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Chapter 1

Introduction

The natural environment is under increasing pressure from global society over the past century (e.g. MA, 2005a; Rockström et al., 2009). Many interrelated factors are contributing to these pressures, including rapid urbanization, increasing human population, growing demand for food, expanding and intensifying agriculture and increasing use of freshwater (UNEP, 2019). Climate change is projected to further intensify these pressures on the natural environment, especially if global temperature cannot be kept below the 1.5°C increase (IPCC, 2018). This has severe impacts on global biodiversity (Johnson et al., 2017), such as exemplified by the recently observed major decline in insect biomass in German nature areas (Hallmann et al., 2017). To mitigate these pressures and promote the sustainable use of the natural environment, there is great need to better incorporate the effects of human actions in policy and decision-making.

To this end, the ecosystem services concept has been developed (de Groot et al., 2002; Fisher et al., 2009). The ecosystem services approach aims to conceptualize the interactions between ecosystems and society, which may help to capture and quantify the human-ecosystem relationship. The approach has been rapidly adopted by policymakers. In order to apply the ecosystem services concept in environmental policy and decision-making, it is necessary that the services that ecosystems provide are quantified, such that policymakers can incorporate environmental effects into policy considerations. Hence, researchers have developed various methods to quantify ecosystem services, both in biophysical and in social and economic terms.

However, parallel with the attempts to quantify ecosystem services, there is an ongoing academic debate about the desirability of such valuation. Especially economic valuation in monetary terms is controversial (Gómez-Baggethun et al., 2010). Some argue that monetary valuation is "inappropriate and fundamentally alien" to the "deeply held nonmonetizable value" of nature (Harvey, 1996: 156). They belief that nature has intrinsic value, which cannot be approximated in monetary terms. Furthermore, researchers debate whether monetary valuation is in fact possible with the current set of economic valuation methods (e.g. Chee, 2004; Ludwig, 2000; Spangenberg and Settele, 2010). Amongst others, it has been argued that although economic valuation methods are based on the same economic assumptions, they take different angles to estimate the value of ecosystem services which could lead to diverging values (Spangenberg and Settele, 2010). Also, it has been noted that it may be challenging to grasp the underlying complexity in ecological functioning of the ecosystem that produces the services with monetary value estimation (Chee, 2004; Polasky and Segerson, 2009).

Despite these concerns, however, most scientists and practitioners have adopted a pragmatic view upon monetary valuation of ecosystem services, according to a recent survey (Ainscough et al., 2019). It is felt that monetary valuation is generally non-problematic, although caution should be in place about potential misuse. As such, while being aware of the shortcomings, researchers largely seem to accept the idea of quantifying ecosystem services in economic terms (e.g. Schröter et al., 2014b; Schröter and van Oudenhoven, 2016), probably in recognition of the opportunities it offers to incorporate environmental effects into decision-making (e.g. de Groot et al., 2010a; Guerry et al., 2015; Hauck et al., 2013). After all, monetary valuation offers a relatively straightforward and transparent way of quantifying ecosystem services.

Yet, while most researchers thus take a moderatly positive stance towards monetary valuation, and many studies have estimated the value of ecosystem services in monetary terms (de Groot et al., 2012), relatively little attention has been paid to the validity of the various monetary valuation approaches used. Researchers have typically analyzed the methodological reliability of valuation methods, primarily for summarizing ecosystem service values and (potential) application in benefit transfer, while simultaneously also testing for the main determinants of ecosystem service values, such as socio-economic conditions of beneficiaries and spatial aspects of service delivery (e.g. Brander et al., 2006; Ghermandi et al., 2010). Although these studies have sometimes taken characteristics of the ecosystem along in their analyses, they do not address whether the ecological status of the ecosystem that produces the services is reflected in ecosystem service values in a consistent manner. As such, it remains unclear whether estimated monetary values for ecosystem services actually reflect the ecological status of the ecosystem.

The lack of attention to the validity of monetary valuation methods with respect to the ecosystem's ecological status, is problematic given the promise of the ecosystem services approach to better integrate environmental effects into policy and decision-making. In order to be effective as a policy tool, monetary valuation methods should produce numbers that provide meaningful information about the ecological status of the ecosystem that delivers the service. For example, the estimated monetary values for ecosystem services from degraded ecosystems need to reflect their decreased capacity to provide services. Only then, valuation is meaningful for policies that aim to prevent land degradation and to support sustainable land management practices. Similarly, when the provided services are used or enjoyed by beneficiaries, estimated monetary values for these services also need to reflect the demand or need for them. In degraded ecosystems the need for, for instance, water provisioning services may be higher than in well-functioning ecosystems. Only when estimated monetary values adequately reflect such differences in benefits from ecosystem services, policy measures will automatically steer towards promoting sustainable practices, while simultaneously discouraging environmental degradation.

Hence, it must be clear whether the monetary values that are estimated for ecosystem services with the current set of monetary valuation methods adequately reflect the benefits that they are aiming to capture. As such, there is an urgent need to evaluate the estimated monetary values for ecosystem services, especially since monetary valuation is now starting to be widely applied in policy and practice. This thesis will do so in the context of water-related ecosystem services.

1.1 The ecosystem services approach

The ecosystem services approach is a concept which aims to articulate the relation between ecosystems and human society by capturing the benefits that ecosystems provide to society (MA, 2005a). It aims to grasp both the benefits that ecosystems provide to people as well as the dependencies of people on ecosystems. Ecosystem services consist, for instance, of the production of food from arable land, purification of water by wetlands, pollination of crops by insects and recreation opportunities by natural areas (see e.g. Goldman, 2010).

The Millennium Ecosystem Assessment (MA) has proposed a typology of provisioning, regulating and cultural services, which has been widely adopted and has been worked out in detail by CICES (2018). Several conceptual frameworks have been proposed to visualize the beneficial relations between ecosystems and society (e.g. Daily et al., 2009; de Groot et al., 2010a). These frameworks offer the potential to integrate environmental effects of human actions into policy and decision-making (see Figure 1.1; Daily et al., 2009). As such, the concept of ecosystem services can help to integrate environmental effects in different types of decision contexts, such as the generation of public understanding about the dependence of society on ecosystems, cost-benefit analyses in environmental decision-making, landscape management, nature conservation or a mix of the above (Balmford et al., 2002; Fisher et al., 2009).

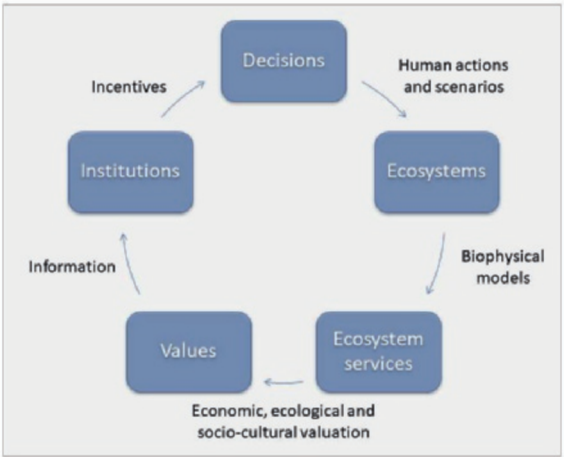

Figure 1.1 Framework visualizing how ecosystem services can be integrated into decision-making (after: Daily et al., 2009).

Since the ecosystem services concept was proposed by the MA as a policy tool for sustainable natural resource use, the ecosystem services approach has been gaining momentum in policy and practice (de Groot et al., 2012; Dick et al., 2018). Up to now, the concept has been integrated into various international policy initiatives. In the EU biodiversity strategy for 2020, ecosystem services are a central component, on which member states are required to report in their national reporting systems (EC, 2011). To this end, the 'Mapping and Assessment of Ecosystems and their Services (MAES)' framework has been developed (Maes et al., 2013).

Concrete national initiatives that are being initiated to assess ecosystem service benefits are, amongst others, the 'Atlas Natuurlijk Kapitaal' in the Netherlands (Rijksoverheid, 2015), and 'National Ecosystem Assessments' in the United Kingdom (UK NEA, 2011a) and Germany (Albert et al., 2017).

Also, ecosystem services are central to other major international policy initiatives, such as the 'Sustainable Development Goals' (SDGs; e.g. being integral to goal number 15 'Life on land' that focusses on conserving ecosystem services provided by terrestrial and inland freshwater ecosystems; UN, 2018) and an independent United Nations panel has been erected to provide assessments and policy-relevant tools about benefits from ecosystems (i.e. the 'Intergovernmental Science-Policy Panel on Biodiversity and Ecosystem Services' (IPBES); Díaz et al., 2015). Other initiatives focus on integrating the economic benefits of ecosystem services into decision-making processes both for public and private sector organizations. These initiatives include, for example, 'The Economics of Ecosystems and Biodiversity' (TEEB) which is aiming to make decision-makers recognize the (economic) benefits from ecosystems (TEEB, 2010) and the Natural Capital Protocol which provides guidelines on how to account for the impacts and dependencies of organizations on natural capital (Natural Capital Coalition, 2016).

1.2 The valuation of ecosystem services

To operationalize the ecosystem services concept for decision-making, the values of ecosystem services need to be quantified (Boerema et al., 2016). Different types of valuation have been developed over time, as ecosystem service values may span over highly different value dimensions (see e.g. Martín-López et al., 2014; Pascual et al., 2010). Biophysical and economic valuation have been the most common types of valuation, while socio-cultural valuation has also been gaining attention more recently (Scholte et al., 2015).

Using *biophysical valuation* (or: ecological valuation or biophysical quantification), ecosystem services are measured in terms of the contribution of ecological functions and processes to the provision of services (in biophysical units; de Groot et al., 2010a). For example, carbon sequestration can be measured as the amount of tonnes carbon dioxide (CO_2) stored and water purification as the amount of dissolved nitrogen in the water. Different types of biophysical valuation have been developed, such as indicator-based methods (e.g. Egoh et al., 2012; van Oudenhoven et al., 2012) and mapping and modelling approaches (e.g. Bagstad et al., 2013b; Burkhard et al., 2012; Nelson et al., 2009; Stürck et al., 2014). Although the measurement of ecosystem services with the biophysical approach is relatively straightforward, the complexity of their integration into decision-making processes may be challenging, as it demands a comprehensive understanding of the ecosystem services involved and the demand for them.

To facilitate such understanding, *economic valuation* has been developed as a means to quantify ecosystem services (Fisher et al., 2008). The approach of monetary valuation focusses on estimating the values of the benefits that are provided by ecosystem services in monetary terms (Bateman et al., 2011). This approach generates monetary value estimates that can be relatively easily incorporated into economic decision-making processes, such as cost-benefit analyses (Fisher et al., 2009). Monetary values are typically estimated using economic valuation methods that are based on the economic principle that the value (of ecosystem services) can be estimated based on the welfare that they generate (Bateman et al., 2011; Bockstael et al., 2000; Polasky and Segerson, 2009). From this perspective, the benefits from ecosystem services are seen as use or instrumental values (Gómez-Baggethun et al., 2010). As such, monetary valuation is mainly based on individual preferences, utility values and rational choices, which may tend to overlook communal, non-use and cultural values (Chan et al., 2011).

As it has been recognized that monetary valuation may have difficulty to capture social and cultural value dimensions, *socio-cultural valuation* has also been developed (Cowling et al., 2008; Iniesta-Arandia et al., 2014; Scholte et al., 2015). In socio-cultural valuation, social and cultural aspects are combined in an attempt to capture the importance that people attach to ecosystem services (Daniel et al., 2012). A wide variety of methods has been developed to this end, such as social mapping and social preference surveys (Kelemen et al., 2016; Scholte et al., 2015). These are typically characterized by consultative, participatory and deliberative processes and generate qualitative and/or quantitative measures for ecosystem service values (Christie et al., 2012). Due to the large heterogeneity in socio-cultural valuation methods, valuation outcomes obtained with different methods are often difficult to compare (Kelemen et al., 2016), which may hamper their integration into decision-making processes.

In view of these different types of valuation, recently a more pluralistic view on ecosystem services valuation has been emerging. This view considers that ecosystem services may contain different value dimensions and advocates to integrate the different types of valuation (i.e. so-called 'integrated valuation'; Jacobs et al., 2016). It argues that based on the value dimensions contained in the ecosystem services under consideration, relevant valuation approaches can be selected and – if need be – integrated using synthesizing methods, such as deliberative valuation and multi-criteria decision analysis (Jacobs et al., 2018). For example, ecological, economic and social values of alternative scenarios for water pollution control were weighed in an integrated manner using multi-criteria analysis (Liquete et al., 2016). This approach has been adopted by IPBES as 'holistic valuation' that aims to capture and integrate the diversity of values (IPBES, 2016; Pascual et al., 2017). While doing so, IPBES has also rephrased 'ecosystem services' into 'nature's contributions to people' (Díaz et al., 2018). This way, IPBES aims, amongst others, to take the cultural context of stakeholder groups better into account in valuation. The IPBES approach, however, also has been subject to considerable debate (e.g. Braat, 2018; Kenter, 2018; Maier and Feest, 2016; Peterson et al., 2018). Some argue, for instance, that by replacing 'ecosystem' by 'nature', ecosystems can be disregarded that are not seen as nature, such as urban or agro-ecosystems (Peterson et al., 2018).

While the integrated approach makes the valuation of ecosystem services more balanced across the several valuation approaches, it does not solve specific issues of the underlying valuation methods. This thesis focuses on the methods to estimate the value of ecosystem services in monetary terms.

1.3 Monetary valuation methods

Different monetary valuation approaches have been developed that try to estimate the monetary value of ecosystem services, while accommodating for the different settings in which the various ecosystem services prevail (Bateman et al., 2011; Farber et al., 2006; Freeman III, 2003). Some services, such as food and raw materials provision, provide benefits that have direct use value which are directly traded on the market. In this case, the monetary values of these services can be estimated using the ‘direct market pricing’ method. Some ecosystem services provide indirect benefits to end products that are directly traded on the market. For these services, valuation can be based on indirect means of value assessment (Farber et al., 2002). Their value may be estimated by the extent to which they contribute to these end products, using a ‘production function’ method. This may, for instance, be the case for services such as water purification and soil erosion control that contribute to agricultural production. A different approach of indirect measurement relates to estimating the costs of preventing environmental damage or degradation, for example, for services such as climate or flood regulation. These cost-based methods estimate the costs of not degrading the environment: they include ‘avoided cost’, ‘replacement cost’ and ‘mitigation and restoration cost’ approaches and typically estimate optional use values.

When values cannot be directly or indirectly deduced from market prices, the monetary value of ecosystem services can be estimated based on hypothetical markets. This type of valuation can typically accommodate the estimation of services that have optional use value, such as recreation and tourism, or non-use value, such as cultural values. Two approaches of hypothetical markets have been developed. The first approach is based on revealed preferences, estimating the amount of money spent on ecosystem services and goods, such as expenditures on recreational activities or buying farmland with associated soil fertility. The second approach is based on stated preferences, asking individuals directly what they would be willing to pay for an ecosystem service, such as water quality improvement or landscape aesthetics. Finally, a method that potentially can estimate the value of all types of ecosystem services is ‘benefit transfer’ (Plummer, 2009). This secondary valuation method estimates the monetary value of ecosystem services by transferring ecosystem service values based on previous estimates from primary valuation studies.

1.4 The role of the study context

Regardless of the economic valuation method used, the monetary values should reflect relevant characteristics of the study context in order to provide a reliable estimate of the value of its ecosystem services. The study context comprises the ecosystem’s capacity to produce ecosystem services (capacity side), the demand by beneficiaries for ecosystem services (demand side) and the spatial flow of ecosystem services within the landscape from the ecosystem to the beneficiaries (see Figure 1.2; Serna-Chavez et al., 2014; Villamagna et al., 2013). The capacity to deliver ecosystem services depends primarily on the conditions of the ecosystem that provides services (‘ecosystem factors’), while the demand for ecosystem services depends mainly on the conditions of (groups of) beneficiary individuals ('beneficiary factors'; Villamagna et al., 2013). Here, the distinction made between ‘capacity’ and ‘demand’ – following the classification by Villamagna et al. (2013) – is largely similar to other terms used in specific research fields, such as ‘ecosystem function’ and ‘ecosystem service’ in ecological economics (de Groot et al., 2002), ‘supply’ and ‘demand’ in ecosystem services mapping (Crossman et al., 2013) and ‘capacity’ and ‘flow’ in ecosystem accounting (Schröter et al., 2014a). Furthermore, spatial characteristics of the flow of ecosystem services from the site of production by the ecosystem to the location of use by the beneficiaries are relevant ('spatial factors'; Serna-Chavez et al., 2014).

Apart from these characteristics of the study context, given that the monetary value of ecosystem services can be estimated with different valuation methods, characteristics of the selected methodological approach may have an impact on monetary value estimation (‘methodological factors’). Ideally, however, such methodological factors do not influence the estimates, as it is the aim of valuation studies to quantity the spatially distributed services of the ecosystem for its beneficiaries (i.e. the factors of the study context).

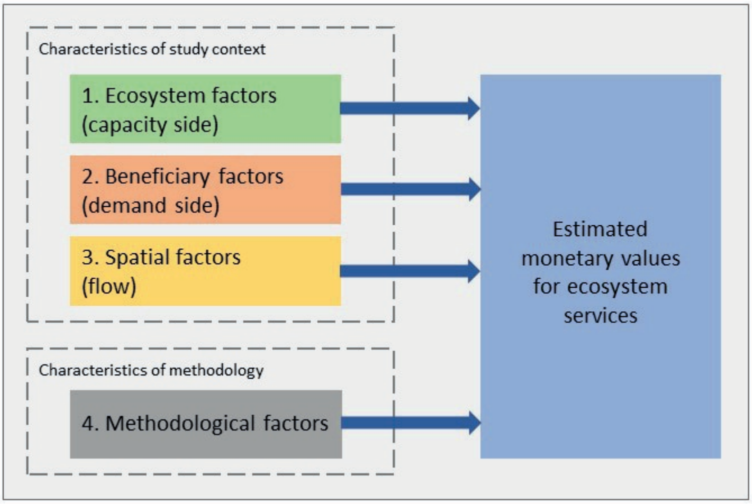

Figure 1.2 Simplified theoretical framework of factors considered in the analysis of estimated monetary values for ecosystem services in this thesis, including characteristics of the study context: (1) ecosystem, (2) beneficiary and (3) spatial factors, and characteristics of the methodology: (4) methodological factors.

Besides methodological factors, characteristics of the study context, i.e. ecosystem, beneficiary and spatial factors, thus determine the estimation of monetary values for ecosystem services. *Ecosystem factors* play a crucial role, as the type and number of ecosystem services that can be delivered and in what quantity and quality depends on the ecological status of the ecosystem. Ecosystem factors that can play a role are related to conditions of the soil, vegetation, water, climate and biodiversity, amongst others. Degraded ecosystems, for instance, are likely to deliver different or less services that will be of less quality than those delivered by well-functioning ecosystems. For example, the service of recreational angling will be less (if existent at all) in surface water bodies of poor water quality than in water bodies of good quality. Hence, it is expected that monetary value estimation of ecosystem services will depend on the ecological status of the ecosystem.

Some researchers have emphasized the importance of ecosystem characteristics for the delivery of ecosystem services (Fisher et al., 2009; Maes et al., 2012). Many studies have taken ecosystem structure as the inherent basis for ecosystem services delivery when conceptualizing ecosystem service frameworks (e.g. Carpenter et al., 2009; de Groot et al., 2002). Several studies have systematically reviewed the role of (a)biotic attributes in the delivery of ecosystem services (Harrison et al., 2014; Smith et al., 2017). However, only few studies have explicitly investigated the role of ecosystem characteristics in the monetary valuation of ecosystem services in systematic reviews (Johnston et al., 2005) or the impact of restoring ecosystems on the valuation of ecosystem services in specific case study contexts (Acuña et al., 2013; Vermaat et al., 2016).

Beneficiary factors refer to the characteristics of (groups of) individuals who benefit from the provided ecosystem services that are expected to affect monetary values estimation (Boyd and Banzhaf, 2007; Fisher et al., 2009). Socio-economic characteristics of ecosystem service beneficiaries, including personal and socio-economic characteristics, are therefore likely to influence the values that are associated to ecosystem services. Personal characteristics can affect how people experience or enjoy ecosystem services, such as their perception about the ecological status of the ecosystem or their commitment towards environmental protection (e.g. Orgill et al., 2013). Also, socio-economic characteristics, such as the welfare status of a group of beneficiaries, can influence what kind of value will be attached a particular ecosystem service – such as the value that people from different societal groups as anglers, boaters and bird watchers attach to water recreation services. Previous studies have found that indicators, such as income level, GDP per capita and population density, may significantly affect monetary values for ecosystem services (e.g. Brander et al., 2006; Genius et al., 2008; Vermaat et al., 2016).

Spatial factors characterize the flow of ecosystem services from the site of production (‘capacity’) to that of enjoyment ('demand'; Bagstad et al., 2013a; Serna-Chavez et al., 2014). For example, due to the downstream movement of water within a river basin, spatial dependencies would be important to weigh in the valuation of water-related services. As ecosystem conditions, provided services and beneficiaries are all spread heterogeneously across the landscape (Boyd and Banzhaf, 2007; de Groot et al., 2010a; Fisher et al., 2009), factors that capture this spatial heterogeneity may explain variation in monetary values. Some services may be produced and enjoyed locally, while others may flow from the site of production to the site of enjoyment on a larger spatial scale, such as water-related services within a river basin (Hein et al., 2006; Serna-Chavez et al., 2014). As such, the distance between the location where the service is produced and where the beneficiaries reside can be relevant, for example in case of water quality-related ecosystem services (see: Jørgensen et al., 2013; Schaafsma et al., 2012). Although the effect of distance has been relatively well investigated, other spatial factors that are expected to be relevant to affect monetary value estimations have not yet received much attention. For instance, the ecological status of the ecosystem that provides the service in relation to the distance of the beneficiary to that resource can be of relevance.

Previous studies have, thus, investigated the role of these characteristics of the study context to varying extents. While the role of beneficiary factors has received relatively much attention, spatial factors and, in particular, ecosystem factors have often been overlooked. This thesis aims to fill this gap in the literature by analyzing the joint role of all characteristics, with particular attention for the – often neglected – role of ecosystem factors and a specific focus on water-related ecosystem services.

1.5 Water-related ecosystem services

To investigate the role of study context characteristics in monetary valuation – and in particular those of the ecosystem – ecosystem services that are related to water provide an excellent case. Next to light and temperature, water is a vital element for life on earth, as it determines the net productivity of ecosystems (Molles Jr., 2008). However, the availability of global freshwater resources is limited and will become more limited due to climate change (Collins et al., 2013; Mekonnen and Hoekstra, 2016). On an ecosystem scale, the availability of water is primarily controlled by ecosystem factors, e.g. of the soil, flora and fauna. A particularly powerful example that illustrates this is whether there will be rainfall or not is determined by the presence of forests (Sheil and Murdiyarso, 2009). Another clear example shows that the infiltration of rainfall into crusted dryland soils is promoted by the activity of termites (Mando et al., 1996). Because ecosystem factors control the availability of water in the ecosystem, these factors are vital in the provision and valuation of ecosystem services. Hence, it is expected that if ecosystem factors play a role in the monetary valuation of ecosystem services, they will be especially relevant in case of water-related ecosystem services due to the key role of water in ecosystem services delivery.

Water can play both a direct and indirect role in the delivery of water-related ecosystem services. The direct role lies in those services that are provided by water itself. These range from drinking water provision, flood regulation, water quality control, salinization prevention to water recreation (Brauman et al., 2007; CICES, 2018; Keeler et al., 2012). In the indirect role of water, water indirectly contributes to the delivery of ecosystem services, as many ecosystem functions and processes such as soil functioning, vegetation productivity, animal abundance and human activity are controlled by water (Molles Jr., 2008). For example, water is indispensable for provisioning services, such as food and raw materials. Water also contributes to regulating services, such as providing water to maintain soil processes (i.e. soil functioning) or local humidity levels (i.e. climate regulation). Finally, water is of relevance in maintaining cultural values, such as aesthetic values, recreation and spiritual well-being. This way, water serves as a key intermediary in the delivery of many ecosystem services. Both the ecosystem services in which water plays a direct and indirect role will be referred to as water-related services in this thesis.

The main pressures on water-related services are alterations in water quantity and quality (Grizzetti et al., 2016). When there is little water available or the water is of poor quality, water quantity and quality may be primarily determining ecosystem functioning and the delivery of water-related services (Brauman et al., 2007; Keeler et al., 2012). Water quantity affects water-related services in various ways, which is particularly visible when the availability of water is limited. Altered water quantity levels occur in regions where the water availability has become artificially low due to high demand for consumption and irrigation or is naturally low due to potential evapotranspiration outweighing precipitation (i.e. drylands; Leemans and Kleidon, 2002; Mekonnen and Hoekstra, 2016). In these regions, water availability may become more limited due to climate change, which is expected to increase the number of periods with lower water availability (Collins et al., 2013; Mekonnen and Hoekstra, 2016). Water quantity affects the delivery of multiple water-related services, both in a direct and indirect manner (Brauman et al., 2007; Grizzetti et al., 2016). The delivery of many services is directly affected, such as the provision of water for consumptive, non-consumptive, irrigation and industrial use, hydropower generation, waste dilution, flood control and salinization prevention (Karabulut et al., 2016). Also, ecosystem functioning is affected by water quantity, which indirectly affects the delivery of many other services, such as commercial or recreational fishing by providing fish habitat as a nursery or refuge, local climate regulation by maintaining humidity patterns and wastewater attenuation through natural dilution (Grizzetti et al., 2016).

Besides water quantity, water quality may also significantly impact the delivery of water-related ecosystem services (Brauman et al., 2007; Grizzetti et al., 2016; Keeler et al., 2012). Water quality directly influences ecosystem services, such as water purification by filtering and absorbing pollutants, for example, through nitrogen removal by micro-organisms and prevention of soil erosion by sediment retention of (aquatic) vegetation (Lautenbach et al., 2012; Vigiak et al., 2016). Indirectly, water quality also affects multiple ecosystem services, such as drinking water provision through water purification and waterborne disease prevention and water recreation opportunities (Keeler et al., 2012; Rankinen et al., 2016). Water-related services may come particularly under pressure in intensively used regions where the water quality is compromised due to either point discharges from highly urbanized and industrialized areas or diffuse discharges from intensively farmed land (Carpenter et al., 1998). Because of climate change, water quality may more easily deteriorate due to larger fluctuations in water availability and higher water temperatures that, for example, may cause increased frequencies of algal blooms (Whitehead et al., 2009).

1.6 Research objectives of the thesis

The overall aim of this thesis is to investigate whether and to what extent characteristics of the study context play a role in the estimation of monetary values for water-related ecosystem services. In particular, the role of ecosystem factors will be investigated, while simultaneously examining other factors that have previously been proven to be relevant in monetary value estimation, being beneficiary and spatial factors. Besides, characteristics of the selected methodology may also affect estimated values (i.e. methodological factors). The role of the study context in monetary valuation of water-related services will be investigated for two types of pressures on the delivery of water-related services, i.e. water quantity and water quality, and at two types of spatial scales, i.e. the global and river basin scale. Estimated monetary values for water quantity-related services will be investigated for regions that experience water scarcity, being drylands across the world. Estimated monetary values for water quality-related services will be investigated in a river basin that faces poor water quality, being the Scheldt river basin in western Europe.

This research aim is addressed in the following research questions:

(1) Do ecosystem and beneficiary factors play a role in estimated monetary values for water quantity-related ecosystem services in drylands?

(2) Do estimated monetary values for water quantity-related ecosystem services differ when estimated with different valuation methods or provided by different dryland ecosystem types?

(3) Do ecosystem and spatial factors play a role in the willingness to pay (WTP) for water quality-related ecosystem services in the Scheldt river basin?

(4) Is the perception of beneficiaries about the water quality affected by actual, empirically

measured water quality, when stating their preferences for water quality-related ecosystem services? And is this relation affected by beneficiary factors?

1.7 Outline of the thesis

The role of the study context – and in specific of ecosystem factors – in the monetary valuation of water-related ecosystem services is examined in different ways in the following chapters. In chapters 2 and 3, the role of local ecosystem and beneficiary factors in the monetary valuation of ecosystem services is investigated for water quantity-related services in drylands. For these drylands, it is analyzed whether local ecosystem conditions, such as vegetation productivity and water availability, play a role in the estimated monetary values for water quantity-related services, next to indicators for local socio-economic conditions of beneficiaries and selected methodological approach. This is done by means of a meta-analysis that summarizes more than 500 observations of estimated monetary values for ecosystem services from 66 valuation studies in drylands worldwide (chapter 2). As differential effects are found for the type of ecosystem and valuation method, this analysis is followed by an in-depth study of the effects of the type of dryland ecosystem and the type of valuation method on estimated monetary values for individual water quantity-related services (chapter 3).

While chapter 2 and 3 look at water quantity-related ecosystem services, chapters 4 and 5 focus on the estimation of monetary values for water quality-related ecosystem services. These are specifically investigated by means of the stated preferences valuation method of WTP for water quality in the Scheldt river basin in western Europe. The role of three ecosystem and spatial factors of the study context (i.e. actual measured water quality of the nearest water body, distance to the nearest water body and local land cover) on WTP for water quality improvements are investigated (chapter 4). The role of these factors is investigated in addition to beneficiary and methodological factors. As public perception of the actual water quality may be an important determinant of the WTP for water quality-related services, the public perception of water quality is investigated in relation to the actual measured water quality and how this affects preferences for water quality improvements (chapter 5). Specifically, the role of beneficiary factors is investigated in explaining differences in water quality perception. In the final chapter, the findings from the four previous chapters are discussed (chapter 6). In this chapter, also research gaps and implications are identified and the conclusion of this thesis is presented.