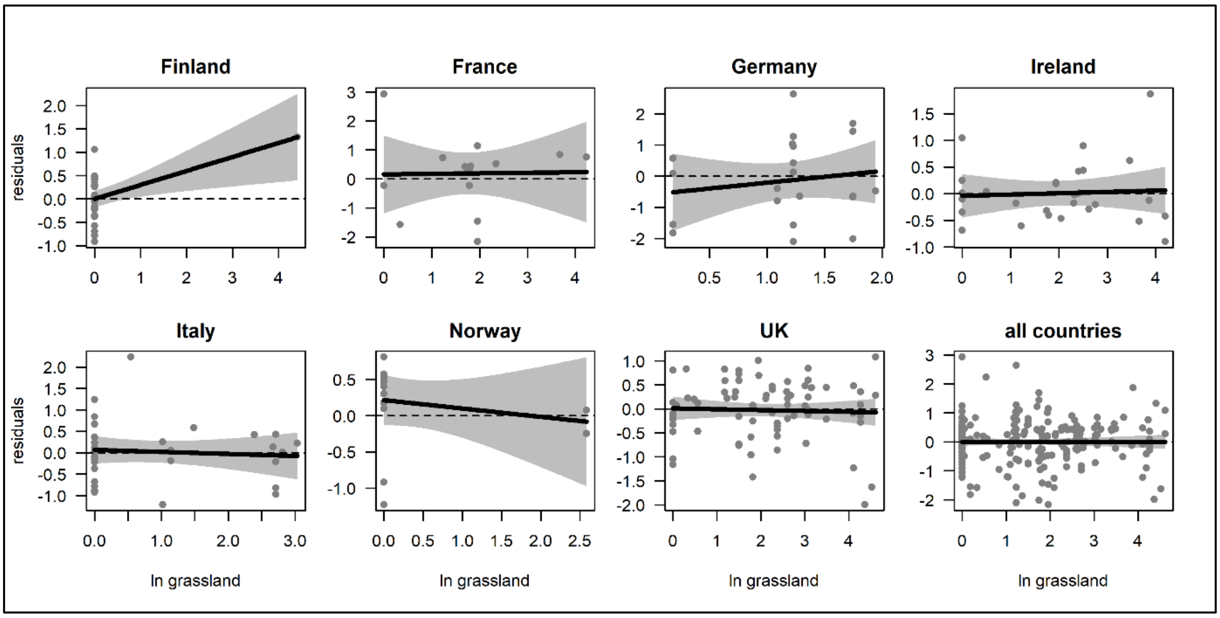

Figure A 6.12: Final mixed model residuals against predictor ln grassland for all observations and for countries with multiple observations.

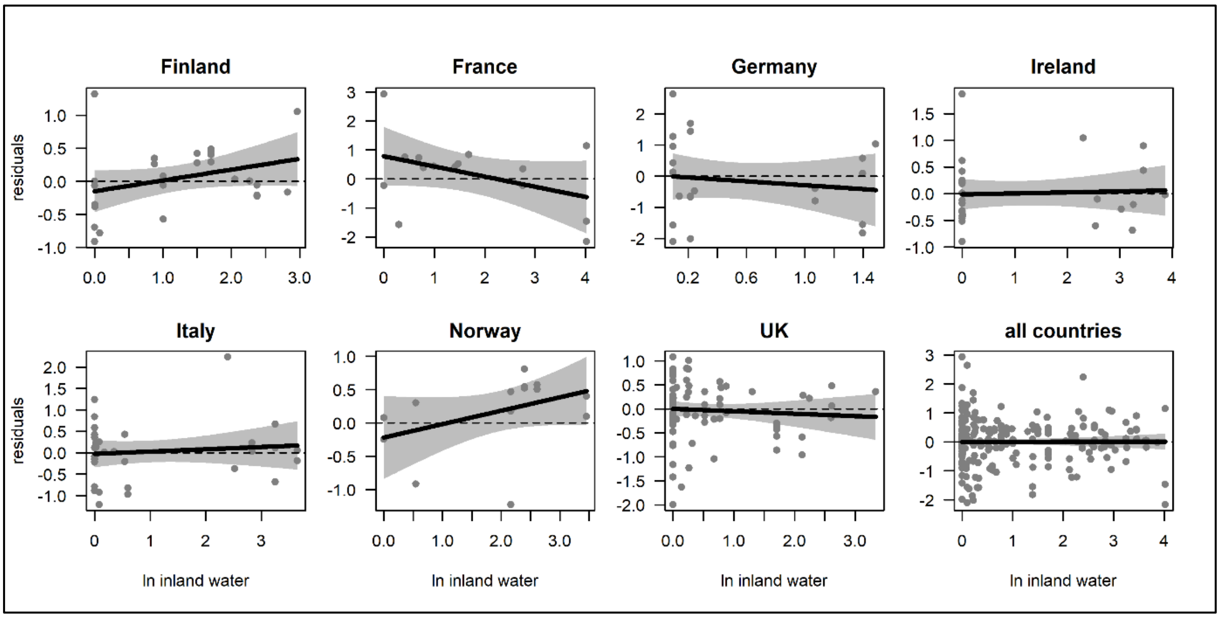

Figure A 6.13: Final mixed model residuals against predictor ln inland water for all observations and for countries with multiple observations.

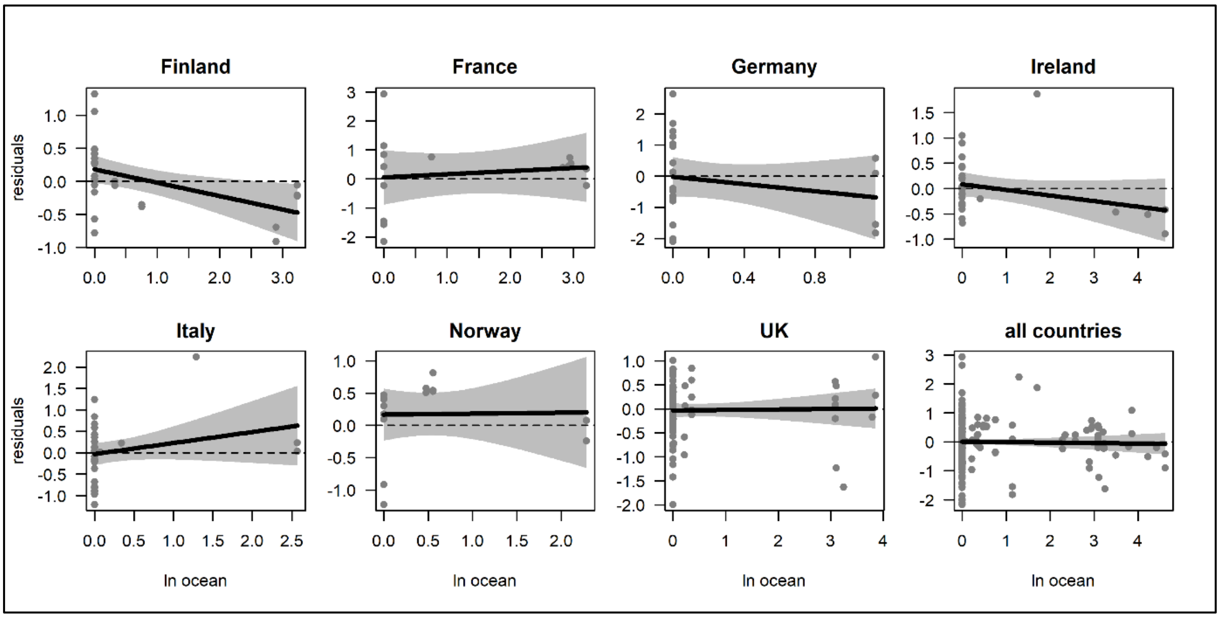

Figure A 6.14: Final mixed model residuals against predictor ln ocean for all observations and for countries with multiple observations.

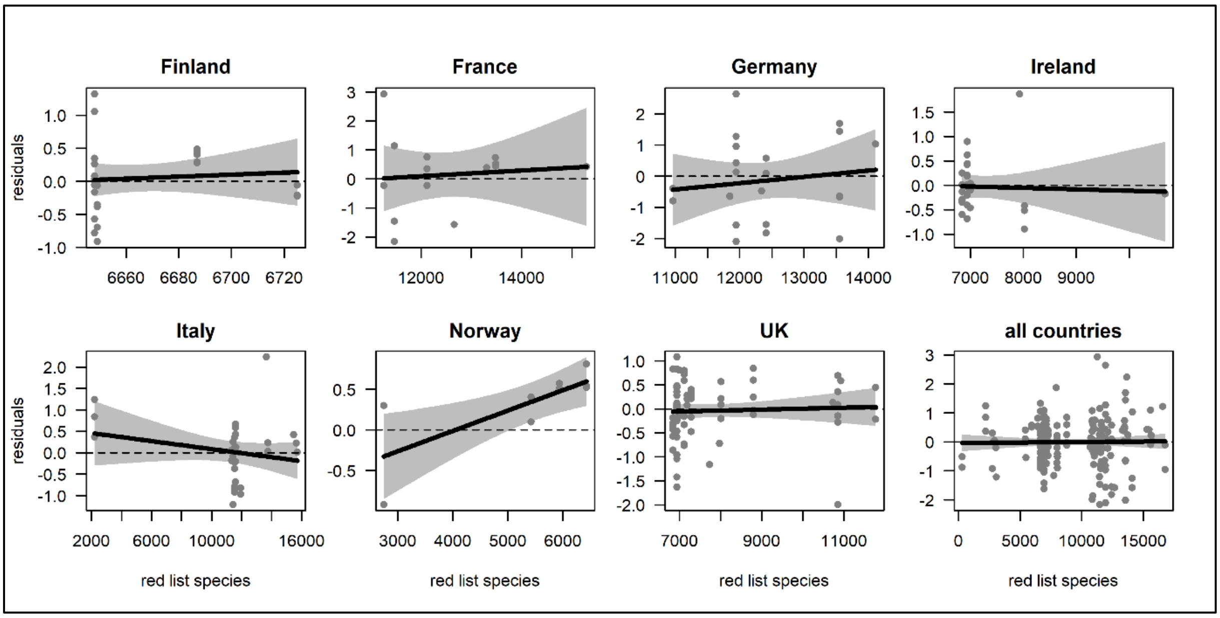

Figure A 6.15: Final mixed model residuals against predictor red list species for all observations and for countries with multiple observations.

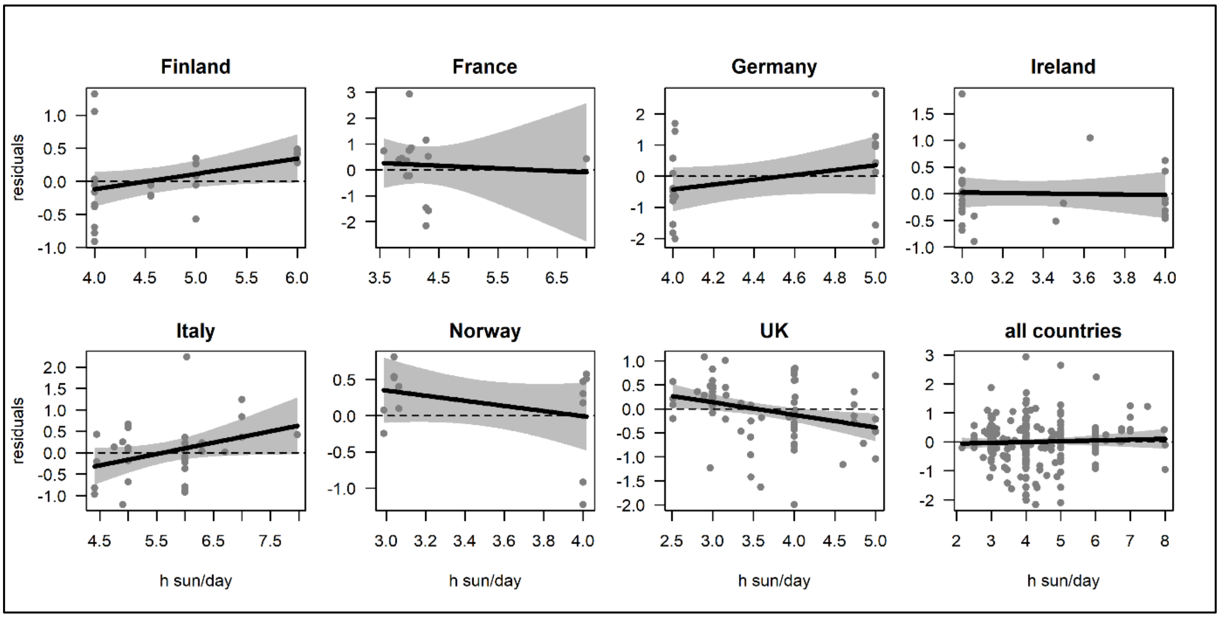

Figure A 6.16: Final mixed model residuals against predictor h sun/day for all observations and for countries with multiple observations.

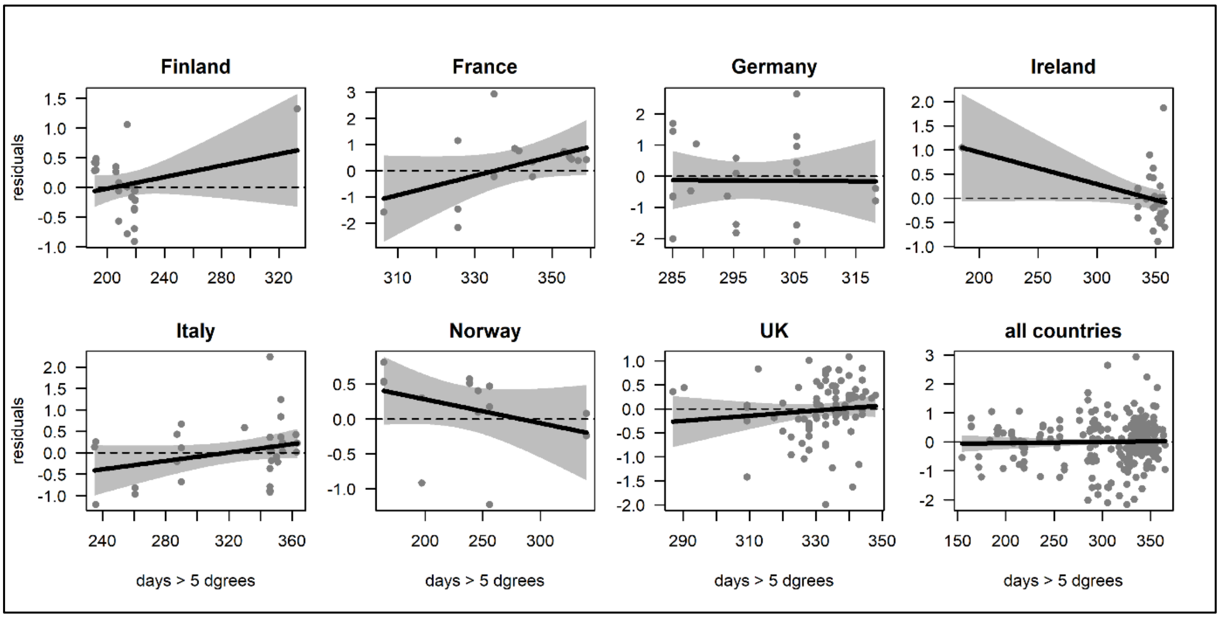

Figure A 6.17: Final mixed model residuals against predictor days>5 degrees for all observations and for countries with multiple observations.

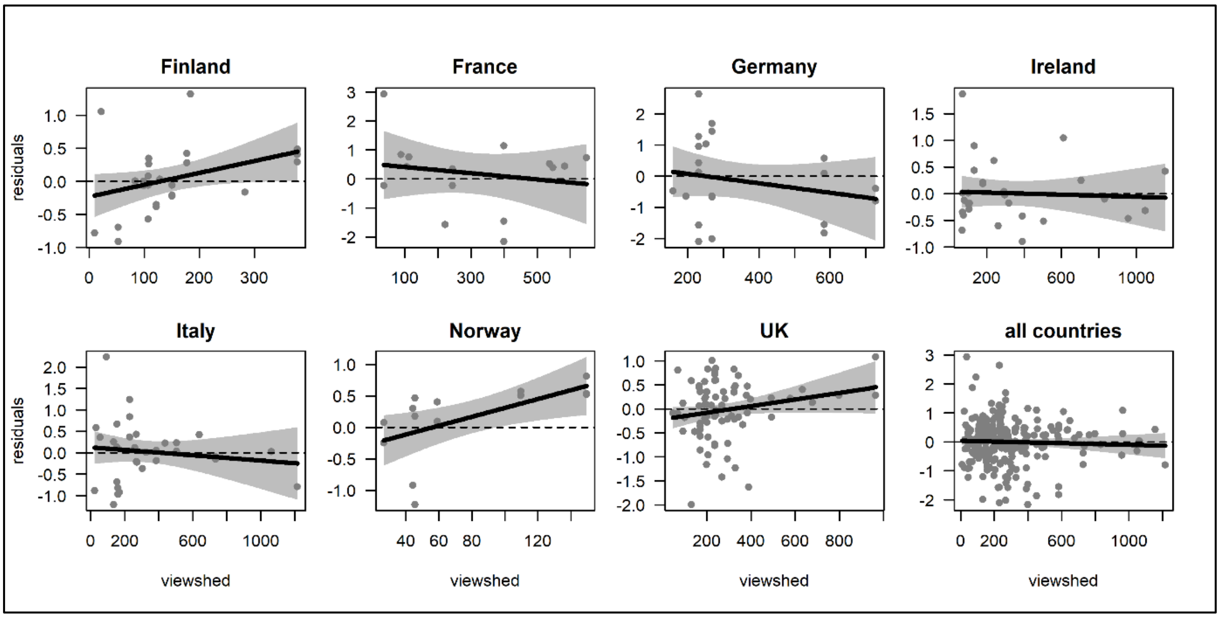

Figure A 6.18: Final mixed model residuals against predictor viewshed for all observations and for countries with multiple observations.

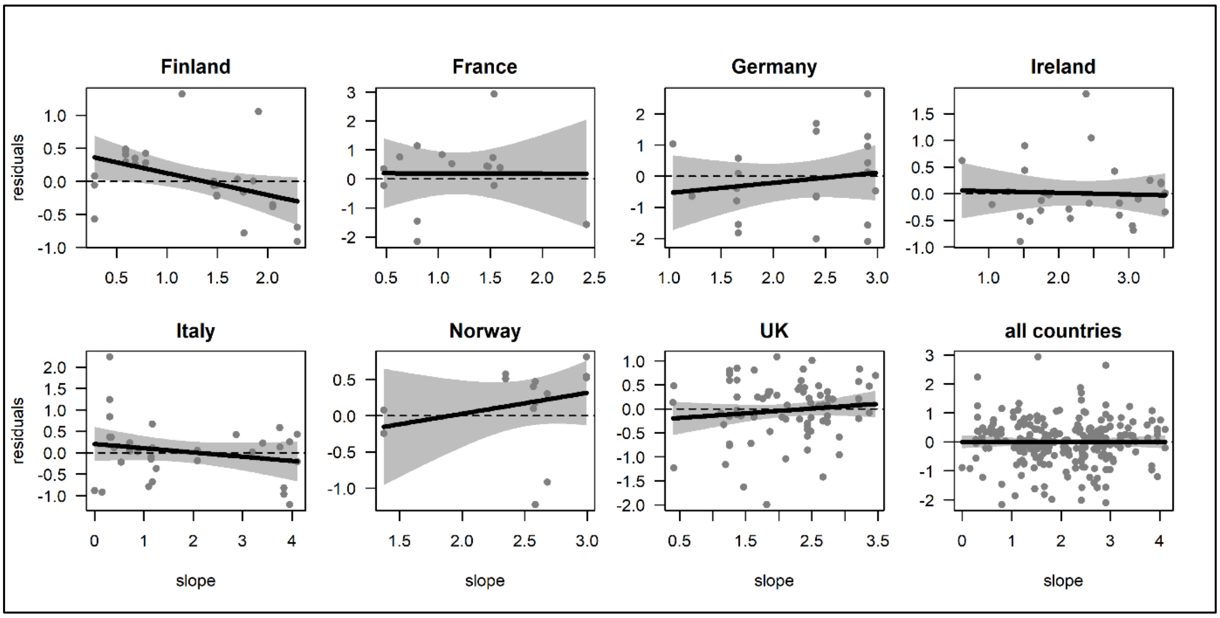

Figure A 6.19: Final mixed model residuals against predictor slope for all observations and for countries with multiple observations.

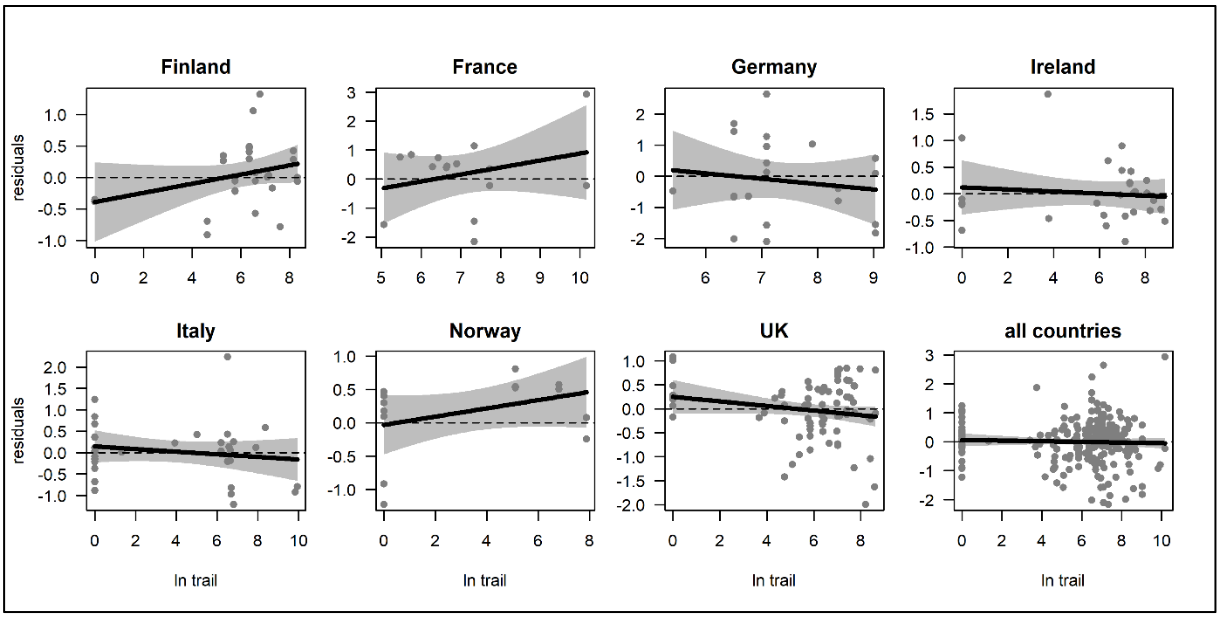

Figure A 6.20: Final mixed model residuals against predictor ln trail for all observations and for countries with multiple observations.

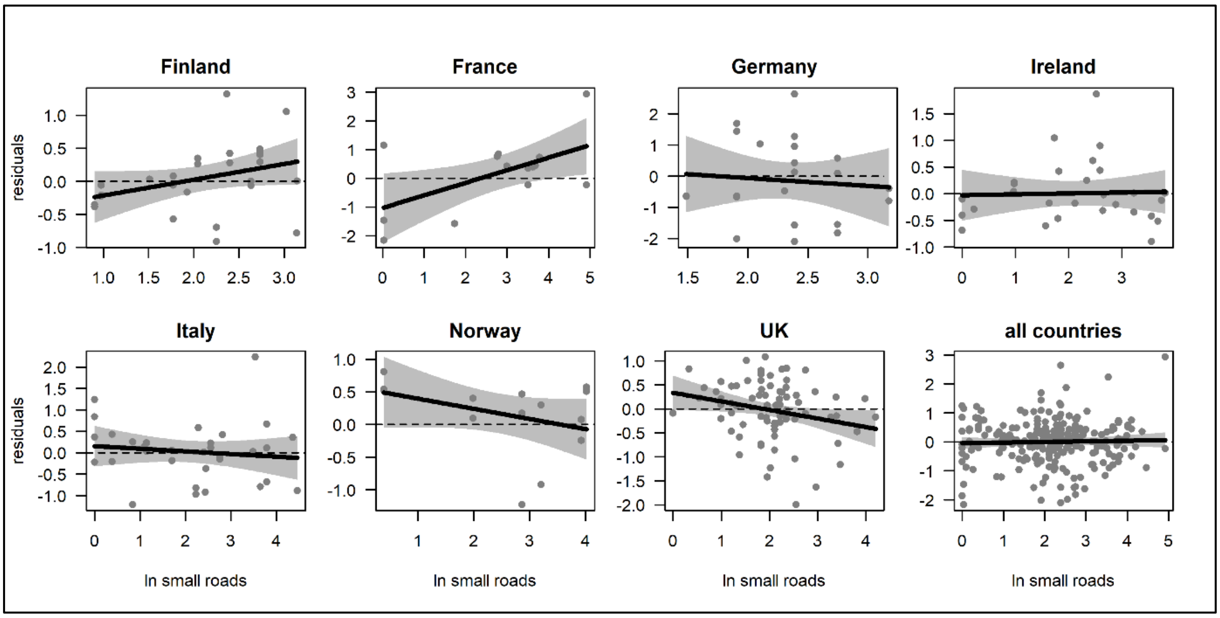

Figure A 6.21: Final mixed model residuals against predictor small roads for all observations and for countries with multiple observations.

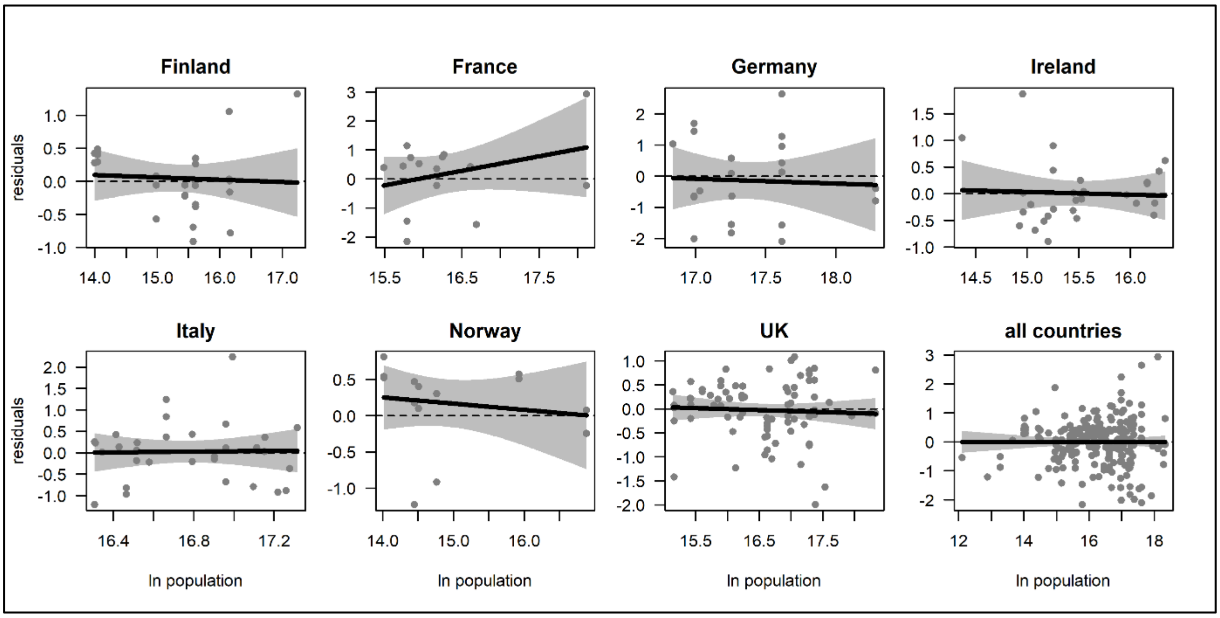

Figure A 6.22: Final mixed model residuals against predictor ln population for all observations and for countries with multiple observations.

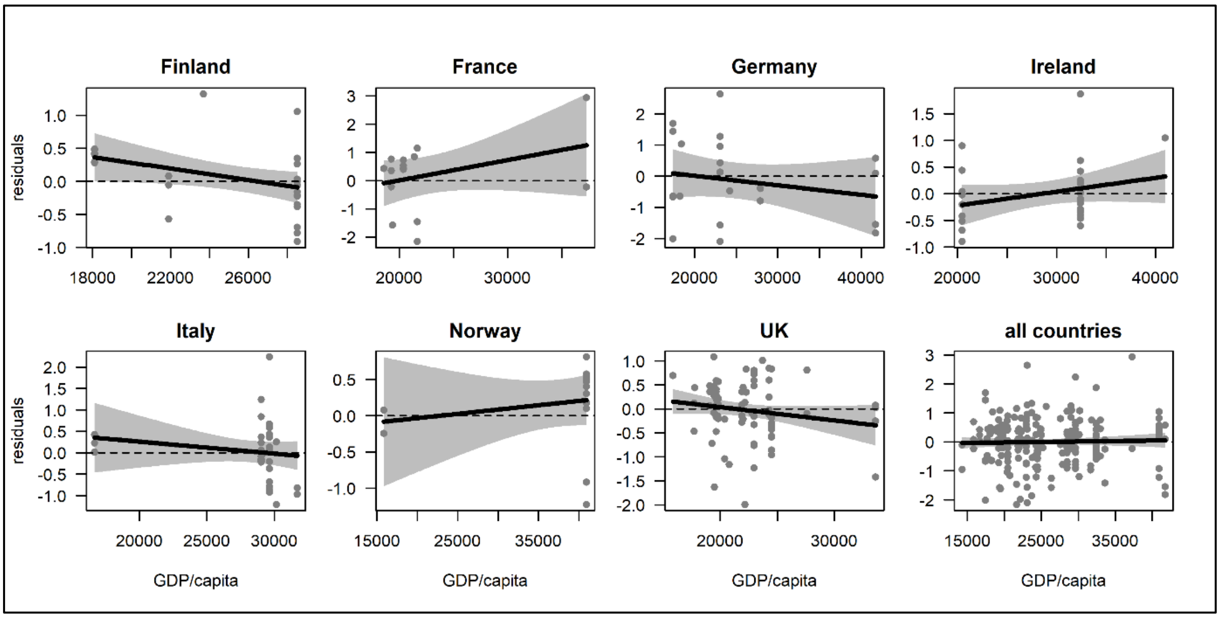

Figure A 6.23: Final mixed model residuals against predictor GDP/capita for all observations and for countries with multiple observations.

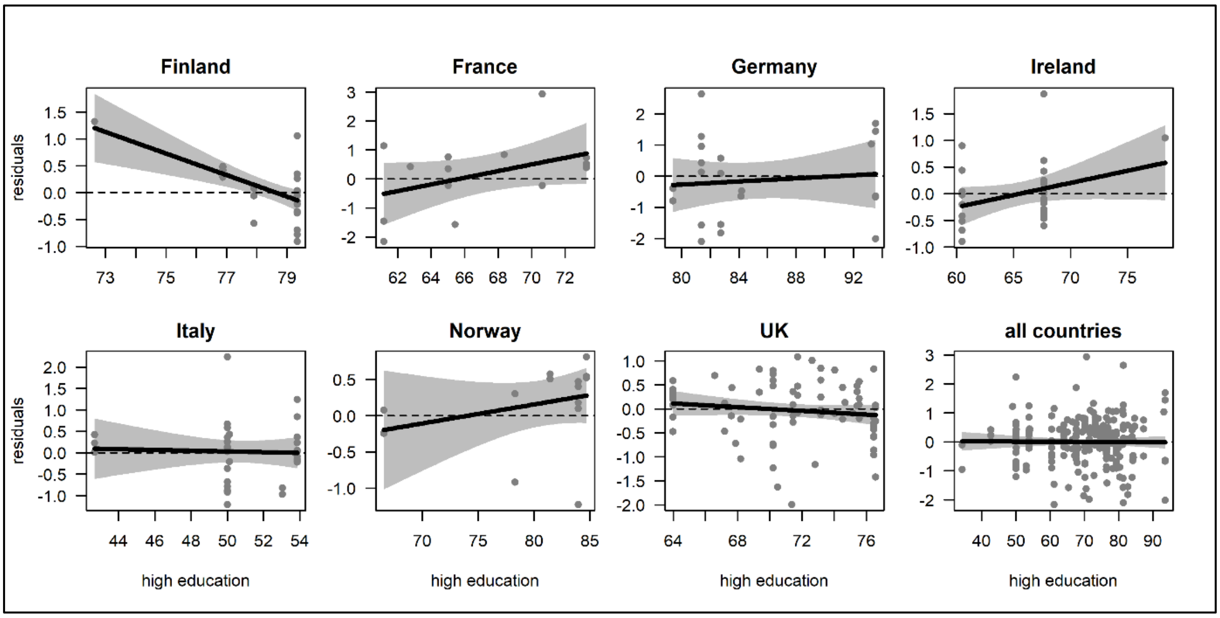

Figure A 6.24: Final mixed model residuals against predictor high education for all observations and for countries with multiple observations.

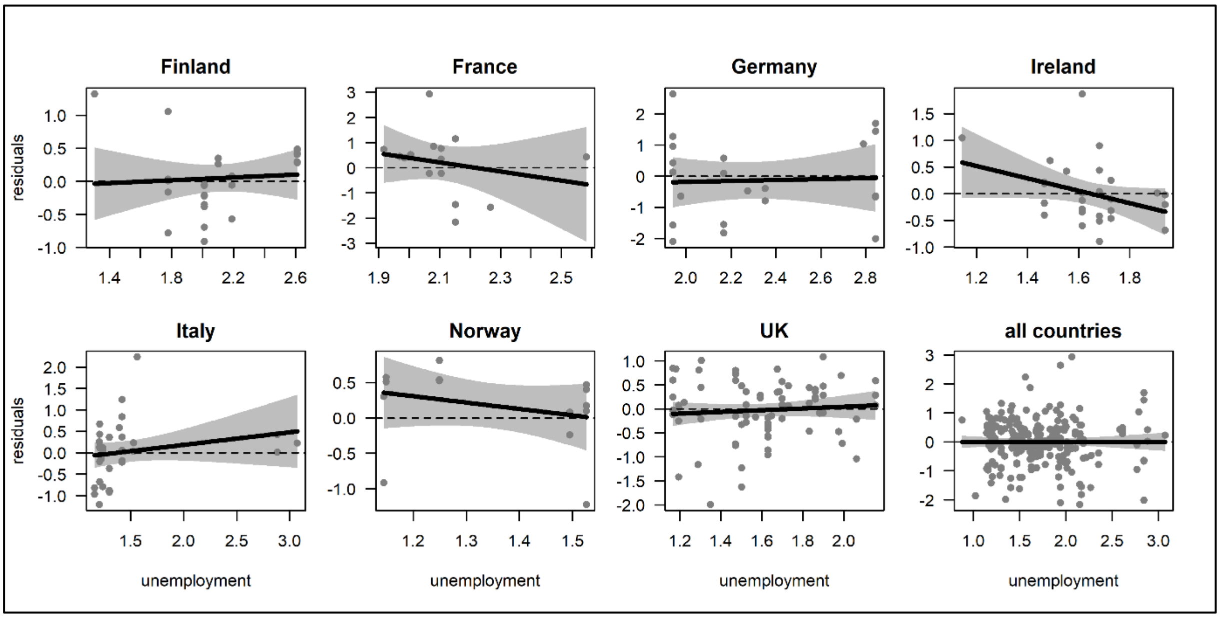

Figure A 6.25: Final mixed model residuals against predictor unemployment for all observations and for countries with multiple observations.

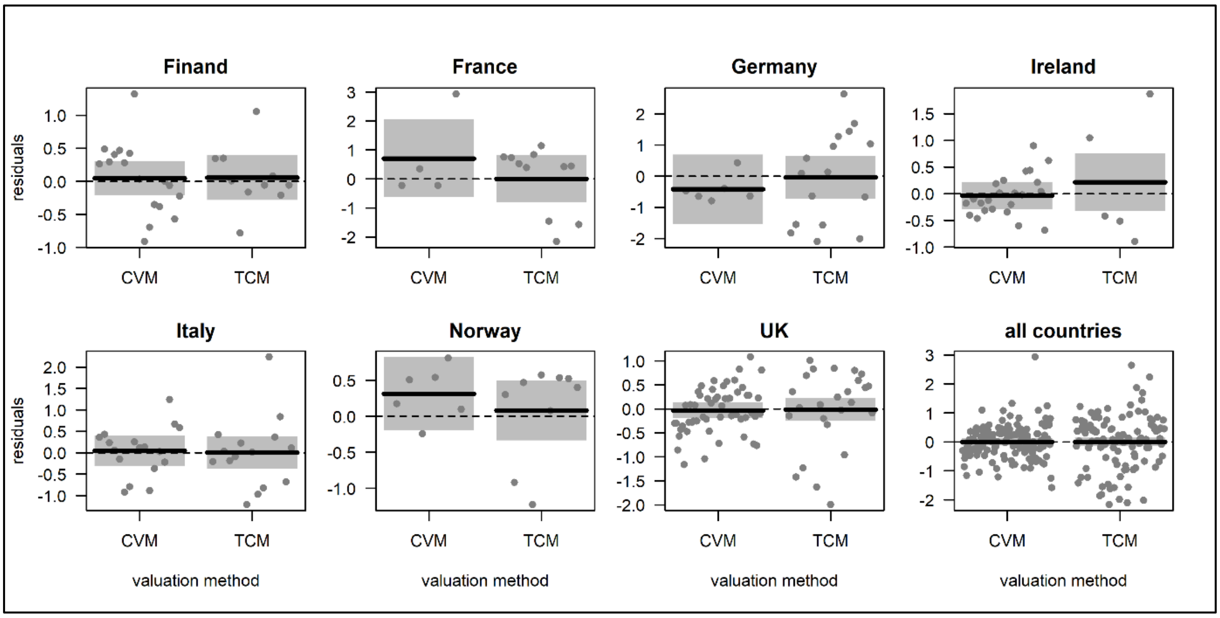

Figure A 6.26: Final mixed model residuals against predictor valuation method for all observations and for countries with multiple observations.

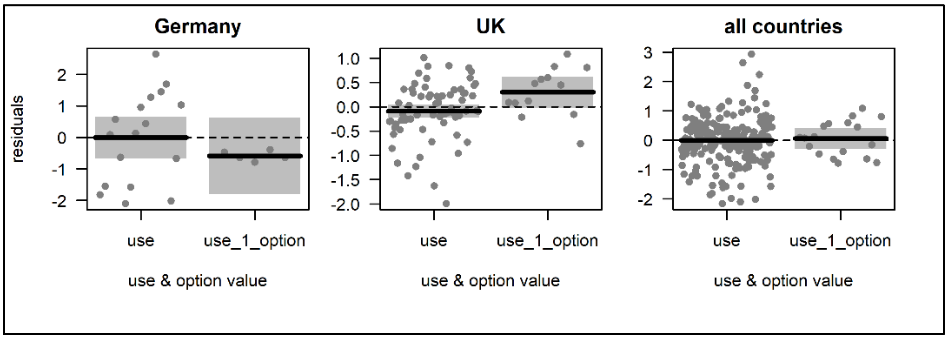

Figure A 6.27: Final mixed model residuals against predictor use & option value for all observations and for countries with multiple observations.

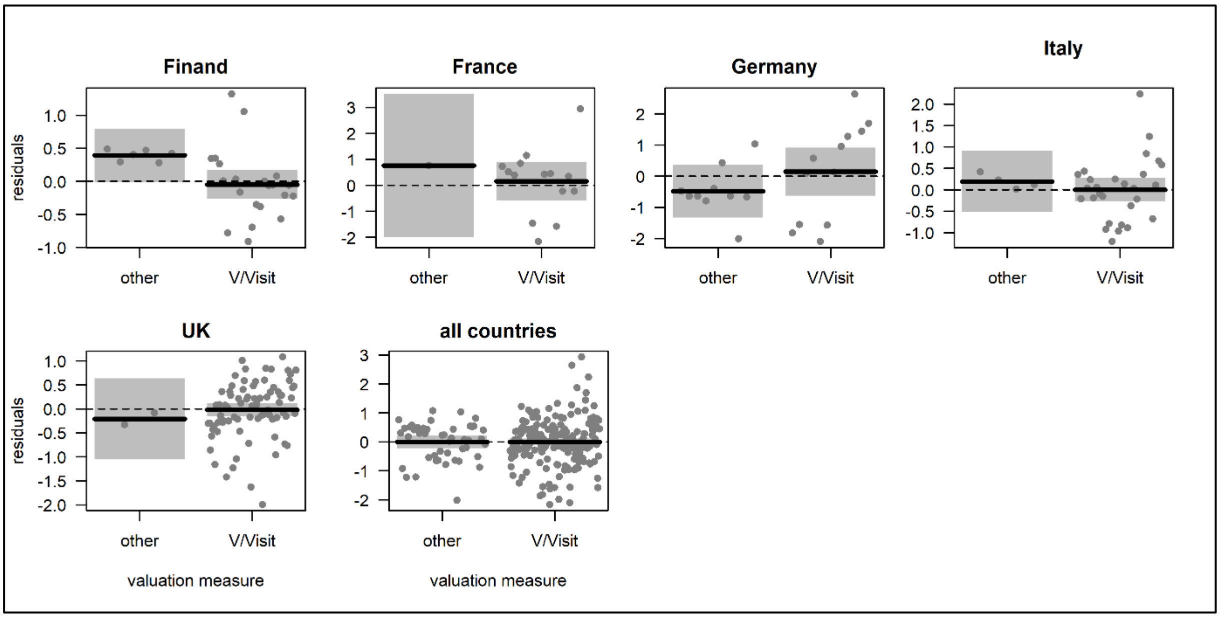

Figure A 6.28: Final mixed model residuals against predictor value-measure v/visit for all observations and for countries with multiple observations.

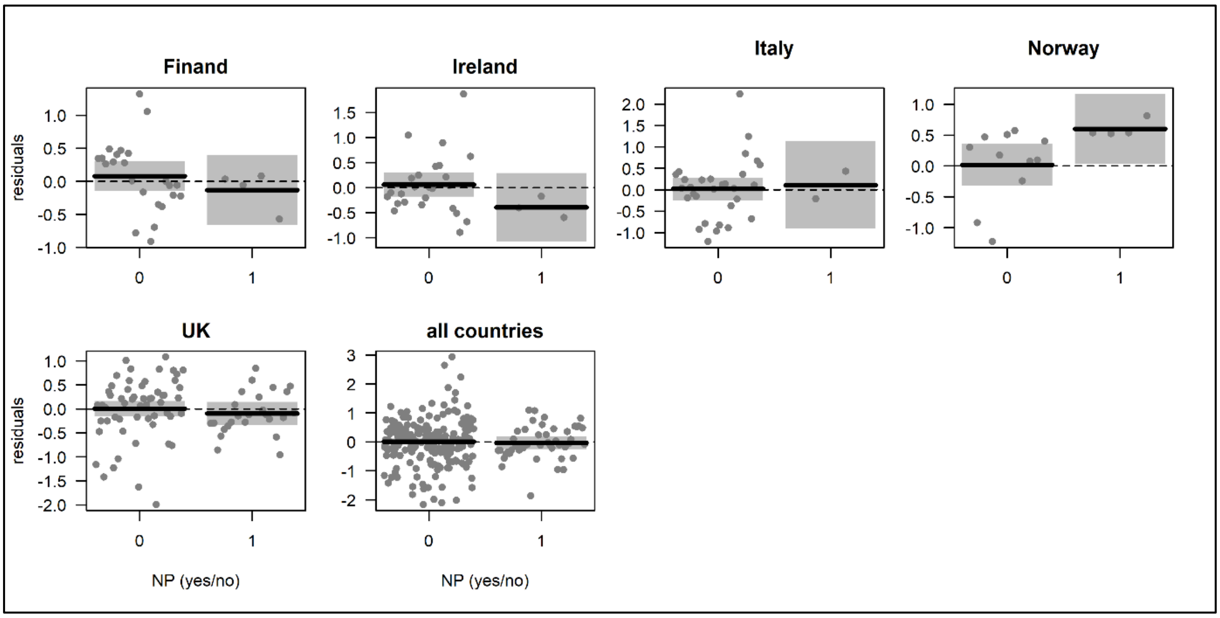

Figure A 6.29: Final mixed model residuals against predictor national park (NP) for all observations and for countries with multiple observations.

Valuation Methods

Even though both methods, CVM (Contingent Valuation Method) and TVM (Travel Cost Method), are well established environmental economic methods for the valuation of the recreational value per visit (the same response variable), someone might be concerned that the two different valuation methods do not measure the same thing and thus, result in different response variables. This would question the criterion validity of the two valuation methods. Even though this question is discussed elsewhere in literature (Rolfe and Dyack 2010, Clarke 2001) and is far beyond the scope of this paper, we conduct additional statistical investigations on this issue. Figure A6.30 shows the residuals of the final mixed model for the valuation methods CVM and TVM. The valuation method does not show any considerable effect on the residual.

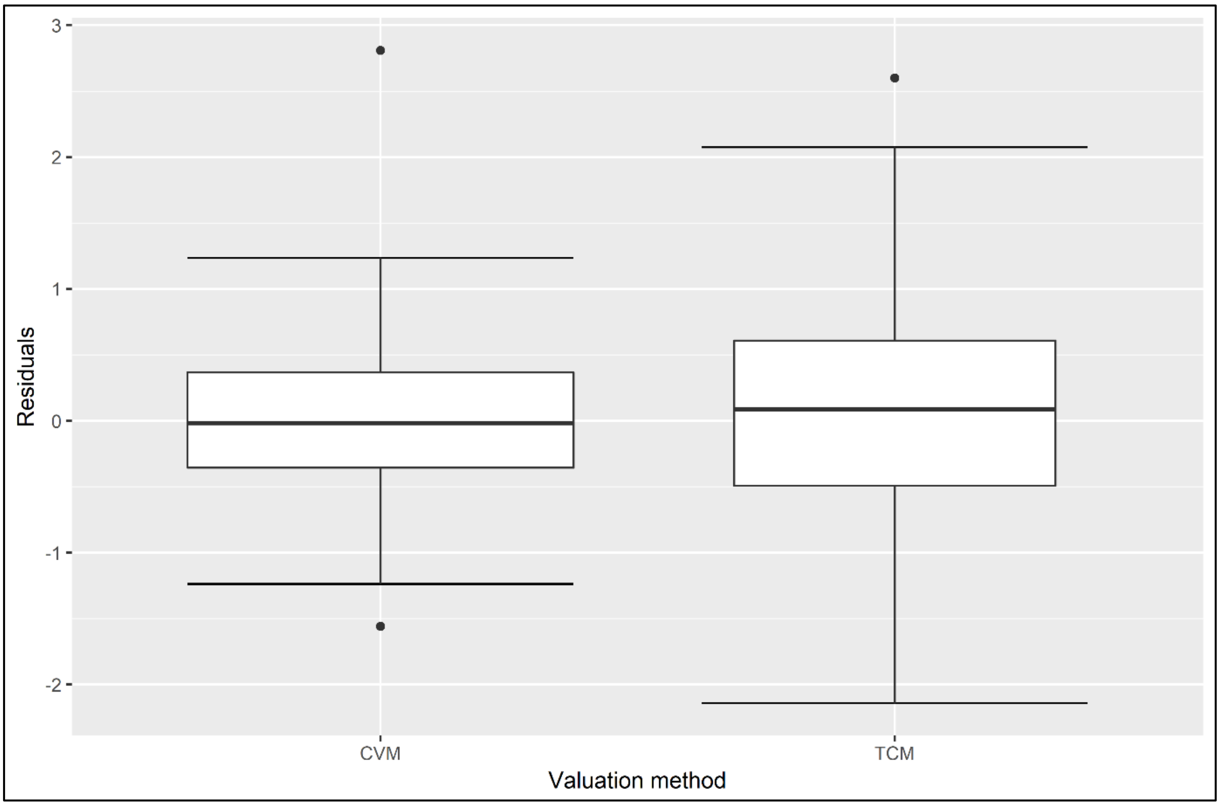

Figure A 6.30: Residuals of the final mixed model by valuation method.

Using subsets of our primary data containing observations obtained either only by CVM or by TCM allows us estimating models like the models presented in this paper. One would expect similar patterns in terms of predictor signs and significance levels, even though the reduced sample size causes p-values to climb in the subset models and hence significance levels to decrease. The reduced sample size by approximately half causes insufficient observations for each level of some factor variables and thus, the models presented in this paper can only be approximated. Due to limited degrees of freedom, variable selection procedures cannot be repeated.

Table A6.2: Linear fixed and mixed effects model after stepwise variable selection with the ln of the value per visit as dependent variable (€, 2013) and first author as random intercept in the mixed effects model for the full data set, and similar models for CVM studies only and TCM studies only.

	Full linear mixed effects model			Linear mixed effects model (CVM only)			Linear mixed effects model (TCM only)		
Variable	**Coefficient**	**p-value**		**Coefficient**	**p-value**		**Coefficient**	**p-value**	
Intercept	5.13	3e-04	***	5.66	1.00E-04	***	6.06	3.28E-02	*
TCM	0.68	<1e-16	***	—	—		—	—	
Value-measure v/visit	-0.35	5.1e-02	●	-0.71	4.00E-04	***	-2.55E-02	0.94	
Ln(ha)	7.3e-02	8.1e-03	**	1.48E-02	0.60		0.13	1.87E-02	*
Ln(forest)	-0.18	4.7e-02	*	-2.04E-02	0.80		-0.54	6.90E-03	**
Ln(grassland)	-0.11	6.4e-02	●	2.19E-02	0.68		-0.27	2.60E-02	*
Rain days	-5.9e-03	7.4e-03	**	-2.60E-03	0.26		-0.01	2.65E-02	*
Slope	0.15	3.4e-02	*	0.18	8.80E-03	**	0.15	0.27	
Ln(pop)	-0.23	9e-04	***	-0.25	5.00E-04	***	-0.22	0.12	
Unemployment	0.27	9.2e-02	●	-8.20E-02	0.67		0.34	0.26	
N	244			140			104		

We report significance levels by indicating p-values of up to 0.001, 0.01, 0.05 and 0.1 by "*", "**", "*" and "•".**

Table A6.2 shows the final mixed effects model for the full data set that was already presented earlier in this paper and the models for the subsets including observations estimated either only based on CVM or TCM (the factor variable TCM vs. CVM had to be excluded from these models). All variables but unemployment for the CVM model (not significant) show the same signs across all three models, which indicates the robustness of our models. Significance levels decrease for the subset models due to reduced degrees of freedoms. Interestingly, values per visit obtained by the TCM method are far more sensitive to site and context characteristics than values per visit obtained by the CVM method. The mean absolute parameter value for those covariates (the last 7 in Table A6.2) is three times bigger for the TCM model (0.6) than for the CVM model (0.2). This may indicate that the fictive nature of stated preference methods (such as CVM) results in difficulties in detecting effects for site and context characteristics as compared to reveal preference methods (such as TCM).

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Table A6.3: Models' summary and spatial predictors used in other meta-analyses, their signs and significance levels.

Gerneral Information	Author	Smith&Kaoru 1990						Rosenberger & Loomis 2001	Shresta et al. 2007				Brander et al. 2007	Zandersen & Tol 2009						Londoño & Johnston 2012		Sen et al. 2011	Sen et al. 2012	Sen et al. 2013	Hysková 2013	
	Model nr.	1	2	3	4	5	6	1	1	2	3	4	1	1	2	3	1	2	3	1	1	1	1	1	1	2
LC/U continuous*[2]	Fraction open land															+ ***			+							
	Forest area (%)																								+	+ **
	Species diversity index															–			+							
	Tree age diversity															– ***			–							
	Share of live reef area																			+ ***	+ **					
Other site chara-cteristics (continuous)	Size												+ ***	+	+	– ***	– **	– ***	– **	–	–					
	Size²													–	– *	– *	+ *	+ **	+ *							
	Visits per day												– **													
	Latitude															–			–							
Context characteristics (continuous)	Population density														+ **	+ ***		+	+			+			– ***	
	GDP / capita														+	– ***		+	+							
LC/U dummies	Mountain & heathland																					+ *	+ *	+		
	Forest	– */●	– **/*	– /*	– /	– /*	+ /	–			+ *	– *														
	Woodland & forest																							+ *		
	Woodland																									
	Farmland & wood																						+ ***			
	Grassland, farm & wood																					+ *				
	Wetland																							+ **		
	Freshwater & wetland																						+	+		
	Wetland & lake																									
	Lake	– **/*	– ***/*	– **/*	– ***/*	– */–	– ***/*	– *	– *	– *																
	River	– **/●	– */●	– */●	– ●/	+ /	– * / ●	+ *	+ *		– *	+ *														
	Freshwater & floodplain																							+		
	Freshwater, marine & coastal																					+				
	Coastal marine																							+ **		
	Marine & coastal																						+ *			
	Ocean																									
	Beach																									
	Artificial or natural reef																			+ ***	+ ***					

Gerneral Information	Author	Smith&Kaoru 1990						Rosenberger & Loomis 2001	Shresta et al. 2007				Brander et al. 2007	Zandersen & Tol 2009						Londoño & Johnston 2012		Sen et al. 2011	Sen et al. 2012	Sen et al. 2013	Hysková 2013	
	Model nr.	1	2	3	4	5	6	1	1	2	3	4	1	1	2	3	1	2	3	1	1	1	1	1	1	2
	National Park	+ /	– /	+ */*	+ /	– /	+ /																			
	Protected area																									
	Marine protected area																			+ *	+ *					
	State Park	+ ***/**	+ ***/**	+ */	+ ***/**	+ ***/**	+ ***/**																			
Other site chara-cteristics (dummies)	USDA or Chinese National Forest							– *	– *		–	– *														
	Designated site																					+				
	Developed facilities										– *															
	Public								+ *	+ *	+ *															
	Urban fringe																							+ ***		
	East Africa												+ *							– *	–					
	Caribbean																									
	Indian Ocean																									
	Southeast Asia																									
Regional dummies	Non-UK																					+ ***		+ ****		
	Australia																									
	Mediterranean bio-geographical region																									
	Northern Italy																									
	Eastern Europe																								+ ***	+ **
	Dependent (y)	CSP[°3] / unit of use						CSP / activity day	CSP or WTP[°4]/ day visit				Value/visit	CSP/visit			CSP/visit/ha			WTP / day visit		Value/visit			CSP/visit	
	Observations / studies	399 / 77	399 / 77	405 / 77	399 / 77	399 / 77	399 / 77	760 / 163	682 / 131				100 / 52	up to 189 / 12						27 / 85	71 (CVM[°5] only)	193 / 98	245 / 106	297 / 98	57 / 17	
	Scale / study area	US						North America	all US	North-east	South- east	interm. West	global coral reefs	forest recreation in Europe						global coral reefs		global	global	global	European forest	
	Model type	GLM[°7]						GLM	GLM				LMM[°8] (by author)	GLM						LMM (by study)		GLM			GLM	
Model description	Model estimation	OLS[°10]						OLS	OLS				max. loglikelihood	OLS						max. loglikelihood		OLS			OLS	
	Transformation y	log						|	|				log	log						log		log			log	
	Heteroskedasticity, autocorrelation	Behind backslash (/) Newey-West (1987) variant of White's (1980) consistent covariance matrix; before none						White's standard error	Newey-West version of White correction				|	robust Huber–White variance estimator						Huber–White corrected standard errors		Huber-White-adjusted standard errors		cluster-robust standard errors	Huber-White robust standard errors	
	R^2 (a.: adjusted; m.: multiple)	0.15	0.42	0.15	0.45	0.3	0.43	a. 0.27	a. 0.26	a. 0.28	a. 0.66	a. 0.36	|	0.76	0.76	0.85	0.28	0.28	0.31	|	|	a. 0.75	a. 0.66	a. 0.72	0.91	0.69
	Transfer error[°12]	|						73% to 319% of raw average values [°13]	37%	average of 26%			186%	|						98.82%	93.77%	|			|	

Gerneral Information	Author	Wang et al. 2013	Brander et al. 2015	De Salvo & Signorello 2015									Fitzpatrick et al. 2017*2																	
	Model nr.	1	1	1	2	3	4	5	6	7	8	9	3*1	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
LC/U continuous*2	Fraction open land																													
	Forest area (%)																													
	Species diversity index																													
	Tree age diversity																													
	Share of live reef area												+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	+ ***	– ***	+
Other site chara-cteristics (continuous)	Size		+ *										+	+ *	–	–	+	+	+ **	–	+ **	+	– *	–	–	+ **	– **	–	+ ***	+
	Size²																													
	Visits per day		– ***																											
	Latitude																													
Context characteristics (continuous)	Population density																													
	GDP / capita																													
LC/U dummies	Mountain & heathland																													
	Forest																													
	Woodland & forest	+ **																												
	Woodland			+ **	+	+	– ***	–		+	–	–																		
	Farmland & wood																													
	Grassland, farm & wood																													
	Wetland			– **	–	–	+**	+		–	+	+																		
	Freshwater & wetland																													
	Wetland & lake	– **																												
	Lake																													
	River	+ **																												
	Freshwater & floodplain																													
	Freshwater, marine & coastal																													
	Coastal marine	– **																												
	Marine & coastal																													
	Ocean	+ **																												
	Beach	+ **																												
	Artificial or natural reef												+	+ **	+ **	+ **	+ **	+ **	+ ***	+ *	+ ***	+	+ **	+ **	+	+ **		+ **		+

Gerneral Information	Author	Wang et al. 2013	Brander et al. 2015	De Salvo & Signorello 2015								
	Model nr.	1	1	1	2	3	4	5	6	7	8	9
Other site chara-cteristics (dummies)	National Park	+ **										
	Protected area	+		–	–	– **	–	– •	+ ***	–	–	– •
	Marine protected area											
	State Park											
	USDA or Chinese National Forest	+ **										
	Designated site											
	Developed facilities											
	Public											
	Urban fringe											
Regional dummies	East Africa											
	Caribbean		+ **									
	Indian Ocean		+ ***									
	Southeast Asia		+ *									
	Non-UK											
	Australia		–									
	Mediterranean bio-geographical region			– **	–	– •		–		–	– **	– **
	Northern Italy			– ***	– **	– ***	– ***	–	– ***	–	– •	– *
	Eastern Europe											
Model description	Dependent (y)	CSP/day	value/visit	CSP / day visit								
	Observations / studies	|[°6] / 706	73 / 33	46 / 265								
	Scale / study area	U.S. /China	global coral reefs	Italy								
	Model type	GLM	GLM	GLM			LFE[°9] (by study)	LMM (by study)	LFE (by site)	LMM (by site)	LFE (by author)	LMM (by author)
	Model estimation	OLS	|	OLS			|					
	Transformation y	log	log	linear	log	Box-Cox	|					
	Heteroskedasticity, autocorrelation	|	White's heteroskedasticity-consistent standard errors	Huber–White robust standard errors								
	R^2 (a.: adjusted; m.: multiple)	0.48	a. 0.41	0.32	0.33	|						
	Transfer error[°12]	19%	|	|								

Gerneral Information	Author	Fitzpatrick et al. 2017°2																	
	Model nr.	3[°1]	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Other site chara-cteristics (dummies)	National Park																		
	Protected area																		
	Marine protected area	+	+	+	+ *	+	+	+ ***	–	+ ***	–	+ **	+ **	+	+	+ ***	+		+
	State Park																		
	USDA or Chinese National Forest																		
	Designated site																		
	Developed facilities																		
	Public																		
	Urban fringe																		
Regional dummies	East Africa	– ***	–	–	+	–	+		– *		–	– ***	–	– ***	+	+ **	– ***		– **
	Caribbean																		
	Indian Ocean																		
	Southeast Asia																		
	Non-UK																		
	Australia																		
	Mediterranean bio-geographical region																		
	Northern Italy																		
	Eastern Europe																		
Model description	Dependent (y)	WTP / person day																	
	Observations / studies	91	76	85	71	91	76	85	71	91	76	85	71	91	76	85	71	91	76
	Scale / study area	global coral reefs																	
	Model type	GLM		LM single variable threshold (shaare of live reef)				LM threshold sample splitting (share of live reef)				LM single variable threshold (reef area)				LM threshold sample splitting (reef area)			
	Model estimation	RSS[°11]		maximum likelihood															
	Transformation y	log																	
	Heteroskedasticity, autocorrelation	|																	
	R^2 (a.: adjusted; m.: multiple)	|																	
	Transfer error[°12]	110%	97%	96%	93%	101%	117%	149%	146%	160%	149%	97%	93%	107%	98%	106%	97%	121%	98%

Significance levels are indicated as follow: p-values up to 0.001, 0.01, 0.05 and 0.1 by "***", "**", "*" and "•".

°1 - First two models are not reported as they are a replication of Londoño & Johnston 2012.

°2 - LU/C: land use or land cover

°3 - CSP: consumers surplus

°4 - WTP: willingness to pay

°5 - CVM: contingent valuation method

°6 - |: no information available or not applicable

°7 - GLM: general linear model

°8 - LMM: linear mixed model

°9 - LFE: linear fixed effect model (including a panel variable)

°10 - OLS: ordinary least square

°11 - RSS: residual sum of squares

°12 - Transfer error: relative mean prediction error or out of sample test if nothing else is declaired

°13 - In-sample convergent validity: forecast values ranged from 73% to 319% of the raw average values

7 Mapping the Recreational Ecosystem Services and its Values across Europe: A Combination of GIS and Meta-Analysis

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Keywords:

Ecosystem service mapping
GIS
Meta-analysis
Value transfer
Nature recreation

Abstract

We map recreational visits and the economic value per visit spatially explicit across Europe's non-urban ecosystems using GIS, meta-analysis and geostatistical modelling techniques.

Therefore, we develop a meta-analytic visitor arrival function and a meta-analytic value transfer function by regression analysis. Primary data on the dependent variables are collected from visitor monitoring and valuation studies. We analyse more than 225 studies including visitor counts and value estimates to more than 550 separate case study areas.

Focusing on continuous spatial biophysical and socio-economic predictor variables, we identify underlying spatial drivers of recreational ecosystem service values. By combining our models with spatial explanatory variable layers we predict annual recreational visits and the value per visit on a one km² resolution across Europe. The resulting maps illustrate spatial variations of recreational visitor numbers and the value per visit. In total we predict about 11 billion annual visits to Europe's non-urban ecosystems amounting an economic value of € 57 billion. Comparing our estimates with mean/unit value transfers reveals that the spatial variations of visitor numbers are substantially more important for determining the recreational value per ha than variations in the value per visit.

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7.1 Introduction

Recreation is an ecosystem service supplied by non-urban ecosystems that is of substantial economic value and offers considerable economic opportunities for local communities in terms of income and employment (MA 2005; Maes *et al.* 2012b; Nahuelhual *et al.* 2013; Paracchini *et al.* 2014; Peña *et al.* 2015). To acknowledge this ecosystem service and to integrate it into land-use planning and resource allocation policies, spatially explicit information on the flow of visitors as well of the recreational economic value is fundamental (Maes *et al.* 2012a; Schägner *et al.* 2013; TEEB 2011). In this study we map the value of recreational ecosystem services spatially explicit across all of terrestrial non-urban Europe by estimating a meta-analytic visitor arrival and meta-analytic value transfer function.

The number of studies assessing ecosystem service values spatially explicit has grown exponentially over recent years. Nevertheless, most studies use a relatively simple approach by applying mean value estimates to land cover classes. This is also the case for studies mapping of recreational services. Only a few studies assess the spatial variations of visitor flows and the economic value associated with these flows more in detail by applying some sort of spatially explicit modelling (Schägner *et al.* 2013). Some studies map recreational ecosystem service values by spatially explicit models that are parameterized and validated based on primary data. However, only a few of them assess spatial variations in both, the number of visits and the value per visit (VV) separately and focus on a large continuous area covering various ecosystems. Ghermandi and Nunes (2013) for example map global costal recreation by applying spatially explicit meta-analytic value transfer. In their model, however, the dependent variable is the recreational value per ha and thus no separate information on the visitor numbers and VV is available. Bateman *et al.* (1995) and Bateman *et al.* (1999) model visitor numbers to several forest sites in Wales using linear regression analysis. The predicted visits are combined with a constant mean value estimate per visit (unit value transfer) to derive value estimates for the considered forest sites and thus, spatial variations in the VV are not accounted for. In contrast, using an approach similar to the one we apply, Brander *et al.* (2015) map both, recreational visits and VV throughout coral reef in Southeast Asia spatially explicit to assess the overall recreational value of different locations. They focus, however, only on one land cover type and use a limited set of spatial predictor variables. Within the UK National Ecosystem Assessment, recreational services are mapped at a national scale by using a count data model, which is based on data from a national recreation survey and by a meta-analytic value transfer function to predict the value per recreational visit (Bateman *et al.* 2011b; Sen *et al.* 2013). Several other studies use survey data based models to predict visitors to certain locations by choice models or random utility models. A similar approach is applied by Brainard (1999) to map the value of alternative forest sites, but values per recreational visit are estimated based on estimations of the travel cost between origin zone and the destination of each recreational trip (travel cost method, TCM). Moons *et al.* (2008) instead combine (TCM) for estimating the value per recreational visit with a choice model that is based on survey data and predicts the probability of an individual visiting a certain site. The method is applied to evaluate the value of alternative hypothetical new forest sites in Flanders, Belgium. Also Termansen *et al.* (2008) use a choice model to predict visitor numbers to forest sites in the area of Copenhagen and a random utility model to estimate the value per recreational visit. Termansen *et al.* (2013) model visitor numbers to Danish forest sites by modelling, first, the total demand for forest recreation and second the choice among alternative forest sites and the value per trip in a random utility model. The models are parametrized based on regression analysis of survey data and respondents to such survey may not be representative for the society as a whole. Furthermore, survey data do not capture visitors living beyond the scope of the surveyed area. Typically, such surveys data is available only at a regional and national level and thus, studies relying on such data do map recreation only at regional to national scale.

In this study, we map both: recreational visits and the value per recreational visit spatially explicit throughout all non-urban ecosystems across Europe by means of meta-analytic regression models. Primary data was collected from visitor monitoring studies and primary valuation studies. In total our databases are comprised of 1,267 visitor estimates of 529 separate sites and 245 value estimates from 147 case study areas. Multiplying the predictions of our two models allows us to predict the recreational value per ha at any location throughout non-urban Europe at a one 1 km² resolution. These models can be used to support several policies: (1) the ex-ante evaluation of land-use policies, (2) efficient resource allocation by conservation prioritization of areas of high recreational value, (3) the design of recreational facilities in accordance with expected recreational visitor numbers or (4) the development of a green GDP or a System of Environmental Accounts (SEA) at different spatial scales. To our knowledge, we present the first study mapping recreational visitor numbers and recreational values across all land-cover classes at a continental scale. To estimate our models, we used large sets of innovative continuous biophysical and socio-economic predictor variables that we developed as spatial GIS raster layers.

The paper is organized as the following. In section 2, we present the primary data on our dependent variables as well the spatial predictor variables we use. Section 3 describes the statistical regression techniques we use to estimate our models. Results are presented and discussed in consecutive sections. Finally, we conclude.

7.2 Data

7.2.1 Primary Data

Our primary data that presents the dependent variables of our models consists of two separate data sets. The first data set represents recreational visitor estimates and the second data set consists of estimates of the VV. Both data sets are developed by a broad literature review of the literature on recreational visitor monitoring and primary valuation studies. Studies are identified through internet searches, a review of relevant literature and by contacting researchers involved in this field.

Our primary data on the recreational use consists of 1,267 observations of the total annual visitor estimates at 529 separate case study areas throughout Europe. We derived the data from a review of 150 visitor monitoring studies and data bases as well as governmental reports. We divided the total number of visits by the hectare size of each case study area to obtain the visitor density (visitors per ha). The visitor density ranges from almost zero (0.03) up to 158,740 visits per ha and year. The distribution is however, highly skewed with a mean of 2,362 and a median of 35.8 (see Table 20).

Table 20: Descriptive statistics of visitor density (visits / ha/a).

N	mean	standard deviation	median	min	max
529	2,362	12,061	35.8	0.03	158,740

Our primary data on recreational value consists of 245 estimates of monetary values per recreational visit for 147 separate nature areas in Europe. We obtain the data from 75 valuation studies using either Travel Cost Method (TCM) or Contingent Valuation Method (CVM). We transfer all value estimates to € values and to the 2013 price level using purchasing power parity and country specific inflation data. From the total data set we exclude one outlier, showing an extreme deviation of 60 times the mean value. The remaining 244 value estimates range from € 0.16 to 64.7 per visit with a mean of € 7.17 and a median of € 2.8 (see Table 21).

Table 21: Descriptive statistics of value per visit estimates in €, 2013.

Min.	1st Qu.	Median	Mean	3rd Qu.	Max.	St. dev.
0.16	1.54	2.8	7.17	7.76	64.7	10.98

For each case study site we obtain or create a spatial layer in vector format, containing the boundaries of the surveyed nature area. Polygons for some nature areas are obtained from official sources (EEA 2013; IUCN and UNEP 2015) or from the case study authors. Most polygons are drawn manually in ArcGIS, based on information supplied by the case study publications, the study authors or based on information from internet inquiries. In several cases, we are not able to get any approximation of the location and shape of the case study area and thus could not include those studies in our data base. The case study areas differ widely in terms of size, location, the estimated VV and ecosystem characteristics. The size of case study area ranges from 1.9 ha, for a small Nature Park, east of Padova, Italy, up to 18,048 km² for the Jämtland mountain range in North Sweden. The distribution of the case study areas are presented in Figure 30.

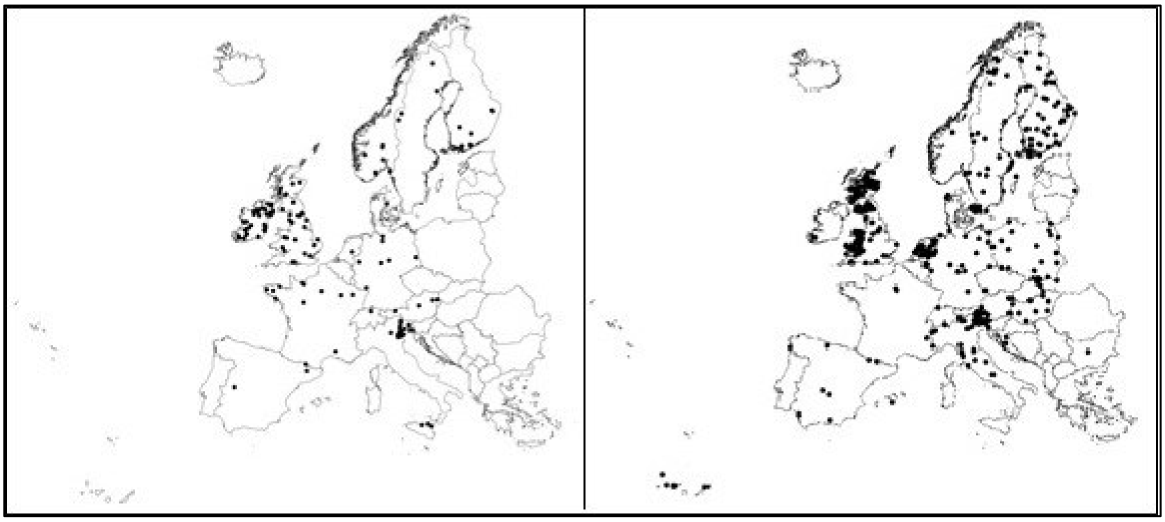

Figure 30: Location of case study areas, right: represented in the primary valuation studies, left: represented in the visitor monitoring studies

7.2.2 Predictors

For the statistical regression analysis, we develop a number of predictor variables, divided into three categories: (1) study characteristics, describing the methodology of primary data collection, (2) site characteristics, describing the study area itself and (3) context characteristics, describing the spatial context of the study area. We select the variables based on a review of past meta-analyses on recreational valuation studies and regression analysis of recreational visitor flows. A complete list of all predictors used in our analysis is presented in Table 22.

For the valuation we chose three methodological characteristics, which we assumed to have a strong influence on the valuation results (see also Table 16 in chapter 6). First, we distinguish studies by their valuation method, which is relatively equally distributed with 140 studies using CVM and 104 using TCM. Second, we distinguish whether studies consider use-values only (226) or if they consider use and option values. Finally, we consider whether studies estimate VV, which is the case of the majority of studies (197), or if studies estimate the value per day visit, value per party visit or the value per month or year of access. For visitor data, we encountered difficulties to define methodological characteristics, because reporting on study methodologies was poor in most studies. Therefore, we classified all visitor monitoring studies according to primary data collection quality using one as the lowest and ten as the highest quality. The quality judgment represents a composite indicator of different quality dimensions: the type of publication (scientific vs. grey literature), the purpose of the visitor monitoring study (scientific vs. political), the institution conducting the study (academic, national park management, others) and the methodological documentation of study (full, incomplete, none). If the documentation of the study was available, we assessed the quality of methodologies based on details such as the temporal and spatial counting resolution, manual or electronic counting devices and the temporal and spatial up-scaling methodology. Finally, a very important aspect for the visitor monitoring studies quality is the description of the study area. Some publications do not supply maps and only rough descriptions of the study area. If the area of the study area is uncertain, then the visitor density is uncertain as well. In addition, we coded all observations in respect to the year of primary data collection.

The main focus of our analysis, however, is to identify the effects of spatial determinants of recreational values in order to produce spatially distributed predictions. Therefore, we prepare several EU wide geospatial layers of site and context characteristics in raster format. It is worth noting that limitations in the availability, accuracy, comprehensiveness and consistency of Europe-wide data sets restrict our choice of predictors and is one limitation to our statistical analysis. We use available GIS data sets and, if necessary the layers are processed in order to derive our predictor variable raster layers. The GIS processing is done with ArcGIS 10.2. We use the following site characteristics in our regression analysis:

(1) Land cover: We can only use a limited number of land cover combinations in our analysis because several CORINE land cover classes occur only rarely, which would result in many zero values in our spread sheet, and thus, the detection of significant effects would become more difficult and vulnerable to outliers. In addition, aggregates of land cover classes are often correlated with each other and thereby can cause problems of collinearity. Based on the analysis of past meta-analyses of recreational valuation studies and visitor modelling studies, we choose the following land cover/use classes as predictor variables for our analysis. To account for forest, we use the Joint Research Centre forest cover map (EC 2006) and compute the mean number of forest pixels (25m resolution) per hectare that are classified as either coniferous or broadleaved forest within each study site. For other land cover types we used the CORINE data set (EEA 2006) to determine the percentage of several land cover classes in the study sites. In particular we determine the share of all natural vegetation, agricultural area and grassland.

(2) Land cover diversity: From the CORINE land cover data set we compute the Simpson Diversity Index (Magurran 1988) of land cover types within a 3 km radius for each pixel of 100 m resolution raster map covering all Europe. We assume that more diverse landscapes are perceived as more beautiful and may therefore positively affect the VV and the visitation frequency.

(3) Water bodies: We compute two 300 m resolution grids of the share of surface area covered with rivers and lakes or ocean using the Euro Regional Map as input data set (EG 2010). Then we apply a kernel density function tool to compute the amount of surface covered with water within a 3 km radius of each pixel. The density function allows a water area that is more remote to be weighted less than water nearby, thereby incorporating a distance decay effect. We believe that water bodies attract visitors and cause the VV to be higher.

(4) Biodiversity: We use the total number of red list species encountered in a study area as an indicator for biodiversity (IUCN 2013). We assume that biodiversity may attract visitors from distant locations and result in higher VV. In addition, we use a dummy variable to indicate whether at least 50% of the study site is designated as a national park (NP).

(5) Climate: We use three climatic variables in our model, under the assumption that better climate in terms of higher temperature, less rain and more sunshine attracts visitors from distant locations for longer recreational trips. As a temperature indicator, we apply a data set from Biavetti *et al.* (2014) indicating the mean number of days per year with maximum temperature above five degrees Celsius. We use a similar data set from Burek (unpublished) indicating the mean number of days per year with at least some precipitation and the mean hours of sunshine per day.

(6) Topography: We use the slope of the digital elevation map from the European Environmental Agency (EEA 2015a) for two indicators describing the topography of the landscape. First, we use the slope value of the 100 m digital elevation map. Second, we compute the area visible from each pixel within a 30 km radius using the viewshed tool. In order to accelerate the viewshed processing we aggregated the digital elevation map to a 1000 m resolution raster grid. We expect that mountain regions and regions offering large viewsheds imply special attraction for recreation and generate higher values and number of visits.

(7) Trail density: We use trail density as a proxy for overall recreational facilities, which may enhance the recreational experience. From the Open Street Map (OSM) data set (OSM 2012), we extract all vector elements that can be classified as non-motorized traffic infrastructure. We use five OSM classes: trails, foot paths, bike paths, bridle paths and steps. On a 100 m resolution we apply the line density tool to compute an indicator for trail availability. Again, the trails are weighted less with increasing distance from the pixel under analysis.

(8) Street density: Similar to trail density we compute an indicator for street density for all minor roads (Tele Road Atlas road classes 4-6) based on the Tele Road Atlas data set (TS 2006). Roads are an important infrastructure for accessing remote locations and are thereby expected to increase visitor numbers. However, the relationship between road density and the enjoyment of nature recreation is unclear. We do not have a specific hypothesis on the effect of streets on recreational values and our analysis has an exploratory character.

(9) Accessibility: The number of people that can access a specific location within a certain time is likely to have an effect on the visitation rate (Schägner *et al.* 2016b; Schägner *et al.* 2016a), which may in turn negatively affect the quality of nature recreation due to crowding effects (Kalisch 2012).

We use the weighted sum of the total population living within a 130 km radius around each pixel, using population data from Batista e Silva *et al.* (2013). In order to account for distance decay, we applied a Gaussian weight function, so that the population is weighted less with increasing distance from the pixel under analysis. The weight function was calculated so that 95% of its integral is located within the 130 km radius.

(10) Socio-economic effects: We use GDP per capita, the unemployment rate and the share of population with upper secondary or tertiary education as proxies for visitors' income and their recreational preferences. For these variables, we extract mean values for the last ten years (as far as available) and the highest data resolution available, which is either NUTS2[45] or NUTS3 level from the Eurostat database (EC 2013).

(11) Share of national park area: We computed the share of each case study area that is designated as a NP. Information on NPs were derived from the World Database of Protected Areas (WDPA). National parks are considered to receive higher visitor flows (Fredman *et al.* 2007) and higher VV.

[45] NUTS is referred to as Nomenclature of Units for Territorial Statistics, which is a hierarchical system defined by Eurostat for dividing up the EU territory in order to produce regional statistics at resolution of different administrative levels.

Table 22: Predictor variables used in the regression analysis.

Type	Variables	Explanation*	Mean / sd* (visitor data)	Mean / sd (valuation data)
Study Character-istics:	TCM	1 if TCM; 0 if CVM	—	0.4 / 0.5
	Use & option	1 if use value; 0 if use & option value	—	0.9 / 0.3
	V/visit	1 if V/visit; 0 otherwise	—	0.8 / 0.4
	Study quality	Quality of data collection method	7 / 1.8	—
	Survey Year	Year of data collection	2004 / 4.7	—
Site Character-istics:	Ln (ha)	Natural log of the study site area in ha	7.3 / 2.7	7.8 / 2.8
	Ln (sri)	Simpson Diversity Index of Corine land use/cover within a 3 km radius (100 m resolution raster)	1.1 / 0.4	1.1 / 0.3
	Ln (forest)	Natural log of the share of forest cover of the study area (100 m resolution raster)	—	1.8 / 0.8
	Ln (conifer forest)	Natural log of the share of conifer forest cover of the study area (100 m resolution raster)	1.4 / 1	—
	Ln (broad-leaved forest)	Natural log of the share of broadleaved forest cover of the study area (100 m resolution raster)	0.8 / 0.9	—
	Ln (natural LC)	Natural log of natural land cover of the study area (100 m resolution raster)	—	2 / 1.6
	Ln (agriculture)	Natural log of agricultural land cover of the study area (100 m resolution raster)	—	2.1 / 1.6
	Ln (grassland)	Natural log of grassland land cover of the study area (100 m resolution raster)	—	1.44 / 1.35
	Ln (inland water)	Natural log of inland water body area within 3 km distance weighted by a kernel function (300 m resolution raster)	—	1 / 1.2
	Ln (ocean)	Natural log of ocean area within 3 km distance weighted by a kernel function (300 m resolution raster)	0.6 / 1.3	0.5 / 1.1
	Red list species	Total number of red list species found in study area (1 km resolution raster)	9,462 / 3,337	8,991 / 3,144
	National park	Share of NP area in percentage; 1 if site is a national park; otherwise 0	37 / 44	0.19 / 0.39
	Rain days	Mean number of days with rain per year (1 km resolution raster)	144 / 35	144 / 34

	H sun/day	Mean hours of sunshine per day (1 km resolution raster)	—	4.19 / 1.12
	Days 5^0 C	Mean numbers of days with an average temperature of above 5 degrees (1 km resolution raster)	—	304 / 53
	Viewshed	Area visible from each location within in a 30 km radius (1 km resolution raster)	307 / 251	276 / 214
	Slope	Slope (100 m resolution raster)	13.9 / 16.5	11 / 12.2
	Ln (trail)	Natural log of trail density using density function in order to account for distance decay effect (100 m resolution raster)	5.6 / 2.8	1.37 / 0.97
	Ln (small roads)	Natural log of small roads density using density function in order to account for distance decay effect (100 m resolution raster)	1.3 / 1.1	2.09 / 1.11
	Ln (Streets large)	Natural log of large roads density using density function in order to account for distance decay effect (100 m resolution raster)	0.9 / 0.9	—
Context Characteristics:	Ln (population)	Population living within 130 km radius of the study area using a Gaussian weight function in order to account for distance decay (100 m resolution raster)	16.1 / 1.4	16.2 / 1.08
	GDP/capita	GDP/ capita in the NUTS 2 or 3 region in which the study area is located	—	25,768 / 6,593
	High education	Share of population with higher education in the NUTS 2 or 3 region in which the study area is located	72.7 / 10.7	70.4 / 11.5
	Unemployment	Unemployment rate in the NUTS 2 or 3 region in which the study area is located	6.2 / 3.9	6.2 / 3.3

* For all spatial predictors mean values per study site area are computed.

7.3 Methodology

To map recreational values per ha spatially explicit we transfer values from study sites (sites for which real world observations exist) to policy sites (sites for which we make predictions) by two models: one on the number of recreational visits and one on the economic value per recreational visit. Both models are developed by regression analysis of real world observations of the depend variables and by using comprehensive and continuous raster layers as predictor variables. By employing the estimated models to make predictions, we intra- and extrapolate visitor numbers and the VV across space (see Figure 31). By multiplying the predicted recreational visits per ha with their predicted value at each location of the map, we drive the overall recreational value per ha.

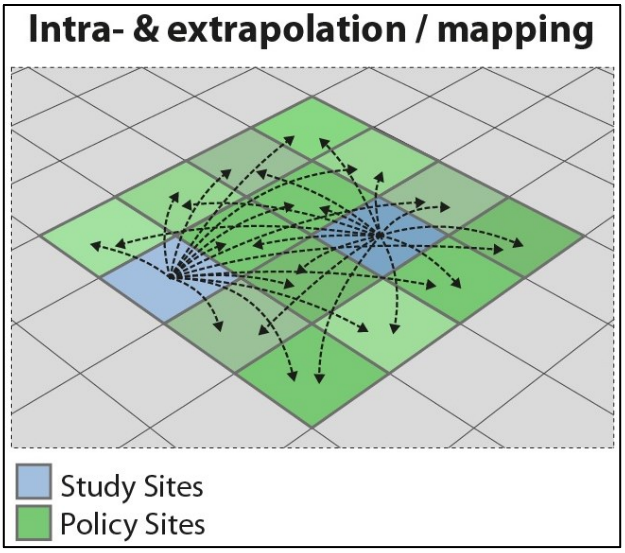

Figure 31: The concept of value transfer and ESS value mapping.

Before conducting the regression analyses to model the visitor density of the different case study areas and to model the value per recreational visit by using the predictor variables described above, we conduct an exploration of our data following the recommendations of Zuur *et al.* (2010) in order to gain initial insights into distributions and dependencies. For some predictors we use logarithmic transformations either because they show a skewed distribution or with the aim to approximately linearize an expected non-linear relationship. We test all our predictors for multicollinearity, but do not identify anything of concern.

We apply a number of regression techniques to identify a model that fits the assumptions of linear regression best. All models are estimated using the open source statistical software R. We start our analysis with a general linear regression (GLM), but it shows a wider spread of the residuals for large fitted values for both our analysis and it is therefore a violation of the homogeneity assumption. We control for this effect by using a linear log-transformed models of the following form:

$$\mathrm{Ln}(Y_i) = \alpha + \beta * X_i + \mu_i \qquad \text{where} \qquad \mu_i \sim N(0, \sigma^2)$$

Y stands for the dependent variables (either the number of visits per ha or the monetary value per recreational visit), α is a constant, β represents a vector of parameters, X is a vector of explanatory variables and μ is the residual, which is normally distributed with the mean of zero and a variance σ. The results are shown on the left (columns 2-4) in Table 23.

We validate our final model against the assumptions of linear regression analysis. Therefore, we plot our residual against fitted values and also against each predictor used in the model as well as predictors not used in the model. One concern is the potential for spatially related residuals.

Among our data base of the valuation studies, several studies use different valuation methodologies to value recreation at the same site. Therefore, it cannot be assumed that these observations are independent. In consequence, we use a general linear mixed model[46] (GLMM) introducing the study site as a random intercept. However, as the introduction of the study site as a random intercept has

[46] In other disciplines, mixed models are also referred as to multilevel analysis, nested data models, hierarchical linear models, and repeated measurements.

almost no effect on the model's results, we abandon this approach. Another concern is in regard to author effects. Several authors conduct multiple valuations of our data base and their specific approach may have an effect on the studies result. We therefore introduce the most common first authors as random intercepts in a mixed model of the following form:

$$\mathrm{Ln}(Y_{ij}) = \alpha + \beta * X_{ij} + \gamma_j + \mu_{ij} \quad \text{where} \quad \mu_{ij} \sim N(0,\ \sigma_\mu^2) \quad \text{and} \quad \gamma_j \sim N(0,\ \sigma_\gamma^2)$$

The random effect is specified by γ_j, representing the correlation of observations from the same sites and which is normally distributed with mean of zero and variance σ_γ. In terms of Akaike Information Criteria (AIC) and Bayesian Information Criteria (BIC) the random effect improves the model considerably. The results of the mixed effects model are shown to the right of Table 23.

We do a similar analysis for the model on the visitor numbers and found that a linear model containing a spatial autocorrelation (SAC) structure performed best in controlling for spatial patterns in the residuals and in regards to AIC and BIC scores. The model's formula remains the same as the formula (1) presented above, but this time we assume that the residuals μ_i of different locations are correlated based on the function f and their distance.

$$\mathrm{cor}\,(\mu_a, \mu_b) = \begin{cases} 1 & if\ a = b \\ f(\mu_a, \mu_b, \rho) & else \end{cases}$$

For all models we conduct stepwise variable selection using maximum likelihood and restricted maximum likelihood estimators to compare AIC and BIC and the likelihood ratio test until all remaining variables are significant at the 0.1 level (see Table 23 and Table 18). We validate our final model against the assumptions of linear regression analysis. Therefore, we plot the residuals against fitted values and against each predictor. We do not identify any linear or non-linear patterns of concern. Models and validation plots are estimated using statistical software R and lme (Bates *et al.* 2015), lattice (Sarkar 2015) sp (Pebesma *et al.* 2015) and gstat R-packages (Pebesma and Graeler 2015).

In order to test the predictive use of the estimated model, we use the model characterized by the lowest AIC and BIC values to map the number of recreational visits per ha and the VV across rural Europe on a one km² resolution. We use a data set of urban morphological zones (EEA 2015c) to cut out all urban areas, because our primary data covers only non-urban ecosystems. The maps indicate how the number of visits and the VV differ across space. By multiplying the two raster maps (number of visits per ha and VV) we obtain the total recreational value per ha of any location throughout Europe.

In a second step, we investigate the contribution of the spatial variation of the number of visits and the VV to the spatial variations of the overall recreational value per ha. Therefore, we conduct two analyses: (1) we compute the mean relative deviation of the predicted pixel scores for two raster data sets, the predicted visits and the predicted VV. Within a product of two variables, the mean relative deviation at each pixel determines the relative influence of the two variables on the mean relative deviation of the output variable; the value per ha. (2) We compare our final results of the predicted recreational value per ha with two alternative estimations of the value per ha, for which we substituted either the predicted number of visits or the predicted VV by the arithmetic mean of our primary data on either the number of visits or the VV. Such a methodology is also referred as to a mean - or unit value transfer. In consequence, the alternative methods to estimate the overall recreational value per ha are characterized by spatial variations which depend either only on the spatial variations in the number of visits or only on spatial variations in the VV, because the other input variable is kept constant across space by taking its mean value. By estimating the correlation between our initial predictions with the two alternative methods, we identify to what extent the number of visits and the VVs drive the total recreational value per ha.

7.4 Results

The results of our regression analysis after variable selection are displayed in Table 23 for our visitor arrival model and in Table 24 for our meta-analytic value transfer function.

7.4.1 Visitor Arrival Model

The GLM of our visitor arrival model using a log-transformed dependent variable shows 14 predictor variables significant at the 0.1 level after variable selection (see left of Table 23). Most coefficients have the expected sign. However, the residual plots of the model show some spatial patterns. The residual bubble plot in Figure 32 shows the spatial distribution of the full model's residual without a spatial correlation structure. It shows clustering of positive and negative residuals across Europe. We applied a SAC to control for these patterns in our second model, which improved the models AIC and BIC values considerably. We compared different SAC structures in our pre-analysis to control for the SAC best. The best model in terms of AIC and BIC values as well as in controlling for the spatial residual patterns, applies an exponential spatial correlation structure of the residuals. The model shows 12 predictor variables significant at the 0.1 level after variable selection (see right of Table 23) and spatially correlated residuals up to a distance of 364 km. The nugget refers to differences between observations, which can neither be explained by the model nor by the SAC due to measurement errors or micro variability.

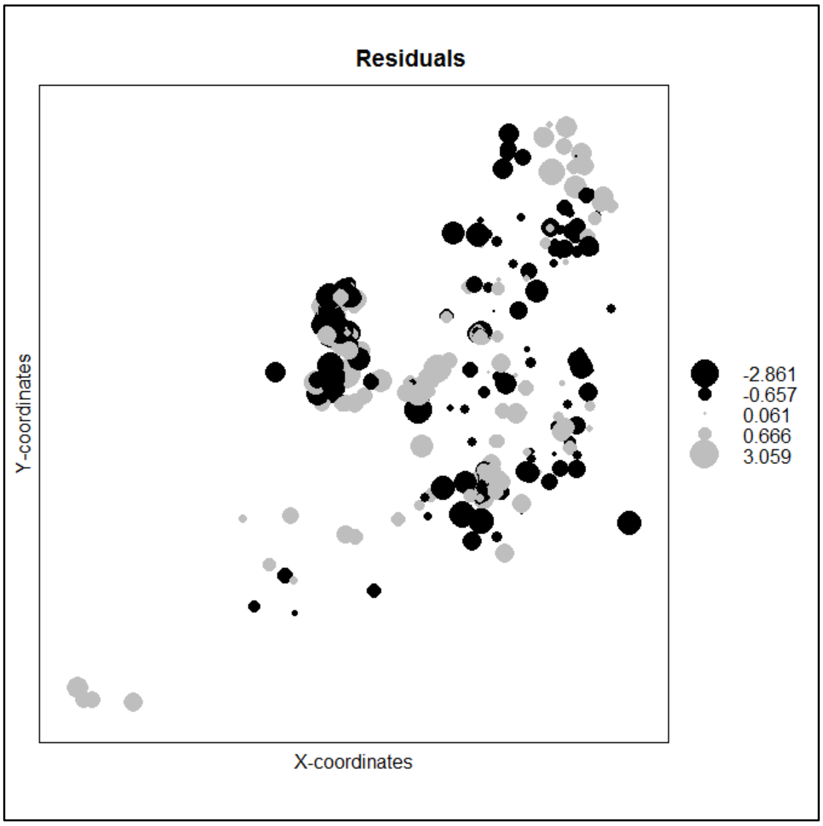

Figure 32: Bubble plot of the spatial distribution of the full model's residual without spatial autocorrelation structure.

A highly significant negative impact on the number of visits per ha can be found for the size of the study area of the visitor monitoring study, which supports theoretical considerations that larger recreational areas act as a substitute in itself. Visitors can spread across larger areas, which results in lower visitors numbers per ha. Interestingly, even though we did not have strong prior expectations regarding the signs of predictors representing forests, both the share of broadleaf and coniferous forests have negative significant signs in the GLM. However, only coniferous forests show a significant effect after introducing the SAC. The proximity to the coast shows a positive significant effect in both models, but only at the 0.1 level in the GLM. The GLM shows also a positive significant effects for land cover diversity and the view shed, but both variables are not significant in the model including the SAC. On the contrary, whether the study site is a NP shows a positive significant effect in the model containing the SAC, but not in the GLM after variable selection. The mean slope value indicating mountainous areas shows significant negative effect in the GLM, but also this variable is not significant in models containing SAC. We find a significant negative effect in both models for the average numbers of days with rain. All our predictor variables representing traffic access infrastructure show positive effects. The availability of large streets is however, only significant in the model containing SAC. Small streets and in particular the availability of trails are highly significant in both models. A highly significant effect is also shown in both models by the population living in the proximity of the study sites areas. The share of the population having upper secondary or tertiary education shows a positive significant effect in the GLM at the 0.1 level, but not after introducing the SAC. To our surprise the unemployment rate shows a significant positive effect in both our models. Our quality judgment of the visitor monitoring study shows a significant negative effect in both our models, whereas the year of the data collection in the visitor monitoring study shows a significant positive effect, but only after introducing the SAC. The negative effect of our quality judgment may indicate that visitor monitoring studies of higher quality result in more accurate estimates, whereas poor quality studies that rely on more assumptions tend to overestimate visitor numbers.[47]

Table 23: Linear fixed model and model containing a spatial residual structure after stepwise variable selection (ln(visits per ha) as dependent variable).

	Linear fixed effect model		**Linear fixed effect model containing spatial residual structure**	
Variable	**Coefficient**	**Sig. level**	**Coefficient**	**Sig. level**
Intercept	-2.79	*	-75.94	*
Ln (ha)	-0.53	***	-0.53	***
Ln (conifer forest)	-0.17	.	-0.21	*
Ln (broadleaved forest)	-0.34	***	—	—
Ln (ocean)	0.10	.	0.21	***
Land cover diversity	0.48	*	—	—
National park	—	—	3.84E-03	.
Viewshed	6.41E-04	*	—	—
Slope	-1.16E-02	*	—	—
Rain days	-7.03E-03	**	-7.75E-03	***
Ln (trails)	0.29	***	0.20	***
Ln (streets large)	—	—	0.21	**
Ln (streets small)	0.16	**	0.29	***
Ln (pop)	0.57	***	0.49	***
Pop high education	1.41E-02	.	—	—
Ln (unemployment)	0.31	*	0.46	.
Study quality	-0.15	***	-0.15	**
Survey Year	—	—	3.75E-02	*
	AIC: 1997		AIC: 1916	
	R^2: 0.74		Range: 364 km	Nugget: 0.48

*** $p \leq 0.001$, ** $p \leq 0.01$, * $p \leq 0.05$, . $p \leq 0.1$

7.4.2 Meta-Analytic Value Transfer Function

For statistical analysis of the VV, the most important predictor is whether the valuation method is TCM or CVM, which shows significantly higher value estimates for TCM in both models. Whether the study assesses “*use values*” or “*use and option values*” does not show a significant effect in our analysis. It is

[47] No significant effects are found in both models for variables describing the share of grassland, the share of arable land, proximity to inland water bodies, the number of IUCN red list species, the mean number of days per year above 5 degrees Celsius, the mean hours of sunshine per day and the GDP per capita.

noteworthy, that only a small share of all studies in our data base consider option value in the valuation approach, and it may therefore be difficult to identify a significant effect. Whether the study estimates the value of single visit or another valuation object (value per party visit, month of access etc.) does show significant negative effects in the mixed effect model, an outcome confirming expectations. It is however, not significant in the fixed effect model.

With respect to spatial predictor variables, six predictors show a significant effect in the GLM and seven in GLMM, which is considerably less than in our visitor arrival model. The explained variations of the GLM on the VV is considerably lower than the GLM of the visitor numbers with and R^2 of only about 0.39 as compared to 0.74.

Contrary to the visitor arrival model, the study site size shows a positive significant effect for both models of the meta-analysis. This confirms our hypothesis that larger areas tend to have a higher recreational VV. Similar to the visitor arrival model, the share of forest cover shows significant negative effects in both models. Grassland also shows a negative significant effect in both our analysis, which is not the case for the visitor arrival models. The availability of water cover, either inland or ocean, shows no significant effect in the meta-analysis. On the contrary, in the visitor arrival model, the proximity to the ocean shows a positive effect. The number of days with precipitation has a significant negative effect on the VV in all our models, as it is also the case for the number of visits. The mean slope value of the study sites — an indicator for mountainous areas — shows a significant positive effect in the GLM, but not in the GLMM. This indicates that people derive greater pleasure from visiting mountains relative to other types of landscape. However, the GLM of the visitor arrival model indicates that fiewer people tend to visit mountainous regions. Population pressure shows a strong and significant negative effect in all our models. This could indicate that people prefer nature recreation in areas with lower population density and value such trips higher. Nevertheless, population pressure shows strong positive effects on the number of visits. Unemployment shows a significant positive effect in both models on the VV, which again contradicts our expectations. However, a possible explanation could be that people travel far and have high values for visits to rural areas, which tend to have higher unemployment rates (Copus *et al.* 2006). Nevertheless, it has to be considered that data on unemployment is only available on the NUTS2 or 3 level, and are not spatially explicit. The average unemployment rate for all years available in Eurostat shows high values in our database for sites located in Finland, France, Germany Spain and Sweden and low for Italy. The variable could also pick up some regional or other unobserved effects, but could not identify any systematic pattern allowing for an explanation.

Table 24: Linear fixed and mixed effect models after stepwise variable selection (ln(value per visit as dependent variable).

	Linear fixed effect model		Linear mixed effect model	
Variable	**Coefficient**	**Sig. level**	**Coefficient**	**Sig. level**
Intercept	6.103	***	5.128	***
TCM	0.772	***	0.678	***
V/visit	—	—	-0.353	.
Ln (ha)	8.92E-02	***	7.25E-02	**
Ln (forest)	-0.174	*	-0.178	*
Ln (grassland)	-0.194	***	-0.105	.
Rain days	-6.27E-03	**	-5.89E-03	**
Slope	—	—	0.146	*
Ln (population)	-0.316	***	-0.230	***
Unemployment	0.424	**	0.273	.
	AIC: 697.9		AIC: 661.1	$\hat{\sigma}_{\gamma}^{2}$ = 0.57
	BIC: 729		BIC: 702.5	$\hat{\sigma}_{\mu}^{2}$ = 0.79

*** $p \leq 0.001$, ** $p \leq 0.01$, * $p \leq 0.05$, . $p \leq 0.1$

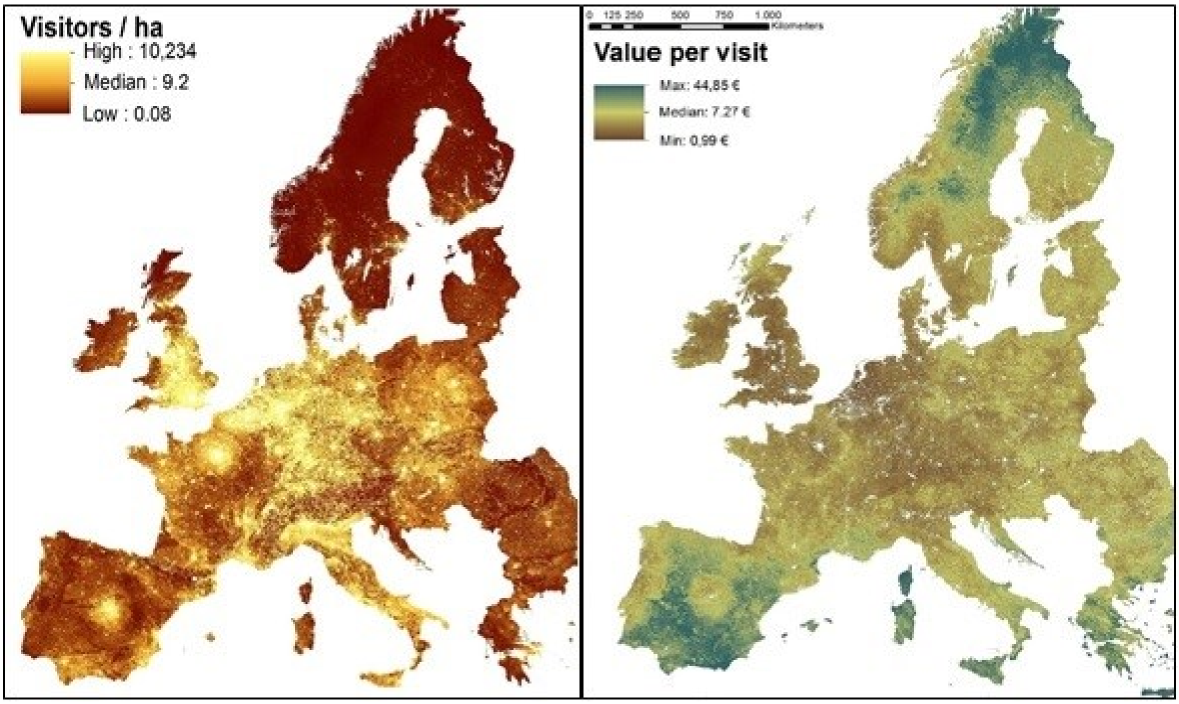

Figure 33: Left: predicted visitors per ha and year, right: predicted value per visit.

Using our best models characterized by the lowest AIC values — the visitor arrival function containing a SAC and the meta-analytic value transfer function containing a random author effect — to make predictions result in two maps, one displaying the predicted visitors per ha and year and one displaying the predicted VV (see Figure 33). Both predictions are skewed with an average number of 27 visits per ha and a median of about 9. The VV shows a mean of € 8.34 and a median € 7.27 [48]. Due to the fact that the response variables are log-transformed we expect to have such skewed distribution and higher prediction errors for large values. It is obvious that the tow predictions, the number of visits and the VV are negatively correlated. High visitation rates are found mainly in the population centres of the Southeast of the UK, the Netherlands, Belgium, in the west of Germany and around Paris but also in parts of Italy. High VV are in contrast found in northern Scandinavia, southern Italy and Spain. By multiplying the two maps (number of visits with the VV), we obtain the overall value per ha (see Figure 34). The mapped values are again highly skewed with a mean value per ha of € 151 and a median value per ha of € 64. Making a prediction to aggregate recreational values across the entire part of Europe covered by our maps by assuming the median size of study sites in our primary data as a resolution results in 11 billion recreational visits to non-urban nature areas, which account for an annual value of about € 57 billion annually.[49]

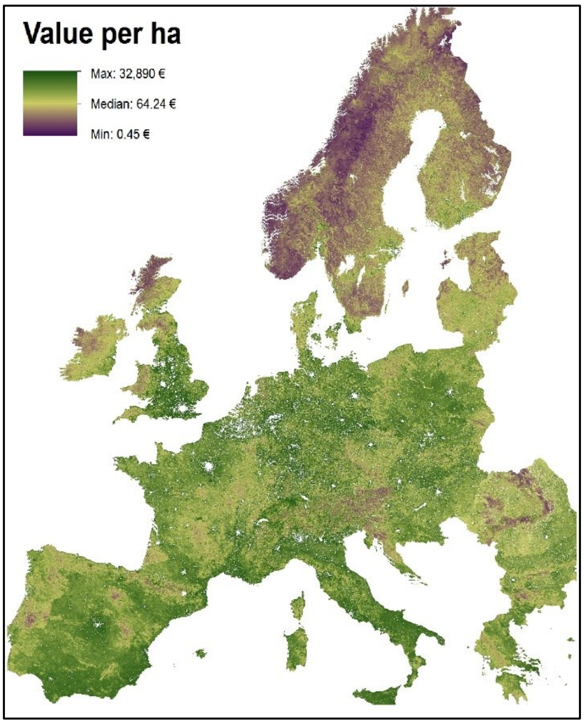

Figure 34: Predicted recreational value per ha.

[48] Note that for illustrative purpose the color shades of all maps are set to cover the same amount of pixels per color shade.

[49] It should be noted that in a log transformed model, it is not possible to take the linear mean to aggregate values across a larger area. For the aggregated prediction, we assume the median study site size as a resolution for the prediction of both the visits per ha and the value per visits.

Comparing the three maps in Figure 33 and 34 reveals that the value per ha is positively correlated with the predicted number of visits, but negatively correlated with the VV. Locations for example in the north of Scandinavia or Scotland receive low numbers of visits and also show a low recreational value per ha. Areas such as the southeast of the UK, the Netherlands, Belgium, and the west of Germany are characterized by high visitor numbers and do show also high values per ha. The correlation between the visits per ha and the value per ha is 0.8, whereas the correlation between the VV and the value per ha is -0.1. If we substitute the predictions of one of our models by the mean estimate of our primary data — either the mean VV or the mean number of visits per ha — to compute the overall recreational value per ha, the resulting maps would be strictly proportional to either the predicted number of visits or the predicted VV. Thus, the correlations between the map of the value per ha and either the map of the predicted visitor number or the VV are the same. The mean relative deviation of the predicted visitor number is 112% whereas mean relative deviation of the predicted VV is only 39%.

7.5 Discussion

Our estimated models fit the data reasonably well and therefore offers valuable information on the main drivers of recreational use and its value across European non-urban ecosystems. The modelling approach can be used to support several policy applications. All predictors with statistically significant effects on the number of recreational visits have signs that are in line with our interpretations or theoretical expectations. Nevertheless, there are also uncertainties related to the model results and prediction accuracy which may be improved by further research.

7.5.1 Spatial Modelling

One major uncertainty in modelling our primary data is related to the independency of single observations, in particular spatial independency, which is one assumption of linear regression analysis. For the visitor data we identified SAC of the residuals, which we accounted for by introducing a spatial residual structure. The question remains, what the source of the SAC is. In an optimal statistical textbook world, introducing SAC in a model would not influence parameter estimates, but only reduce the degrees of freedom of the model. However, looking at real world spatial data, this is hardly ever the case. If parameter estimates are affected as in our case, this may indicate some common spatial econometric problems, such as missing predictors, which are picked up by the spatial error term, a spatial weight matrix or a non-linear relationship (Diggle *et al.* 2000; Smith and Lee 2011; Fingleton and Gallo 2010). A likely explanation could be that unobserved determinants of recreational visits exist, which are spatially related. Such determinants could be manifold and include everything from site, context and methodological study characteristics as well as their interactions. One important aspect could be related to the social-cultural context and path dependencies, which may result in specific recreational patterns in certain countries and regions. Also differing property rights could play an important role. Investigating human recreational behaviour across a study area as big as Europe is such a complex issue that all of these econometric problems may arise. There may hardly be any model that can incorporate all relevant drivers of recreational use, their interactions and non-linear effects.

Encountering such problems is common for modelling spatial data and therefore, we have to be cautious in interpreting p-values and parameter estimates. An option to gain further insights and confidence in model result interpretations is to try different spatial modelling approaches and compare their results. In particular, compare the confidence intervals of the parameter estimates (Bivand 2011; Brunsdon *et al.* 1996; Elhorst 2010; Gerkman 2011; O'Hara and Kotze 2010). However, there are only a very limited number of statistical R-packages that are fully developed to allow for advanced geostatistical approaches as described in literature.

For the meta-analytic value transfer function the question of independence is of even greater concern. As several valuation studies use different valuation methodologies to value recreation at the same site, it cannot be assumed that these observations are independent. Even though, SACs of the residuals are obvious, controlling for these correlations is rejected, because the distance between several observations from virtually the same site is zero. Statistical packages however, require a positive distance. This problem has so far been ignored in meta-analyses of recreational values (Brander *et al.* 2007; Brander *et al.* 2015; Ghermandi and Nunes 2013; Londoño and Johnston 2012; Sen *et al.* 2013; Shrestha *et al.* 2007). We aimed at controlling for spatial dependencies by introducing a random intercept for each study site, but it did not improve the model (as identified by the AIC and BIC values). Correlations across sites may be difficult to identify, because multiple observations per study site are only available for some of the observed sites. In addition, if multiple observations per site exist, the number of observations per site is typically small. Besides, we (and also the past meta-analyses cited above) find that the valuation method has a very strong impact on the value estimate, which also complicates the identification of correlations across observations from the same site. The fact that methodological variables are found to be the most important predictors explaining differences across the VV makes it also difficult to identify robust effects of spatial socio-economic and biophysical variables. We identify seven spatial variables with statistically significant effects on recreational VV. However, reviewing past meta-analyses on studies estimating the VV Schägner *et al.* (submitted) find that the identified significance levels and also the predictors' signs differ strongly across studies for spatial variables. Results of meta-analysis are therefore to be interpreted with caution.

7.5.2 Primary Data Representativeness

Another aspect of uncertainties within our predictions may be related to the representativeness of the sites of primary data collection. All primary data used for our models is collected from literature and publication bias may be an issue. Visitor monitoring and recreational valuation literature may not be representative for non-urban ecosystems across Europe (Thornton and Lee 2000). Ideally, study sites of our primary data would be randomly selected as done in ecology for estimating species distributions (Keirle 2002). However, to the knowledge of the author, primary data has hardly been collected for randomly selected sites. According to our experiences from collecting studies, it seems rather likely, that our primary data is biased towards sites of relatively high recreational supply, such as protected areas and sites that are particularly managed for recreational purpose. One exception represents a visitor monitoring study of the UK Forestry Commission, in which several sampled forest blocks were randomly selected. Nevertheless, the study focuses on forest only (TNS and FCS 2006a; TNS and FCS 2008; TNS and FCW 2005). However, there is little data available for any ordinary rural landscape, which is not drawn to receive a lot of recreational visits. This may result in an overestimation of total visitors and VV, because we are not certain about the extent to which our predictors capture all dimensions of recreational ecosystem services and to what extent relationships between dependent and explanatory variables are constant.

Similar to Brander *et al.* (2007), Brander *et al.* (2015) and Eagles (2014) we conclude that there are still insufficient high quality primary data available. Quality aspects of primary studies are often related to insufficient reporting, which hampers their use in secondary research. Often it is difficult to identify methodologies used for primary data collection. Quality and reporting standards for primary data collection have been repeatedly proposed in order to allow easier statistical assessments (Eigenbrod *et al.* 2010b; Johnston and Rosenberger 2010; Rosenberger and Phipps 2007; Schägner *et al.* submitted). We find that in addition to past proposed reporting standards, spatial information on study sites gain increasing importance. With the advancement of GIS technology, the spatial dimension of ecosystem services has received increasing attention (Maes *et al.* 2012a; Maes *et al.* 2013; Schägner *et al.* 2013). Therefore, exact reporting on the investigated case study area is fundamental, including coordinates, size and map illustration of the precise borders of the study area. If the only available information on the study site is, for example, a name of a forest in a certain country, it may not be possible to identify the location without additional information from the study authors. However, if the site can be identified, researchers can assess site information ex-post, display sites in GIS maps and relate them to other spatial data. Several primary valuation studies in our data base could not be included in our analysis due to insufficient spatial information on the study sites. Several other sites are only approximated, which may add substantial random noise to our statistical analysis.

7.5.3 Drivers of the overall Recreational Values

Our predictions of the overall recreational value correlate strongly positive with the predictions of the number of recreational visits. On the contrary, the correlations between the predicted VV and the value per ha is low and negative. This indicates that spatial variations of the overall recreational value per ha are predominantly determined by the visitor numbers and not by variations in the VV. This is also supported by the mean relative deviation of the predicted visitor numbers and VV, which determines the influence of the two variables' deviations on the deviation of their product, the value per ha. Also the primary data that is used to estimate the models show similar mean relative deviation for these two variables. Consequently, we conclude that accurate estimates of recreational visitor numbers are by far more important than accurate estimates of the VV for deriving accurate estimates of the overall recreational value per ha. This finding has implications for allocating future research priorities. Our results represent empirical support for similar conclusions by Bateman *et al.* (2006a).

7.5.4 Policy Implications

The two models estimated in this studies can be used for a number of policy applications: (1) They may contribute to the fulfilments of the EU Biodiversity Strategy 2020, which requires EU member states to "*map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020*" (EC 2011b) and the achievement of the Aichi Targets, which aim at *"reflecting the values of biodiversity in spatial planning and resource management exercises including through the mapping of biodiversity and related ecosystem services"* (CBD 2013). (2) The mapped recreational visitor numbers and the economic value of recreational ESS can act as a spatial value data base that can be used for value transfers. Policy makers can quickly derive a value estimate of the recreational services of any NP across Europe by consulting the map. (3) The maps may contribute to an efficient resource allocation by allowing policy makers to prioritize areas for conservation due to their high recreational value. In addition, recreational infrastructure may be designed to match the needs of the expected visitor numbers. Furthermore, it may be valuable to compare the models' predictions with real world observations on recreational use and values (if available) and, for example, investigate why some nature areas might remain below their recreational potential and how the recreational use and its value could be increased. However, it should be noted that the model allows only for assessments of NPs. Even if predictions can be made for a new hypothetical NP, no conclusion can be made on whether NPs designation results in an in- or decrease of recreational use and its values. (4) The model allows us to evaluate the effect of land use policies throughout Europe on recreational services and values. (5) Finally, the estimated recreational service values may contribute to set up a green GDP or a System of Environmental-Economic Accounting (SEEA) as proposed by the UN (2014), which may act as a counterpart to traditional GDP accounts and represent an additional measure for the impacts of human action on human well-being.

7.6 Conclusion

Within this study we model recreational visitor numbers and the VV across non-urban ecosystems throughout Europe using a large number of spatially explanatory variables. Our models fit the data reasonably well and we identify spatial drivers for recreational services and its value. Nevertheless, we also highlight uncertainties related to such statistical modelling approaches.

Since all our predictors are obtained from GIS raster layers, which cover the entirety of Europe, the model can be applied for ex-ante evaluation of alternative policy scenarios of change for existing NPs and on the creation of new NPs at a European scale. This information may be useful for several policy applications such as in planning the supply of recreational facilities such as parking and accommodation as well as for resource allocation within conservation prioritization by identifying locations of high recreational value and for setting up green accounting.

Investigating the effect of spatial variations in the number of visits and the VV for determining the overall recreational value per ha, we find that variations in the visitor number are of substantially higher importance. Consequently, we conclude that accurate estimates of recreational visitor numbers are by far more important than accurate estimates of the VV. This finding may have important implications for allocating future research priorities in recreational ecosystem service valuations.

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8 Synthesis and Outlook

The overall aim of this thesis is to contribute to the understanding of spatial variations in ESS supply and demand, and their contribution to human well-being as well as to advance the methodologies used to map ESS values by modelling the interactions between its biophysical and socio-economic drivers. In this final chapter we exemplify how the thesis at hand achieves this aim. The above presented articles are brought together and the inner overall context is pointed up and discussed. The thesis' contribution to the scientific community is exemplified as well as how the individual chapters contribute to solving the above raised research questions. In section 8.1, we give a summary of the methodologies applied in the different chapters and exemplify how they are interconnected and build on one another. The results are summarized. In the subsequent section, we highlight how the different chapters contribute to answering the research questions raised in section 1.2.1 and exemplify the contribution to the international field of research. Finally, in section 8.3, we present an outlook on future research prospects for spatial ESS assessments by pointing out the challenges, limitations and difficulties of current ESS value mapping exercises.

8.1 Methodologies and Results

Mapping ESS values is an interdisciplinary research topic and so is this thesis. The methodologies applied relate to different scientific fields of expertise, which are mainly: ecosystems service modelling, environmental economic valuation, recreational visitor monitoring, geography and GIS as well as geostatistical modelling.

The economic value of an ESS is determined by its supply and demand, which both vary across space (see section 1.1.1, 1.1.2 and chapter 2). As part of this thesis, recreation is chosen as an ESS to exemplify how supply and demand differ across space. The supply side of an ESS is determined by the biophysical characteristics of an ecosystem. It is modelled spatially explicit in chapters 4, 5 and 7. The demand side off a certain ESS is determined by human preferences. It is assessed in the chapters 4, 5, 6 and 7 by different valuation methodologies, which incorporate the spatial dimension of ESS demand to different degrees. Within the statistical modelling of recreational use and its economic value, geostatistical methods are applied to explore their spatial dimensions in depths (see chapters 4, 6 and 7). For the parameterisation of the model of recreational use presented in chapters 4, 6 and 7 primary data on recreational visitor monitoring and primary valuation studies are collected. Thereby, recreational visitor monitoring science is investigated in depth (see chapter 3). For the development of spatial predictors used in our models and for the illustration of the developed ESS maps knowledge of geography and GIS technology was applied. In- and output data in chapters 3, 4, 5, 6 and 7. are all stored, managed, analysed, manipulated and visualised using GIS software. The methodologies and results of the different chapters are summarized in detail in the following.

8.1.1 Chapter 2: Mapping Ecosystem Services' Values: Current Practice and Future Prospects

In chapter 2 of this thesis we review all studies mapping ESS values that we could find by searching the relevant online literature databases. Spatial ESS assessments appear to be a very timely research topic characterised by an exponential growth of the number of publications in recent years. This is related to the fact that ESS value mapping is considered to provide specific advantages for policy analysis as compared to traditional ESS valuation studies such as to support efficient resource allocation across space, to conduct green accounting at different spatial scales and to evaluate trade-offs and any synergies of land-use policies. Considering the biophysical dimension of ESS supply and the economic dimension of ESS demand, we analyse and classify the applied methodologies. Therefore we develop a classification matrix, in which we locate every study depending on the methodology used to assess spatial variations in ESS supply and demand. Our review reveals, that many studies still rely on approaches which account only roughly for spatial variations in ESS supply and demand by using only one spatial indicator, mainly land-use data. However, a trend towards more sophisticated approaches such as geostatistical modelling in combination with value functions can be observed. We identify strengths and weaknesses of the different methodologies and give guidance for future studies on ESS value mapping. We propose to model ESS service supply and demand separately by applying highly spatial models that are calibrated and validated based on regression analysis of primary data. Nevertheless, we also acknowledge the difficulties in conducting such an approach due to extensive primary data requirements as well as GIS and geostatistical expertise. The main shortcomings we identify in recent ESS value mapping studies relate to a limited integration of the scientific disciplines involved. Still, many studies take rather a monodisciplinary perspective by either focusing only on the quantification of ESS supply or on its economic valuation. Furthermore, the policy orientation of many studies is still poor. Only about 35% of the reviewed studies evaluate some kind of scenario that may offer guidance for policy makers. Another important aspect is the assessment of uncertainties involved with developed ESS value maps. If policy makers want to base their decisions on such maps, they need to know about the accuracy and potential error margins of the value estimates. Even though findings indicate that uncertainties can be high, only a minor share of the reviewed studies give quantitative information on the error margins of their results. The given recommendations of our literature review are exemplified by the case studies in the chapters 3, 4, 5, 6 and 7 focusing on the spatial assessment of recreational ESS.

8.1.2 Chapter 3: Recreation Across European Nature Areas: A Review of Monitoring Activities and a Geo-database of Total Annual Visitor Estimates

In this chapter, we present a geo database of recreational visitor numbers to non-urban ecosystems across Europe, review visitor monitoring literature, propose reporting standards for recreational visitor counting and introduce a new online data sharing platform. The lack of real world observations on ESS supply and demand is identified in chapter 2 as one major obstacle for the development of spatially explicit ESS value maps. It is required for parameterization and validation of ESS models. In total we collected 1,267 observations on total annual visitor estimates at 518 separate case study areas all across Europe. Visitor monitoring study areas differ widely and are unevenly distributed. Most study areas are located in North Europe. The study areas' sizes range from only 1 to almost 1 million ha and the visitation rate range from three annual visits per km² up to 15.7 million in small visitor hot spot areas. Each case study area in our database is presented as a spatial layer in vector format, which allows it to locate the area within a map and derive further information of the area via GIS technology. While collecting data from visitor monitoring studies, the relevant literature is reviewed and analysed. Spatial distributions of such activities across Europe as well as temporal and methodological trends are identified. In recent years visitor monitoring has moved on from an instrument mainly used for recreational site management to a vibrant science focusing on a variety of aspects such as visitor experiences, needs, attitudes and perceptions as well as activities, movement patterns, crowding effects, conflicts and user groups' effects on wildlife. Nevertheless, to our surprise, visitor monitoring studies are typically characterized by relatively rudimentary reporting of the monitoring methodology. However, the methodologies used in order to estimate total recreational visitors are manifold and may introduce a systematic bias to the study results and may thus, affect the accuracy of the estimates. Quality and reporting standards for primary data collection have been repeatedly proposed in other disciplines in order to ease statistical assessments of such influences. Based on our findings, we propose reporting standards for visitor monitoring studies, which may ease the use of study results for secondary research. We invite researchers to share their data via the ESP Visualisation tool (http://esp-mapping.net/Home/) and submit their visitor counting data including comprehensive methodological reporting via web interface at http://rris.biopama.org/visitor-reporting.

8.1.3 Chapter 4: Mapping Recreational Visits and Values of European National Parks by Combining Statistical Modelling and Unit Value Transfer

In chapter 4, the database presented in chapter 3 is used to estimate a spatially explicit model on recreational visitor numbers of European national parks by using regression analysis. The dependent variable is the natural log of the total annual number of visits per ha and the predictors used to describe the biophysical and socio-economic characteristics of the visitor monitoring sites and its context (see section 4.2.2 and 6.2) are all obtained from GIS raster layers, which cover the entirety of Europe. Therefore, the model can be used for predictions all across Europe. The resulting map may be applied for ex-ante evaluation of alternative policy scenarios such as the land-use change in existing national parks and on the creation of new national parks. The raster layers of the predictors were either taken from available GIS data sets or we computed them by reprocessing or combining existing data sets using ArcGIS and Python programming. A number of different regression techniques are explored in order to estimate a robust model including general and generalised linear models, additive models, mixed models and SAC structures. In our starting model 14 of the 19 predictors show statistically significant effects at the 0.1 level. The multiple R^2 of 0.68 indicates a relatively high explained variance. However, the residual plots of the model show spatial patterns, which we control for by introducing a spherical spatial correlation structure of the residuals. The beta coefficients of our final model (after variable selections) indicates that the availability of trails shows a positive and strongest effect on the number of recreational visits followed by the population living in the proximity and the availability of water bodies within the national park. Strong negative effects are found for the availability of substitute national park area in the surrounding but also for the share of forest land cover. Other significant negative effects are found for the availability of wetland area and the national park size. Significant positive effects are also found for the number of days per year with a temperature above five degrees, the availability of small roads, the age of the national park and the land cover diversity. We use our final model — a log linear model with spherical SAC structure — to make predictions of the number of visitors to all European national parks combined with residual kriging. We use the resulting maps to evaluate the effects of two national park designation scenarios. Finally, the predicted number of visits is combined with a constant unit value transfer of the value per recreational visit in order to estimate the total recreational value per ha and to highlight the significance of the economic value of recreational ESS across European national parks. Using our model to predict the number of visits to 449 national parks across Europe, we estimate a total of more than 2 billion annual visits with an total recreational value of € 14.5 billion annually. For a proposed national park (Teutoburger Forest) in the west of Germany we predict about 283 annual visits per ha on average. However, the peripheral locations close to the cities of Detmold and Paderborn may receive up to 24,000 visits per ha and year, whereas the hardly accessible centre of the area may receive less than one visit/ha/year. In total we predict about 5.8 million annual visits for the entire area, which accounts for an annual monetary value of € 41.5 million. Results are illustrated by maps that show how visitor numbers varies across space. The mapped predictions of the potential annual number of recreational visits to fictive newly designated national parks anywhere in Europe indicate a wide variety of the visitation rates. Low numbers of visits per ha with minimum values close to zero are predicted for remote areas, which are characterized by low population and little access infrastructure, high forest cover and little land cover diversity, mainly in Scandinavia. On the contrary, high visitation rates are predicted for densely populated areas as for example in the Netherlands, Belgium, the southeast of the UK and the west of Germany.

8.1.4 Chapter 5: GIS-Based Mapping of Ecosystem Services: The Case of Coral Reefs

The methodology of chapter 4 is developed further in chapters 5, 6 and 7. Spatially explicit modelling of recreational visitors is combined with spatially explicit meta-analytic value transfer. Consequently, recreational values can be predicted across space by accounting for spatial variations in the number of visits as well as spatial variations in the value per visit. Both the visitor models and the meta-analytic value transfer functions are parameterised based on regression analysis of primary data. Thereby, the methodological recommendations developed in chapter 2 are put into practice. The approach is used to evaluate the effect of expected future coral reef loss on recreational values in Southeast Asia.

8.1.5 Chapter 6: Exploring the Spatial Dimension of Recreational Ecosystem Service Values: A Combination of Meta-Analytic Value Transfer and GIS

In Chapter 6, we conduct a meta-analysis of recreational values per visit throughout Europe's non-urban ecosystems. We focus on a larger and refined set of spatial biophysical or socio-economic predictors, which are all taken from continuous GIS raster layers in fine resolutions (1ha-100ha) in order to gain additional knowledge of the spatial dimension of recreational ESS values. Similar to chapter 4, a number of different regression analyses are conducted to estimate a robust model. The results of the meta-analysis are critically evaluated against the results of all available other meta-analysis studies that we review in the chapter. Uncertainties involved with the geostatistical analysis are discussed in depth. The starting model of our meta-analysis shows eight predictor variables that are significant at the 0.1 level and with an adjusted R^2 of 0.44. However, the residuals are clustered according to the authors of the single valuation studies, which we control for by introducing a random intercept. The final model after variable selections has eight significant predictors. The strongest effect is shown by the valuation methodology of the primary valuation study. The travel cost method results in significantly higher estimates of the value per visit. Also several spatial predictors that describe the study area or its surrounding show significant effects. Population pressure does have a strong and negative effect on the value estimate. Negative effects are also found for the average number of rainy days per year and the share of forest and grassland. We find positive effects for the study area size and for the mean slope value of the area. A small positive effect is also found for the unemployment rate in the area of the study site. Nevertheless, the statistical uncertainties involved in meta-analyses on value per visit estimates turn out to be substantial. The meta-analyses we review show contradicting results for several spatial variables such as study area size or population density or forest land cover. The final model — a log linear mixed model with authorship of the primary valuation studies as a random intercept — is used to map the value of a recreational visit across non-urban ecosystems in Europe. To derive the total recreational value per hectare, the predicted values per visit are combined with the number of recreational visits per hectare predicted in chapter 4. Finally, the results of the meta-analytic value transfer are compared to results of a unit value transfer in order to illustrate the effects of different ESS value mapping methodologies, which are identified in chapter 2. We investigate to which extent the total recreational value of an ecosystem is determined by variations in the number of recreational visits and by variations in the value per visit. Our results are again illustrated by maps computed using the software R and ArcGIS. Using the estimated meta-analytic value transfer function for the prediction of the entirety of rural Europe results in a recreational value per visit of € 1 to € 45 with a median of € 7.27. High values are found for remote mountainous areas in the North of Scandinavia and in the dry Southern part of Europe, which show little forest and grassland cover and high slope values. Comparing the spatial variations of the predicted values per visit and the number of visits per ha predicted in chapter 4, indicates that the total recreational value per ha is mainly determined by visitor numbers and not by the value per visit. The standard deviation of the predicted values per visit is by far lower than the standard deviations of the number of visits per ha, which is about 360 times higher. When comparing two maps showing the recreational values per ha across Europe, which are based on an estimate of the value per visit by either a constant unit value transfer or a meta-analytic value transfer, differences can hardly be spotted. This also indicates that the total recreational value per ha is highly dominated by the number of visits.

8.1.6 Chapter 7: Mapping the Recreational Value of Non-Urban Ecosystems across Europe: Combining Meta-Analysis and GIS

In this final article, we do a similar regression analysis as presented in chapter 4, but apply it to the entire visitor monitoring data base presented in chapter 3. Thereby, we do not only focus on national parks, but on all non-urban protected and non-protected ecosystems across Europe. Thereby the estimated model allows us to predict recreational visits all across Europe and not be limited to national parks. The general linear model using again a log-transformed dependent variable shows 14 predictor variables significant at the 0.1 level after variable selection. Variable coefficients have the same signs as presented in chapter 4, but some different predictors show a significant effect and the parameters differ. The model shows again a clustering of positive and negative residuals, which we control for by using an exponential spatial correlation structure of the residuals. The model then shows 12 predictor variables significant at the 0.1 level after variable selection and spatially correlated residuals up to a distance of 364 km. We use the model to predict recreational visitor numbers across the entirety of non-urban Europe. In total we estimate about 11 billion recreational visits to Europe's non-urban ecosystems. We then combine our predicted visitors with spatially explicit predictions of values per visit from the meta-analytic value transfer function presented in chapter 6. With that we derive a spatially explicit estimate for the recreational value of non-urban ecosystems. In total we estimate an annual recreational value of about € 57 billion. Again we conduct an analysis of the importance of the predicted number of visits and the predicted values per visit for determining the overall recreational value per ha. Visual interpretation of the different European maps — the predicted visits, the values per visit and the value per ha — reveal that the overall value per ha is positively correlated with number of visits, but not with the value per visit. The correlation between the value per ha and the number of visits is 0.8 but -0.1 between the value per ha and the value per visit. The mean relative deviation — a measure to determine the impact of the deviations of different variables on the deviation of the product of the variables — is 112% for the visitor numbers but only 39% for the value per visit. Therefore, we conclude that accurate assessments of the visitor numbers are by far more important for deriving accurate estimates of the recreational value of an area than accurate assessments of the value per visit.

8.2 Discussion of Results and Research Questions

The thesis at hand contributes significantly to the understanding and methodological development of spatial assessments of ESS and their values. It is identified how the spatial interactions of natural, human, social, and man-made capital as constituents of ESS supply and demand can be captured and modelled for ESS value mapping. A best-practice approach is derived, which is exemplified for recreational ESS. Thereby, the presented research contributes substantially to the scientific field of spatial ESS assessments and to answer the research questions raised in section 1.2.1.

Central for answering the methodological and policy related research questions is chapter 2. The broad literature review on studies mapping ESS values addresses the importance and advantages of assessing the spatial dimension of ESS and their values for designing efficient environmental policy instruments. Even though several reviews on the spatial assessment and mapping of ESS have emerged in recent years (Crossman *et al.* 2013; Maes *et al.* 2012; Martínez-Harms and Balvanera 2012; Egoh *et al.* 2012; Egoh *et al.* 2008; Ayanu *et al.* 2012; Seppelt *et al.* 2011; Liquete *et al.* 2013), none of these reviews take a specific environmental economic perspective, but are rather natural science orientated. To our knowledge, chapter 2 presents the only review focusing specifically on the mapping of monetary ESS values. It received a great resonance in the research community. With 123 Google Scholar citations, it is ranked among the most cited article on the Ecosystem Services Journal web page (Elsevier 2015).

Research question 1: What makes the space-related perspective on ESS and on their values to be of particular interest and what advantages arise from spatially explicit ESS assessment as compared to traditional ESS valuation?

The papers we review in chapter 2 mention specific policies that are related to the mapping of ESS values. Besides communicative and illustrative purposes, these policy applications are an opportunity to (1) support efficient resource allocation, (2) evaluate land use policies at different spatial scales, (3) design spatially explicit payment schemes for ESS and (4) to set up a green accounting at different spatial scales. The space-related interest arises from these particular advantages of ESS value mapping. Policy makers do not only need to know what *is*, but also where it is and why it is there.

Research question 2: What methods are applied or may be applied for spatial assessments of ESS values and what is the state of the art in ESS value mapping?

In chapter 2 we find that a broad variety of methods is applied for mapping ESS values, which can be classified into two broad categories: (1) methods that are used to assess spatial variations in ESS supply and (2) methods that are used to assess spatial variations of the demand for ESS. Methods belonging to the former group use spatial biophysical variables to develop more or less complex models to quantify the ESS supply. Typically, these methods originate from natural sciences such as ecology, biology or hydrology. In contrast, the latter group of methods usually originates from environmental economics and are closely linked to environmental value transfer methodologies. In chapter 2 we present a methodology matrix and classify all reviewed studies with respect to the methodologies applied to assess spatial variations in either ESS supply (the biophysical domain) or ESS demand (the economic domain). Thereby, we give quantitative findings on the developments and state of the art in ESS value mapping. Researchers can consult our review to identify which studies apply a certain combination of methodologies to map values of a specific ESS.

Research question 3: What data is required to best map ESS and their values? What data gaps exist and how should available data be presented and organized?

Within our literature review in chapter 2 and also within the case studies presented in the consecutive chapters we identify insufficient data availability as one main barrier in developing highly accurate ESS value maps. ESS values emanate from the spatial interaction of natural, human, social and built capital. To model these interactions quantitative, comprehensive and high resolution spatial input data for all kinds of capital is required. Furthermore, to calibrate and validate models, real observations on ESS supply and its value are required. In chapter 3 we present a database on real world observations of recreational visitor numbers across Europe's non-urban ecosystems. With this information we fill in an important gap in data availability required for the parameterisation and validation of spatially explicit models on recreational use. The models developed in chapter 4, 5, 6 and 7 are parameterized by conducting regression analysis of this data. The data base is shared with the international research community via the ESP Visualisation tool (http://esp-mapping.net/Home/). Nevertheless, we also emphasize the difficulties involved with collecting primary data. The review on visitor monitoring studies in chapter 3 highlights the development in this relatively new research domain and gives guidance for future primary data collection on recreational use. In particular, the proposed reporting standards and the given recommendations for data sharing across researchers and disciplines may ease the use of such data for future secondary research and thereby contribute to spatial ESS assessments. The online map browser at http://esp-mapping.net/Home/ and the web interface http://rris.biopama.org/visitor-reporting for sharing data on recreational visitor counting and valuation studies may support the sharing of available data, the comprehensive methodological reporting for data collection and improve the quality of future data collection. We invite researchers to submit and share their data via the ESP Visualisation tool (http://esp-mapping.net/Home/) and our data submission tool http://rris.biopama.org/visitor-reporting.

Research question 4: What are the advantages and disadvantages of the different methods for mapping ESS values?

Within our review in chapter 2 we investigate the different methodologies used to map ESS values and evaluate them with respect to their accuracy and precision, the required knowledge, and the time and resource intensity. We score each combination of methodologies applied to assess spatial variations in ESS supply and demand with respect to the different evaluation criteria. Thereby, we contribute to the understanding of how to constitute the functional linkages of the production chain from habitat characteristics to ESS values, which are illustrated in the cascade model (see section 1.1.1), the concept of the total economic value (see section 1.1.2) and the concept of mapping ESS values (see section 1.1.4) in the introduction of this thesis.

Research question 5: Which method appears to be the best for mapping values of a specific ESS under consideration of specific circumstances and study purposes? How to define a best-practice approach?

Based on the findings of our literature review, we propose a best-practice approach for ESS value mapping that applies statistical modelling of both ESS supply and demand in distinct models. Thereby, spatial variations in both dimensions, ESS supply and demand, can be assessed separately and the capacities of all involved disciplines, natural sciences and socio-economic sciences can be accounted for. Nevertheless, we also acknowledge the difficulties of our proposed best-practice approach, which are mainly related to spatial data availability, statistical modelling capacity and its interdisciplinary nature. As a result, we also identify substantial future research prospects in ESS value mapping (see section 8.3) and by highlighting strengths and weaknesses of different methodologies (see chapter 2) we give guidance for future ESS value mapping exercises. For the specific case of mapping recreational ESS values, we show in chapter 4, 6 and 7 that accurate assessments of the visitor numbers are far more important than accurate assessments of the value per visit. This finding may be important for setting future priorities in this field.

Research question 6: How to exemplify a best-practice approach for mapping ESS values through a case study?

The methodological findings of chapter 2 are exemplified by case studies in chapters 4, 5, 6 and 7 focusing on the mapping of recreational ESS and its value. In these case studies, models to predict either recreational visitor numbers or the value per visit are developed by statistical regression analysis of primary data using comprehensive continuous raster layers as predictor variables. Thereby, we supply spatially explicit information on ESS supply and demand. By multiplicative combination of this information, we derive predictions of the recreational value per ha. To our knowledge chapter 5 presents the first study mapping recreational visitor numbers and the value per visit on an international scale. Chapter 4 and 7 present the first studies mapping recreational visitor numbers on an international scale that covers all land cover types. The same applies for mapping the value per recreational visit in chapter 6.

Research question 7: How to map recreational services and its values and what are their main drivers?

In chapters 4, 5, 6 and 7 we model recreational visits per hectare and year as well as the value per recreational visit spatially explicit by applying the best-practice approach proposed in chapter 2. Through our regression analysis we identify the most important explanatory variables. As compared to past studies (e.g. Neuvonen *et al.* 2010; Loomis 2004 and Loomis *et al.* 1999) we use a more international and comprehensive recreation use data set. By combining and reprocessing existing GIS data sources we develop new explanatory variables that have not yet been used in this form to model recreation visitor numbers in general and/or on a continental scale. We omit commonly used dummy variables (such region, land-cover type etc.), but use continuous explanatory variables in order to identify the underlying effects of biophysical and socio-economic characteristics. We introduce for example a distance decay effect for an European wide accessibility-indicator and a substitute-indicator, climatic data and user-generated contents from social media, such as Open Street Map data.

A special focus is also given to the geostatistical modelling of recreational use and the challenges in addressing observed and unobserved spatial dependencies in primary data structure. By accounting for SAC across the residuals we investigate a common geostatistical problem, which has so far been ignored in statistical analysis of recreational use data, even though it is of importance to derive robust model results (Bivand *et al.* 2013). Our approach may stimulate the use of more advanced geostatistical modelling techniques in the field of recreational use modelling.

Nevertheless, we also highlight uncertainties involved with geostatistical modelling approaches. Typically, real world spatial data differ from the ideal statistical text book world. Even minor violations of the assumptions of regression analysis may inflate standard errors and p-values. Thereby the chance of type I and type II errors is increased. To our knowledge, we conduct the first review of meta-analyses on recreational ESS values and compare our results with past meta-analysis (see chapter 6). Parameter estimates and significance levels show contradicting results in different meta-analyses for several predictors, which reveals the uncertainties related to such large scale modelling.

Research question 8: How to integrate ESS maps into environmental policy and how to evaluate policy scenarios with ESS maps?

Within our literature review presented in chapter 2 we identify the advantages of ESS value mapping for specific policy applications. However, we also identify a lack of policy orientation in many studies by not applying the presented methods and results for policy analysis. In the case studies presented in chapters 4, 5, 6 and 7 we exemplify the applications of ESS value mapping for such policy analysis. In chapter 4 and 7 we predict recreational visitor numbers and the recreational value of a newly designated national park. In addition, we estimate expected visitor numbers and recreational values of a hypothetical new national park anywhere in Europe. Thereby, we contribute to the efficient allocation of resources by supporting the prioritisation of conservation and by supporting the allocation of recreational facilities that match recreational demand. Land-use change is evaluated in chapter 5, by evaluating the forgone recreational value due to shrinking coral reef cover as a result of climate change. The chapters 4, 5, 6 and 7 all contribute to the development of a green GDP by estimating the total recreational value of all national parks in each European country (chapter 4 and 6), of coral reefs in Southeast Asia (chapter 5) and of all non-urban ecosystems in Europe (chapter 7). In addition, the studies may contribute to the fulfilment of the EU Biodiversity Strategy 2020, which requires the EU member states to map and value their ESS spatially explicit and the achievement of the Aichi Targets, which aim at "*reflecting the values of biodiversity in spatial planning and resource* *management exercises including through the mapping of biodiversity and related ecosystem services*" (CBD 2013). Nevertheless, the integration of spatial ESS assessments into policies remains a timely and widely discussed research questions (see also chapter 8.3.7 Policy Integration).

Research question 9: How can spatial ESS maps contribute to biodiversity protection?

This thesis was financed by the European Commission and the work was conducted in the light of the EU Biodiversity Strategy 2020 as part of the PRESS project (PEER [Partnership for European Environmental Research] Research on EcoSystem Services) and the MAES working group (Mapping and Assessment of Ecosystems and their Services). The Biodiversity Strategy "*reflects the commitments taken by the EU in 2010, within the international Convention on Biological Diversity"* and "*aims to halt the loss of biodiversity and ecosystem services in the EU and help stop global biodiversity loss by 2020.*" Among other aspects, Action 5 of the Biodiversity Strategy requires member states to "*map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020*" (EC 2011b). The first results of the PRESS project were shared with the Environment Directorate-General - Environment (EU DG ENV) in preparation of the Biodiversity Strategy and the final results are considered supporting its implementation (Maes *et al.* 2011a; Maes *et al.* 2012b). In the follow up, the MAES working group was established under the Common Implementation Framework — the governance structure to underpin the effective delivery of the strategy — to assist member states with the fulfilment of the requirements (EC 2011a; Maes *et al.* 2013). Within this context the thesis at hand may contribute to the methodological development of spatial ESS assessments and thereby to development and implementation of the Biodiversity Strategy. To maintain and restore ecosystems, spatial information of the status-quo as well as trends and scenarios are critical. Target 2 of Biodiversity Strategy requires that "*ecosystems and their services are maintained and enhanced by including green infrastructure in spatial planning and restoring at least 15% of degraded ecosystems*" (EC 2011b). Nevertheless, despite all these efforts biodiversity and natural ecosystems within the EU and around the world are still under threat and the targets of the Biodiversity Strategy are yet far from being achieved (EC 2015; EEA 2015). Linkages between ESS and biodiversity are mixed and not always easy to prove and the challenge of halting biodiversity loss is still one major research prospect that will be discussed more in detail in the following section.

Research question 10: What are future research prospects in mapping of ESS and their values?

We have confidence in delivering a substantial contribution to the methodological development of spatial ESS value assessments and in particular to the mapping of recreational ESS value by the research presented in this thesis. Nevertheless, there are several issues of interest for future research within the domain of ESS value mapping and we believe that this field will remain a dynamic, active and vibrant research topic for a long time. The main challenge is to make ESS value maps more accurate, precise and comprehensive and to integrate them into the design of environmental policy instruments and into decision-making processes. If such maps are used to consult policy makers, mapping errors may lead to suboptimal policy decisions. So far little evidence exists on the accuracy of different methods used for ESS value mapping, but error margins are generally large (see chapter 2) and this applies as well for the case studies presented in this thesis. Given the scope, scale and complexity of our analysis, our modelling exercises perform reasonably well as compared to past research. But still, the ESS value maps presented in this thesis may only allow for the identification of broad trends at landscape scale. For small scale assessments, estimates may still be relatively inaccurate and show considerable mapping errors. They may give an indication of ESS supply and values, but may not be used for policy recommendations on its own. The difficulties in developing more accurate and precise

ESS value maps are manifold and finally, the role of biodiversity in ESS provision and how the ESS concept can be used best to protect biodiversity remains yet insufficiently understood. These aspects bear a broad set of future research prospects that are discussed more in detail in the following.

8.3 Limitations and Future Research Prospects

8.3.1 Integration of Disciplines

Accounting for both dimensions of ESS values — its supply and its demand — as well as accounting for their spatial dimension requires a deep integration of several disciplines (Bockstael 1996). Still, many studies take rather mono-disciplinary approaches and only a few studies combine the strengths of multiple disciplines. However, suitable mapping of ESS values requires expertise in the domain of economic valuation, ecology, geography and geostatistical modelling. Nevertheless, studies dominated by an ecological perspective tend to use sophisticated ESS models, but apply rudimentary valuations. Several studies use quickly derived value estimates, such as expenditure data, replacement costs and market prices for different ESS, but without any reference to the meaning and accuracy of these different value measures. On the other hand, studies dominated by an economic perspective tend to focus on the valuation process, but give little attention to ESS supply modelling. In recreational ESS valuation studies for example, often focus is put on the economic valuation process and little attention is given to accurate visitor estimates (Bateman *et al.* 2006a). Attention is also demanded by the applied ESS classifications and definitions that are often not transferable across disciplines and may either cause omitting or double counting of certain ESS in final environmental economic value estimates. Within this thesis we aim to assess recreational services spatially explicit by considering the point of view of all disciplines involved, but at the same time we cannot claim to have an expert of each discipline in the department's team in which the thesis at hand was elaborated.

8.3.2 Data Availability

ESS values originate from the spatial interaction of natural, human, social and built capital. The principal challenge in ESS value mapping is to capture these interactions by a functional form, which quantifies the cascade from certain habitats over ecosystem functions and ESS onto ESS values as exemplified in section 1.1.1 and 1.1.2. To model these interactions spatially explicit, appropriate explanatory variables are required that represent the determinant of ESS supply and demand.[50] Therefore, accurate, comprehensive and high spatial and temporal resolution data of all kinds of capital is needed. In addition, real world observation of ESS quantities and values are necessary for model calibration and parameterisation. However, generating such data is a costly and time consuming procedure and its limited availability is a major obstacle to the development of high quality ESS maps, including also the case studies presented in chapters 4, 5, 6 and 7. Nature recreation may strongly be determined by for example recreational facilities such as parking lots, benches and on travel time based accessibility (Cullinan *et al.* 2008b; Sen *et al.* 2013). However, such data sets do not exist on a European scale. The distance based accessibility indicator and the trail density indicator we used to approximate such predictors (see chapters 4, 6 and 7) took months of computer processing time. The primary data that we use for model calibration was collected by a timely literature review and is still limited in scope and quality (see also chapter 3). The availability of more adequate predictor variable

[50] Explanatory variables in this domain are also referred to as indicators or ecosystem service indicators in literature and the scientific debate.

layers as well as a higher quantity and quality of measurement of recreational visitor numbers and values would considerably improve our modelling and mapping approach. With progressing remote sensing and computer processing technologies as well as continuous ESS sampling, the pool of required data can be expected to grow in quantity, quality and spatial resolution. In recreational science for example, new remote controlled visitor monitoring technologies, GPS tracking and social media may enable the collection of recreational use data in higher quality and quantity at lower costs. Monetary valuation studies on recreational ESS are conducted continuously all over the world and methodologies are developed further (Fezzi *et al.* 2014; Rolfe and Windle 2015; Sánchez *et al.* 2015; Windle and Rolfe 2013). New data sources may be explored, such as social media and citizens' science, which have only been discovered recently for ESS value mapping. For example the Open Street Map project, in which registered users can contribute to mapping multiple map features, is continuously growing and has become much more than "*just*" a road map. It may offer valuable indicators for spatial assessments of various ESS. Satellite Earth observation offers great opportunities in supplying required spatial data sets from recent (e.g., Landsat 8, surface water dynamics and Sentinel-2) and planned (e.g., EnMAP, GEDI, Tandem-L, and FLEX) projects. Cloud computing platforms such as Google Earth Engine offer new opportunities for modelling and combining large scale spatial data sets (Cord et al 2017; Pekel et al. 2016). Efforts are required to aggregate, harmonise and share available data. For primary data collection, quality and reporting standards are of great importance and have been repeatedly proposed in order to ease secondary research (see chapter 3). Recently, interactive online meta-databases have been established to enhance data sharing, such as the "*ESP Visualisation tool*" (http://www.es-partnership.org/esp), ‘‘*Earth Economics*’’ (http://www.eartheconomics.org/) and the "*DOPA (Digital Observatory for Protected Areas) explorer*" (http://ehabitat-wps.jrc.ec.europa.eu/dopa_explorer/).

8.3.3 Geostatistical Modelling

The statistical methods applied for estimating models to map ESS and their values may be improved in several studies. Often the specific nature of spatial data is not captured because economists and ecologists may lack advanced geostatistical knowledge. However, the modelling of spatial data demands for particular investigations of spatial correlations and interactions across input- and output data. Simple linear regression analysis may fall short in identifying such interactions and may therefore result in misleading conclusions (Bivand *et al.* 2013). In chapter 4 and 7 we account for spatial patterns in the residuals of our model by comparing a number of spatial random effects and residual correlation structures. However, typically, for the analysis of real world spatial data, uncertainties about the model results' robustness are high (Bivand *et al.* 2013; see chapters 4, 6 and 7). To improve confidence in spatial modelling results, alternative modelling approaches could be compared and incorporate for example SACs either within the fixed or the random part of the model, such as a spatial lag model, a Durbin model, spatial autoregressive models with autoregressive disturbances, geographically weighted regressions or even by using Bayesian approaches. However, it is still under discussion, which is the best model to use for a specific purpose and required statistical software packages are still under development (Bivand 2011; Brunsdon *et al.* 1996; Elhorst 2010; Gerkman 2011).

8.3.4 Non-linearity

Another issue of importance when defining the functional form of interrelation between habitat characteristics and ESS values are non-linear relationships. Typically, the spatial interactions of natural, human, social, and man-made capital that determine the value of ESS are far more complex than can be captured by linear models. For example, the rates of substitution across different ESS and man-made capital, which are implied by monetary valuation, may change drastically as a result of non-marginal changes in ESS supply and demand. When ecosystems hit tipping points, beyond which ecosystems shift into a less desirable state is a critical question in ESS mapping and valuation. The often non-linear and multi-scale relations between ESS values and all sorts of capital are not yet sufficiently understood. Their incorporation into environmental valuation and policy scenario analysis is of critical concern for ensuring sustainable policy recommendations (de Groot *et al.* 2010; Nelson and Daily 2010). In chapters 4, 6 and 7 we at least aim at accounting for non-constant rates of substitutability by the use of national park and land cover substitute indicators. However, substitution relationships between different recreation sites may be far more complex. Substitutability may not only exist between national parks and land cover types but also among all types of ecosystems and man-made recreational sites such as museums or amusement parks. The entire spectrum of human preferences and recreational behaviour is highly diverse and complex and may be characterised by multiple local maxima, substitute relationships, interactions, feedbacks and non-stationarity. Incorporating such effects in ESS models and value functions presents a great challenge for future research.

8.3.5 Comprehensiveness

To deliver meaningful policy recommendations, accurate assessments of all relevant ESS values are of great importance. Only comprehensive ESS value maps allow it to identify trade-offs and synergies between all different ESS values and deliver policy guidance to maximise total ecosystem service values (see also section 1.1.2). However, due to the complexity of ESS value mapping, there is a trade-off between comprehensive inclusion of ESS and the accuracy of the analysis. To this end, the case studies presented in the thesis at hand focus solely on the mapping of recreational ESS values. The results may deliver relevant information for policy makers, but may hardly be used as sole basis for decision making. It is a challenge to combine multiple ESS models, but also to link them by creating dynamic meta-ESS models that include feedbacks and linkages between different ESS. Finally, it needs to be considered how changes in ESS values trigger human responses that in turn affect pressures on ecosystems.

8.3.6 Non-monetary Values

An important, but often neglected aspect in ESS value mapping are values that cannot be expressed in monetary terms. By focusing on monetary ESS values only, the case studies presented in this thesis may leave out part of the picture. Monetary valuation implies a rate of substitutions across different goods and services, which leave the individual with the same level of utility. However, not all ESS and biodiversity values can be substituted for by means of monetary compensation, such as religious, moral and intrinsic ecosystem values. Participatory ESS value mapping approaches try to incorporate such effects by assessing the needs, perceptions and attitudes of different stakeholders with respect to ESS values. It may be worth investigating how values differ across space due to differences in institutions and attitudes and how they may be instrumentalised to influence ESS values and human interaction with nature (Kotchen and Reiling 2000; O'Neill and Spash 2000; Pritchard Jr. *et al.* 2000). The strengths and weaknesses of both, monetary and non-monetary ESS value mapping, as well as methods to combine them for developing efficient policies to halt ecosystem degradation and biodiversity loss remains to be explored.

8.3.7 Policy Integration

The policy orientation of many studies that map ESS values is still poor. Only the minority of the studies we review in chapter 2 use their analysis to give guidance for a specific policy question, such as the evaluation of alternative policy scenarios. In chapters 4, 5, 6 and 7, we aim to do so by evaluating predicted land use changes and the effect of national park designation. Nevertheless, besides general statements and illustrative purpose, the application of ESS value maps for the design of spatially explicit real world policy instruments and decision making processes is still limited (Burkhard *et al.* 2013; EC 2014; Fisher *et al.* 2008). It is still to be addressed how such maps can be integrated best in political decision making processes and how results must be presented to satisfy the needs of policy makers. To provide useful information for decision makers, ESS mapping needs to cover the whole range of ESS, their interactions and societal valuations. However, so far high-quality ESS studies (including the studies presented in this thesis) choose only sectoral approaches and thus, they focus either on a specific ESS or a specific linkage within the ecosystem service cascade (Müller et al. 2010).

The question of policy integration is discussed intensively in the MAES working group (Mapping and Assessment of Ecosystems and their Services) and at the regular MAES workshops, which are meant to assist member states in fulfilling the EU Biodiversity Strategy 2020 requirements (MAES WS 2017). Still ongoing Horizon 2020 research projects were established with budgets of several € millions in order to address these questions such as OpenNESS (Operationalisation of natural capital and ecosystem services) and ESMERALDA (Enhancing ecoSysteM sERvices mApping for poLicy and Decision mAking). A new project on “Integrated Natural Capital Accounting” (INCA) has just been launched. The development of guidance and practical tools for integrating ecosystem services into planning and decision-making processes is one priority within the recent “Action Plan for nature, people and the economy” of the European Commission (EC 2017).

8.3.8 Biodiversity

Finally, the role of biodiversity for ESS provision and how the concept should be interpreted and used best for its protection is so far insufficiently understood and the need for extensive research is widely recognised such as in the upcoming IPBES report, which is currently under review (IPBES, forthcoming). The total biodiversity value is indubitably infinite, because humans depend on its existences, but how marginal changes in biodiversity are to be valued remains controversial. Evidence on correlations between biodiversity and ESS supply and its values are mixed (Benayas *et al.* 2009; Haines-Young and Potschin 2010; Maes *et al.* 2011b; Maes *et al.* 2012c). The contribution of biodiversity to ecosystem resilience and its insurance values (the value of ensuring future ESS supply) are yet hardly quantified. Provisioning ESS such as from intensified timber and agricultural production may often depend on few species only, but diversity may contribute to ecosystems' capacities to absorb and adapt to external shocks and thereby ensure ESS supply in the long run (Mace *et al.* 2012; Maes *et al.* 2012c; Müller et al. 2015; Müller et al. 2016).

The case studies presented in this thesis may be used as an argument in favour of nature conservation and thereby indirectly support biodiversity protection. Nevertheless, a direct link between biodiversity and nature recreation and its economic value is difficult to capture. Within our modelling exercise, we cannot identify significant positive relationships between endangered species and recreational ESS supply and values. However, land cover diversity shows a positive significant effect on recreational visitor numbers (see chapter 4) and national parks tend to receive higher visitor numbers (see chapter 7). Diverse landscapes may offer habitat for a greater variety of species, but typically landscapes of diverse land cover are characterised by high human intervention. For the most of Europe, the highest level of hemeroby is found for forested land covers, which are not diverse and in our models shows negative effects on both, recreational visitor numbers and values.

Quantifying the direct linkages between biodiversity and ESS supply is one major difficulty and a hot research question. Even though natural ecosystems provide recreational opportunities such as hiking and bird watching, which contribute to human well-being, the link may be difficult to prove by a general linear model. Human preferences are complex and may allow for multiple maxima in the underlying utility function. Whereas some people may benefit from bio-diverse recreational sites the effect may be masked by others who do not appreciate it. Depending on local supply of alternative recreational opportunities, higher biodiversity may increase recreational ESS values, but the relationship must not necessarily be continuously positive. Nevertheless, recreational ESS may depend only on few charismatic key species, such as certain birds or mammals, but not on many other species such as insects and soil organisms which may play a far more significant role for the overall ecosystem functioning.

The challenges in biodiversity valuation are manifold and to what extent the ESS concept on its own is suitable to halt biodiversity loss and how it can be used best is questioned by various studies. Even though several studies highlight correlations between biodiversity and ecosystem functions and ESS supply (Balvanera *et al.* 2006; Duarte 2000), the linkage between biodiversity and ESS values is less clear. In particular provisioning ESS, which benefit local communities such as agriculture or timber tend to correlate less positively with biodiversity (Bullock *et al.* 2011; Dalberg and WWF 2013; Swift *et al.* 2004). Therefore, distributional effects play an important role in conservations policies. Almost every ecosystem management strategy involves trade-offs among different ESS and biodiversity.

The question, whether biodiversity is treated as a precondition of ESS supply that is to be considered an ESS itself by means of existence, bequest, information and insurance values or if biodiversity is to be considered as an instrumental value independent from human being, is debated in science and may have a crucial effect on the interpretation of ESS valuation studies and biodiversity management practice (Mace *et al.* 2012). Ecosystem processes and functions that are the basis of ESS depend on combinations of certain biotic and abiotic ecosystem components. However, the complexity of their interactions are far from being understood, even in simple ecosystems and it is so far not foreseeable how these interactions will be effected by external shocks such as climate change. Therefore, the contribution of biodiversity to secure ESS supply can only roughly be quantified and valued in economic terms. The economic concept of the Total Economic Value (see section 1.1.2) is purely anthropocentric and thus, values are based only on human preferences, which are measured by empirical methods. However, the complexity of biodiversity is difficult to grasp, in particular for the average, less informed citizens. We use the number of red list species and land cover diversity as biodiversity indicators in our case studies of chapter 4, 6 and 7, because we believe that it is relatively easy to be recognised and understood by people participating in nature recreation. However, other measures exist, such as total species counts, genetic diversity or taxonomic diversity which may be more important for ecosystem functioning but less easy to be recognised by average recreational visitor (Purvis and Hector 2000). In times where most people live in large agglomerations and vegetables and meat is bought in supermarkets, little may be known about food supplies' dependence on ecosystem functions and about biodiversity in general. In consequence, preferences for biodiversity may hardly be developed, and thus, it is only appreciated little (Sattler *et al.* 2007). On the contrary, people that do appreciate biodiversity may do so for reasons that are not necessarily captured by monetary values. Monetary values imply their substitutability by other means of man-made goods and may thereby support a perception of biodiversity commodification. However, life has moral, intrinsic and religious values, which cannot be substituted by any kind of market good.

A purely science-based approach, relying only on observable and measurable truths may therefore fail to halt ecosystem degradation and biodiversity loss. If biodiversity cannot be valued sufficiently, neither by markets nor by non-market valuation techniques, we may need to treat it like a merit good, which is to be insured by some governmental or non-governmental institution. A broader perspective including precautionary principles as well as the acknowledgement of the intrinsic value of life, as to be independent from the existence of human beings, may be required (Lee et al. 2017). The ESS concept may help to exemplify part of the value of nature, but the nature of value is complex and the ESS concept may never capture the entire significance and meaning of life and its diversity on Earth. Interwoven with society's needs, attitudes, beliefs and institutional arrangements, ESS "*valuation is a way of organizing information to help guide decisions, but not a solution or end in itself. It is one tool in the much larger politic of decision-making*" (Daily *et al.* 2000).

8.4 References

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9 Appendix

9.1 Statement of contributions

The core chapters of this thesis (chapter 2 - 7) were conceptually discussed between the authors of the chapters. They are the result of collaborations among the authors. The author of this thesis has made substantial contribution to all of the chapters, which are described more in detail in the following:

Chapter 2: Schägner, Jan Philipp, Luke Brander, Joachim Maes, and Volkmar Hartje. 2013. 'Mapping Ecosystem Services' Values: Current Practice and Future Prospects.' Ecosystem Services, Special Issue on Mapping and Modelling Ecosystem Services, 4 (June): 33–46. doi:10.1016/j.ecoser.2013.02.003.

Jan Philipp Schägner developed the research question and the conceptual design of the paper and presented it to the co-authors. He received fruitful comments and suggestions from all co-authors. Jan Philipp Schägner wrote the paper and the co-authors added valuable comments. Luke Brander revised the paper and added some sentences and a paragraph.

Chapter 3: Schägner, Jan Philipp, Joachim Maes, Luke Brander, Maria Luisa Paracchini and Gregoire Dubois. submitted. 'Monitoring Recreation Across European Nature Areas: A Geo-database of Visitor Counts, a Review of Literature and a Call for a Visitor Counting Reporting Standard', Journal of Outdoor Recreation and Tourism, 18 (June): 44–55. doi.org/10.1016/j.jort.2017.02.004

Jan Philipp Schägner developed the research question and the conceptual design of the paper and presented it to the co-authors. He received fruitful comments and suggestions from all co-authors. Jan Philipp Schägner wrote the paper and the co-authors added valuable comments and revisions. Gregoire Dubois drafted one paragraph.

Chapter 4: Schägner, Jan Philipp, Luke Brander, Joachim Maes, Maria Luisa Paracchini and Volkmar Hartje. 2016. 'Mapping Recreational Visits and Values of European National Parks by Combining Statistical Modelling and Unit Value Transfer', Journal for Nature Conservation, 31: 71–84. DOI: 10.1016/j.jnc.2016.03.001.

Jan Philipp Schägner developed the research question and the conceptual design of the paper and presented it to the co-authors. He received fruitful comments and suggestions from all co-authors. Jan Philipp Schägner wrote the paper and the co-authors added valuable comments and revisions. Joachim Maes suggested the extension of the paper by adding value estimates. Luke Brander and Maria Luisa Paracchini revised the paper.

Chapter 5: Brander, Luke M., Florian V. Eppink, Jan Philipp Schägner, Pieter J. H. van Beukering, and Alfred Wagtendonk. 2015. 'GIS-Based Mapping of Ecosystem Services: The Case of Coral Reefs.' In Benefit Transfer of Environmental and Resource Values, edited by Robert J. Johnston, John Rolfe, Randall S. Rosenberger, and Roy Brouwer, 465–85. The Economics of Non-Market Goods and Resources 14. Springer Netherlands. http://link.springer.com/chapter/10.1007/978-94-017-9930-0_20.

Jan Philipp Schägner was part of a team working on the spatial assessment of ecosystem services and on the development of methodologies for the best ecosystem service mapping practice. He engaged in fruitful discussions on this topic, which were one important basis for the development of the paper. He mainly wrote the subsection on the methodologies for mapping values of ecosystem services.

Chapter 6: Schägner, Jan Philipp, Luke Brander, Joachim Maes, Maria Luisa Paracchini and Bastian Bertzky. submitted. 'Spatial Dimensions of Recreational Ecosystem Service Values: A Review of Meta-Analyses and a Combination of Meta-Analytic Value-Transfer and GIS', Ecosystem Services.

Jan Philipp Schägner developed the research question and the conceptual design of the paper and presented it to the co-authors. He received fruitful comments and suggestions from all co-authors. Jan Philipp Schägner wrote the paper and the co-authors added valuable comments and revisions.

Chapter 7: Schägner, Jan Philipp, Luke Brander, Joachim Maes, Maria Luisa Paracchini and Volkmar Hartje. 2016. 'Mapping the Recreational Value of Non-Urban Ecosystems across Europe: Combining Meta-Analysis and GIS', European Association of Environmental and Resource Economists 22nd Annual Conference. Zurich, Switzerland.

Jan Philipp Schägner developed the research question and the conceptual design of the paper and presented it to the co-authors. He received fruitful comments and suggestions from all co-authors. Jan Philipp Schägner wrote the paper and the co-authors added valuable comments and revisions.

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