Essays on the Economics of Ecosystems and Biodiversity

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Contents

Introduction

Intact ecosystems are fundamentally important for the people living on this planet. They secure livelihoods, provide basic materials for life, and contribute to well-being and health. This insight has been known for millennia (see, e.g., Fisher et al., 2009), and the importance of ecosystem services (ES) has been discussed in science implicitly and explicitly for decades (see Daily et al. (1997) for an overview). However, it was the Millennium Ecosystem Assessment (MA, 2005) which has prominently put forward the role of biodiversity and ecosystems for human well-being and has conceptualized the term “ecosystem services” to categorize in which ways humans benefit socially and economically from biodiversity and ecosystems.

According to the MA (2005), ES are “the benefits people obtain from ecosystems” and can be grouped into the following four categories:

- Provisioning services, such as food, water, timber, and fiber,
- Regulating services, such as climate regulation, flood protection, and water purification,
- Cultural services, such as recreation, aesthetic enjoyment, and spiritual fulfillment, as well as
- Supporting services, such as soil formation, and nutrient cycling.

Figure 1 visualizes the interrelations between ecosystems, biodiversity, ES, and human well-being on the one hand and drivers of change as well as governance and decision-making on the other hand. It takes into account a more recent definition of ES, which are now seen more as “the direct and indirect contributions of ecosystems to human well-being” (TEEB, 2010). This is close to the original definition of the MA but makes a finer distinction between ES and benefits (see also Fisher et al., 2009). Within ecosystems, biophysical structures, processes, and functions form the basis for the provision of ES. The benefits of these ES are generated where people are directly affected and enjoy the services. In many cases, the realization of the benefits necessitates the input of other forms of capital, i.e., labor or physical capital.

ES can thus also be understood as a flow of services generated by the stock of natural capital, which benefit humans often only after some form of production or processing (Fisher et al., 2009). The value attached to different benefits, i.e., their importance or worth, can vary with different sets of preferences or norms. Finally, information on the multiple benefits and values of ES can be incorporated in decision-making and influence governance structures. This in turn influences the direct and indirect drivers of change, which feed back into biophysical structures, processes, and functions (TEEB, 2010).

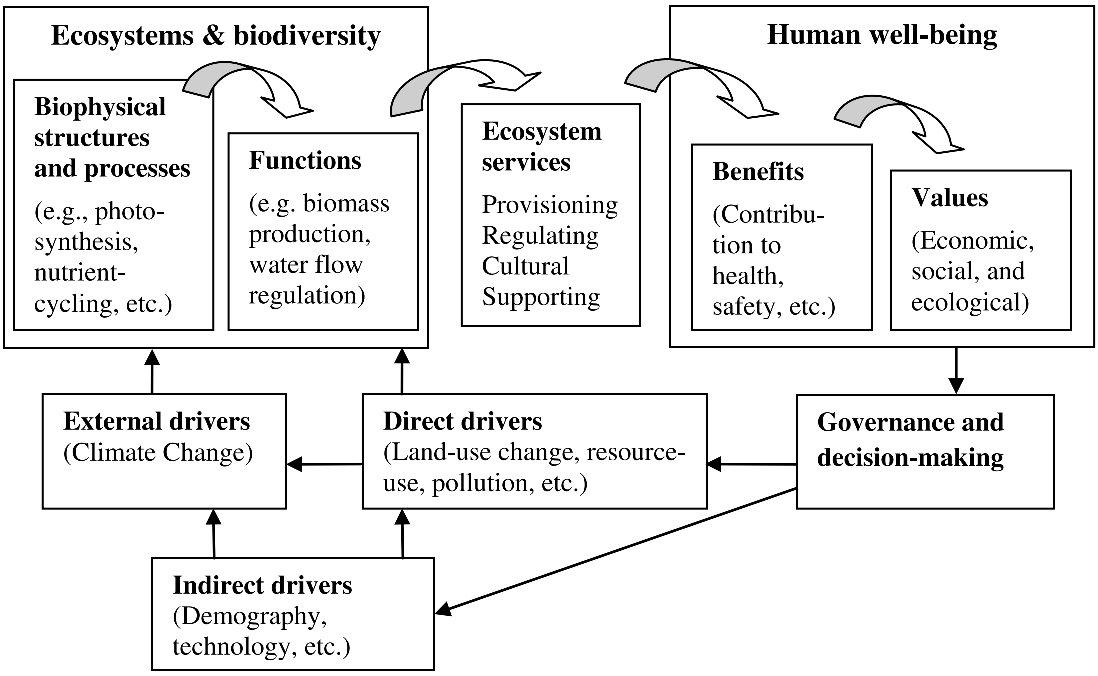

Figure 1. Relationship between ecosystems, biodiversity, ecosystem services, and human well-being. Own presentation, based on TEEB (2010) and MA (2005).

The role of biodiversity for ES provision and human well-being is important but still subject to research in many respects. It is widely acknowledged that biodiversity underpins the functioning of ecosystems and is essential for a sustained flow of ES (CBD, 2010; MA, 2005). However, there remains a vast uncertainty about the exact links between biodiversity, ES, and human well-being (CBD, 2010). One reason for this is the broad meaning of the term “biodiversity”. It is defined as “the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part” (UN, 1992). This includes variability at multiple scales of biological organizations (genes, species, and ecosystems) and at different geographic scales (local, regional, or global). The relationship between biodiversity, ES, and human well-being thus depends on contexts and scales, which requires a case-wise appreciation of their interrelations and impedes an easy “one-size-fits-all” global assessment (Sukhdev et al., 2014).

A major problem of our time is that biodiversity and ecosystems have been degrading with increasing speed over the last decades and still continue to degrade. To give just a few examples, over half of the 14 biomes assessed by the MA have experienced a 20-50% conversion to human use (MA, 2005). The extent and integrity of natural habitats are thus continuously declining in most parts of the world, and extensive fragmentation and degradation of habitats are contributing to biodiversity loss (CBD, 2010). Over the past few hundred years, human actions have led to an increase in species extinction rates by at least 100 times the background rates that were typical throughout Earth’s history (MA, 2005). Therefore, the target agreed on by the world’s governments in 2002 under the umbrella of the United Nations Convention on Biological Diversity (UN CBD) “to achieve by 2010 a significant reduction of the current rate of biodiversity loss at the global, regional and national level” has not been met. Instead, biodiversity is continuing to decline in all three main components – genes, species, and ecosystems (CBD, 2010).

As the MA acknowledges, people all over the world have benefitted from the conversion of natural ecosystems to human-dominated systems. Agriculture, fisheries, and forestry, for example, are often major pillars of national development strategies, providing revenues that allow investments and economic growth. Still, this contributes to biodiversity loss and habitat degradation, and the costs of these activities extend beyond the direct costs of conversion and use. First, there are direct trade-offs between the provision of different ES. Managing practices that increase harvests in aquaculture, for example, may reduce rice yields of nearby subsistence farmers (MA, 2005). This example also shows that, second, the benefits and costs of conversion often accrue to different groups of people, which raises questions of equity and fair compensation. Third, resource extraction and use can diminish the natural capital base and impede a sustainable future provision of ES.

Many of these indirect costs that are induced by the conversion and economic use of ecosystems are not factored into decision-making. In economic terms, the main reason for this is that markets for many ES are missing or imperfect. There are market failures due to external effects and the public good nature of biodiversity and many ES. The presence of external effects implies that an economic agent only carries the direct costs and benefits of ecosystem use or conversion, while the utility of other agents is also (negatively) affected but not compensated. The costs related to a – present or future – impairment of an ecosystem, for example, thus partly fall upon society as a whole or on subgroups of the population who do not benefit from the respective actions. The public good problem implies that people cannot be reasonably excluded from the use of a resource, which leads to free-riding and under-supply of this resource in unregulated situations. Even where markets exist, they thus do not signal the true, social costs of ecosystem conversion and use to individual decision-makers, which frequently leads to inefficient resource allocation and over-use (e.g., Perrings et al., 2009; Perman et al., 2009).

There are several ways in which economic methods are helpful to make explicit and internalize such external effects. One way is to incorporate the non-marketed benefits of ES and external effects in theoretical economic models, which may be used to inform policy-making, particularly if the models are validated and calibrated according to sound empirical findings. Theoretical (bio-)economic models are frequently used to analyze the efficient and optimal allocation of scarce natural resources at one point in time as well as over time. When market failures such as externalities exist, privately optimal allocations diverge from efficient and socially optimal allocations. Theoretical models can help identify the effects of such market failures on market outcomes. The resulting findings can be used to develop management rules and governance schemes to induce the efficient and socially optimal behavior of individual economic agents and to internalize external effects (Perman et al., 2009).

A second way is the economic valuation of the non-marketed benefits related to ES or the costs of their impairment. Economic valuation helps to uncover the trade-offs from decisions affecting ecosystems and biodiversity, i.e., it makes explicit the relative scarcity of ES by uncovering the opportunity costs of their use. In addition, funds for conservation are scarce and limited such that there will often be a need to choose between different conservation projects or to weigh conservation against economic development. Economic valuation of ES may thus be an additional tool to inform policy-makers about the benefits that ecosystems and biodiversity entail, such that appropriate incentive or compensation schemes can be created (Pearce, 1997; TEEB, 2010).

This dissertation uses both of these approaches to consider selected economic aspects of ES and biodiversity and derive recommendations for policy-making in different contexts. It is structured by the types of ecosystems considered and methodological approaches used. Part 1 of this dissertation comprises two papers that deal with the economic valuation of urban ES. The first paper uses the travel cost method and a random utility framework to value the benefits derived from recreation in urban parks. The second paper uses an alternative approach, i.e., the life satisfaction approach for valuing the contribution of urban green spaces to human well-being. Part 2 of this dissertation comprises two papers that consider marine ES and biodiversity. The first paper of part 2 analyzes the implications of environmental valuation for policy-making in a marine context. The second paper of part 2 integrates the non-use value of biodiversity into a dynamic optimization model to analyze how this influences the optimal management of multi-species ecosystems.

Part 1 of this dissertation deals with different aspects of urban ES in a European context. By now, 75% of the European population lives in cities (World Bank, 2013). Their well-being and quality of life depends on a continued flow of ES from ecosystems outside but also within city boundaries. Urban green spaces (UGS) such as urban parks thus are important for city inhabitants, in particular because they provide a range of regulating and cultural ES. Some examples for regulating services in cities are micro-climate regulation, storm water retention, and air quality improvements. The major cultural urban ES provided by UGS is the opportunity for city inhabitants to recreate and experience nature within the city (CBD, 2012; TEEB, 2011; Bolund and Hunhammar, 1999).

The first paper, which is titled “Recreation decisions in urban environments: Evidence from participation and choice models”, is an application of environmental valuation methods to urban environments with a focus on recreation. In particular, it uses choice modeling approaches to elicit the trade-offs people are willing to make when choosing between different urban parks to visit. In addition, it uncovers individual determinants of the participation in recreational activities in urban parks.

The frequent use of UGS and parks for recreational purposes is one important channel through which people benefit from UGS (see Bowler et al. (2010) for a review). Consequently, it is important to investigate which characteristics drive their use, also from a policy perspective. The first paper focuses on the recreation decisions of the inhabitants of the city of Berlin, Germany. It is divided into two parts: The first part addresses the question of which individual characteristics influence the frequency of park use (participation model). The second part addresses the question of which park attributes influence the choice to visit a certain park (choice model). Moreover, we analyze the relationship between individual perceptions of the natural environment and objective environmental indicators, and how both influence park choice. Finally, we derive marginal willingness-to-pay (WTP) estimates for improvements in various park attributes.

There are several ways in which this paper adds to the abundant literature on recreation in urban parks. First, we add a detailed case study for the city of Berlin, highlighting the determinants of participation in park recreation and of park choice. Second, to our knowledge, this is the first study to apply a random utility framework to investigate the relative importance of natural and non-natural park attributes on park choice. Third, we estimate marginal WTP for park attributes based on revealed preference data and travel costs in an urban context. Proposing a novel approach, we calculate the travel cost measure by taking into account the probability of each respondent choosing a certain transport mode. This is important in an urban context in order not to overestimate travel costs. Finally, we show how the perception of the natural environment in urban parks relates to objective environmental indicators and how both aspects influence park choice.

The results of the participation model confirm earlier findings that both the objectively measured and the subjectively perceived availability of UGS increase the frequency of park use. In addition, we find that people from higher socio-economic backgrounds, i.e, with higher income and education, tend to use urban parks more often than people from lower socio-economic backgrounds, even if we control for the availability of UGS. In the choice model, we observe a significantly negative impact of travel costs on park choice. The effects of almost all park attributes have the expected positive sign. This includes non-natural park attributes such as playgrounds and cafés as well as the perceived tidiness and naturalness of a park. The resulting estimates of marginal WTP are small but reasonable. The results from both models open room for policy advice. If it is a policy goal to increase actual use of UGS, for example, because of its positive health effects, then increasing the supply of UGS overall is one important prerequisite. However, it is also necessary to check whether the quality of the green space provided meets the needs of the target population, and whether there are ways to promote outdoor recreation for people from lower socio-economic backgrounds.

The second paper, which is titled "The role of urban green space for human well-being", takes an alternative approach to the valuation of urban ES. We use self-reported information on life satisfaction and different green space measures on the individual level to explore how UGS affect the well-being of the residents of Berlin, Germany, while controlling for a number of socio-economic and demographic characteristics. We combine individual data from an internet survey with spatially highly disaggregated geographical data on UGS. The estimation results are used to calculate marginal rates of substitution (MRS) between income and UGS as an indicator for the value of UGS for well-being.

Economic valuation of UGS has so far mostly been carried out using contingent valuation or hedonic pricing (see, e.g., Brander and Koetse (2011) for a review). The life satisfaction approach (LSA) is an alternative and increasingly popular method to value environmental amenities. It is based on the assumption that environmental (dis)amenities are among the factors that directly determine subjective well-being or life satisfaction. Unlike stated preference methods, this method does not ask people to place a monetary value on a complex environmental good in a hypothetical situation. Survey respondents are not aware of the fact that the answer to a well-being question will be used to value an environmental amenity. Compared to contingent valuation, this may reduce biases resulting from the hypothetical nature of the decision and from potentially strategic behavior.

Examples for applications of the LSA include, e.g., the valuation of air quality, climate, noise, or scenic amenities (see Welsch and Kühling (2009) or Frey et al. (2010) for reviews). Unlike early studies, which look at nationwide or cross-country data sets and suffer from a lack of disaggregated environmental data (e.g., Welsch, 2006), we use highly disaggregated urban land cover data. In addition, there is so far only one study that uses the LSA to value UGS. Ambrey and Fleming (2013) investigate the role of UGS for the well-being of people in major Australian cities. We thus add to the literature by using the LSA to value UGS in a European city. In addition, we test whether data gathered in a small-scale internet survey can be used to employ the LSA.

We observe a significant, inverted U-shaped effect of the amount of and distance to UGS on life satisfaction. According to our results, the amount of UGS within a 1 km buffer around the respondents' residential addresses that leads to the largest positive effect on life satisfaction is 36 ha or 11.5% of the buffer area. As three-quarters of the respondents have less than this amount of UGS available in their living environments, green space is, overall, in insufficient supply in the case study area in Berlin. This also implies positive MRS estimates evaluated at the means of green space area and income. For city management, our results imply that policies should aim at increasing the supply of UGS in areas where they are particularly scarce. Moreover, a more homogenous supply of UGS should be aimed at.

Summing up, part 1 of this dissertation uses different approaches to reveal the value of urban ecosystems and ES for human well-being. The travel cost method is used to show how increasing the quality of urban parks leads to an increase in their recreational value. In addition, the life satisfaction approach is used to infer the direct effects of UGS on human well-being. Both approaches reveal significant values and contributions of urban ecosystems and ES to human well-being.

Part 2 of this dissertation shifts the focus from the consideration of urban ecosystems to marine ecosystems and biodiversity in two further papers. Marine and coastal ES comprise the provision of food such as fish and seafood, as well as regulating services such as climate regulation, water purification, and flood protection. In addition, the ocean provides recreational opportunities, offers inspiration, and aesthetic enjoyment. But in spite of increased awareness, the ocean "remains chronically undervalued, poorly managed and inadequately governed" (GOC, 2014). The services marine and coastal ecosystems provide have received far less attention than those provided by terrestrial ecosystems. This might be due to differences in accessibility and direct experience (TEEB, 2009). In addition, current ocean governance schemes do not ensure sufficient protection of marine biodiversity, and they do not foster the sustainable use of marine living resources (Visbeck et al., 2014).

The third paper, which is the first paper of part 2 of this dissertation, is titled "On the environmental effectiveness of the EU Marine Strategy Framework Directive" (MSFD). The paper focusses on the economic requirements of the MSFD (EU, 2008) and analyzes the role of environmental valuation in a marine policy context. It asks to which degree the requirements of the MSFD to carry out cost-benefit analysis (CBA) can be fulfilled given the current state of knowledge, and it describes potential consequences and problems related to an effective implementation of the MSFD.

From a European policy perspective, increasing threats to the marine environment resulting from human use have been recognized. There are several regulations that aim at managing the human impact on the marine environment. Most recently, the European Union (EU) adopted the MSFD, which is to guide future maritime policy and aims at achieving or maintaining a good environmental status (GES) of Europe's seas by 2020. The MSFD requires an assessment of how humans use the marine environment and the development of action plans including explicit measures to achieve a GES by 2020. Before their implementation, these measures, *inter alia*, need to be assessed by examining their cost-effectiveness and by carrying out CBA.

The aim of this paper is to discuss the challenges of valuing marine ES in the context of the MSFD. The paper contributes to the existing literature by assessing the limitations of environmental valuation and CBA in the marine context and by highlighting the possible consequences: the environmental effectiveness of the MSFD might be hampered and the GES might not be achieved. Existing valuation studies, for example, tend to look at changes in tangible benefits like recreation and food provision but mostly ignore changes in more intangible benefits from, e.g., ecosystem functioning or resilience. However, it might be these services that are more important for sustainable development and societal welfare. A CBA that ignores such services will most likely underestimate the true value of marine ES. Since the costs of improvement measures are easier to determine, this in turn might reduce the probability of measures being implemented.

The fourth paper, which is the second paper of part 2 of this dissertation, is titled "Biodiversity and optimal multi-species ecosystem management". This paper looks at the economics of ecosystems and biodiversity from a different angle. While in environmental valuation, one is concerned with eliciting the non-market values of ES and biodiversity, the last paper in this dissertation answers the question of how the consideration of such non-market values influences the optimal management of a multi-species ecosystem. The paper is set in the context of fisheries economics but the findings can be extrapolated to other cases and ecosystems.

Fisheries management has been forced to deal with declining catch rates and standing stocks in many fisheries all over the world during the last decades. To some extent, it has reacted to these developments by adopting the goal of employing ecosystem-based approaches. This implies that not only economic profits should be maximized but that conservation goals also need to be taken into account (Pikitch et al., 2004). Against this background, we reconsider optimal multi-species ecosystem management in a bio-economic model taking into account both harvesting profit and biodiversity value. More specifically, we analyze how optimal management decisions change when a biodiversity index is introduced in the objective function of a bio-economic dynamic optimization model to capture the value of biodiversity.

Since multi-species applications are increasingly prominent in bio-economic modeling, and biodiversity conservation is high on the international political agenda, it is important to investigate the properties of a bio-economic model when an aggregate biodiversity index based on species abundances is included to capture biodiversity values. We explore the effects of including such an index in a multi-species model of a harvested ecosystem on the optimal steady-state, which has, to our knowledge, not been investigated before. In addition, we exemplify the effects in a more complex age-structured model applied to a predator-prey system of three Baltic Sea fish species (cod, sprat, and herring), and we illustrate the role of the elasticity of substitution for optimal management.

Within the analytical model, we show that extinction is never optimal when a global biodiversity value is taken into account, i.e., if species are imperfect substitutes for one another. Moreover, a stronger preference for species diversity leads to a more even distribution of stock sizes in the optimal steady state, and a higher value of biodiversity increases steady state stock sizes for all species when they are ecologically independent or symbiotic. For a predator-prey ecosystem, the effects may be positive or negative depending on relative prices and the strength of species interaction. In the quantitative application to the Baltic Sea predator-prey system, we find that using stock biomass or stock numbers as abundance indicators in the biodiversity index may lead to opposite results.

To sum up, the four papers of this dissertation analyze different aspects of the economics of ecosystems and biodiversity. A broad array of methodological approaches is used, ranging from intensive literature review and policy analysis to various econometric techniques and optimal control approaches. In addition, this dissertation considers different types of ecosystems, focusing on urban ecosystems as well as marine ecosystems and biodiversity. Taken together, the papers of this dissertation reveal the importance of selected ecosystems for human well-being, uncover the value of non-marketed ES, address the implications of environmental valuation for policy-making, and analyze optimal management when non-market values are integrated in economic decision problems.

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Part I: Valuing urban ecosystem services

Paper 1: Recreation decisions in urban environments: Evidence from participation and choice models

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Abstract:

In this paper, we consider two aspects related to recreation decisions in urban environments applied to the case of Berlin, Germany. First, we investigate the effects of individual green space availability and socio-economic characteristics on frequencies of park use (participation model). Second, we estimate the effects of several non-natural and natural park attributes on park choice using a random utility framework (choice model) and compare the effects of subjective and objective indicators for naturalness. Based on the choice model and travel costs, we calculate marginal willingness-to-pay (WTP) for park attributes using a novel approach to account for the probability of the respondents using different means of transport within the city. The most important results of the participation model are that income and education levels have significantly positive effects on park use, even if the availability of urban green space (UGS) is controlled for, which implies that people from higher socio-economic backgrounds tend to benefit more from the provision of UGS. The results of the choice model demonstrate significantly positive effects for the majority of the non-natural park attributes. In addition, both subjective and objective indicators for naturalness have significantly positive effects on park choice. The resulting WTP estimates are comparable to those of other studies.

Keywords: recreation, travel costs, random utility model, urban green space

JEL classification: Q26, Q51

1 Introduction

Urban green spaces (UGS) provide important ecosystem services (ES) for people living in cities, which are the everyday environment for 75% of Europeans (World Bank, 2013). UGS provide regulating services such as air filtration, micro-climate regulation, and storm-water retention. However, they also provide space for recreation, contributing to the well-being and quality of life of the urban population (TEEB, 2011; Bolund and Hunhammar, 1999). The frequent use of UGS and parks for recreational purposes is thus one important channel through which people benefit. Consequently, it is important to investigate which characteristics drive participation in outdoor recreation in urban environments also from a policy perspective. In this context, James et al. (2009) identify, among others, two important research questions: First, what are the personal and social influences that result in greater use of UGS? And second, what are the necessary quantities, qualities and configurations of UGS that contribute to its regular use?

These two main questions guided the design of the current study, which focuses on the recreation decisions of the inhabitants of Germany's capital city, Berlin. The study is divided into two parts. The first part addresses the question of which individual characteristics influence the frequency of park use. The second part addresses the question of which park attributes influence the choice to visit a certain park. Moreover, we address the question of how the individual perception of the natural environment is related to objective environmental indicators, and how both influence park choice. Finally, we derive marginal willingness-to-pay (WTP) estimates for improvements in single park attributes.

Individual determinants of participation in park-related recreation have been studied before. Examples of studies analyzing the individual determinants of park use from countries other than Germany are from the UK (Dallimer et al., 2014), Denmark (Schipperijn et al., 2010), Finland (Neuvonen et al., 2007), and Turkey (Yilmaz et al., 2007). Some earlier studies, however, only include comparatively few explanatory variables (e.g., Payne et al., 2002). Moreover, there are no studies that evaluate the individual determinants of park use in German cities and relate the findings to the literature. For the particular case of Berlin, there are studies analyzing the availability of UGS within the city (e.g., Kabisch and Haase, 2014), but they do not investigate the individual determinants of park use. While the availability of UGS is one important determinant of actual use (e.g., Schipperijn et al., 2010), individual characteristics provide additional explanatory power. Consequently, it is important to analyze the individual factors that influence the use of UGS and parks beyond their objective availability to find promoting factors and potential barriers to actual use.

Regarding park attributes, there is a broad spectrum of literature regarding the stated importance of park attributes for visitors (e.g., Kabisch and Haase, 2014; Özgüner, 2011; Tyrväinen et al., 2007). However, these studies are mostly descriptive in nature. Other studies focus on physical activity, analyzing how park attributes influence the level of physical activity (e.g., Kaczynski et al., 2008; Giles-Corti et al., 2005), using mostly simple logistic regressions. However, there is almost no study that investigates the influence of park attributes on park choice using a random utility framework in urban contexts. This framework, however, is particularly well-suited when there are many substitute sites, which holds for urban parks. The study by Bullock (2006), which specifically focuses on recreation in urban parks, is the only study in this context we are aware of. He uses a stated choice experiment to identify the relative importance of park attributes for park choice in Dublin, Ireland. Applying the random utility framework to park choice has the potential to reveal the trade-offs people are willing to make and to take into account substitution possibilities between several parks within the city.

Using random utility models (RUM) combined with revealed preference travel cost data to value cultural ES, such as recreational opportunities, has been popular in environmental economics for the last few decades. It would be beyond the scope of this paper to review these studies in detail, but recent reviews and meta-analyses give an overview of existing valuation studies for different contexts (e.g., Ghermandi and Nunes, 2013; Wang et al., 2013). As becomes obvious from these reviews, most studies using RUM approaches refer to rural recreation and do not consider urban contexts. In urban contexts, environmental valuation has relied more on contingent valuation or hedonic pricing methods (see Brander and Koetse (2011) and Perino et al. (2013) for recent reviews). Fleischer and Tsur (2003) is one rare example of a travel cost study that estimates consumer surplus per trip to urban parks based on seasonal trip demand. However, they treat all trips to urban parks as one generic class of trips and do not differentiate between single parks. Kinnell et al. (2006) and Binkley and Haneman (1978) provide related analyses, but they concentrate on the recreation of urban residents outside the city.

There is also a substantial body of literature on UGS and parks regarding aspects such as preferences and perception (e.g., Hofmann et al., 2012; Zhang et al., 2013; Bjerke et al., 2006; Van den Berg and Koole, 2006). Many of these studies analyze how perception influences the stated appropriateness of UGS and parks for recreation (see, e.g., Bjerke et al. (2006) or Zhang et al. (2013) and references cited therein). In addition, some studies analyze how perceptions of the natural environment of lay persons relate to expert judgments (Hofmann et al., 2012). Some studies also analyze how perception relates to objective environmental indicators (e.g., Real et al., 2000), but they are mostly carried out in rural, not in urban, environments (Hofmann et al., 2012). However, to the best of our knowledge, there is no study that includes individual perceptions and objective environmental indicators in a random utility framework to explore how both actually affect park choice in urban environments.

Considering this brief literature overview, we find several ways in which we add to the abundant literature on recreation in urban parks. First, we add a detailed case study for the city of Berlin, highlighting the determinants of participation in park recreation and of park choice. Second, we investigate the individual determinants of participation separately for summer and winter. Third, to our knowledge, this is the first study to apply a random utility framework to investigate the relative importance of natural and non-natural park attributes on park choice and to estimate marginal WTP for park attributes based on revealed preference data and travel costs in an urban context. Proposing a novel approach, we calculate the travel cost measure by taking into account the probability of each respondent’s choice of a certain transport mode. This is important in an urban context in order not to overestimate travel costs.

Finally, we show how the perception of the natural environment in urban parks relates to objective environmental indicators and how both aspects influence park choice.

The remainder of the paper is organized as follows. Section 2 gives an overview of the empirical methods used. Section 3 describes the data used for the analysis, which are derived from an individual survey and complemented with geographical data processed with a geographical information system (GIS). Section 4 presents the results on the determinants of participation in park recreation and park choice as well as WTP for park attributes. Section 5 discusses the results and the approach, and section 6 presents the conclusions.

2 Empirical methodology

2.1 Modeling park use frequencies

We use an ordered logit model to estimate the effects of individual characteristics on park use frequencies.[1] The ordered logit model is built around a latent regression model, which implies that a continuous range of preferences underlies the observed discrete response (Greene, 2012), with:[2]

$$U_i^* = \boldsymbol{x_i}'\boldsymbol{\beta} + u_i \tag{1}$$

The latent, i.e., unobserved variable U_i^* represents the "strength of preferences" (Greene, 2012) or utility that each individual $i=1,\ldots,N$ holds for visiting parks. This utility is influenced by $k=1,\ldots,K$ observable individual characteristics such as age and gender that are contained in the vector $\boldsymbol{x_i}' = (x_{i1}, \ldots, x_{iK})$. Utility is also influenced by a random component, captured in the idiosyncratic error term u_i. Actual utility, U_i^*, cannot be observed. The realized discrete choice, however, can be observed and is used as an indicator for the underlying preferences and utility levels.

[1]Park use frequencies are given in six categories from "never" to "(almost) daily". We thus have realizations in ordered categories such that an ordered logit model is appropriate for the estimation.

[2]It is unclear, *a priori,* how the individual characteristics enter the utility function, but it is conventional to use a linear function, which results in the linear random utility function (1) (Greene, 2012).

For a model with J classes $j=1,\ldots,J$, it is assumed that:

$$U_i = j \quad \text{if} \quad \mu_{j-1} < U_i^* \leq \mu_j \quad \text{for} \quad j = 1, \ldots, J \tag{2}$$

with $\mu_0 = -\infty$ and $\mu_J = \infty$. The parameters μ_j are unknown threshold parameters to be estimated along with the parameter vector $\boldsymbol{\beta}' = (\beta_1, \ldots, \beta_K)$. The probability of individual i choosing class j is given by:

$$Pr(U_i = j|\boldsymbol{x}) = F(\mu_j - \boldsymbol{x}'\boldsymbol{\beta}) - F(\mu_{j-1} - \boldsymbol{x}'\boldsymbol{\beta}) \tag{3}$$

with F being the cumulative distribution function (cdf) of u_i. For the ordered logit model, u is logistically distributed with $F(z) = e^z/(1+e^z)$. The parameters are estimated using maximum likelihood. The parameters do not represent the marginal effects of the individual characteristics on the outcome variable. However, their sign can be interpreted, and it determines whether the latent variable increases or decreases with the individual characteristics. If β_k is positive, an increase in $\boldsymbol{x_{ik}}$ necessarily decreases the probability of choosing the lowest class, while it increases the probability of choosing the highest class.

2.2 Modeling park choice

We use the random utility framework first developed by McFadden (1974) to model the choice of an urban park for recreation. This framework suggests that each individual $i=1,\ldots,N$ chooses between several alternatives $j=1,\ldots,J$ to maximize her utility, U_{ij}. It thus holds that for the chosen alternative, say park j^*, the utility is greater than for all other available parks j, i.e., $U_{j^*} > U_j$ for all $j \neq j^*$. Utility, U_{ij}, is influenced by K observable park attributes, x_{ikj} with $k=1,\ldots,K$, collected in the vector $\boldsymbol{x_{ij}}$, and individual travel costs to each park, c_{ij}; but it is also influenced by an unobservable random component, ε_{ij}:

$$U_{ij} = \alpha c_{ij} + \boldsymbol{\beta}'\boldsymbol{x_{ij}} + \varepsilon_{ij} = V_{ij} + \varepsilon_{ij} \tag{4}$$

Utility can thus be divided into a systematic component, V_{ij}, and an unobservable random component, $\varepsilon_{ij,}$. The random error terms, $\varepsilon_{ij,}$ are assumed to be independently and identically distributed following Gumbel (type 1 extreme value) distributions. This model is called a conditional logit model[3] and can be estimated using maximum likelihood. The regression model estimates the $K+1$ parameters α and $\boldsymbol{\beta}' = (\beta_1, \dots, \beta_K)$ such that the likelihood of the observed pattern of park choices is maximized. The probability that individual i chooses alternative $j*$ is given by:

$$Pr(j^*) = \frac{\exp\left(\alpha c_{ij^*} + \boldsymbol{\beta}' \boldsymbol{x}_{\boldsymbol{ij^*}}\right)}{\sum_{j=1}^{J}(\exp\left(\alpha c_{ij} + \boldsymbol{\beta}' \boldsymbol{x}_{\boldsymbol{ij}}\right))} \tag{5}$$

This probability is given by the exponential of the systematic component of the utility of alternative j^* divided by the sum of the exponentials of the systematic components of the utilities of all alternatives in the choice set. The probability to choose one alternative thus depends on the attributes of the chosen alternative and on the attributes of all other alternatives, such that substitution possibilities are accounted for.

Based on this linear specification of the conditional logit model, the marginal WTP for single attributes, k, per person per visit can be calculated as the simple ratios of the attribute coefficients and the negative cost coefficient:[4]

$$WTP_k = -\frac{\partial U_{ij}/\partial x_{ijk}}{\partial U_{ij}/\partial c_{ij}} = -\frac{\beta_k}{\alpha} \tag{6}$$

Estimating a standard conditional logit model, however, is often not very realistic because the necessary assumption of independence of irrelevant alternatives (IIA) is frequently violated. One alternative is to fit a mixed logit model with random

[3]This term is often used interchangeably with the term multinomial logit model. There are, however, slight differences, as the multinomial logit model incorporates individual-specific characteristics (e.g., age, income), while the conditional logit model incorporates alternative-specific attributes (e.g., size and other site attributes). Individual-specific characteristics can be incorporated into the conditional logit model via interaction terms.

[4]This, of course, only holds if travel costs and other attributes enter the utility function in a linear way. In a non-linear specification, WTP would be a function of the current levels of the attributes.

parameters for some or all of the attributes.[5] The mixed logit model relaxes the IIA assumption and allows for unrestricted substitution patterns. The utility function underlying the mixed logit model changes in that the parameters are not assumed to be fixed anymore but to be individual-specific and randomly distributed as specified, e.g., normally or log-normally (Train, 2003). Equation (4) thus changes as follows in the mixed logit specification:

$$U_{ij} = \alpha_i c_{ij} + \boldsymbol{\beta_i}'\boldsymbol{x_{ij}} + \varepsilon_{ij} \tag{7}$$

Collecting all random parameters in the vector $\boldsymbol{\delta_i}$,[6] these parameters are allowed to vary according to the following equation:

$$\boldsymbol{\delta_i} = \boldsymbol{\delta} + \boldsymbol{L}\boldsymbol{v_i} \tag{8}$$

where $\boldsymbol{\delta}$ is a constant vector, and $\boldsymbol{v_i}$ is a vector of random variables with $\boldsymbol{L}$ being the corresponding lower triangular Cholesky matrix defining the standard deviations and covariances of $\boldsymbol{\delta_i}$. Assuming without loss of generality that Var[$\boldsymbol{v_i}$]=$\boldsymbol{I}$, it follows that Var[$\boldsymbol{\delta_i}$] $= \boldsymbol{LL'} = \boldsymbol{\Sigma}$. The mixed logit model can be estimated by simulating the log-likelihood function rather than direct integration to compute the probabilities.[7]

In the mixed logit model, the WTP estimates are no longer straight-forward if at least one of the parameters used for their computation is assumed to be random. In that case, the mean expected WTP and related confidence intervals can be computed using simulation methods such as the delta method or the method developed by Krinsky and Robb (see Hole (2007b) for an overview of the applicable simulation methods).[8] If the travel cost parameter is considered as random in addition to the parameters of the other attributes, WTP is ultimately the ratio of two randomly

[5]The mixed logit model is also often called the random parameters logit model (Greene, 2012).

[6]If, e.g., all explanatory variables were assumed to be random, it would follow that $\boldsymbol{\delta_i}' = (\alpha_i, \beta_1, \dots, \beta_K)$.

[7]See Hole (2007a) for a description of how the mixlogit command can be used to fit mixed logit models using Stata and for details on how simulation is implemented.

[8]These methods are based on the unconditional population distributions of the parameters. It is also possible to compute mean WTP and confidence intervals based on the individual parameters, which implies using conditional distributions in which known choices are taken into account. See Hensher et al. (2006) for a discussion of the differences between these methods.

distributed variables, e.g., of two normally distributed variables. Consequently, the distribution of WTP may no longer have well-defined moments, which can result in unreasonably large WTP estimates (Greene, 2012).

One solution to this problem is to consider the travel cost parameter as fixed and only allow other attribute parameters to be random. As has been noted in the literature, however, this is often not realistic, as heterogeneity in preferences for money can be expected across a population, even beyond observable socio-economic characteristics (see, e.g., Scarpa et al. (2008) or Hole and Kolstad (2012)). One possibility to cope with this issue is to present several sets of estimations, comparing WTP estimates and confidence intervals derived from regressions with fixed and random travel cost parameters to control for unrealistically high WTP estimates and large confidence intervals (see Doherty et al. (2013) for an example).

3 Case study city and data

3.1 Case study city Berlin

Berlin is the capital city of Germany. It is a federal city state located in the east of Germany, and it forms the center of the metropolitan area of Berlin-Brandenburg. Berlin covers an area of 892 km^2 (SSUB, 2013), of which 7.4% is covered with urban green spaces, 17.5% is forests, 7.2% is agricultural areas and 5.6% is rivers and lakes (see Figure 1; based on EEA, 2012). Overall, 37.7% of the city is thus covered by natural areas. Most of the forest, agricultural, and water areas, however, are located in the outer districts of the city, while UGS are spread over the whole city. Berlin had a total population of 3.50 million as of December 2011 (ASBB, 2012). The population is currently increasing and is estimated to peak at approximately 3.76 million in 2030, mainly due to medium-term positive net in-migration (SSUB, 2012a).

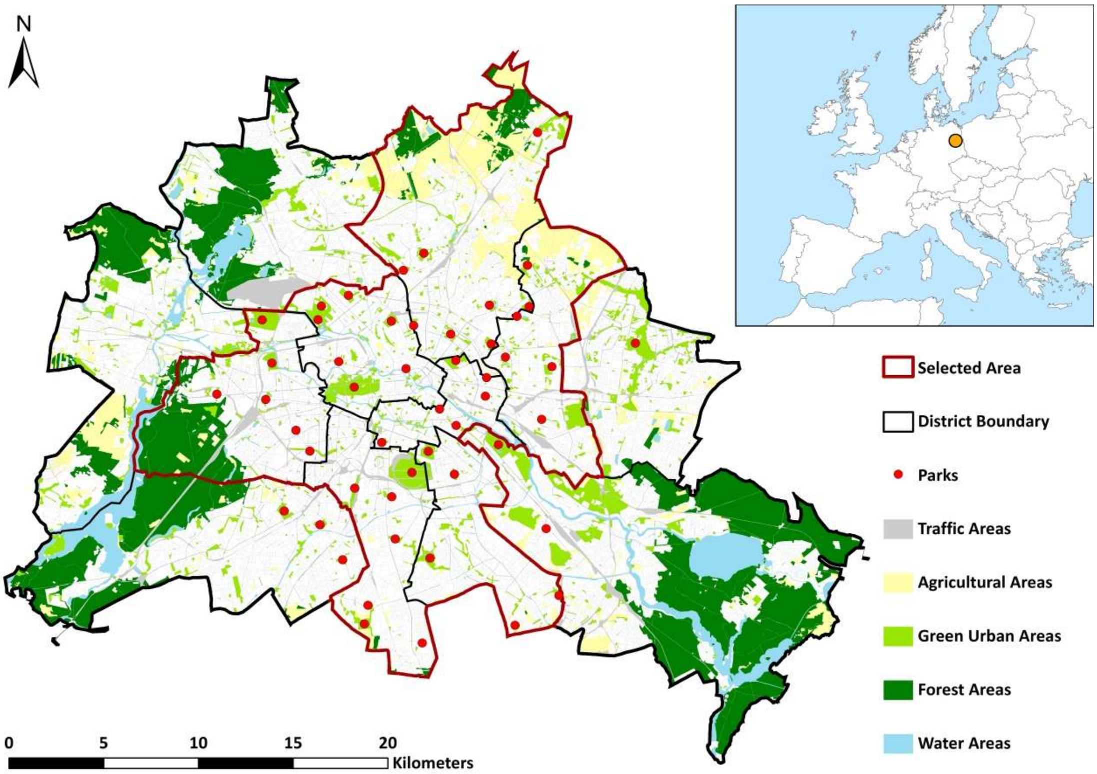

Figure 1. Distribution of urban green spaces and other natural and semi-natural areas in Berlin. Own presentation based on Urban Atlas data (EEA, 2012).

3.2 Individual-level data

3.2.1 Survey data

Most of the individual-level data used for our analysis originate from a web survey carried out in Berlin, Germany, in September 2012. One objective of the survey was to investigate people's preferences for recreation in urban parks. The survey included a number of questions on park use patterns such as frequency of park visits in summer and winter, activities carried out and perception of the visited parks. In addition, the survey included questions on socio-economic and demographic characteristics of the respondents as well as housing and neighborhood characteristics.

The survey was implemented and executed by a professional polling agency using a pre-selected web panel with members aged 18 years or above. Potential survey participants were screened to ensure that they had been living in Berlin for more than one year. Only residents of the districts of Mitte, Kreuzberg-Friedrichshain, Pankow, Charlottenburg-Wilmersdorf, Tempelhof-Schöneberg, Neukölln, and Lichtenberg were included in the survey (see Figure 1).[9] The final sample consists of 485 usable observations (see Bertram and Rehdanz (2014) for more details on the survey).

Respondents were asked to provide their residential address to enable us to link the survey data with geographical data on UGS and parks. Of the 485 usable observations, 11% provided their full address, and 67.2% provided at least the name of their street (see also Bertram and Rehdanz, 2014). We limit our analysis to these respondents and do not include respondents who gave more ambiguous information regarding their residential address to reduce measurement error in the GIS data while keeping a sample size that is not too small. Respondents were also asked to indicate whether there was a park within walking distance of their home to have a subjective indicator for the perceived availability of UGS and parks in the living environments of the respondents. Table A-1 in Appendix A reports descriptive statistics of individual-level data of the final sample included in the ordered logit regression in section 4.1.

3.2.2 Land cover data

In addition to the perceived individual availability of UGS, we calculate the objective availability of UGS based on spatial land cover data from the Urban Atlas (EEA, 2012). This database provides pan-European comparable land use and land cover data for large urban areas with more than 100,000 inhabitants at a 50x50 m grid level

[9]These districts were selected because they include the densely populated inner-city districts of Berlin with a relatively homogenous distribution of green space. The districts were also selected to be comparable with the whole population of Berlin with regard to age and gender and to have a balanced distribution between the formerly Eastern and Western German parts of the city of Berlin.

(EU, 2011). We use the areas designated as "green urban areas"[10] as an indicator for individual green space availability for the analysis in section 4.1.

To calculate the individual availability of UGS, we created buffers with a diameter of 1 km around the respondents' residential addresses and calculated the amount of green space within the buffer area for each respondent. This gave us an objective measure of green space availability at the respondent level. We excluded observations in the 1st and 99th percentiles of the variable "urban green space" from our sample to mitigate the influence of outliers. This led to the exclusion of 10 observations from the sample. The total area for the buffer with a 1 km radius is 314.2 ha (or approximately 3.14 km^2). Individual green space availability is between 0.5% and 31.2% of this buffer area, i.e., between 1.1 ha and 98.1 ha. Based on the same Urban Atlas data set, we also calculated the distance from each respondent's individual residential address to the nearest UGS larger than 5 ha. The mean distance is 656 m, ranging from a minimum of 0.5 m to a maximum of 2,495 m.

3.3 Park-level data

3.3.1 Park facilities

The random utility park choice models estimated in section 4.2 of this paper require knowledge about the different parks that are visited by the residents of Berlin. Based on data availability, the final choice set of urban parks in Berlin considered in the choice model in section 4.2 contains 50 parks ranging in size from 1.2 to 258.5 ha. Table B-1 in Appendix B gives an overview of the 50 parks that form the choice set.

One important aspect for the decision on outdoor recreation in urban parks besides their natural quality is the equipment of these parks with facilities such as playgrounds or sport facilities (e.g., Kaczynski et al., 2008). We carried out an intensive desktop research to identify which facilities are present in which of the 50

[10]This land use category includes public green areas for predominantly recreational use such as gardens, zoos, parks, and castle parks. Not included are private gardens within housing areas, cemeteries, buildings within parks, such as castles or museums, patches of natural vegetation or agricultural areas enclosed by built-up areas without being managed as green urban areas. We also include the lawns belonging to the former Tempelhof airport, which are now used for recreation, in the measure of urban green space. This was not the case in the original data set (EU, 2011).

parks in the choice set. This included internet-based research analyzing information provided by the City of Berlin but also the inspection of other web sites including Google Earth. The facilities included in the final analysis are picnic areas, barbecue areas, playgrounds, and sport facilities, as well as restaurants and cafés (see Table D-1 in Appendix D).

3.3.2 Biotope values

A second important aspect for the choice to visit a park is its natural quality. It is very difficult to find objective, disaggregated data at the park level to describe their natural quality or, e.g., their biological diversity. The data we use in our analysis to account for the natural quality of the parks are so-called biotope values. These values are based on a comprehensive biotope mapping carried out by the City of Berlin. The biotope mapping envisages a detailed description of landscapes via distinguishable and separable biotope types. The current biotope mapping documents the present state and distribution of particularly valuable biotopes in Berlin (SSUB, 2012b).[11]

The biotope values are calculated based on the different biotope types. The calculation takes into account base factors, including human impact on the natural environment (hemeroby), the presence of endangered species, the scarcity of the biotope type, and the diversity of animal and plant species. Also included are risk factors, including the time needed to restore species communities and the potential to restore abiotic conditions of the habitat. The biotope values in the visited parks range from one to 60. The variability of the biotope values (measured by their standard deviation in each park) ranges from zero to 18.6 (see Table D-1 in Appendix D).

[11]The biotope mapping is based on data from 2003 to 2012. The latest update is from June 2012. The biotopes of all forest areas, Natura 2000 areas, other protected areas and other particularly valuable natural areas in Berlin were mapped via visual inspection of the areas. Other areas not covered by forests were mapped by inspection of aerial photographs, or existing secondary data were used and transposed into biotope types (SSUB, 2012b).

3.3.3 Individual perception of park attributes

One aim of this paper is to contrast the effect of subjective indicators for the natural quality of urban parks with objective indicators. We thus include in the analysis information on the individual perception of the natural quality of urban parks derived from the survey in addition to objective information. Moreover, it is not possible to gather objective data on all attributes that influence park choice, which is why we also include information on the perception of several non-natural park attributes.

Approximately two-thirds of the respondents in Berlin visit parks at least once a month in summer and in winter.[12] These regular park visitors were asked which park was most important for their leisure time activities and how they perceived the park regarding a number of natural and non-natural attributes. The attributes had to be rated on a 4-point scale, from “does not apply at all” to “does fully apply”. This procedure, however, only produces information on the perception of the chosen park and not on the non-chosen alternatives. In addition, information on single attributes of the chosen park can be missing.

To fill missing values and observations on non-chosen alternatives, we averaged the individual ratings of the different park attributes over all respondents who chose the same park to be able to assign one value per attribute to each of the parks. This procedure was necessary given that it was not possible to gather a rating for each of the attributes for each of the 50 parks from each of the respondents. Although not perfect, this strategy is frequently followed to generate data that can be used in choice modeling approaches with revealed preference data (Hensher et al., 2005). For our data, we observe that for the majority of parks and attributes, the perceptions are very similar for those people who visited the same parks and gave a rating. Consequently, we are confident that the average ratings are good proxies for overall perception and the actual state of a park.

[12]We asked respondents to indicate their average number of park visits differentiated by seasons. “Summer” was framed as the last summer semester, lasting approximately from April 2012 to September 2012, and “winter” was framed as the last winter semester, lasting approximately from October 2011 to March 2012.

We then carried out a factor analysis to extract the main factors that influence park choice. The results of the factor analysis can be found in Appendix C. The factor analysis revealed three factors with eigenvalues larger than one. The first factor can be called naturalness and is influenced by scenic beauty, biological diversity, a natural design, and a varied landscape. The second factor is influenced by perceived tranquility and cleanliness as well as by a low density of visitors, and it can be called tidiness. The third factor is slightly more diverse. It is influenced not only by good accessibility and reachability of the park but also by low crime and good opportunities to meet people. We call this factor convenience. Descriptive statistics of the three factors over the 50 parks can be found in Appendix D.

3.4 Travel costs

Travel costs are needed as an explanatory variable in the park choice model in section 4.2 to estimate the marginal WTP for single park attributes. Travel costs are quite sensitive to the mode of transport chosen, particularly within the city, where much travel can be assumed to take place by walking, which is basically costless. In our sample, approximately 51% of the respondents get to their favorite park on foot, 21% use public transport, 19% go by bike, and 9% go by car. However, it has been observed in the literature that transport mode choices depend, among other things, on urban form and structure, distance to be travelled, and socio-economic and demographic characteristics of the respondents (see Heinen et al. (2009) for a review of the literature on travel mode choice for bicycle commuting). It can thus be assumed that respondents would choose different modes of transport depending on the distance to the park they intend to visit. However, we do not observe the mode of transport for the non-chosen alternatives in our choice set.

As a solution to this problem, we propose a novel approach, i.e., to calculate travel costs based on each respondent's probability to choose a certain mode of transport. To this end, we estimate a multinomial logit model with transport mode as the dependent variable and several individual demographic and socio-economic characteristics as well as distance to the favorite park as explanatory variables to determine the influence of these characteristics on transport mode choice.[13] The results of this regression can be found in Appendix E. We find that distance is the most important predictor of transport mode choice. We use this information, i.e., the estimated parameters, to predict the probability of each respondent to get to each park in the choice set using a certain transport mode depending on her individual characteristics and the actual distance to each of the parks.[14] We then weigh the costs of the different transport modes with the respective probabilities.[15] For car travel, we assume a cost of 0.30 € per kilometre; for bike travel, we assume a cost of 0.10 € per kilometre; and for public transport, we assume a lump sum of 5.20 € per round trip. The cost of walking is assumed to be zero.[16]

Travel costs are then calculated based on the distance from the respondents' residential addresses to all the parks in the choice set. The simple Euclidean distance ranges from 87 m to 15 km over all respondents and parks in the choice set. The simple Euclidean distance is multiplied by two to account for the route to the park and back home. We add entrance fees to the travel costs for those parks where such a fee is levied. The resulting direct travel costs in the sample range from 0.05 € to 12.94 € over all respondents and all parks.[17]

In addition to these direct trip costs, travel costs can also include opportunity costs of time. There is a long-standing debate in the literature on whether opportunity costs of time should be included in travel costs, a debate that goes back to an article by Clawson and Knetsch (1966) and remains unresolved today (Phaneuf and Smith, 2005). In an urban context, distances tend to be relatively small such that travel costs are sensitive to adding opportunity costs of time. On the one hand, one argument for

[13]The estimation is based on the random utility framework described in section 2.2, only that the travel mode is now influenced by individual-specific characteristics and not by alternative-specific attributes.

[14]Probabilities are calculated in analogy to the way described in section 2.2 for the site choice model using equation (5).

[15]See section 4.2.3 for sensitivity analyses with respect to different assumptions on travel costs.

[16]Costs for car travel are based on ADAC (2013) and the German tax law, which allows tax reimbursements for an amount of 0.30 € per kilometer for travel to work. Costs for bike travel are based on Brühbach (2009), who calculates costs between 0.03 € and 0.12 € per kilometer based on depreciation and annual maintenance costs depending on annual distance travelled.

[17]Note that when using the probability-weighted travel cost measure, travel costs are always larger than zero because the probability to walk is never equal to one.

not including opportunity costs of time would be that travel time can be part of recreation if one enjoys, for example, walking to the park or going there by bike. On the other hand, recreation in urban parks tends to be an activity that is pursued in day-to-day life, where time is scarcer for working people than on weekends, when more time is already planned for recreational activities. This would, to our mind, call for including opportunity costs of time, particularly in an urban context.

We decided to present two sets of results, one for travel costs excluding opportunity costs of time and one including them, to show the sensitivity of our results regarding the chosen travel cost measure. Opportunity costs of time are calculated based on an hourly wage rate of one-third of net individual income for those people who work full-time or part-time. While this is an ad-hoc assumption, it has been used in many applications and has been shown to yield reasonable results (Parsons, 2013). For people who are employed full-time, we assume that the number of working hours per year is 1920, which is standard in the literature, while we assume that it is half this amount for part-time workers. Travel time is calculated based on Euclidean distance assuming a speed of 4 km/h for walking, 15 km/h for biking and using public transport, and 25 km/h for going by car.[18] Travel times are also weighed by the probabilities to choose different modes of transport and are doubled to account for traveling to the park and back. The resulting travel costs, including opportunity costs of time in the sample, range from 0.05 € to 38.57 € over all respondents and all parks.

4 Results

4.1 Determinants of participation in park recreation

In this section, we analyze the determinants of participation in park recreation. We investigate how individual socio-economic characteristics influence park use in Berlin. In addition, we look at the impact of the availability of parks and UGS on park use. Figure 2 gives an overview of park use frequencies in Berlin divided by

[18]Average travel times per transport mode are chosen based on observations of travel time in the survey as well as average transport times assumed by Google maps for travel in the city of Berlin.

visits in summer and winter. As expected, use frequencies are higher in summer than in winter. In summer, most respondents use urban parks one to three times a month, while this decreases to less than once a month in winter. However, the share of respondents who use parks at least once a week, which amounts to 35% in summer, still amounts to 22% in winter. This indicates that urban parks are important for recreation even during the colder season.

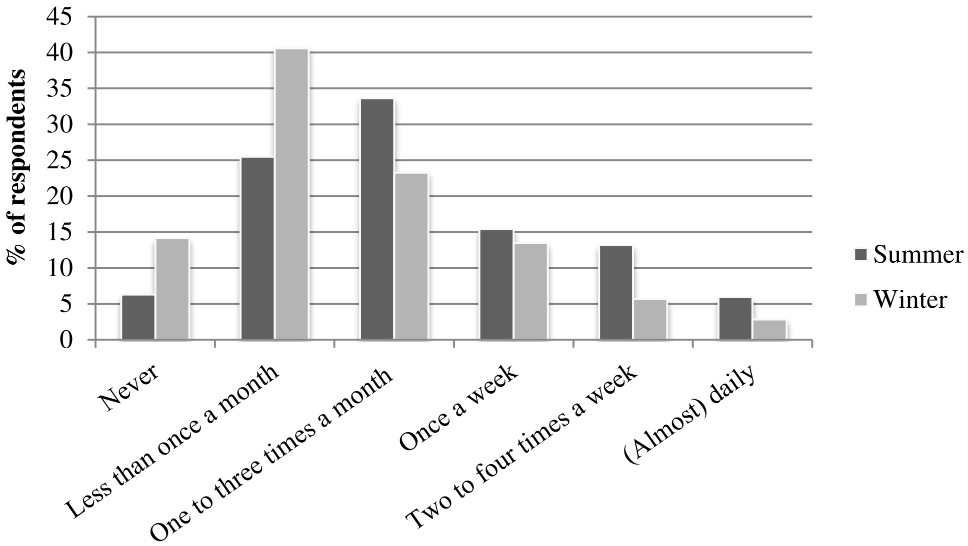

Figure 2. Frequency of park use in summer and winter.

In the following analysis of the determinants of park use frequencies, we use different indicators to represent the individual availability of UGS and parks. This includes two objective measures: i.) the amount of UGS in the living environment of the respondent, and ii.) the distance to the nearest UGS larger than 5 ha. We are also interested in the question of whether the effect of these objective measures is mirrored by the effect of a subjective measure. We thus also include a dummy variable that captures whether respondents find that there is a park within walking distance of their home in the regression.[19]

[19]Walking distance was not further specified in the survey, yet correlation between the objective and subjective measures is substantial. Consequently, the three variables are included one by one in different sets of regressions.

The following regression analysis is carried out using a multivariate ordered logit model as described in section 2.1. We assume a linear random utility function (see equation (1)). Error terms are clustered on the district level to account for the fact that we combine variables on the individual level with variables on the district level in the estimation (Moulton, 1990). The results are summarized in Table 1.

Effects of environmental characteristics

The results show that the availability of UGS has a significantly positive effect on park use frequencies. First, the amount of UGS in the living environment of the respondents positively influences their probability to use parks more frequently (models 1a and 1b). Using smaller buffer areas, i.e., the amount of UGS in buffers with radiuses of 500 m and 300 m instead of a buffer with a radius of 1000 m, the effect of green space availability on park use frequencies is still positive and significant. Effect sizes increase with smaller buffer areas, but significance levels decrease.[20] Second, the distance to UGS shows the corresponding significantly negative effect (models 2a and 2b).[21] This confirms earlier research that has also found significantly negative relationships between the distance to and use of UGS (e.g., Schipperijn et al., 2010). Third, the effects of the objective measures are mirrored in the effect of self-reported distances to urban parks, which is significantly positive (models 3a and 3b).

Regarding the differences between summer and winter, there is no clear evidence to conclude whether the availability of green spaces is more important for determining park use frequencies in either season. For the objective measures, the effects are slightly stronger in winter than in summer. This could indicate that a sufficient supply of UGS is particularly important to foster the use of green spaces in the colder season. However, the effect of the subjective measure is smaller in winter than in summer, which would be in contrast to this hypothesis.

[20]Results are not shown but are available from the authors upon request.

[21]Inverse distances are not significant. Using squared distances, the effect is strongly negatively significant, but the model fit, as judged by Pseudo R^2 as well as AIC and BIC, decreases compared to the corresponding model using simple Euclidean distances.

Table 1. Determinants of participation in park recreation (ordered logit).

Frequency of park use	**Model 1a Summer**	**Model 1b Winter**	**Model 2a Summer**	**Model 2b Winter**	**Model 3a Summer**	**Model 3b Winter**
Environmental characteristics						
Urban green space (ha in 1 km buffer)	0.0219*** (0.01)	0.0289*** (0.00)				
Distance to nearest UGS > 5 ha (in 100 m)			-0.0975*** (0.03)	-0.1088*** (0.02)		
Park in walking distance					0.7136*** (0.26)	0.5978** (0.28)
Individual characteristics						
Gender (male)	0.0766 (0.11)	-0.0697 (0.10)	0.1118 (0.12)	-0.0394 (0.11)	0.1034 (0.07)	-0.0224 (0.10)
Migration background	-0.0903 (0.27)	0.0039 (0.24)	-0.0224 (0.26)	0.0909 (0.23)	-0.0257 (0.30)	0.0761 (0.24)
Age	-0.0292*** (0.01)	-0.0136 (0.01)	-0.0296*** (0.01)	-0.0141 (0.01)	-0.0278*** (0.01)	-0.0112 (0.02)
Bad or very bad health	Reference	Reference	Reference	Reference	Reference	Reference
Fair health	-0.1228 (0.23)	0.2059 (0.21)	-0.1121 (0.28)	0.1820 (0.20)	-0.1364 (0.27)	0.1997 (0.25)
Good health	-0.1511 (0.42)	0.0850 (0.34)	-0.1051 (0.39)	0.1324 (0.32)	-0.2034 (0.38)	0.0341 (0.31)
Very good health	1.0230*** (0.32)	1.2721*** (0.22)	0.9080*** (0.35)	1.0669*** (0.24)	0.7609** (0.34)	1.0101*** (0.27)
Single	Reference	Reference	Reference	Reference	Reference	Reference
Married	-0.3181 (0.23)	-0.0947 (0.33)	-0.3213 (0.21)	-0.0685 (0.36)	-0.2644 (0.19)	-0.0385 (0.31)
Partner	-0.3323** (0.16)	-0.3561* (0.21)	-0.3322** (0.15)	-0.3171 (0.20)	-0.3430*** (0.12)	-0.3250** (0.16)
Separated, divorced, or widowed	-0.4954 (0.43)	-0.1377 (0.53)	-0.5366 (0.43)	-0.1749 (0.49)	-0.5499 (0.47)	-0.1728 (0.54)
Individual income	0.0004** (0.00)	0.0005*** (0.00)	0.0003** (0.00)	0.0004*** (0.00)	0.0003** (0.00)	0.0004*** (0.00)
Visits of other green areas in the city	0.5210*** (0.08)	0.5213*** (0.06)	0.5192*** (0.08)	0.5182*** (0.05)	0.5409*** (0.08)	0.5308*** (0.06)
Household characteristics						
Child	0.1587 (0.18)	0.1236 (0.31)	0.2525 (0.16)	0.3316 (0.28)	0.2076* (0.12)	0.2865 (0.25)
Detached, semi-detached, or terraced house	Reference	Reference	Reference	Reference	Reference	Reference
Small apartment building	1.0898** (0.53)	1.2628*** (0.45)	1.0884** (0.51)	1.2379*** (0.45)	0.8478 (0.54)	0.9565** (0.45)
Large apartment building	0.6590 (0.44)	0.7618** (0.38)	0.6569 (0.50)	0.7488 (0.47)	0.4424 (0.41)	0.5117 (0.33)
High rise	0.5341 (0.61)	0.5652 (0.47)	0.6195 (0.68)	0.6365 (0.60)	0.4039 (0.63)	0.4127 (0.45)

Table 1. Determinants of participation in park recreation (ordered logit) (continued).

Frequency of park use	**Model 1a Summer**	**Model 1b Winter**	**Model 2a Summer**	**Model 2b Winter**	**Model 3a Summer**	**Model 3b Winter**
District controls						
Mitte	Reference	Reference	Reference	Reference	Reference	Reference
Friedrichshain-Kreuzberg	0.1227	0.1960	0.0022	0.0272	0.0517	0.0753
	(0.13)	(0.14)	(0.11)	(0.14)	(0.13)	(0.13)
Pankow	-0.4118***	-0.3990***	-0.4362***	-0.4770***	-0.5619***	-0.5894***
	(0.08)	(0.05)	(0.08)	(0.05)	(0.08)	(0.05)
Charlottenburg-Wilmersdorf	-0.7195***	-0.5104***	-0.8856***	-0.6829***	-0.9741***	-0.8390***
	(0.23)	(0.13)	(0.20)	(0.11)	(0.19)	(0.12)
Tempelhof-Schönefeld	0.2262***	0.1863***	0.2296**	0.1550***	-0.0093	-0.1005**
	(0.09)	(0.04)	(0.10)	(0.06)	(0.07)	(0.04)
Neukölln	-0.7628***	-0.5618***	-0.8421***	-0.6707***	-0.8003***	-0.6243***
	(0.14)	(0.12)	(0.16)	(0.14)	(0.14)	(0.12)
Lichtenberg	-0.8910***	-0.7021***	-1.0576***	-0.8570***	-0.8766***	-0.6514***
	(0.16)	(0.12)	(0.19)	(0.14)	(0.12)	(0.10)
Number of observations	317	317	317	317	317	317
Pseudo R^2	0.1105	0.1117	0.1110	0.1055	0.1072	0.0947
AIC	922.4612	865.8601	921.8689	871.8721	925.8419	882.1636
BIC	945.0146	888.4135	944.4223	894.4255	948.3953	904.7170

Clustered standard errors in parentheses. Cutpoints omitted. * p<0.1, ** p<0.05, *** p<0.01.
AIC: Akaike Information Criterion; BIC: Bayesian Information Criterion.

Effects of individual characteristics

Regarding the effects of individual characteristics, we observe a negative effect of age on park use frequencies. This age effect, however, is only significant for park use in summer but not for park use in winter. This could indicate that there is greater divergence between the recreational behavior of younger and older people in summer than in winter. Younger people seem to visit parks significantly more often than older people in summer, while in winter, younger and older people visit parks with comparable frequencies. The effect occurs even though we control for health, and it remains significant if the health dummies are excluded from the regressions. Some earlier studies have found that physical activity levels decrease with increasing age (Schipperijn et al., 2013; Payne et al., 2002), but others show no clear pattern in the relationship between age and park use (Schipperijn et al., 2010; Neuvonen et al., 2007). One reason for this could be that these studies did not differentiate between park use in summer and winter.

Regarding the effects of health, we find that people with very good health use parks significantly more often than people with bad or very bad health. The effect of health is stronger in winter than in summer suggesting that people with better health are more likely to use parks more frequently in winter. While this seems intuitive, it could also be that there is reverse causation in the health variable, such that people who use parks more often and in winter benefit from it in terms of better health. To check for any confounding effects, we also ran the whole set of regressions without the health variables. The effects of all other variables are constant both in terms of effect sizes and significance levels. Regarding marital status, there is a significantly negative impact of living with a partner compared to being single. Other marital statuses do not show a significant influence on park use frequencies compared to being single. We also do not observe significant effects for gender or migration background on the frequency of park use.

In addition, there is a small but strongly significantly positive effect of income on park use frequencies. Given that the individual availability of UGS is already controlled for in the regressions, this indicates that people with a better socio-economic background profit more from UGS because they use them more. This is confirmed when replacing the income variable with dummy variables for education. These regressions show that people with tertiary education use parks significantly more frequently than people with basic education, even after controlling for the individual availability of UGS (results not shown). This finding is in line with findings from a Danish (Schipperijn et al., 2010) and a Turkish (Yilmaz et al., 2007) study, so it does not solely reflect circumstances in the case study city Berlin. In contrast, the positive effect of education on park use is not significant in a Finnish case study (Neuvonen et al., 2007).

Finally, it can be observed that the frequency of visits to other natural areas in the city has a positive effect on park use. This indicates that urban parks and green spaces are not necessarily substitutes for other urban green areas such as forests. Instead, they seem to complement one another such that people with stronger preferences for outdoor recreation use different types of green areas in their leisure time. People thus seem to seek diversity in the ways they spend their leisure time.

Effects of household characteristics

Regarding household characteristics, almost no significant effect can be observed for having a child under the age of 12 in the household. This effect is only positively significant in model 3a. Turning to the housing variables, we observe that people living in small apartment buildings are more likely to use parks than people living in detached, semi-detached, or terraced houses. There is also a positive effect for large apartment buildings, but it is only significant in model 1b. For high rises, there is no significant effect in any of the model specifications. The housing effect is stronger in winter than in summer, suggesting that the need of public green space for people living in apartments increases in the colder season or that people living in detached, semi-detached or terraced houses decrease their use of public UGS in winter while people living in apartments maintain their level of use. It could have been expected that people living in high rises or larger apartment buildings visit parks significantly more frequently, *ceteris paribus*, than people living in detached houses. This is not the case, however, which could be another hint for the finding that people from lower socio-economic backgrounds do not profit as much from UGS as people from higher socio-economic backgrounds.

Effects of district controls

District dummies indicate that people living in Mitte are significantly more likely to use urban parks than the respondents living in all other districts except Friedrichshain-Kreuzberg, which is close to Mitte and exhibits similar characteristics. This effect can be explained by population density. Replacing the district dummies with a variable that captures population density on the district level shows that population density positively influences park use frequencies (results not shown). This underlines that urban parks are of particular importance for the people living in the densely populated inner city districts of Berlin.

4.2 Determinants of park choice

In this section, we present the results of the park choice model described in section 2.2. We analyze which park attributes are particularly important for park choice and how natural park attributes are traded off against non-natural park attributes. We also compare the effects of objective indicators for the natural quality of the parks with the effects of their individual perception. The results of the choice model presented in section 4.2.1 are based on a mixed logit specification with uncorrelated random coefficients. In section 4.2.2, we derive marginal WTP estimates for single park attributes based on the mixed logit specifications.

4.2.1 Results of the mixed logit park choice model

We fit a mixed logit model with uncorrelated coefficients of the random variables to estimate the effects of natural and non-natural park attributes on park choice (Table 2). The random coefficients were chosen after repeated model runs with different model specifications. These revealed that there is no preference heterogeneity with respect to the presence of picnic areas or barbecue areas in the sample. In addition, preference heterogeneity could not be found for the perceived tidiness and convenience of the park or for natural quality expressed either as the variability of the biotope value, or the perceived naturalness of the park. Thus, these variables are considered to have fixed coefficients in the final model specifications.

In contrast to this, there is evidence for strong preference heterogeneity with respect to the presence of cafés/restaurants and with respect to travel costs. There is mixed evidence for preference heterogeneity with respect to the presence of playgrounds and sport facilities. We allow these coefficients to be random to account for potential preference heterogeneity in some of the specifications. All random coefficients are given normal distributions in the final model specifications. The model is estimated using 100 Halton draws.

Table 2. Results of mixed logit park choice models.

Park choice	**Opportunity costs of time excluded**			**Opportunity costs of time included**		
	Model 4a	**Model 4b**	**Model 4c**	**Model 5a**	**Model 5b**	**Model 5c**
Mean						
Travel costs 1	-2.0095***	-1.9586***	-1.9950***			
	(0.23)	(0.21)	(0.21)			
Travel costs 2				-1.2766***	-1.2612***	-1.3007***
				(0.12)	(0.12)	(0.12)
Picnic area	0.4865**	0.3802*	0.2944	0.5778***	0.5054**	0.3946*
	(0.21)	(0.22)	(0.21)	(0.20)	(0.20)	(0.21)
Barbecue area	0.6458***	0.8074***	0.6338**	0.6107**	0.7755***	0.6073**
	(0.24)	(0.25)	(0.25)	(0.24)	(0.25)	(0.25)
Playground	1.2800**	1.2502*	1.6794***	1.2848**	1.2509**	1.6920***
	(0.59)	(0.64)	(0.62)	(0.54)	(0.57)	(0.57)
Sport facilities	0.6528*	0.3821	0.3777	0.7329**	0.5210*	0.4696
	(0.34)	(0.30)	(0.30)	(0.33)	(0.30)	(0.30)
Café/restaurant	4.1867***	4.0809***	4.5010***	3.5166***	3.4032***	3.8856***
	(1.07)	(1.01)	(1.11)	(0.86)	(0.84)	(0.86)
Tidiness	0.4022***	0.3474**	0.3400**	0.4830***	0.4378***	0.4121***
	(0.14)	(0.14)	(0.14)	(0.14)	(0.14)	(0.15)
Convenience	-0.2726	-0.2490	-0.2261	-0.2895*	-0.2841*	-0.2507
	(0.17)	(0.17)	(0.17)	(0.16)	(0.17)	(0.17)
Naturalness		0.3408***			0.3601***	
		(0.12)			(0.12)	
Biotope value			0.1161***			0.1243***
(std. dev.)			(0.02)			(0.02)
Standard deviation						
Playground	1.1299	1.1759	1.2036	1.2553	1.3412*	1.3040*
	(1.00)	(1.15)	(0.98)	(0.77)	(0.81)	(0.75)
Sport facilities	1.3339	0.6879	0.9739	1.5810*	1.2031	1.2980
	(1.00)	(1.35)	(1.09)	(0.83)	(0.86)	(0.80)
Café/restaurant	3.7510***	3.6832***	4.1207***	2.9852***	2.8610***	3.3659***
	(1.12)	(1.07)	(1.19)	(1.01)	(1.00)	(0.94)
Travel costs 1	1.0589***	1.0055***	0.9763***			
	(0.22)	(0.20)	(0.19)			
Travel costs 2				0.5528***	0.5437***	0.5479***
				(0.08)	(0.08)	(0.08)
Number of obs.	211	211	211	211	211	211
Log likelihood	-456.29	-452.00	-444.64	-447.86	-443.24	-434.61
AIC	936.58	930.00	915.27	919.72	912.47	895.21
BIC	1023.75	1024.43	1009.70	1006.89	1006.90	989.64

Standard errors in parentheses. * $p<0.1$, ** $p<0.05$, *** $p<0.01$.
"Travel costs 1" exclude opportunity costs of time; "travel costs 2" include opportunity costs of time.
AIC: Akaike Information Criterion; BIC: Bayesian Information Criterion.

Effects of travel costs

Travel costs have a strong and significantly negative effect on park choice in all model specifications. It can be observed, however, that the parameter estimate is larger in absolute terms in the first set of specifications (models 4a to 4c) than in the second (models 5a to 5c). The reason for this is that models 4a to 4c use the travel cost estimate without opportunity costs of time, while models 5a to 5c use the travel cost estimate including time costs. Travel costs are thus considerably smaller in the first set of specifications than in the second. The travel cost coefficients have a strongly significant standard deviation in all specifications, which underlines that the effect of travel costs on park choice significantly varies over individuals in the estimation sample. The models with travel costs that include time costs yield a better model fit than the models using travel costs without time costs according to the log likelihood values at convergence as well as AIC and BIC.

Effects of non-natural park attributes

Models 4a and 5a present baseline estimations in which we only include travel costs and non-natural park attributes in the regressions. On the one hand, this includes objective information on the equipment of a park with facilities such as playgrounds and sport facilities. On the other hand, this includes subjective perceptions of park attributes such as tidiness and convenience. In Models 4b and 4c as well as 5b and 5c, we add natural park attributes to the regressions.

Regarding the objective attributes, it can be observed that the presence of almost all park facilities significantly positively influences park choice. The relatively largest effect on park choice can be observed for the presence of cafés/restaurants. This effect is significant at the 1% level in all model specifications. Overall, playgrounds and barbecue areas are the second and third most important facilities, respectively. The relative importance of picnic areas and sport facilities varies with the model specifications, and effects are not significant in some of the specifications.

Turning to the effects of subjective perceptions, we find that perceived tidiness has a strong and significantly positive effect on park choice. Convenience, however, does not have a significant impact on park choice in most model specifications and even shows a slightly negative effect in two of the model specifications. One reason for this could be that most of the attributes contained in the factor convenience, such as accessibility, can also be related to crowdedness and intense use, which might be a negative factor when deciding which park to visit.

In addition, we find, as expected, that there is strong preference heterogeneity for cafés/restaurants in the estimation sample, as indicated by the strongly significant standard deviation. There is mixed evidence for preference heterogeneity regarding the presence of sport facilities and playgrounds.

Effects of natural park attributes

Models 4b and 4c as well as models 5b and 5c vary in the way the natural attributes of the parks are captured as explanatory variables for park choice. Models 4b and 5b include a variable that reflects how the naturalness of a park is perceived by park visitors, while models 4c and 5c include the variability of the biotope values of the parks. The natural variables are added to the model one by one because they are strongly correlated with one another. The Pearson correlation coefficient between perceived naturalness and the variability of the biotope value is 0.42 and is significant at the 1 % level.

In all cases, model fit increases when comparing the models with natural attributes to the baseline models without natural attributes. Both natural variables show a strong and significantly positive effect on park choice. The strongest effect in terms of relative effect size can be observed for the perceived naturalness of the parks, followed by the variability of the biotope value. Model fit is best when the variability of the biotope value is used as an indicator for naturalness in the mixed logit regressions. These results show that the natural quality of a park is important for people and that it positively influences park choice. In addition, individual perception of naturalness seems to match well with objective natural quality, as measured by the variability of the biotope value. Consequently, both objective and subjective indicators for natural quality can be used as determining factors in park choice models.

4.2.2 Estimates of marginal willingness-to-pay

We calculate marginal WTPs for the single park attributes based on the estimations of park choice using the mixed logit model. WTP is calculated using simulation by dividing draws from the unconditional distribution of the attribute parameter by draws from the unconditional distribution of the travel cost parameter. Figure 3 shows mean estimates of WTP as well as 95% confidence intervals for model specifications 5b and 5c.[22] The WTP estimates calculated using the travel cost measure including opportunity costs of time are consistently larger in absolute terms than the WTP estimates based on travel costs excluding opportunity costs of time (see Appendix F). This is due to the larger travel cost coefficient in the case without time costs. In general, the marginal WTP estimates are quite low, with values below one Euro in most the cases. This, however, is reasonable given that these are marginal WTP estimates per person per park visit.

Regarding marginal WTP for non-natural park attributes, we find that the marginal WTP for the presence of cafés/restaurants is considerably larger than the WTP for all other non-natural and natural attributes, ranging from 2.70 € to 2.99 €. The second most important non-natural attribute is the presence of playgrounds, with the marginal WTP ranging from 0.99 € to 1.30 €. The marginal WTP for the other attributes is smaller but still significant at the 5% level in most cases. Exceptions are the WTP for picnic areas, which is not significant at the 5% level in model 5c, and the WTP for sport facilities, which is not significant at the 5% level in models 5b and 5c. In addition, the marginal WTP for changes in the convenience of the park is not significantly different from zero at the 5% level in any of the model specifications. Increasing the tidiness of a park, in contrast, entails a marginal WTP between 0.32 € and 0.38 € in the case with time costs.

[22]Given that the models including opportunity costs of time yield better model fits than those excluding them, we focus on the former in the presentation of WTP. A full set of numerical values for WTP estimates can be found in Appendix F. Confidence intervals are calculated using the delta method (see Hole (2007b) for a description and evaluation of the method).

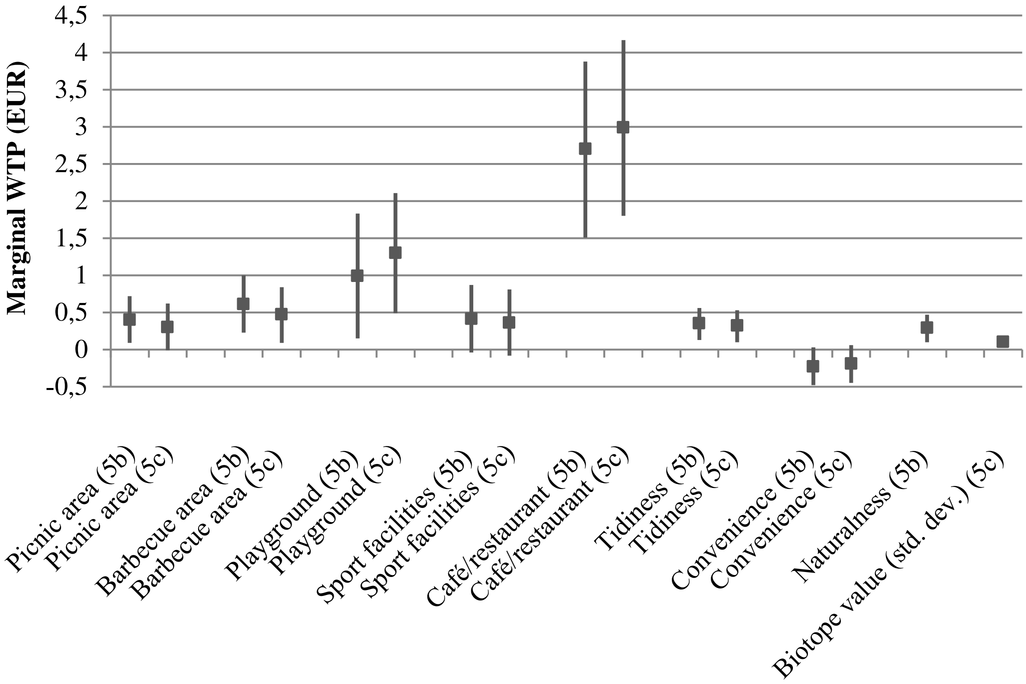

Figure 3. Mean WTP estimates for park attributes per person per visit with 95% intervals based on model specifications 5b and 5c (including time costs).

Regarding the natural attributes, marginal WTP is significant but smaller than that for the non-natural attributes. Increasing the perceived naturalness by one index point, for example, only entails a marginal WTP of 0.29 €, and the marginal WTP to increase natural variability amounts to 0.10 €. This is considerably smaller than the marginal WTP for non-natural attributes. However, the scale of the underlying variable has to be kept in mind. Non-natural attributes are dummy variables such that WTP refers to the existence versus non-existence of an attribute in a park. The biotope value, in contrast, is measured as a continuous variable, and WTP refers to a marginal increase in the variability of the biotope value, which explains the lower value.

To our best knowledge, there is only one study in an urban context to which our results can be more or less directly compared. Bullock (2006) estimates marginal WTPs for attributes of urban parks based on a discrete choice experiment for the case of Dublin, Ireland. Travel costs are calculated based on average travel time to the parks multiplied by the opportunity costs of time without direct costs of transport. The magnitude of the WTP estimates is smaller than for our estimates in the model specifications with opportunity costs of time, i.e., models 5a to 5c. For play facilities, for example, Bullock (2006) estimates a marginal WTP between 0.65 € and 0.70 €. Kinnell et al. (2006) also estimate the effect of park attributes on recreational choice but they do not report marginal WTP.

In addition, to gain an idea of the implied aggregate WTPs, consider that Berlin has approximately three million inhabitants aged 18 or older. According to our survey data, Berlin residents of this age visit parks, on average, twice a month, i.e., 24 times per year.[23] Based on these numbers and the mean WTP estimates of Model 5c, which has the best model fit, the aggregate WTP would be within a range of 7.2 million Euros per year for increasing the variability of the biotope values of the parks in Berlin (lowest marginal WTP over all attributes) and 215.3 million Euros per year for providing cafés and restaurants in the parks in Berlin (highest marginal WTP over all attributes). Note, however, that these numbers have to be interpreted with care, as the survey was only carried out in parts of the city such that the numbers may not be representative for the whole city.

4.2.3 Sensitivity analysis

We carry out several sensitivity analyses to validate our estimation results and WTP estimates. First, we calculate an alternative travel cost measure that is not based on a probability weighting for the transport modes but that assigns the travel costs of the transport mode with the largest probability to each combination of respondent and park. Using this measure, travel costs are between zero and 15.38 € in the case without opportunity costs of time and between zero and 38.81 € in the case with opportunity costs of time. These values are close to our probability-weighted

[23]The category “1 to 3 times a month” is the median outcome for the frequency of park visits in our survey aggregated over visits in summer and winter.

measure of travel costs (0.05-12.94 € and 0.05-38.57 €, respectively), with a slightly larger range.

Regarding the estimation results, we find that the model fit decreases compared to the model using the probability-weighted travel cost measure as judged by AIC, BIC, and log likelihood at convergence.[24] The results for the effects of natural and non-natural park attributes on park choice do not show any systematic differences between the two approaches. While some effect sizes are slightly larger and some are slightly smaller, overall, the results are comparable regarding effect sizes and significance levels. Travel costs still have a significantly negative effect on park choice, but the effects are smaller in absolute terms. This results in slightly larger WTP estimates using this alternative approach. The full set of WTP estimates using this alternative approach are displayed in Table G-1 in Appendix G.

Second, we also carry out the mixed logit choice regressions assuming a fixed travel cost parameter while using the probability-weighted travel cost measure. The parameters of the non-natural park attributes "playground", "sport facilities", and "café/restaurant" are still assumed to be random. The main reason for carrying out this sensitivity analysis is to check whether allowing the travel cost parameter to be random increases significance levels and thus reduces the validity of the WTP estimates.

It first has to be noted that the regressions with the fixed travel cost parameters yield a worse model fit than the models with the random travel cost parameters as judged by AIC, BIC, and log likelihood at convergence.[25] The parameter estimates of all natural and non-natural park attributes are smaller in absolute terms using the specifications with fixed travel cost parameters. The significance levels are equal for nearly all the attributes and are very similar for the remaining ones. Travel costs still have a significantly negative impact on park choice, but again, the effect sizes are smaller in absolute terms. This results in WTP estimates that are comparable but slightly larger in the case with fixed travel cost parameters for some of the attributes.

[24]Estimation results are not shown but are available from the author upon request.
[25]Again, estimation results are not shown but are available from the author upon request.

In particular, we do not find evidence that the confidence intervals of the WTP estimates are larger when a random travel cost parameter is used. On the contrary, in our application, confidence intervals are larger when a fixed travel cost parameter is assumed. This is comparable to findings by Doherty et al. (2013), who also find larger and more dispersed WTP estimates as well as a poorer model fit for the specifications assuming fixed travel cost parameters. The full set of WTP estimates using this alternative approach is displayed in Table G-2 in Appendix G.

5 Discussion

5.1 Participation model

In section 4.1 of this paper, we analyzed the effects of individual characteristics on the decision to engage frequently in recreation activities in urban parks. The findings from this participation model are broadly in line with the literature and add some interesting details. First, the findings regarding the effects of the availability of UGS on park use levels confirm earlier research. The probability to use urban parks more frequently is unambiguously increased by lesser distance to UGS (e.g., Schipperijn et al., 2010) and a higher amount of UGS in the vicinity of one's home (e.g., Neuvonen et al., 2007).

Regarding the effects of socio-economic variables, we find that both income and education positively influence the level of park use. While income is often missing in studies, the positive effect of education has been found in a few (Schipperijn et al., 2010; Yilmaz et al., 2007). In addition, we do not observe a positive effect of living in high rise buildings or large apartment buildings compared to living in detached, semi-detached or terraced houses as could have been expected. Taken together, these findings suggest that, for the case of Berlin, people from higher socio-economic backgrounds benefit more from an increased supply of UGS because they use them more. The differences in park use between people from different socio-economic backgrounds could be due to differing preferences or due to differences in the attractiveness of UGS in areas with lower and higher income in the city.

Regarding the effects of age on park use, findings in the literature are mixed. Some studies find a significantly negative effect (Schipperijn et al., 2013; Payne et al., 2002), others find a hump-shaped effect for men (Schipperijn et al., 2010), and still others find no statistically significant effect at all (Dallimer et al., 2014; Neuvonen et al., 2007). In our case study, the effect of age is significantly negative, but this can only be observed for the summer months. This difference between seasons could be one reason why other studies have not found significant effects.

The effects of age, income, and education on park use open room for policy advice. If it is a policy goal to increase the actual use of green spaces because of its positive effects on health, for example, then increasing the supply of UGS overall is one important prerequisite. However, it is also necessary to check whether the quality of the green spaces provided meets the needs of the target population. In particular, it would have to be checked whether a different green space design could encourage the elderly to use parks more often. In addition, it would be necessary to explore whether there are ways to promote outdoor recreation for people from lower socio-economic backgrounds.

The land cover data from the Urban Atlas, which we use in the participation model to investigate the influence of green space availability on use frequencies, do not allow a deeper investigation of the qualities of UGS. There is no differentiation beyond the categorization in different land cover classes. In section 4.2, we thus extended our analysis to investigate how various natural and non-natural park attributes influence park choice. We also investigated whether there is preference heterogeneity with respect to single park attributes.

5.2 Choice model

In the choice model, we observe a significantly negative impact of travel costs on park choice, as expected. In addition, the effects of almost all park attributes have the expected positive sign. The perceived factor convenience, however, has a negatively significant effect in two specifications. This could be explained by the fact that, e.g., good accessibility to urban parks is often associated with noisiness and crowdedness.

The estimates of marginal WTP for park attributes resulting from the choice regressions are small but reasonable and significant for the majority of the specifications.

To our knowledge, this is the first study to use a RUM with revealed preference data and travel costs for valuing the site attributes of urban parks. One likely reason for this is that travel costs within the city have been considered negligible. However, our survey data show that a large proportion of the respondents travel within the city by bike, public transport, or car. For these modes of transport, at least some costs occur. We thus propose including the costs of using these transport modes in the analysis by calculating a travel cost measure that takes into account the probability of each respondent choosing a certain transport mode. We suggest that this is a reasonable approach to account for the travel costs that occur for travel within cities without overestimating them.

The resulting travel costs are quite small because of the relatively short distances that need to be travelled to get to a park as well as the relatively large probabilities of using costless modes of transport. Still, the ranges seem reasonable for inner city travel (see section 3.3), and the marginal WTP estimates are not negligible in our application. In fact, they are comparable to other travel cost studies on recreation outside of cities where distances are larger. To give one example, Bujosa Bestard and Riera Font (2009) find a marginal WTP between 0.11 € and 0.16 € for picnic sites in Spanish forests. This is comparable to our estimates without time costs and is smaller than our estimates with time costs. In addition, including opportunity costs of time is as reasonable in an urban context as it is for applications outside the city, which would allow using the travel cost approach even in the absence of direct travel costs.

It has to be noted, however, that underlying our analysis is the assumption that people always start from home to get to their favorite park in the choice set. In reality, however, people do, in some cases, visit different parks depending on from which location in the city they intend to go there. Based on information gathered in the survey, we calculated that actual distances travelled amount to approximately two-thirds of the distance from home, on average. Consequently, the WTP estimates calculated in section 4.2 would have to be corrected by this factor. However, again, this is knowledge we only have for the visited parks and not for the non-chosen alternatives. Consequently, we are unable to incorporate this knowledge systematically in our analysis. However, this is an issue that needs to be considered by future research.

Another issue is that objective data on urban parks are missing to a large extent. For example, there is no objective information available regarding the cleanliness, maintenance, or crime rates of the urban parks in Berlin. In this application, we circumvent this issue by including subjective perceptions of some of these attributes in the choice regressions. While subjective perceptions are more likely to influence actual choices, this approach can complicate the interpretation of WTP, as it is not clear how improvements in perception, which are valued, translate into actual states of the world (Adamowicz et al., 1997). However, we have subjective and objective data for the attribute “naturalness”, which could be used to calculate how changes in the objective indicator translate into changes in perception. Moreover, the strong correlation between the objective indicator and subjective perception underlines findings from the literature that report a good correspondence between subjective and objective indicators when landscape amenities are considered (see Phaneuf and Smith (2005) for an overview).

Establishing a link between objective and subjective indicators would, in our application, not be possible for those attributes for which only subjective data are available. Still, including perceptions on “tidiness” and “convenience” in the choice model is the only way to account for these attributes, which have been shown to influence park choice. We thus decided to leave them in the set of explanatory variables to avoid an omitted variable bias of the estimates, even though WTP might be more difficult to interpret. Future research would benefit from an improved database, particularly regarding objective indicators for park attributes.

Finally, we find that preference heterogeneity is much less pronounced than what was anticipated in the mixed logit choice model. For most of the park attributes, improvements would thus increase the attractiveness of the parks for all respondents.

However, there is slight evidence for preference heterogeneity regarding the presence of playgrounds and sport facilities, and there is strong evidence for preference heterogeneity regarding the presence of restaurants and cafés as well as for travel costs. It would be interesting to investigate in more detail whether this preference heterogeneity could be explained by changes in age and other socio-economic characteristics, which are the individual characteristics that explain differences in use frequencies. In addition, it would be interesting to see whether the quality of the green spaces provided in the city vary systematically between areas with different socio-economic backgrounds. This, however, would require a much more detailed analysis that is beyond the scope of this paper, but it opens an interesting path for future research.

6 Concluding remarks

This paper successfully applied a travel cost analysis using a RUM approach to value park attributes in an urban context. It demonstrates that marginal WTP for improvements are small but significant and are comparable to other studies in more rural contexts. The paper has also revealed the influence of socio-economic characteristics on the participation in recreational activities in urban parks. Both analyses underline that merely increasing the provision in UGS is not enough to encourage their frequent use. Instead, the quality of the green spaces, including their equipment with non-natural facilities as well as their natural quality, crucially influences the recreation decisions of city inhabitants.

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Appendix A. Overview of individual-level data used for the analysis.

Table A-1. Definitions and summary statistics of individual-level data.

Variable name	Definition	Mean	Std. dev.	Min	Max
Dependent variables					
Frequency of park visits (summer)	Frequency of park visits in summer 2012; measured on a 6-point scale from 0 „never“ to 5 „(almost) daily“	2.22	1.30	0	5
Frequency of park visits (winter)	Frequency of park visits in winter 2011/12; measured on a 6-point scale from 0 „never“ to 5 „(almost) daily“	1.64	1.21	0	5
Environmental characteristics					
Urban green space (UGS)	Hectares of urban green space in a 1km buffer area around the respondent's home	24.44	18.22	1.09	98.12
Distance	Distance to nearest urban green space > 5 ha; measured in metres	656.0	432.2	0.5	2495.3
Park in walking distance	Dummy variable; 1 if respondent reports to have a park in walking distance to her home, 0 else	0.52	0.50	0	1
Visits of other green areas in the city	Frequency of visits over last 12 month; measured on a 6-point scale from 0 „never“ to 5 „(almost) daily“	1.63	1.28	0	5
Individual demographic and socio-economic characteristics					
Gender	Gender dummy; 1 if „male“, 0 if „female“	0.51	0.50	0	1
Age	Age; measured in years	45.9	14.34	18	78
Very bad health[a]	Health dummy; 1 if „Very bad health“, 0 else; reference category	0.03	0.18	0	1
Bad health[a]	Health dummy; 1 if „Bad health“, 0 else	0.18	0.38	0	1
Fair health	Health dummy; 1 if „Fair health“, 0 else	0.27	0.45	0	1
Good health	Health dummy; 1 if „Good health“, 0 else	0.38	0.49	0	1
Very good health	Health dummy; 1 if „Very good health“, 0 else	0.13	0.34	0	1
Single	Marital status dummy; 1 if „single“, 0 else; reference category	0.31	0.46	0	1
Married	Marital status dummy; 1 if „married“, 0 else	0.34	0.47	0	1
Partner	Marital status dummy; 1 if „living in a relationship“, 0 else	0.23	0.42	0	1
Separated[a]	Marital status dummy; 1 if „separated“, 0 else	0.02	0.14	0	1
Divorced[a]	Marital status dummy; 1 if „divorced“, 0 else	0.09	0.28	0	1
Widowed[a]	Marital status dummy; 1 if „widowed“, 0 else	0.01	0.11	0	1
Migration background	Dummy variable; 1 if at least one parent of the respondent is of a nationality other than German, 0 else	0.34	0.47	0	1
Individual income	Total net monthly individual income in Euros	1439.8	855.5	50	7500

Table A-1. Definitions and summary statistics of individual-level data (continued).

Variable name	Definition	Mean	Std. dev.	Min	Max
Basic education	Education dummy; 1 if education on ISCED level 1 or 2[b]; reference category	0.07	0.26	0	1
Secondary eductaion	Education dummy; 1 if education on ISCED level 3 or 4[b]	0.45	0.50	0	1
Tertiary education	Education dummy; 1 if education on ISCED level 5 or 6[b]	0.48	0.50	0	1
Household characteristics					
Child	Dummy variable; 1 if at least one child under the age of 12 is living in the household, 0 else	0.11	0.31	0	1
Detached, semi-detached, or terraced house	Housing dummy; 1 if „detached, semi-detached or terraced house“, 0 else; reference category	0.06	0.25	0	1
Small apartment building	Housing dummy; 1 if „apartment building with 3 to 8 apartments“, 0 else	0.29	0.46	0	1
Large apartment building	Housing dummy; 1 if „apartment building with 9 or more apartments (but no high rise)“, 0 else	0.55	0.50	0	1
High-rise building	Housing dummy; 1 if „high rise“, 0 else	0.09	0.29	0	1
District controls					
Mitte	District dummy; 1 if respondent lives in this district, 0 else; reference category	0.17	0.37	0	1
Friedrichshain-Kreuzberg	District dummy; 1 if respondent lives in this district, 0 else	0.09	0.29	0	1
Pankow	District dummy; 1 if respondent lives in this district, 0 else	0.15	0.36	0	1
Charlottenburg-Wilmersdorf	District dummy; 1 if respondent lives in this district, 0 else	0.14	0.34	0	1
Tempelhof-Schöneberg	District dummy; 1 if respondent lives in this district, 0 else	0.19	0.39	0	1
Neukölln	District dummy; 1 if respondent lives in this district, 0 else	0.11	0.32	0	1
Lichtenberg	District dummy; 1 if respondent lives in this district, 0 else	0.15	0.36	0	1

The number of observations included is 317 for all variables.
[a] In the regression analyses in section 4.1, the dummy variables „separated“, „divorced“, and „widowed“ as well as „very bad health“ and „bad health“ are merged into one variable, respectively, due to the low number of observations.
[b] For an overview of how the German educational achievements translate into the internationally comparable ISCED levels see Statistisches Bundesamt (2010).

Appendix B. Urban parks contained in the choice set.

Table B-1. Urban parks in Berlin, Germany, contained in the choice set.

Park name	District	Size (ha)
Großer Tiergarten	Mitte	195.9
Volkspark Rehberge	Mitte	75.7
Park am Plötzensee	Mitte	37.1
Schillerpark	Mitte	31.4
Görlitzer Park	Friedrichshain-Kreuzberg	18.3
Viktoriapark	Friedrichshain-Kreuzberg	16.1
Volkspark Friedrichshain	Friedrichshain-Kreuzberg	52.9
Schlosspark Buch	Pankow	23.6
Bürgerpark Pankow	Pankow	14.5
Mauerpark	Pankow	7.3
Park am Weißen See	Pankow	32.7
Schlosspark Charlottenburg	Charlottenburg-Wilmersdorf	58.0
Volkspark Jungfernheide	Charlottenburg-Wilmersdorf	159.8
Volkspark Wilmersdorf	Charlottenburg-Wilmersdorf	23.3
Lietzenseepark	Charlottenburg-Wilmersdorf	18.7
Volkspark Hasenheide	Neukölln	51.7
Britzer Garten	Neukölln	84.2
Tempelhofer Park	Tempelhof-Schöneberg	258.5
Volkspark Mariendorf	Tempelhof-Schöneberg	15.9
Malchower See Park	Lichtenberg	57.9
Fennpfuhlpark	Lichtenberg	15.0
Treptower Park	Treptow-Köpenick	100.2
Gärten der Welt	Marzahn-Hellersdorf	26.3
Volkspark Humboldthain	Mitte	29.3
Monbijoupark	Mitte	6.7
Schlosspark Niederschönhausen	Pankow	25.9
Stadtpark Steglitz	Steglitz-Zehlendorf	32.3
Botanischer Garten	Steglitz-Zehlendorf	37.1
Kleiner Tiergarten	Mitte	5.1
Orankeseepark	Lichtenberg	128.2
Preußenpark	Charlottenburg-Wilmersdorf	6.0
Volkspark Prenzlauer Berg	Pankow	30.5
Ernst-Thälmann-Park	Pankow	20.7
Rudower Höhe	Neukölln	56.8
Forckenbeckplatz	Friedrichshain-Kreuzberg	2.7
Franckepark	Tempelhof-Schöneberg	8.4
Freizeitpark Marienfelde	Tempelhof-Schöneberg	80.7
Gutspark Marienfelde	Tempelhof-Schöneberg	8.3
Volkspark Lichtenrade	Tempelhof-Schöneberg	4.7
Mariannenplatz	Friedrichshain-Kreuzberg	9.8
Fauler See Park	Pankow	26.3
Boxhagener Platz	Friedrichshain-Kreuzberg	1.2
Dörferblick	Neukölln	25.7
Gemeindepark Lankwitz	Steglitz-Zehlendorf	15.0
Georg-Kolbe-Hain	Charlottenburg-Wilmersdorf	7.5
Hans-Baluschek-Park	Tempelhof-Schöneberg	23.7
Johannisthaler Park	Treptow-Köpenick	3.6
Körnerpark	Neukölln	10.8
Landschaftspark Herzberge	Lichtenberg	106.7
Grünfläche Rummelsburger Straße	Lichtenberg	16.0

Appendix C. Factor analysis of perceived park attributes.

Table C-1. Factor analysis of perceived park attributes.

Perceived park attribute	Factor 1 (Naturalness)	Factor 2 (Tidiness)	Factor 3 (Convenience)
Varied landscape	0.8704		
Naturalness/natural design	0.8607		
Scenic beauty	0.8212		
Biological diversity	0.7967		
Tranquility		0.8894	
Cleanliness		0.7633	
Low density of visitors		0.7218	
Easy to get to			0.6954
Good opportunities to meet people			0.6546
Low crime			0.5235
Good accessibility			0.4510

Extraction method: Principal factors. Rotation method: Varimax with Kaiser normalization. Three factors with eigenvalues greater than one retained. Factor loadings >0.45 displayed.

Appendix D. Overview of park-level data used for the analysis.

Table D-1. Definitions and summary statistics of park-level data.

Variable name	Definition	Mean	Std. dev.	Min	Max
Biotope value	Variability of the biotope value in the urban parks; measured as the standard deviation	6.64	4.40	0	18.57
Naturalness	Perception of naturalness of the urban parks; principal factor	0	0.96	-2.52	2.00
Tidiness	Perception of tidiness of the urban park; principal factor	0	0.95	-3.15	1.92
Convenience	Perception of convenience of the urban park; principal factor	0	0.87	-2.76	1.70
Picnic area	Dummy variable; 1 if picnic area is present in the urban park, 0 else	0.24	0.43	0	1
Barbecue area	Dummy variable; 1 if barbecue area is present in the urban park, 0 else	0.14	0.35	0	1
Playground	Dummy variable; 1 if playground is present in the urban park, 0 else	0.76	0.43	0	1
Sport facilities	Dummy variable; 1 if sport facilities are present in the urban park, 0 else	0.46	0.50	0	1
Café/restaurant	Dummy variable; 1 if a café or restaurant is present in the urban park, 0 else	0.56	0.50	0	1

The number of observation is 50 for all variables.

Appendix E. Choice of transport mode.

Table E-1. Multinomial logit regression on the choice of transport mode.

Transport mode	**Model E-1**		
	Walk (Base outcome)		
	Bike	**Car**	**Public transport**
Distance to favorite park	0.0008*** (0.00)	0.0009*** (0.00)	0.0010*** (0.00)
Individual income	0.0000 (0.00)	-0.0006 (0.00)	-0.0002 (0.00)
Age	0.0089 (0.02)	0.0314 (0.02)	-0.0195 (0.02)
Gender	-0.5565 (0.42)	0.2521 (0.59)	-0.3859 (0.44)
Single	Reference	Reference	Reference
Married	-1.3140** (0.60)	0.1588 (0.76)	-1.0503* (0.63)
Partner	-0.2233 (0.54)	0.6846 (0.80)	-0.3229 (0.57)
Separated, divorced, or widowed	-0.3838 (0.75)	-13.2747 (492.81)	0.3139 (0.76)
Constant	-2.5142*** (0.91)	-3.7424*** (1.32)	-1.3758 (0.97)
Number of obs.	210		
Pseudo R^2	0.2050		
AIC	447.14		
BIC	527.47		

Appendix F. WTP estimates.

Table F-1. WTP estimates for park attributes per person per park visit with 95% confidence intervals (probability-weighting of transport mode).

WTP: Mean (95% CI)	Model 4a	Model 4b	Model 4c	Model 5a	Model 5b	Model 5c
Picnic area	**0.24 (0.04; 0.44)**	0.19 (-0.02; 0.41)	0.15 (-0.06; 0.36)	**0.45 (0.14; 0.76)**	**0.40 (0.09; 0.72)**	0.30 (-0.01; 0.62)
Barbecue area	**0.32 (0.08; 0.56)**	**0.41 (0.17; 0.66)**	**0.32 (0.08; 0.56)**	**0.48 (0.11; 0.85**	**0.61 (0.23; 1.00)**	**0.47 (0.09; 0.84)**
Playground	**0.64 (0.08; 1.20)**	**0.63 (0.02; 1.26)**	**0.84 (0.26; 1.43)**	**1.00 (0.21; 1.80)**	**0.99 (0.15; 1.83)**	**1.30 (0.49; 2.11)**
Sport facilities	**0.32 (0.01; 0.64)**	0.20 (-0.09; 0.48)	0.19 (-0.09; 0.47)	**0.57 (0.08; 1.06)**	0.41 (-0.04; 0.87)	0.36 (-0.08; 0.81)
Café/restaurant	**2.08 (1.13; 3.03)**	**2.08 (1.16; 3.00)**	**2.26 (1.25; 3.27)**	**2.75 (1.56; 3.95)**	**2.70 (1.51; 3.88)**	**2.99 (1.80; 4.17)**
Tidiness	**0.20 (0.06; 0.34)**	**0.18 (0.04; 0.31)**	**0.17 (0.03; 0.31)**	**0.38 (0.16; 0.59)**	**0.35 (0.13; 0.56)**	**0.32 (0.10; 0.53)**
Convenience	-0.14 (-0.30; 0.02)	-0.13 (-0.29; 0.04)	-0.11 (-0.28; 0.05)	-0.23 (-0.47; 0.02)	-0.23 (-0.48; 0.03)	-0.19 (-0.45; 0.06)
Naturalness		**0.17 (0.05; 0.29)**			**0.29 (0.10; 0.47)**	
Biotope value (std. dev.)			**0.06 (0.03; 0.08)**			**0.10 (0.06; 0.13)**

Notes: Mean WTP based on simulation. 95% confidence intervals calculated using the delta method as implemented in Stata; see Hole (2007b). Numbers in bold represent WTPs that are significant at the 5%-level. Calculations based on travel cost measure with probability-weighting of transport mode.

Appendix G. Sensitivity analyses for WTP estimates.

Table G-1. WTP estimates for park attributes per person per park visit with 95% confidence intervals (transport mode with largest probability).

WTP: Mean (95% CI)	Model 4a	Model 4b	Model 4c	Model 5a	Model 5b	Model 5c
Picnic area	**0.35** **(0.09; 0.60)**	**0.32** **(0.04; 0.59)**	**0.29** **(0.04; 0.56)**	**0.80** **(0.26; 1.33)**	**0.71** **(0.15; 1.26)**	0.54 (-0.00; 1.07)
Barbecue area	**0.47** **(0.17; 0.78)**	**0.58** **(0.23; 0.93)**	**0.47** **(0.15; 0.79)**	**0.91** **(0.27; 1.55)**	**1.12** **(0.44; 1.80)**	**0.86** **(0.22; 1.51)**
Playground	**1.08** **(0.06; 2.11)**	**1.09** **(0.16; 2.03)**	**1.28** **(0.37; 2.20)**	**1.31** **(0.47; 2.15)**	**1.27** **(0.44; 2.10)**	**1.78** **(0.89; 2.66)**
Sport facilities	**0.31** **(0.07; 0.56)**	**0.28** **(0.01; 0.56)**	**0.26** **(0.00; 0.51)**	**0.76** **(0.02; 1.49)**	0.52 (-0.14; 1.17)	0.63 (-0.22; 1.47)
Café/ restaurant	**6.22** **(1.82; 10.62)**	**4.13** **(1.80; 6.47)**	**4.25** **(1.98; 6.51)**	**5.28** **(2.00; 8.56)**	**5.03** **(2.02; 8.05)**	**5.14** **(2.53; 7.74)**
Tidiness	**0.19** **(0.02; 0.37)**	0.16 (-0.01; 0.33)	0.14 (-0.03; 0.31)	**0.58** **(0.21; 0.95)**	**0.54** **(0.16; 0.91)**	**0.47** **(0.10; 0.84)**
Convenience	-0.13 (-0.32; 0.06)	-0.10 (-0.31; 0.10)	-0.09 (-0.29; 0.11)	-0.42 (-0.86; 0.01)	-0.40 (-0.86; 0.06)	-0.37 (-0.82; 0.08)
Naturalness		**0.21** **(0.05; 0.38)**			**0.46** **(0.12; 0.80)**	
Biotope value (std. dev.)			**0.06** **(0.03; 0.09)**			**0.14** **(0.08; 0.21)**

Notes: Mean WTP based on simulation. 95% confidence intervals calculated using the delta method as implemented in Stata; see Hole (2007b). Numbers in bold represent WTPs that are significant at the 5%-level. Calculations based on travel cost measure with single transport mode selected according to largest probability.

Table G-2. WTP estimates for park attributes per person per park visit with 95% confidence intervals (fixed travel cost coefficient).

WTP: Mean (95% CI)	**Model 4a**	**Model 4b**	**Model 4c**	**Model 5a**	**Model 5b**	**Model 5c**
Picnic area	**0.28 (0.03; 0.53)**	0.23 (-0.03; 0.48)	0.16 (-0.10; 0.41)	**0.53 (0.16; 0.90)**	**0.46 (0.09; 0.84)**	0.35 (-0.03; 0.73
Barbecue area	**0.36 (0.06; 0.65)**	**0.46 (0.15; 0.76)**	**0.31 (0.01; 0.62)**	**0.50 (0.05; 0.94)**	**0.64 (0.18; 1.10)**	**0.48 (0.02; 0.93)**
Playground	**0.59 (0.14; 1.04)**	**0.55 (0.17; 0.92)**	**0.83 (0.23; 1.42)**	1.21 (-0.42; 2.84)	1.31 (-0.18; 2.80)	1.52 (-0.07; 3.11)
Sport facilities	**0.33 (0.02; 0.65)**	0.23 (-0.09; 0.54)	0.16 (-0.11; 0.44)	**0.77 (0.16; 1.39)**	**0.61 (0.01; 1.21)**	0.51 (-0.06; 1.07
Café/restaurant	**2.12 (1.20; 3.03)**	**2.14 (1.21; 3.07)**	**2.38 (1.31; 3.44)**	**3.09 (1.77; 4.42)**	**2.97 (1.71; 4.23)**	**3.47 (1.95; 4.99)**
Tidiness	**0.24 (0.07; 0.41)**	**0.21 (0.04; 0.38)**	**0.18 (0.01; 0.35)**	**0.43 (0.17; 0.69)**	**0.39 (0.14; 0.65)**	**0.35 (0.09; 0.61)**
Convenience	-0.15 (-0.35; 0.06)	-0.15 (-0.37; 0.06)	-0.12 (0.33; 0.09)	-0.28 (-0.57; 0.02)	-0.29 (-0.59; 0.02)	-0.24 (-0.55; 0.07)
Naturalness		**0.23 (0.08; 0.38)**			**0.34 (0.11; 0.56)**	
Biotope value (std. dev.)			**0.08 (0.05; 0.11)**			**0.12 (0.08; 0.16)**

Notes: Mean WTP based on simulation. 95% confidence intervals calculated using the delta method as implemented in Stata; see Hole (2007b). Numbers in bold represent WTPs that are significant at the 5%-level. Calculations based on travel cost measure with probability-weighting of transport mode.

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Paper 2: The role of urban green space for human well-being[1]

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Abstract:

Most people in Europe live in urban environments. For these people, urban green space is an important element of well-being, but it is often in short supply. We use self-reported information on life satisfaction and different individual green space measures to explore how urban green space affects the well-being of the residents of Berlin, the capital city of Germany. We combine spatially explicit survey data with spatially highly disaggregated GIS data on urban green spaces. We observe a significant, inverted U-shaped effect of the amount of and distance to urban green space on life satisfaction. According to our results, the amount of green space in a 1 km buffer that leads to the largest positive effect on life satisfaction is 36 ha or 11.5% of the buffer area. In our sample, 75% of the respondents have less green space available. Our results are robust to a number of robustness checks.

Keywords: life satisfaction, urban ecosystem services, urban green space, well-being

JEL classification: I31, Q51, Q57, R00

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1 Introduction

Approximately 75% of Europeans live in urban areas (World Bank, 2013). One important element for their well-being and quality of life is the availability of urban green space. There are different ways in which urban green space can positively influence well-being and health (see Tzoulas et al. (2007) for an overview). Benefits can accrue from increased activity levels as a result of being in contact with nature (see Bowler et al. (2010) for a review). In addition, Kaplan (2001) shows that natural elements in the view from a window can have positive effects. Further benefits are brought about by the moderation of adverse environmental conditions such as air pollution, high temperatures, or noise (e.g., Gidlöf-Gunnarsson and Öhrström, 2007). However, in most urban areas, and particularly in inner-city areas, green spaces are in insufficient supply (Kabisch and Haase, 2011).

Individual countries and/or cities have begun to take an increasing responsibility in developing urban green space and improving the services provided by different forms of urban green. Following the Convention on Biological Diversity (CBD; UN, 1992), these countries and cities have formulated national, regional, or local action plans to integrate urban biodiversity and ecosystem services (ES) provided by urban green space, among others, into management.[2] The German National Strategy on Biological Diversity, for example, calls for an increase in green space in settlement areas (BMU, 2007).[3] At the city level, some German cities have defined minimum targets for per capita supply of urban green space.[4]

City development, however, always has to address trade-offs and conflicting interests between inner development, e.g., for housing, and the development or preservation of green and open spaces (Schetke et al., 2012). Information on the benefits and costs of alternative land uses can therefore be valuable in supporting

[2]See TEEB (2011) for an overview of the ES concept and its application in an urban context.
[3]This objective is integrated into federal law by requiring that open spaces in urban and peri-urban areas have to be preserved and developed where they are not sufficiently available (§ 1 Abs. 6 BNatSchG, 2009).
[4]The City of Berlin, e.g., has the goal to provide 6 m^2 of public green space per inhabitant (SSUB, 2013a).

decision-making and ensuring that land is used sustainably, meeting the needs of the inhabitants.[5] Information on the benefits of ES, which are mostly not traded on markets, however, is often not available. Despite the relevance of urban ES for a large share of the population, environmental valuation studies have so far focused on ES in rural contexts. Existing studies on the economic valuation of urban green space have mostly used traditional techniques such as stated or revealed preference approaches (see Brander and Koetse (2011) and Perino et al. (2014) for recent meta-analyses).

A recent alternative in the field of environmental valuation is the life satisfaction approach (LSA).[6] The two existing economic studies analyzing the effect of urban green on life satisfaction cover cities in Australia (Ambrey and Fleming, 2013) and China (Smyth et al., 2008). Unlike Ambrey and Fleming (2013) and Smyth et al. (2008), we use an individual green space measure that captures the area of green space surrounding a respondent's home. We further add to this literature by exploring and comparing different ways in which urban green might affect the well-being of city inhabitants. Moreover, we offer the first application of the LSA to value urban green in a European city, namely Berlin, the capital city of Germany.

Berlin, located in the Eastern part of Germany with an area of 892 km^2 (SSUB, 2012a) and a population of 3.4 million (ASBB, 2013), is particularly interesting. The expected population growth and the trend towards smaller household sizes will exert strong pressure on existing green spaces in the inner-city districts, particularly if a densification strategy is to be followed and urban sprawl is to be avoided. Such a conflict can currently be observed in the case of the Tempelhofer Feld.[7] On the other

[5]The EU Biodiversity Strategy (EC, 2011), for example, requires all member states to assess their ecosystems and the economic value of those systems by 2020.

[6]Self-reported life satisfaction is used as a proxy for subjective well-being. Please note that we use the terms life satisfaction and well-being interchangeably throughout the paper.

[7]The so-called Tempelhofer Feld is located on the area of the former airport Berlin Tempelhof. The associated free areas including the former airfield have a size of 303 ha. Currently, it can be used by the public, e.g., for recreational purposes and is left more or less in its original state (GrünBerlin GmbH, 2013). There are conflicting interests concerning the future development of the area.

hand, there are still many open spaces such as brownfields that might be turned into residential or commercial areas (Simons et al., 2012).

The objective of this paper is to answer the following research questions: (1) In which ways, if any, does urban green space affect the well-being of people? (2) Is more green space always better, or is there a level of urban green at which the positive impact on well-being is maximized? (3) What is the monetary equivalent of a change in the availability of urban green space?

To address these questions, we use spatially explicit survey data of Berlin residents together with spatially highly disaggregated GIS data on urban green spaces. We use several indicators for our analyses. Based on land cover data from the Urban Atlas (EEA, 2012), we calculate the amount of green space available in the living environment of each respondent as well as the distance to the nearest green space. Based on our survey data, we analyze the frequency of park visits and a dummy variable indicating whether the respondent has a view of a park from his/her home.

The remainder of the paper is organized as follows. Section 2 provides a review of the literature on the economic valuation of urban green and the literature on economics and subjective well-being. Section 3 presents the empirical approach and the data. Section 4 reports the results of the main regressions and sensitivity analyses. Section 5 discusses the results and presents the conclusions.

2 Economic valuation of urban green space – literature review

2.1 Stated and revealed preference methods

Despite the relevance of urban ES for city inhabitants, there are relatively few economic studies that elicit the value of urban ES using either stated preference methods such as contingent valuation (CV) or choice experiments (CE) or revealed preference methods such as hedonic pricing (HP) or travel costs (TC). Even fewer environmental valuation studies specifically focus on urban green spaces or parks.

The results of a range of CV and HP studies are analyzed in two recent meta-analyses that focus on different types of urban ecosystems and have different regional foci. Brander and Koetse (2011) provide a meta-analysis of 32 international CV and HP studies valuing different types of urban open spaces with a focus on the USA. They find that most of the CV studies refer to urban forests and urban agriculture, and far fewer studies investigate urban green spaces and parks. HP studies, in contrast, mostly investigate the role of urban parks and green spaces for property prices. A more recent meta-analysis for the UK is provided by Perino et al. (2014). It is based on five studies analyzing the effect of increased distance to formal recreation sites and city-edge green space on property prices using HP, as well as CV, and expert interviews.

With respect to CE, there are even fewer examples of studies analyzing preferences for urban ES. The only study that values urban green spaces or parks that we are aware of is an application for Dublin, Ireland, by Bullock (2006).[8] Two examples of TC studies are Fleischer and Tsur (2003), who use the individual TC method to estimate the economic value of urban parks in Israeli cities, and Chaudhry and Tewary (2006), who use zonal TC to assess the recreational value of urban forests in Chandigarh, India.

2.2 The life satisfaction approach

The LSA is a recent alternative in the field of environmental valuation. It is based on the assumption that environmental (dis)amenities are among the factors that determine subjective well-being (SWB). Following this approach, self-reported life satisfaction is taken as a proxy for SWB and estimated as a function of factors such as environmental amenities and income, while at the same time controlling for other socio-economic, demographic, and geographical information. Based on the assumption that life satisfaction data are an approximation of what Kahnemann et al. (1997) labeled “experienced utility”, this estimated relationship is used to derive the

[8]Examples of CE used in other urban contexts are studies by Lanz and Provins (2013), who focus on local environmental improvements in the UK, or Bae (2011), who analyzes preferences for urban stream restoration in Korea.

implicit marginal rate of substitution (MRS) between income and the environmental amenity in question.

Unlike stated preference methods, this method does not ask people to place a monetary value on a complex environmental good in a hypothetical situation. Survey respondents are not aware of the fact that the answer to the well-being question will be used to value an environmental amenity. Compared to CV, this may reduce biases resulting from the hypothetical nature of the decision and from potentially strategic behavior. In comparison to revealed preference methods, the LSA does not rely on decisions being reflected in actual market transactions. Thus, it is, for example, not affected by biases resulting from the assumption that the housing market is in equilibrium, which is a basic assumption of the HP method but may not always be the case in reality. There are, however, also limitations to the LSA. One of its preconditions is that life satisfaction data, which are used as a proxy for well-being or utility, satisfy appropriate quality requirements (being ordinal in character, consistent, valid, and reliable).[9] For a discussion of the underlying assumptions and implications of the LSA in comparison with CV and HP, see Ferreira and Moro (2010) and Frey et al. (2010).

Research on SWB has identified a number of personal, demographic and socio-economic factors that explain differences in SWB (see, e.g., Dolan et al. (2008) for an overview). With regard to environmental (dis)amenities, most of the studies have looked at air pollution. The most recent examples are studies by Levinson (2012), Ferreira and Moro (2010), Luechinger (2009, 2010), MacKerron and Mourato (2009), and Rehdanz and Maddison (2008). Other environmental issues investigated include climate (Ferreira and Moro, 2010; Brereton et al., 2008; Rehdanz and Maddison, 2005; Frijters and van Praag, 1998), noise (Rehdanz and Maddison, 2008; van Praag and Baarsma, 2005), scenic amenities (Ambrey and Fleming, 2011), protected areas (Ambrey and Fleming, 2012), land cover (Kopmann and Rehdanz, 2013), droughts (Carroll et al., 2009), and floods (Luechinger and Raschky, 2009).

[9]See Welsch and Kühling (2009) for a discussion of conceptual and methodological issues. In section 5, we address additional issues potentially relevant to this analysis.

Many of the earlier environmental studies look at nationwide or cross-country data sets and suffer from a lack of more disaggregated environmental data (e.g., Welsch, 2006 or Rehdanz and Maddison, 2005) and are thus not able to take more disaggregated spatial controls into account. Some studies do include spatial controls, e.g., accounting for the fact whether people live in urban, rural or peri-urban areas (e.g., Ferreira and Moro, 2010). Few studies explicitly address urban environments or data sets customized to urban environments. One exception is MacKerron and Mourato (2009), who look at air quality in London using spatially disaggregated data.

Two studies investigate the amenity value of urban green spaces. Using wave 5 of the HILDA survey, Ambrey and Fleming (2013) investigate the role of public green space for the well-being of people in major Australian cities. The green space measure they use is the percentage of public green space in the resident's collection district.[10] The estimated implicit MRS for a 1% (equivalent to 143 m^2) increase in public green space is AUD 1,168 in terms of annual household income. Smyth et al. (2008) use survey data gathered from the inhabitants of 30 Chinese cities to estimate the effects of pollution, disasters, congestion and green space on human well-being. The green space measure they use is the area of green space per capita on the city-level. They find a statistically significantly positive effect of green space on life satisfaction for the model specification with city dummy variables. However, MRS estimates are not reported and cannot be derived.

3 Methodology and data

3.1 Methodological approach and empirical strategy

We estimate the effects of different demographic, socio-economic, and environmental variables on individual life satisfaction using the following regression

[10]The collection district is the smallest spatial unit in the Australian Standard Geographical Classification. Assuming each collection district takes the shape of a circle, the median radius from the centroid is approximately 750 m (Ambrey and Fleming, 2013).

equation:

$$LS_{ij} = \beta_0 + \beta_y \, ln(Y_i) + \beta_x X_i + \beta_z Z_i + \beta_s \, S_j + \beta_a A_i + \varepsilon_{ij} \tag{1}$$

The dependent variable, LS_{ij}, is the stated life satisfaction of respondent i living in district j. Explanatory variables include Y_i, which is the individual net monthly income of respondent i.[11] Income enters the regression equation in its natural logarithm to account for the declining marginal utility of income. Further explanatory variables are captured in the vectors X_i and Z_i, which contain other demographic and socio-economic characteristics of respondent i and of her household, respectively (see Table A-1 in Appendix A for a description and summary statistics of all variables). The variable S_j contains dummy variables for each district j to control for district-specific effects, and ε_{ij} is the error term.

We estimate a set of different specifications that differ in the way the environmental variable A_i is measured. We analyze i.) the amount of urban green space available in the living environment of respondent i, ii.) the distance to the nearest urban green space bigger than 5 ha, iii.) the frequency of visits to urban parks, and iv.) whether respondent i has a view of a park from his/her home. The indicators for park view and the number of park visits are based on self-reports derived from our survey. The amount of urban green space and the distance to the nearest urban green space are calculated based on the residential address of the respondents and land cover data from the Urban Atlas (EEA, 2012).[12]

In addition to the linear regression model described in equation (1), we also estimate a non-linear form in which the amount of and distance to green spaces enter

[11]Total net monthly individual income was calculated by dividing the corresponding household income by the weighted number of household members according to the OECD-modified equivalence scale (OECD, 2009). Because there was no information about children under the age of 14 in the household in the survey, this was adapted to children under the age of 12. Household income was indicated in ranges. We used the midpoint of the indicated range to calculate the corresponding individual income. The use of midpoints is common in life satisfaction studies if income is given in ranges (Carroll et al., 2009; MacKerron and Mourato, 2009).

[12]See section 3.2 for a more detailed description of these data.

the regression in their linear and in their squared form. In the non-linear case, the estimation equation changes to

$$LS_{ij} = \beta_0 + \beta_y \, ln(Y_i) + \beta_x X_i + \beta_z Z_i + \beta_s \, S_j + \beta_{a1} A_i + \beta_{a2} A_i^2 + \varepsilon_{ij} \qquad (2)$$

This functional form presumes that the effect of urban green may also depend on its current allocation. The parameter β_{a1} is expected to be positive, while β_{a2} is expected to be negative.[13]

The estimated relationships can then be used to derive the implicit MRS between the environmental variable, e.g., the individually available amount of urban green space, A_i, and individual income, Y_i, by dividing the absolute value of the derivative of life satisfaction (LS_{ij}) with respect to A_i by the derivative of LS_{ij} with respect to Y_i. For the linear specification, the MRS, evaluated at the mean of income, can thus be calculated as follows:

$$MRS = \left. \frac{\partial \, LS_{ij} / \partial \, A_i}{\partial \, LS_{ij} / \partial \, Y_i} \right|_{dLS_{ij}=0} = \frac{\beta_a}{\beta_y} \, \overline{Y} \qquad (3)$$

For the non-linear specification, the current allocation of the environmental good has to be taken into consideration. The implicit MRS between A_i and Y_i evaluated at the means of the environmental variable and income can be calculated as follows:

$$MRS = \left. \frac{\partial \, LS_{ij} / \partial \, A_i}{\partial \, LS_{ij} / \partial \, Y_i} \right|_{dLS_{ij}=0} = \frac{\beta_{a1} + 2 \, \beta_{a2} \, \bar{A}}{\beta_y} \, \overline{Y} \qquad (4)$$

[13]Further non-linear specifications tested include i.) both income and the green space variable as natural logarithms, which implies a Cobb-Douglas utility function, and ii.) both as transformations implying a utility function with a constant elasticity of substitution (CES). These specifications are not further considered since the effect of urban green space on life satisfaction is not significant using Cobb-Douglas or CES specifications.

3.2 Data

3.2.1 Survey data

Most of the data used for our analysis come from a web survey carried out in Berlin, Germany, in September 2012.[14] The main objective of this survey was to investigate the role of urban green space, particularly parks, for the well-being of people living in urban environments. For this purpose, the survey included a number of questions on park use patterns and the perception of the environment. The survey also included questions on socio-economic and demographic characteristics of the respondents and their households, including gender, age, marital status, education, occupation, and household income. To take account of the fact that the housing environment is very important for personal well-being in urban surroundings, we also included questions on the housing type and on neighborhood characteristics in the survey (see Table A-1 in Appendix A for summary statistics).

Survey participants were screened to ensure that they had been living in Berlin for more than one year. Only residents of the districts of Mitte, Kreuzberg-Friedrichshain, Pankow, Charlottenburg-Wilmersdorf, Tempelhof-Schöneberg, Neukölln, and Lichtenberg were included in the survey (see Figure 1). These districts were selected because they include the densely populated inner-city districts of Berlin with a relatively homogenous distribution of green space and exclude districts with large shares of water areas and forests. The districts were also selected to be comparable with the whole population of Berlin with regard to age and gender. Moreover, a balanced distribution between formerly Eastern and Western German parts of the city of Berlin has been targeted. The final sample consists of 485 usable observations.

[14] More detailed information about the survey is provided in Appendix B.

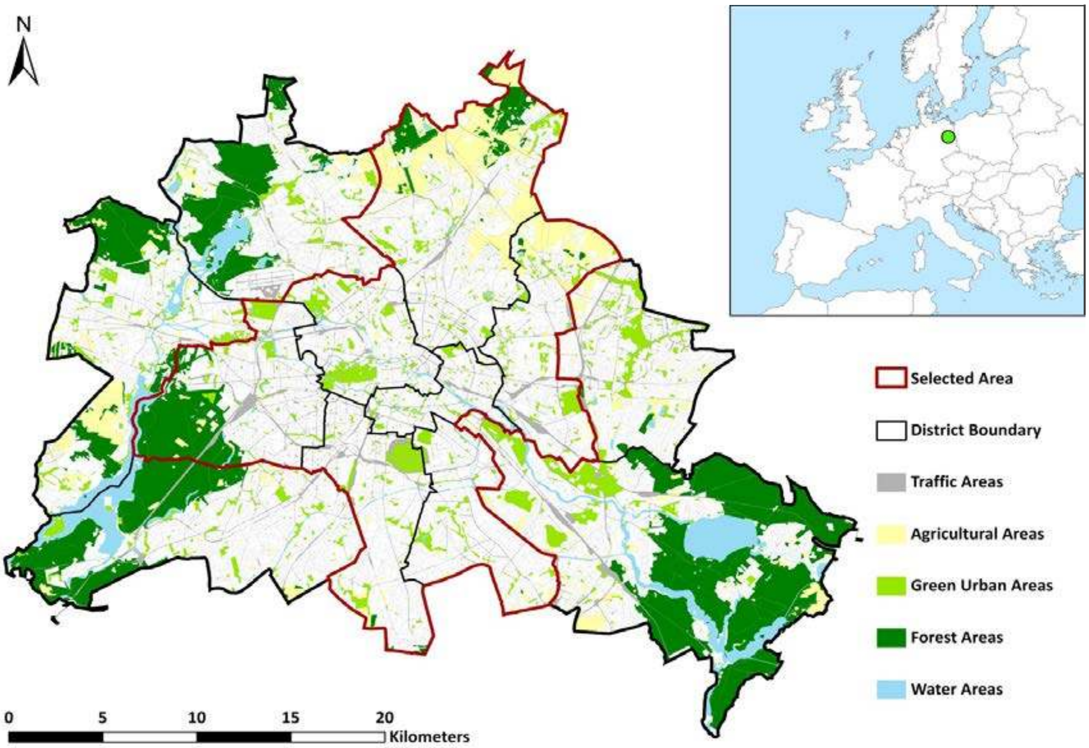

Figure 1. Distribution of urban green spaces and other natural and semi-natural areas in Berlin. Own presentation based on Urban Atlas data (EEA, 2012).

The question about subjective well-being was phrased as follows: “All things considered, how satisfied are you with your life these days?” Respondents were asked to answer the question on an 11-point Likert scale ranging from “0” (very dissatisfied) to “10” (very satisfied). This question mode is in line with many large surveys carried out in different countries and used in the economic literature on life satisfaction (Welsch and Kühling, 2009). The mean value for life satisfaction in our sample is 6.8 with a standard deviation of 2.05, which is comparable to other life satisfaction studies for Germany (e.g., Kopmann and Rehdanz, 2013).

The distribution of gender in our sample is comparable to that in Berlin and in the selected districts of Berlin. Regarding age, people between 50 and 64 are overrepresented by approximately 15% in our sample, while people above 64 are underrepresented to the same extent. This, however, seems unavoidable, given that we use a web survey. The distribution of income in our sample is comparable to that in Berlin and in the selected districts.

As indicated in section 3.1, two of the environmental indicators we use to analyze the well-being effects of urban green are derived from the survey, namely the indicators of the "frequency of park visits" and "park view." The "frequency of park visits" is an ordinal variable measured on a 6-point scale, ranging from "never" to "(almost) daily."[15] The dummy variable "park view" is derived from a question asking respondents whether they have a view of a park from their homes. Approximately 14% of the respondents report having such a view.

We use self-reported information on residential addresses from the survey to locate respondents in the city of Berlin in a spatially explicit way using ArcGIS. This enables us to link the survey data with GIS data on urban green spaces. Out of the 485 usable observations, 11% provided their full address, including street name and number. Table 1 gives an overview of the type of information we gathered from the respondents, the way we located them in ArcGIS, and the location statuses we assigned to them based on a decreasing level of precision. Given that we are interested in questions relating very specifically to the living environment of the respondents, we decided to limit the subsequent analyses to those respondents that indicated at least the name of the street (location statuses 1 and 2).

Table 1. Overview of different location statuses.

Location status	**Description**	**Location in ArcGIS**	**Number of obs.**	**Percentage**
1	Full address	Precise address	53	10.9%
2	Street without number	Middle of the street	326	67.2%
3	ZIP code	Centroid of the ZIP code area[a]	50	10.3%
4	District	Centroid of the district[b]	56	11.5%

[a] The average area of the ZIP code areas in which the respondents live is 3 km^2.
[b] The average area of the selected districts is 54 km^2.

[15]Frequencies were averaged over the stated number of visits in summer and winter during the 12 months preceding the survey.

3.2.2 GIS data and social indicators

The spatial data on urban green spaces are taken from the Urban Atlas (EEA, 2012). This database provides pan-European comparable land use and land cover data for large urban areas with more than 100,000 inhabitants on a 50x50 m grid level (EU, 2011). We use the areas designated as "green urban areas" for our analyses.[16]

To calculate the individual availability of urban green space, we created buffers with a diameter of 1 km around the respondents' residential addresses and calculated the amount of green space within the buffer area for each respondent. This gives us an individual measure of green space availability on the respondent level.[17] However, we have to be aware of potential measurement errors in the green space variable due to the uncertainties related to the location of the respondents described in section 3.2.1. Thus, we excluded observations in the 1st and 99th percentile of the variable "green space" from our sample to mitigate the influence of outliers. This led to an exclusion of 10 observations from the sample. The total buffer area for the buffer with a 1 km radius is 314.2 ha (or approximately 3.14 km^2). Individual green space availability is between 0.5% and 31.2% of this buffer area, i.e., between 1.1 ha and 98.1 ha.[18]

Based on the same Urban Atlas data set, we also calculated the Euclidean distance from each respondent's individual residential address to the edge of the nearest urban green space greater than 5 ha. The mean distance is 661 m, ranging from a minimum of 0.5 m to a maximum of 2,495 m.

[16]This land use category includes public green areas for predominantly recreational use such as gardens, zoos, parks, or castle parks. Not included are private gardens within housing areas, cemeteries, buildings within parks, such as castles or museums, patches of natural vegetation or agricultural areas enclosed by built-up areas without being managed as green urban areas. We also include the lawns belonging to the former Tempelhof airport, which are now used for recreation, in the measure for urban green space. This was not the case in the original data set (EU, 2011).

[17]All respondent addresses are located more than 1 km away from city boundaries such that there is full information regarding land cover for all buffer areas considered.

[18]For comparison, the area of the selected districts in Berlin is between 2,034 ha in the case of Friedrichshain-Kreuzberg and 10,315 ha in the case of Pankow. The area share of public green space is between 6.8% in Pankow and 14.8% in Mitte (SSUB, 2012a).

In addition to land cover data from the Urban Atlas, we use sealing data provided by the City of Berlin for a sensitivity analysis (SSUB, 2012c; see also section 4.3.4). Sealing data do not only reflect the supply of public green spaces but also the existence of other types of public green or private green such as gardens. High degrees of sealing thus often go hand in hand with a mismatch between the number of inhabitants and the supply of green spaces and imply reduced regulating services such as local climate regulation. The GIS data set provides average degrees of sealing in Berlin on the block level for the year 2011.[19] We create the variable "sealing" by calculating the area-weighted average degree of sealing in a buffer area around the respondents' residential addresses.

We also use noise data provided by the City of Berlin for a sensitivity analysis (SSUB, 2013b; see also section 4.3.4) as noise disturbances, particularly at night, may influence subjective well-being in urban environments. Noise levels are calculated based on raster maps aggregating information on noise from different sources, namely street noise from motor vehicles including busses, noise created by trams and subways as well as noise created by airplanes. We use two separate data sets, one for noise disturbances occurring during day and night times and one for noise disturbances at night only. We average the respective noise indices per raster cell over a buffer area surrounding the respondents' residential addresses.

In addition to environmental data, we use highly disaggregated data on the social status of neighborhoods for additional robustness checks (see section 4.3.1). The City of Berlin publishes social indicators such as different measures of unemployment on the level of planning units, which are considerably smaller than the districts and may thus give a good picture of the local social status of a neighborhood (SSUB, 2011).[20]

[19]Blocks have an average size of 3.6 ha.
[20]Planning units have an average size of 2 km^2 and are thus approximately two-thirds the size of the 1 km buffers used to calculate the availability of urban green spaces. The planning units were developed to reflect social, spatial, and architectural aspects of neighborhoods.

4 Estimation results

The results of the regressions estimating the effect of several explanatory variables on life satisfaction using ordered logit models are shown in Table 2.[21] We cluster error terms on the district level to calculate robust estimates as we combine variables on the individual level and on the district level in our regressions (Moulton, 1990). Model 1 presents the results of a baseline specification focusing on demographic and socio-economic variables. Models 2 to 7 additionally include environmental variables to estimate their effect on life satisfaction. We add the environmental indicators to the regressions one by one because there is a significant correlation between them.

Because respondents had the option to not answer single questions in the survey, the final sample consists of 316 observations. Most omissions occurred for the income variable, a potential issue we address in section 4.3.5. For all other variables, the number of omissions was relatively low so that we can safely exclude selection bias resulting from missing data points.

4.2.1 Effects of socio-economic variables

The results of Model 1 reflect the standard findings of the life satisfaction literature regarding the effects of most demographic and socio-economic explanatory variables on life satisfaction (see Dolan et al. (2008) for a review). We find that income has a significantly positive effect on life satisfaction with a declining marginal effect, which is consistent with findings from other studies using cross-sectional micro data on income and life satisfaction within one country (Clark et al., 2008).[22] We also find the well-studied U-shaped relationship between age and life satisfaction (Blanchflower and Oswald, 2004). Both being married and having a partner have a

[21]Ordered logit is used because life satisfaction is measured on an ordinal scale. In accordance with the literature (e.g., Ferreira and Moro, 2010 or Brereton et al., 2008), we find that OLS yields very similar results compared to ordered logit.
[22]In addition to using net monthly individual income, we also used net monthly household income in sensitivity analyses. The results are comparable.

Table 2. Results of main regressions (ordered logit).

Life satisfaction (LS_{ij})	**Model 1**	**Model 2**	**Model 3**	**Model 4**	**Model 5**	**Model 6**	**Model 7**
Individual demographic and socio-economic characteristics (Y_i, X_i)							
Log individual income	0.8577***	0.8545***	0.8332***	0.8686***	0.9020***	0.8247***	0.9720***
	(0.28)	(0.27)	(0.27)	(0.28)	(0.30)	(0.26)	(0.27)
Gender	-0.0465	-0.0409	-0.0852	-0.0719	-0.0431	-0.0427	-0.0578
	(0.19)	(0.20)	(0.22)	(0.20)	(0.21)	(0.23)	(0.22)
Age	-0.1091**[b]	-0.1081***[b]	-0.1093**[b]	-0.1172**[c]	-0.1215***[b]	-0.1151***[b]	-0.1112**[c]
	(0.04)	(0.04)	(0.04)	(0.05)	(0.04)	(0.04)	(0.05)
Age squared	0.0013***[b]	0.0013***[b]	0.0013***[b]	0.0014**[c]	0.0015***[b]	0.0014***[b]	0.0013**[c]
	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)
Single	Reference	Reference	Reference	Reference	Reference	Reference	Reference
Married	0.5032***	0.5009***	0.5273***	0.5119***	0.5827***	0.5353***	0.4809***
	(0.12)	(0.11)	(0.14)	(0.11)	(0.14)	(0.12)	(0.15)
Partner	0.4137*	0.4082	0.4002**	0.4216*	0.4638**	0.4445**	0.3955*
	(0.24)	(0.25)	(0.20)	(0.23)	(0.20)	(0.21)	(0.23)
Separated, divorc-ed or widowed	0.0290	0.0246	0.0902	0.0516	0.1048	0.0663	-0.0160
	(0.44)	(0.44)	(0.44)	(0.41)	(0.43)	(0.37)	(0.44)
Bad or very bad health	Reference	Reference	Reference	Reference	Reference	Reference	Reference
Fair health	1.0441***	1.0445***	1.1657***	1.0619***	1.0716***	1.0781***	1.1328***
	(0.26)	(0.26)	(0.24)	(0.24)	(0.23)	(0.25)	(0.27)
Good health	2.2097***	2.2083***	2.3672***	2.2728***	2.3798***	2.2299***	2.2868***
	(0.36)	(0.36)	(0.36)	(0.36)	(0.42)	(0.35)	(0.33)
Very good health	3.4648***	3.4599***	3.5843***	3.4843***	3.5538***	3.3166***	3.5301***
	(0.27)	(0.26)	(0.27)	(0.25)	(0.21)	(0.28)	(0.24)
Basic education	Reference	Reference	Reference	Reference	Reference	Reference	Reference
Secondary education	1.3072***	1.3182***	1.3730***	1.2554***	1.3027***	1.2663**	1.2805***
	(0.49)	(0.51)	(0.52)	(0.45)	(0.42)	(0.53)	(0.46)
Tertiary education	1.2405**	1.2548**	1.3387***	1.1587**	1.1176**	1.1916**	1.2267***
	(0.49)	(0.49)	(0.52)	(0.46)	(0.47)	(0.49)	(0.38)
Full-time employed	Reference	Reference	Reference	Reference	Reference	Reference	Reference
Part-time employed	0.8963**	0.9092**	0.8510**	0.8360**	0.8637**	0.8335**	0.8857**
	(0.37)	(0.39)	(0.39)	(0.35)	(0.35)	(0.34)	(0.36)
Unemployed	-0.3141	-0.3158	-0.3410	-0.3048	-0.3617	-0.3178	-0.1367
	(0.87)	(0.87)	(0.87)	(0.84)	(0.79)	(0.92)	(0.73)
Unable to work	1.0080	1.0066	1.0547	0.9662	0.9603	1.1763	1.1175
	(1.05)	(1.05)	(0.98)	(1.00)	(1.05)	(1.11)	(0.95)
Retired	0.1599	0.1684	0.1264	0.1374	0.1257	0.1133	0.1591
	(0.79)	(0.81)	(0.79)	(0.79)	(0.78)	(0.80)	(0.81)
Student	-0.0033	0.0074	0.0438	-0.0590	0.0455	-0.0612	0.0512
	(0.43)	(0.42)	(0.46)	(0.37)	(0.40)	(0.42)	(0.38)
Other occupation	-0.5159	-0.4974	-0.5365	-0.6407	-0.6504	-0.5500	-0.4538
	(0.52)	(0.51)	(0.48)	(0.55)	(0.52)	(0.50)	(0.48)
Migration background	-0.0483	-0.0483	-0.0022	-0.0386	-0.0662	-0.0570	-0.0088
	(0.24)	(0.24)	(0.28)	(0.25)	(0.25)	(0.25)	(0.26)
Lifetime	0.9514*	0.9571*	0.9459**	0.8911*	1.0202**	0.9958**	0.7730*
	(0.49)	(0.49)	(0.43)	(0.47)	(0.46)	(0.46)	(0.42)

Table 2. Results of main regressions (ordered logit) (continued).

Life satisfaction (LS_{ij})	**Model 1**	**Model 2**	**Model 3**	**Model 4**	**Model 5**	**Model 6**	**Model 7**
<u>Household characteristics (Z_i)</u>							
Child	-0.0056	0.0096	0.0201	-0.0553	-0.0309	-0.1146	-0.0525
	(0.27)	(0.26)	(0.29)	(0.27)	(0.25)	(0.28)	(0.28)
Detached, semi-detached or terraced house	Reference	Reference	Reference	Reference	Reference	Reference	Reference
Small apartment building	0.0579	0.0520	-0.0072	0.1032	-0.0209	-0.1189	-0.1040
	(0.48)	(0.49)	(0.53)	(0.47)	(0.44)	(0.58)	(0.47)
Large apartment building	0.3095	0.3017	0.2638	0.3627	0.2198	0.2464	0.1668
	(0.51)	(0.53)	(0.53)	(0.48)	(0.48)	(0.57)	(0.50)
High-rise building	-0.8268	-0.8272	-1.0324*	-0.7808	-0.8860	-0.8873	-0.9433*
	(0.56)	(0.56)	(0.56)	(0.51)	(0.61)	(0.54)	(0.56)
<u>Environmental variables (A_i)</u>							
Green space		-0.0018	0.0486***[a]				
		(0.01)	(0.01)				
Green space squared			-0.0007***[a]				
			(0.00)				
Distance (in 100 m)				-0.0446*	0.1273[b]		
				(0.02)	(0.09)		
Distance (in 100 m) squared					-0.0104*[b]		
					(0.01)		
Frequency of park visits						0.2437**	
						(0.10)	
Park view							1.0307***
							(0.26)
<u>District controls (S_j)</u>							
Mitte	Reference	Reference	Reference	Reference	Reference	Reference	Reference
Friedrichshain-Kreuzberg	-0.4734***	-0.4831***	-0.5027***	-0.4745***	-0.4591***	-0.5282***	-0.5141***
	(0.12)	(0.12)	(0.14)	(0.13)	(0.14)	(0.13)	(0.11)
Pankow	-0.1733	-0.1844	-0.1657	-0.1275	-0.0881	-0.1422	-0.1402
	(0.14)	(0.17)	(0.17)	(0.12)	(0.11)	(0.15)	(0.12)
Charlottenburg-Wilmersdorf	-0.1476	-0.1650	-0.0373	-0.1279	-0.0919	-0.0933	-0.0378
	(0.09)	(0.14)	(0.16)	(0.10)	(0.10)	(0.11)	(0.10)
Tempelhof-Schönefeld	-0.0100	-0.0239	0.1729	0.0454	0.0743	-0.0368	-0.0010
	(0.18)	(0.23)	(0.22)	(0.18)	(0.18)	(0.19)	(0.17)
Neukölln	-0.2473*	-0.2477*	-0.2149	-0.2466*	-0.3010**	-0.2255*	-0.1600
	(0.14)	(0.14)	(0.14)	(0.13)	(0.15)	(0.13)	(0.15)
Lichtenberg	0.0080	0.0062	0.0330	-0.0743	-0.0407	0.1181	-0.0113
	(0.17)	(0.17)	(0.16)	(0.17)	(0.17)	(0.16)	(0.16)
Pseudo R^2	0.1286	0.1286	0.1360	0.1308	0.1353	0.1337	0.1368
AIC	1125.5218	1125.4284	1116.0261	1122.6291	1116.8953	1118.9911	1115.0470
BIC	1148.0562	1147.9628	1138.5606	1145.1635	1139.4298	1141.5256	1137.5814

Clustered standard errors in parentheses. Cutpoints omitted. The number of observations is 316 for all models. * p<0.1, ** p<0.05, *** p<0.01, [a] jointly significant at 1% level, [b] jointly significant at 5% level, [c] jointly significant at 10% level.

positive effect as opposed to being single. Moreover, self-reported health has a strongly significant, positive, and monotonously increasing impact on life satisfaction.

The impact of education is significantly positive for secondary and tertiary education compared to basic education. The effect of secondary education is slightly bigger than of tertiary education, which is in line with part of the literature (Stutzer, 2004). Being employed part-time as opposed to being employed full-time also has a significantly positive effect on life satisfaction. Previous evidence on the role of part-time and full-time work on life satisfaction is mixed (Dolan et al., 2008). A negative effect of part-time work as opposed to full-time work has been observed for men (Schoon et al., 2005). Estimating Model 1 separately for men and woman reveals that being employed part-time has a significantly positive effect for women, while there is no significant effect for men (results not shown). We find a negative effect of being unemployed, but the coefficient is insignificant (e.g., as in Clark and Oswald, 1994), which might be because there are only 19 unemployed people in our sample.

Interestingly, we also find that respondents who have lived in a certain district of Berlin for their whole lives report significantly higher levels of life satisfaction than others. Regarding the variables that address the living environment of the respondents, we find weak evidence that the respondents' housing conditions affect their well-being. Model specifications 3 and 7 show that people living in a high-rise building are significantly less satisfied with their lives compared to those living in detached, semi-detached, or terraced houses. Regarding district effects, we find that people living in Friedrichshain-Kreuzberg and Neukölln appear to be less satisfied with their lives than people living in Mitte.

4.2.2 Effects of environmental variables

As described in section 1, there are several ways in which urban green space may influence life satisfaction. Regarding the available amount of urban green space in the living environment, Model 2 suggests that there is no significant linear effect on life satisfaction. This seems plausible because the marginal value of additional green space may depend on the current allocation. In line with this reasoning, Model 3 provides evidence of an inverted U-shaped relationship between urban green space and life satisfaction. This implies that additional urban green space first increases life satisfaction but tends to decrease life satisfaction above a certain threshold.[23]

The maximal positive impact of urban green space occurs at an area of 36 ha or 11.5% of the buffer area. This is well above the sample mean of 24.4 ha or 7.8%. Three quarters of the respondents in the sample have less urban green space in their living environment. Thus, measured at the means, increasing the amount of urban green space in the respondents' living environments, *ceteris paribus*, increases their life satisfaction. The two other published studies investigating the effect of urban green space on life satisfaction find a linear positive effect. However, the possibility of non-linear relationships is not considered in those papers.

In Models 4 and 5, we include the distance to the nearest urban green space greater than 5 ha as an alternative measure of the availability of urban green space in the regressions. The Euclidean distance is found to be statistically significant, with a negative effect of increasing distance (Model 4). Adding the squared distance to the model produces an inverted U-shaped relationship between distance and life satisfaction (Model 5), similarly to Model 3. The distance coefficients are jointly significant at the 5% level. Evaluated at the mean distance of 661 m, the impact is negative.[24]

Turning to the self-reported environmental variables, Model 6 suggests a positive relationship between the frequency of park visits and life satisfaction.[25] This finding can be perceived as a support of medical findings on the effect of outside activities on health. Note, however, that park visits may also depend on health and well-being, giving rise to concerns about the reverse causation and endogeneity of this variable.

[23]The green space coefficients are also jointly significant at the 1% level if self-reported health is excluded from the regression.
[24]For urban green space greater than 5 ha, a distance of about 600 m provides the largest positive impact on life satisfaction. In our sample, 46.8% of the respondents live further away from an urban green space of that size.
[25]Results are comparable if the frequencies of summer or winter visits are used.

Note further that the frequency of park visits is related to the availability of green space and the distance to urban green spaces because both objective measures are significant predictors of park use (Schipperijn et al., 2010).[26] Finally, Model 7 shows a strongly positive and significant effect of having a park view on life satisfaction. This suggests support for psychological findings on the positive effects of natural window views on well-being (Kaplan, 2001). However, park views could be particularly prone to a bias arising from self-selection of residential location, as a park view is an attribute that is very specific to single residences.

4.3 Sensitivity analysis and robustness checks

The effects of main robustness checks are shown in Table 3. We compare the results to our preferred model specification, Model 3, which analyzes the effects of an objective green space indicator. We focus on this model because the subjective indicators used are likely to be more vulnerable to biases. In addition, Model 3 shows the best model fit as judged by the information criteria among the models using objective indicators. Robustness checks have also been carried out for the other main specifications, supporting the findings presented below.

4.3.1 Social indicators

We add social data on the level of the planning unit[27] to the regression to check if the effects of urban green space shown in section 4.2 actually capture other local effects. The reason for this is that the social status of the respondents' living environments might be reflected in their endowment with green spaces. In addition to the district dummies already included, we use data on unemployment rates on the level of planning units as an indicator of social status. Model 8 shows that the effects of urban green space are comparable to those of Model 3. Therefore, even when

[26]Also in our sample, there is a significant effect of the availability of and distance to urban green spaces on their use.
[27]See section 3.2.2 for a description of the planning units.

Table 3. Robustness checks for life satisfaction regressions (ordered logit).

Life satisfaction (LS_{ij})	**Model 8 Unemploy-ment**	**Model 9 Lifetime**	**Model 10 Sealing**	**Model 11 Sealing squared**	**Model 12 Status < 4**	**Model 13 Imputed I**	**Model 14 Imputed II**
Log individual income	0.8608*** (0.27)		0.8716*** (0.28)	0.8987*** (0.27)	0.8289*** (0.25)	0.9006*** (0.30)	0.7611*** (0.19)
Social and environmental variables (A_i)							
Unemployment rate in planning unit	0.0361 (0.03)						
Green space	0.0452***[a] (0.01)	0.0127[a] (0.04)			0.0395***[a] (0.01)	0.0521***[a] (0.01)	
Green space squared	-0.0006***[a] (0.00)	-0.0007*[a] (0.00)			-0.0006***[a] (0.00)	-0.0007***[a] (0.00)	
Mean sealing			0.0047 (0.01)	0.0719**[c] (0.03)			
Mean sealing squared				-0.0006**[c] (0.00)			
Imputed green space							0.0349**[a] (0.02)
Imputed green space squared							-0.0005***[a] (0.00)
Imputation dummy						0.3347 (0.75)	
Imputation dummy							-0.0742 (0.27)
Individual demographic and socio-economic characteristics	Yes	No	Yes	Yes	Yes	Yes	Yes
Household characteristics	Yes	No	Yes	Yes	Yes	Yes	Yes
District dummies	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Number of observations	316	33	316	316	355	341	426
Pseudo R^2	0.1366	0.1546	0.1288	0.1298	0.1393	-	-
AIC	1115.2432	111.8479	1125.2198	1123.8907	1245.2297	-	-
BIC	1137.7777	120.8269	1147.7543	1146.4251	1268.4624	-	-

Clustered standard errors in parentheses. Cutpoints omitted. * p<0.1, ** p<0.05, *** p<0.01,
[a] jointly significant at 1% level, [b] jointly significant at 5% level, [c] jointly significant at 10% level.

controlling for the social status of the respondents' living environments, we find a significant effect of urban green space on life satisfaction.

4.3.2 Self-selection of residential location

We further check if our environmental coefficients are biased upwards as a result of the self-selection of residential location. Individuals with higher preferences for urban green space and parks might move to greener areas of the city. As Ambrey and Fleming (2013) note, however, the evidence seems to be mixed, and several authors find that this selection bias is rather small (e.g., Chay and Greenstone, 2005). To investigate the issue further, we use the lifetime variable to split the sample and analyze the effect of urban green space only for the subsample of respondents that have been living in the same district for their whole lives. Because this subsample only includes 33 observations, we only include district dummies as control variables. The results of Model 9 again show the inverted U-shaped effect of urban green space on life satisfaction that is significant at the 1% level. This may be an indicator that the effect of urban green space expands beyond an effect of pure self-selection, even if the results have to be interpreted carefully due to changes in sample size.

4.3.3 Preference heterogeneity

A further issue is that of preference heterogeneity among subsamples with differing demographic or socio-economic characteristics (Menz and Welsch, 2012; Ambrey and Fleming, 2013). One could hypothesize, for example, that families with children or people in certain age groups have stronger preferences for urban green space close to their homes, or that preferences differ with respect to gender. We follow two strategies to investigate this: i.) adding interaction effects and ii.) carrying out group comparisons (results not shown).

First, for income, age, gender, and children, we include interaction terms between these four variables with the green space variable and add them to Model 3 one by one in separate regressions. Neither of these interaction terms is significant, suggesting that there is no significant difference of the effect of urban green space on life satisfaction between the respective groups. Second, for marital status and occupation, we perform separate analyses for subsamples.[28] Regarding marital status, we find that urban green space has a significant effect for those being single and for those living in a relationship. Regarding occupation, we find that the effect of urban green space is significant for the subsample of full-time employees.

Overall, there is thus no clear evidence of the systematic preference heterogeneity in our sample. This is consistent with findings of Ambrey and Fleming (2013), who also find less preference heterogeneity than anticipated. However, these results have to be treated with care, as differences in sample sizes may influence the results.

4.3.4 Smaller buffer areas and alternative environmental variables

We calculated the area of green space available also for buffers with smaller areas. Replacing the 1 km buffer with a 0.5 km or a 0.3 km buffer leads to insignificant results regarding the effect of the green space variable on life satisfaction. In these cases, the buffer areas are getting much smaller, decreasing from 3.14 km^2 for the 1 km buffer to 0.78 km^2 for the 0.5 km buffer and 0.28 km^2 for the 0.3 km buffer. Overall, these smaller buffer areas might be too small to capture the characteristics of a neighborhood regarding their supply with urban green space and to have a significant effect on life satisfaction.

Regarding distance, we tried simple Euclidean, inverse and squared distances to the edge of the nearest urban green space among urban green spaces of all sizes and among urban green spaces bigger than 1 ha, 5 ha, and 10 ha, respectively. All distance measures have significant effects if the distance to the nearest green space greater than 5 ha is considered. No significant effects can be observed for the distance measures using other size thresholds.

So far, we have looked at the availability of public green space and its effect on life satisfaction. However, private green such as gardens or other green infrastructure

[28] As in section 4.3.2, we use a smaller regression model, only including district dummies and the variable urban green space plus its squared value to account for the smaller sample size.

such as street trees may also influence the life satisfaction of city inhabitants, as they can considerably alter the character of a neighborhood. Models 8 and 9 show the effects of average sealing in the respondent's living environment on life satisfaction. As observed, there is no significant linear effect. This does not change when trying out areas with different sizes over which sealing degrees are averaged. Including average sealing in a 300 m buffer and additionally including its squared value, in contrast, yields an inverted U-shaped effect that is significant at the 10% level. The effect is comparable to that of "urban green space" but significance levels are lower for sealing. These results suggest that the average degree of sealing with the largest positive impact on life satisfaction is approximately 60%. This represents a discontinuous but still quite dense urban fabric. Approximately 54% of the respondents live in areas with higher average degrees of sealing of up to 91%.

We also carried out several regressions including noise disturbances as a control variable in addition to the green space variables. We considered mean and maximal noise disturbances both combined for day and night times and for night times only. Like for the variable sealing, noise levels were calculated for different buffer areas with radiuses of 0.3 km, 0.5 km, and 1 km. There was in no case a statistically significant effect of noise levels on life satisfaction. The effects of urban green space on life satisfaction remains stable and significant (results not shown).[29]

4.3.5 Alternative samples and imputed data

To address the concern of potential selection bias due to missing data points in our sample, we ran robustness checks with larger samples and with imputed data sets. Model 12 shows how parameter estimates change if we enlarge the data set to include respondents for whom only ZIP codes are known, increasing the number of observations to 355. Urban green spaces are still highly significant for life satisfaction even though the size of the effect slightly decreases.

[29]Note that in the case of Berlin, the variable noise can also serve as a good proxy for air pollution, because traffic is captured by this variable and because there is no relevant industry located in the city.

Given that we are faced with 50 missing observations on the income variable, we rerun the main regressions using imputed data as a robustness check. First, we only impute income data, using multiple univariate imputation with truncated regression and 50 imputations.[30] The results of Model 13 show that there is still a significant effect of urban green space on life satisfaction with increased effect sizes. The income coefficient also increases and is still highly significant. Second, we impute income and the area of urban green space together. To do so, we treat all observations with a location status greater than two, i.e., all respondents for whom we only know ZIP codes or districts (see Table 1), as missing and impute also the data on green space area. We use multiple multivariate sequential imputation with chained equations and, again, 50 imputations. This increases the sample size to 426 observations. The effects of urban green space are still highly significant. Both the coefficients of income and urban green space decrease (Model 14).

4.4 Valuation of urban green space using the life satisfaction approach

Following the methodology described in section 3.1, we calculate the implicit MRS between income and the environmental variables of green space area and distance, respectively. To account for the non-linear relationships, we calculate the MRS for different combinations of income and green space availability based on equations (3) or (4), depending on the specification. The resulting implicit MRS are shown in Table 4.

Regarding green space area, the implicit MRS is EUR 25.03 per person per hectare per month based on average green space availability and average income. The MRS estimates range from EUR -30.88 for low income and high green space availability to EUR 110.63 for high income and low green space availability.

Comparing our results to the one existing study that values urban green space using the life satisfaction approach, we find that our estimates are significantly

[30]We use multiple imputations as implemented in Stata 13.

Table 4. Implicit marginal rates of substitution (MRS).

		Individual net monthly income		
		Low (587 €)	Mean (1,444 €)	High (2,301 €)
Urban green space (non-linear) (Model 3)	Low (6.1 ha)	28.22	69.43	110.63
	Mean (24.4 ha)	10.17	25.03	39.88
	High (42.7 ha)	-7.88	-19.38	-30.88
Distance (linear) (Model 4)	All	44.93	64.18	94.14
Distance (non-linear) (Model 5)	Low (228 m)	-51.98	-127.87	-203.76
	Mean (661 m)	6.63	16.31	25.99
	High (1,094 m)	65.24	160.49	255.74

Calculations based on equation (3) for Model 4 and equation (4) for Models 3 and 5. Numbers reflect the monetary equivalent (€) of a one hectare increase in the area of urban green space or a one hundred meter decrease in the distance to urban green space, respectively, per person per month. Low (high) values correspond to the mean value minus (plus) one standard deviation.

smaller. Ambrey and Fleming (2013) find an average MRS of AUD 1,168 per household per year for a 143 m^2 increase in green space in the respondent's collection district. This would translate to AUD 81,678 (or EUR 52,640[31]) per household per year for a 1 ha increase in green space. Multiplying the MRS we calculated for average income and average green space availability by 1.9, which is the average household size in our sample, we find a MRS of EUR 571 per household per hectare per year, which is two orders of magnitude smaller. In the CV studies considered by Brander and Koetse (2011), mean WTP is USD 13,210 (EUR 10,200)[32] per hectare per annum, while the median WTP is USD 1,124 (EUR 868), reflecting a rather skewed distribution. This is closer to our MRS estimates and considerably smaller than the estimates of Ambrey and Fleming (2013). This comparison, however, is limited because the CV studies considered by Brander and Koetse (2011) comprise all types of urban open space, including agricultural land and forests.

[31]Converted with an exchange rate of 1.5516 EUR/AUD as of December 31, 2013.
[32]Converted with an exchange rate of 1.2949 EUR/USD as of December 31, 2011.

Our estimated MRS per hectare can also be compared to results from HP studies. Gibbons et al. (2014) present a recent HP study using UK data. Based on their results, the marginal value for urban green space is GBP 196 (EUR 218) per ha.[33] In addition, Perino et al. (2014) present a marginal value function which can be used to calculate the marginal value of a one percentage point increase in informal green space in a 1 km^2 square based on an HP study by Cheshire and Sheppard (1995). Transferred to our case study, where the mean availability of urban green space is 7.78%, the marginal value would be GBP 193 (EUR 214) per ha.

Regarding distance, Perino et al. (2014) report a mean marginal value of proximity to a formal recreation site of GBP 150 (EUR 167[34]) per meter, ranging from a minimum of GBP -41 (EUR -46) to a maximum of GBP 3,348 (EUR 3,720) in the studies considered. The marginal value function derived from their meta-regression shows that marginal values are approximately in a range from GBP 5 to 65 (EUR 6 to 72) per meter depending on the actual distance to the next formal green space. Transferred to Berlin, our case study city, the marginal value would be GBP 6 (EUR 7) per meter.[35] Our estimates provide a mean value of EUR 4 per household per meter per year.

Note, however, that the marginal values derived in these studies are not directly comparable to our estimates. One reason for this is that the values in the studies by Perino et al. (2014), Gibbons et al. (2014), and Brander and Koetse (2011) refer to one-off payments, while our estimates refer to the annual MRS. In addition, it has to be noted that MRS estimates derived using the LSA and the HP method are only directly comparable when wages, rents, and environmental amenities are included in

[33]Based on a mean ward size of 1,038.5 ha and mean green space availability of 51.1%, a one percentage point increase in green space (+10.4 ha on average) would translate into a house price increase of GBP 2,031.

[34]Converted with an exchange rate of 0.9000 EUR/GBP as of December 31, 2009.

[35]The marginal value per meter is positively influenced by park size and negatively influenced by distance, income and population. For our sample, park size, distance, and income were set to their means and adjusted as required. Population was set to 3.4 million. The low marginal value derived for Berlin is mainly due to the large population; the calculations for the UK are based on a nationwide average population of 200,000 per city.

the life satisfaction regression and when the housing market is in equilibrium. If rents are not included in the life satisfaction regressions, which is mostly the case in the life satisfaction literature, the estimated effect of the environmental amenity on life satisfaction only includes the residual effect, i.e., the part of the externality that is not compensated for in the housing market (Ferreira and Moro, 2010). Consequently, our estimates only capture the residual effect of urban green space on life satisfaction.

5 Discussion and conclusions

We use four individual green space measures to explore how urban green space affects the well-being of the residents of Berlin, Germany. We combine spatially explicit data derived from a customized web survey with spatially highly disaggregated GIS data on urban green spaces to carry out our analyses.

The first measure we analyze is the amount of urban green space in a certain buffer area around a respondent’s residential address. This objective measure combines the notion of distance with the notion of the absolute availability of urban green space. Both aspects are significant determinants of the actual use of urban green space (Schipperijn et al., 2010). This green space measure may thus reflect the degree to which green spaces are actively used but also the degree to which they are able to mediate adverse environmental impacts. The impacts of increased outdoor activity levels may also be captured by the objective indicator “distance to the nearest green space greater than 5 ha” and in the subjective indicator “frequency of park visits,” which we also analyze. Moreover, “park view” is included in a separate regression to see whether we can confirm the positive effects of views onto natural elements found in the psychological and medical literature.

With respect to the objective indicators, our results suggest that the effect of the available amount of urban green space is non-linear with the marginal utility of green space first increasing and then decreasing. This is supported by the effects of distance and sealing, which also show significant non-linear, inverted U-shaped effects. One possible explanation for an inverted U-shaped relationship might be that living very close to urban green spaces may not only be associated with amenities but also with disamenities arising, e.g., from noise, congestion or fear of crime (Bixler and Floyd, 1997; Kuo et al., 1998). This issue has also been discussed in the hedonic pricing literature, suggesting that the positive effect of parks on the values of nearby properties very much depends on the quality and usage of the park (Crompton, 2001). In addition, it seems plausible that people living in urban environments not only have preferences for urban green but also have preferences for living close to infrastructure, shops, schools, or work. Studies using the LSA to investigate scenic amenities (Ambrey and Fleming, 2011) or land cover (Kopmann and Rehdanz, 2013) support the existence of non-linearities.

Further, our results suggest that the amount of urban green space in a 1 km buffer around residential addresses that lead to the largest positive effect on life satisfaction is 36 ha, or 11.5% of the buffer area. As three-quarters of the respondents have less than this amount of urban green space available in their living environments, green space is, overall, in insufficient supply in the case study area in Berlin. This also implies positive MRS estimates evaluated at the means of green space area and income. Based on mean green space availability and mean income, the implicit MRS is EUR 25 per person per hectare per month. For city management, our results imply that policies should aim at increasing the supply of green spaces in areas where they are particularly scarce. Moreover, a more homogenous supply of urban green space should be targeted.

An explanation for the comparably low MRS estimates could be that we only capture the residual effect of urban green space on life satisfaction that is not compensated for in the housing market. Rents are likely to be particularly high in areas with a large supply of urban green space which would tend to decrease life satisfaction. The high provision of urban green space would increase life satisfaction but the residual effect might be low and comparable to that observed in areas with a very low provision of urban green spaces and low rents. Our results suggest that the residual positive effect of urban green space on life satisfaction is largest in areas with an intermediate supply of urban green spaces, maybe because rents are relatively lower in these areas such that the mix between green space provision and rents is perceived to be better and the residual effect on life satisfaction is larger.

The analyses of the subjective green space indicators show that both the frequency of park visits and park views have positive well-being effects, with the effect of park views being particularly strong and significant. The strong effect of park views, however, may also be caused by the self-selection of the residential location. It seems plausible that park views are more prone to bias from self-selection than the variable of urban green space because the latter captures the character of a relatively large area, while a park view is an attribute that is very specific to single residences. Moreover, the effect of park visits may be biased by reverse causation, as life satisfaction itself may influence the frequency of park visits. We thus do not calculate MRS for the subjective green space indicators.

Despite the fact that we can show positive effects of green space on life satisfaction over a broad range of specifications and robustness checks, some issues arise that may bias the estimated income coefficients and, therefore, the estimated implicit MRS.

Endogeneity of income may be a potential issue. One reason for this is that the observed relationship between economic conditions and well-being could be explained by unobserved heterogeneity if personality traits are not considered as control variables (see, e.g., Boyce, 2010). It seems likely that on average, happier individuals tend to lose their job less often, to be re-employed more easily, or to find jobs that are better paid (see Frey and Stutzer (2002) for a discussion). This might bias the income coefficient upwards and thus bias the MRS downwards. Introducing character trait controls may reduce the potential upward bias from unobserved heterogeneity. Unfortunately, it was outside the scope of our survey to include questions about character traits so that we cannot control for unobserved individual heterogeneity in our cross-sectional data set.

Additional bias may arise from the fact that people compare their income to the incomes of other individuals (e.g., Clark et al., 2008). Because comparisons to other individuals' incomes are mostly made upwards, this effect can bias the income coefficient downwards if no appropriate controls are included in the regressions. Some studies try to instrument income to avoid potential biases (e.g., Powdthavee, 2010). However, instrumenting is not without problems either, because it is very difficult to find a valid instrument for income. There is thus no agreement yet on how to instrument income, particularly in cross-sections.

Overall, the directions of potential biases of our income coefficient are ambiguous. Our income coefficients are larger than in studies using panel data (e.g., Levinson, 2012) or repeated cross-sections (e.g., Ferreira et al., 2012) but comparable to other cross-sectional studies (e.g., Ferreira and Moro, 2010). Given that we report relatively high income coefficients, our MRS estimates are likely quite conservative estimates of real MRS. Further limitations to our study arise from the fact that we cannot control for the quality of urban green spaces, which may influence well-being effects. However, currently available GIS data do not contain the necessary information, so a refinement of the analysis in this direction has to be deferred to future research.

Despite these limitations, and even though more research is needed to refine MRS estimates from studies using the LSA, our study underlines the importance of urban green space for city inhabitants. Particularly in inner-city areas, green space is often in insufficient supply. This is also the case for our Berlin case study, where three-quarters of the respondents have less green space available in their living environment than the amount at which the positive impact on life satisfaction would be maximized. Based on our well-being analysis, increasing the average supply of urban green space and aiming at a more homogenous distribution would be preferable to current allocations.

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Appendix A. Overview of the data used for the analysis.

Table A-1. Definitions and summary statistics of demographic, socio-economic, and environmental variables.

Variable name	Definition	Mean	Std. dev.	Min	Max
Life satisfaction (LS_{ij})					
Life satisfaction	Overall life satisfaction; measured on a Likert scale from 0 "very dissatisfied" to 10 "very satisfied"	6.728	2.08	0	10
Individual demographic and socio-economic characteristics (Y_i, X_i)					
Log individual income	Natural logarithm of the total net monthly individual income in Euros	7.117	0.59	3.912	8.923
Gender	Gender dummy; 1 if "male", 0 if "female"	0.509	0.50	0	1
Age	Age; measured in years	45.97	14.31	18	78
Single	Marital status dummy; 1 if "single", 0 else; reference category	0.310	0.46	0	1
Married	Marital status dummy; 1 if "married", 0 else	0.342	0.48	0	1
Partner	Marital status dummy; 1 if "living in a relationship", 0 else	0.228	0.42	0	1
Separated[a]	Marital status dummy; 1 if "separated", 0 else	0.019	0.14	0	1
Divorced[a]	Marital status dummy; 1 if "divorced", 0 else	0.089	0.28	0	1
Widowed[a]	Marital status dummy; 1 if "widowed", 0 else	0.013	0.11	0	1
Very bad health[a]	Health dummy; 1 if "Very bad health", 0 else; reference category	0.035	0.18	0	1
Bad health[a]	Health dummy; 1 if "Bad health", 0 else	0.177	0.38	0	1
Fair health	Health dummy; 1 if "Fair health", 0 else	0.275	0.45	0	1
Good health	Health dummy; 1 if "Good health", 0 else	0.380	0.49	0	1
Very good health	Health dummy; 1 if "Very good health", 0 else	0.133	0.34	0	1
Basic education	Education dummy; 1 if education on ISCED level 1 or 2[b]; reference category	0.073	0.26	0	1
Secondary education	Education dummy; 1 if education on ISCED level 3 or 4[b]	0.453	0.50	0	1
Tertiary education	Education dummy; 1 if education on ISCED level 5 or 6[b]	0.475	0.50	0	1
Full-time employed	Occupation dummy; 1 if "full-time employed", 0 else; reference category	0.462	0.50	0	1
Part-time employed	Occupation dummy; 1 if "part-time employed", 0 else	0.130	0.34	0	1
Unemployed	Occupation dummy; 1 if "unemployed", 0 else	0.060	0.24	0	1
Unable to work	Occupation dummy; 1 if "unable to work", 0 else	0.038	0.19	0	1
Retired	Occupation dummy; 1 if "retired", 0 else	0.155	0.36	0	1
Student	Occupation dummy; 1 if "student", 0 else	0.092	0.29	0	1
Other occupation	Occupation dummy; 1 if "other occupation", 0 else	0.060	0.24	0	1
Migration background	Dummy variable; 1 if at least one parent of the respondent is of a nationality other than German, 0 else	0.335	0.47	0	1

Table A-1. Definitions and summary statistics of demographic, socio-economic, and environmental variables (continued).

Variable name	Definition	Mean	Std. dev.	Min	Max
Lifetime	Dummy variable; 1 if respondent has been living in the same district for her whole life	0.104	0.31	0	1
Household characteristics (Z_i)					
Child	Dummy variable; 1 if at least one child under the age of 12 is living in the household, 0 else	0.111	0.31	0	1
Detached, semi-detached, or terraced house	Housing dummy; 1 if "detached, semi-detached or terraced house", 0 else; reference category	0.070	0.25	0	1
Small apartment building	Housing dummy; 1 if "apartment building with 3 to 8 apartments", 0 else	0.294	0.46	0	1
Large apartment building	Housing dummy; 1 if "apartment building with 9 or more apartments (but no high rise) ", 0 else	0.541	0.50	0	1
High-rise building	Housing dummy; 1 if "high rise", 0 else	0.095	0.29	0	1
Environmental variables (A_j)					
Green space	Hectares of urban green space in a 1 km buffer area around the respondent's home	24.43	18.26	1.094	98.120
Distance	Distance to nearest urban green space > 5 ha; measured in metres	660.84	432.6	0.5	2495.3
Frequency of park visits	Frequency of park visits; measured on a 6-point scale from 0 "never" to 5 " (almost) daily"	1.932	1.20	0	5
Park view	Dummy variable; 1 if respondent has a view of a park from his/her home, 0 else	0.130	0.34	0	1
District controls (S_j)					
Mitte	District dummy; 1 if respondent lives in this district, 0 else; reference category	0.165	0.37	0	1
Friedrichshain-Kreuzberg	District dummy; 1 if respondent lives in this district, 0 else	0.092	0.29	0	1
Pankow	District dummy; 1 if respondent lives in this district, 0 else	0.155	0.36	0	1
Charlottenburg-Wilmersdorf	District dummy; 1 if respondent lives in this district, 0 else	0.136	0.34	0	1
Tempelhof-Schöneberg	District dummy; 1 if respondent lives in this district, 0 else	0.187	0.39	0	1
Neukölln	District dummy; 1 if respondent lives in this district, 0 else	0.114	0.32	0	1
Lichtenberg	District dummy; 1 if respondent lives in this district, 0 else	0.152	0.36	0	1

The number of observations included is 316 for all variables.

[a] In the regression analyses in section 4, the dummy variables "separated", "divorced", and "widowed" as well as "very bad health" and "bad health" are merged into one variable respectively due to the low number of observations.

[b] For an overview of how the German educational achievements translate into the internationally comparable ISCED levels see Statistisches Bundesamt (2010).

Appendix B. Survey details.

The survey was implemented using a pre-selected web panel with approximately 100,000 registered members throughout Germany and 4,000 members in Berlin aged 18 years or above. It was implemented and executed via a professional polling agency. Potential survey participants were invited via email to participate in the survey. In this email, a link to the survey was provided, but the topic of the survey was not further specified. Having clicked on the link, potential participants were directed to a standard starting screen, which clarified the expected length of the survey and the potential reward to be gained. The topic announced on this starting screen was kept very general, only indicating that the survey was related to city life, to avoid self-selection in terms of interest in environmental issues. In total, the questionnaire consisted of between 25 and 45 questions, depending on whether the respondents regularly visited parks or not.

An early version of the questionnaire was pretested with university students. The final version of the survey was pretested with a set of 50 participants. Responses to the survey were checked according to several quality criteria, including the time taken to complete the survey and obvious answer patterns that revealed that respondents had clicked through the survey without paying attention to questions and answers. Approximately 10% of the observations have thus been eliminated from the sample. In addition, a thorough validity test was carried out, checking answers for obvious inconsistencies in content. This led to a further exclusion of 25 observations from the sample.

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Part II: Managing marine ecosystem services and biodiversity

Paper 3: On the environmental effectiveness of the EU Marine Strategy Framework Directive[1]

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Abstract:

Marine and coastal ecosystems – and thus the benefits they create for humans – are subject to increasing pressures and competing usages. For this reason, the European Union (EU) adopted the Marine Strategy Framework Directive (MSFD), which is to guide future maritime policy in the EU and aims at achieving or maintaining a good environmental status (GES) of European seas by 2020. To this end, the MSFD requires the development of improvement measures, which have to be assessed *inter alia* by examining their cost-effectiveness and by carrying out cost-benefit analysis (CBA) before their implementation. This paper investigates the applicability of environmental CBA in the marine context. It identifies and discusses problems that could hamper the environmental effectiveness of the MSFD. For example, the fact that marine ecosystem services are much less tangible than terrestrial ones implies greater challenges for the quantification of benefits for society in a marine context. One finding is that the limitations of environmental valuation methods regarding their ability to capture the whole total economic value of improvement measures are a potential source of problems, as the MSFD allows countries to disregard measures with disproportionately high costs. The transboundary nature of the main European seas adds to the complexity of the valuation task, e.g., due to the danger that benefits that occur outside of national territories are neglected. Moreover, the current state of knowledge on the functioning of complex marine ecosystems and the links to socio-economic impacts and human well-being seem insufficient to meet the MSFD requirements.

Keywords: cost-benefit-analysis, ecosystem services, environmental valuation, EU Marine Strategy Framework Directive, Europe

JEL classification: Q51, Q53, Q57, Q58

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1 Introduction

Marine and coastal ecosystems are important for humans in multiple ways. They provide a number of goods and services which are used directly and indirectly by humans. These goods and services include the provision of food, energetic and mineral resources, but also the regulation of important ecological functions such as the climate system. Moreover, the ocean offers transport routes and recreational opportunities. However, marine and coastal ecosystems – and thus the benefits they create for humans – are subject to increasing pressures and competing usages (Nunes et al., 2009; Luisetti et al., 2011). These pressures result, e.g., from intensified fishing efforts, nutrient enrichment, increasing maritime transport, pollution, noise, sediment sealing and increasing ocean acidification caused by anthropogenic CO_2 emissions. Despite their great importance, goods and services provided by marine and coastal ecosystems have received far less attention than those provided by terrestrial ecosystems – maybe due to differences in access and direct experience (COWI, 2010; TEEB, 2009).

From a European policy perspective, increasing threats to the marine environment resulting from human use have been recognized, and there are several regulations that aim at managing the human impact on the marine environment.[2] Most recently, the European Union (EU) adopted the Marine Strategy Framework Directive (MSFD)[3] in 2008, which is to guide future maritime policy and aims at achieving or maintaining a good environmental status (GES) of Europe's seas by 2020. The MSFD requires an assessment of how humans use the marine environment and the development of action plans and explicit measures to achieve a GES by 2020. Before their implementation, these measures *inter alia* need to be assessed by examining their cost-effectiveness and by carrying out cost-benefit analysis (CBA).

[2]Measures taken include the introduction of marine protected areas, fishing quotas, and measures to prevent pollution. There are two international conventions that focus on the North Sea and the Baltic Sea respectively, the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR, 1992) and the Convention on the Protection of the Marine Environment of the Baltic Sea Area (HELCOM, 1974). The European Water Framework Directive (WFD, 2007) is related to the provisions of OSPAR and HELCOM, as it aims at establishing a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater.
[3]Directive 2008/56/EC (EU, 2008). The MSFD entered into force on 17 June 2008.

While the costs of such improvement measures are often relatively easy to determine, e.g., in terms of foregone revenues, the determination of the associated benefits is more challenging for at least two reasons. The first difficulty is to trace how a change in the marine biosphere (e.g., less marine litter or lower levels of nutrient loads) that leads to a change in the provision of ecosystem goods or services finally affects benefits for humans. Second, the associated benefits need to be quantified in monetary terms to carry out a CBA. Many ecosystem goods and services, particularly those created in a marine environment, are not traded on markets and thus prices, as an indicator for values, do not exist. Environmental valuation methods can be used to value such non-market goods and services.

The aim of this paper is to discuss the challenge to value marine ecosystem goods and services in the context of the MSFD, which requires the application of an ecosystem-based approach to the management of human activities affecting the marine environment (Art. 1.3 MSFD).[4] Some scoping studies have been carried out that examine the economic requirements of the MSFD and review the existing literature on marine ecosystem goods and services and their valuation. COWI (2010) identifies explicit and implicit economic requirements of the MSFD and assesses the possible role that economic analysis can play in its implementation. Turner et al. (2010) present different methodological tools that can be used to analyze the role of socio-economic drivers and responses in environmental-economic systems[5] and provides an overview of valuation studies on marine ecosystem services in European countries.

This paper contributes to the existing literature by assessing the limitations of environmental valuation and CBA in the marine context and by highlighting the possible consequences; the environmental effectiveness of the MSFD might be hampered and the GES might not be achieved. Existing valuation studies, for example, tend to look at changes in tangible benefits like recreation and food

[4]This approach is based on the recommendations of the Millennium Ecosystem Assessment (MA, 2005) as well as the study on The Economics of Ecosystems and Biodiversity (TEEB, 2010), which both call for a holistic valuation approach based on the concept of ecosystem services.

[5]These tools include the Driver-Pressure-State-Impact-Response framework, scenario analysis, and cost-benefit analysis (CBA), including the corresponding theoretical background.

provision but mostly ignore changes in more intangible benefits derived, e.g., from ecosystem functioning or resilience. However, it might be these services that are more important for sustainable development and society as a whole. A CBA that ignores such services will most likely underestimate the true value of marine ecosystem goods and services significantly. Since the costs of improvement measures are easier to determine, this in turn might reduce the probability of measures being implemented.

This reasoning is illustrated in more detail by the example of eutrophication, which is listed as a pressure in the MSFD (App. III, Table 2 MSFD). Unlike other pressures, eutrophication is one of the few pressures identified by the MSFD that is scientifically relatively well understood and for which a number of economic valuation studies exist. Moreover, eutrophication is one of the leading causes of water quality impairment around the world and a major problem in Europe.[6] In this paper, background knowledge from natural sciences is combined with economic methodologies. The concept of the total economic value (TEV) is reconsidered and applied to this complex environmental problem to better demonstrate the challenges for economic assessments. This is most probably the first paper that identifies gaps in knowledge that might affect the environmental effectiveness of the MSFD, based on the most recent studies that evaluate economic benefits of eutrophication reductions, and also taking into account the recommendations prompted by the Millennium Ecosystem Assessment (MA, 2005) as well as the study on The Economics of Ecosystems and Biodiversity (TEEB, 2010) and their reflection in the MSFD requirements. In particular, the paper shows that the complex interactions between ecological effects and human well-being considerably increase the challenge for environmental valuation in the marine context.

The paper is organized as follows. In section 2, the main MSFD requirements are presented with a special focus on the provisions that contain economic terms. In

[6]In 2008, a global review identified 415 areas worldwide which experienced symptoms of eutrophication, of which only 13 were classified as recovering (Selman et al., 2008). Though progress has been made in Europe, eutrophication is still a major problem in Europe’s seas – not only in the Baltic and the North Sea but to some extent also in the Mediterranean Sea and the Black Sea (Coll et al., 2010; Remoundou et al., 2009).

section 3, important concepts underlying economic valuation of ecosystem goods and services are highlighted, economic valuation methods are briefly reviewed and related to the marine context. In section 4, the ecological aspects of eutrophication are sketched, and the complexity of the interactions between ecological eutrophication effects and human well-being is highlighted. Moreover, the valuation literature on eutrophication in European seas is reviewed and the challenges of environmental valuation and CBA in the context of eutrophication are illustrated. In section 5, the implications for the environmental effectiveness of the MSFD that are implied by the economic requirements of the MSFD, by the nature of the environmental valuation methods, and by the interdisciplinary nature of environmental valuation are discussed in detail. Section 6 concludes.

2 Requirements of the MSFD

The aim of the MSFD is to effectively protect the marine environment in Europe and to sustain the associated natural resource base, which is essential for a number of marine-related economic and social activities. To this end, the MSFD aims at achieving or maintaining a GES of Europe's seas (Baltic Sea, Northeast Atlantic, Mediterranean Sea, and Black Sea) by the year 2020 (Art. 1.1 MSFD). The MSFD constitutes an important cornerstone of the EU's future maritime policy and aims at promoting the integration of environmental considerations in all relevant policy areas (Preamble, no. 3 MSFD).

To this end, the MSFD requires EU Member States (MS) to develop marine strategies for their marine waters (Art. 5.1 MSFD) in order to preserve or restore marine ecosystems and prevent their deterioration (Art. 1.2 (a) MSFD). These marine strategies shall apply an ecosystem-based approach to the management of human activities affecting the marine environment and ensure a sustainable use of marine goods and services by present and future generations (Art. 1.3 MSFD). The marine strategies shall include i.) an initial assessment of the current environmental status of the marine waters, including the environmental impact of human activities thereon, ii.) a description of the GES, including the selection of a series of environmental targets and associated indicators, iii.) a monitoring program for the ongoing assessment and regular updating of targets, and iv.) a program of measures designed to achieve GES (Art. 5.2 (a-b) MSFD).

To take account of the transboundary nature of marine waters, the MSFD defines marine regions and subregions according to geographical and ecological criteria. MSs sharing a marine region or subregion shall cooperate in developing their national marine strategies to ensure coherence and coordination (Art. 5.2 MSFD). The MSFD also requires MSs to take into account transboundary effects of measures in the same marine region or subregion (Art. 2.1; also Art. 8.3(b), 14.1(d), 13.8).

The MSFD explicitly requires MSs to take into account social and economic aspects when preparing and implementing their marine strategies. The four key economic requirements of the MSFD are presented in the following:[7]

- Initial assessment of a MS's marine waters, including an economic and social analysis (ESA) of the use of those waters, and of the cost of degradation of the marine environment (Art. 8.1(c) MSFD)
- Establishment of environmental targets and associated indicators describing GES, including due consideration of social and economic concerns (Art. 10.1 in connection with Annex IV, no. 9 MSFD)
- Identification and analysis of measures needed to be taken to achieve or maintain GES, ensuring cost-effectiveness of measures and assessing the social and economic impacts including cost-benefit analysis (Art. 13.3 MSFD)
- Justification of exceptions to implement measures to reach GES based on disproportionate costs of measures taking account of the risks to the marine environment (Art. 14.4 MSFD)

[7]See COWI (2010) for a more detailed review of the economic requirements of the MSFD.

Economic considerations are thus central for developing the marine strategies required by the MSFD. For example, CEA and CBA have to be carried out before the implementation of any new measure to reach GES. Moreover, economic considerations are likely to play a major role for justifying exceptions from the requirement to reach GES. Several reports (COWI, 2010; Eftec/Enveco, 2010), including a guidance document published by the European Working Group on the Economic and Social Assessment (EU WG ESA) in December 2010 (EC, 2010), aim at clarifying the role of economic analysis for the implementation of the MSFD. Still, in a number of cases, it is not yet clear how economic considerations interact with each other and with other disciplinary considerations required by the MSFD. This is discussed in more detail in section 5 of this paper.

3 Environmental valuation in the marine context – underlying concepts and valuation methods

3.1 Underlying concepts

As mentioned in the previous section, the MSFD requires the application of an ecosystem-based approach to the management of human activities. This approach should also be followed when marine strategies, including the programs of measures to achieve a GES, are designed (Art. 1.3 MSFD). It acknowledges that intact marine ecosystems provide a wide variety of benefits to society through the goods and services they offer. Moreover, it emphasizes that ecosystems as a whole are important for humans. There are different approaches used to categorize ecosystem goods and services and the benefits they create for humans; two very important ones are the approach of the MA (2005) and the approach of the total economic value (TEV) (Pearce and Turner, 1990).

The MA approach highlights the complex interactions between ecosystem services, human behavior, and well-being. While humans impact on ecosystems directly and indirectly and on different scales, this alters the services provided by ecosystems, which then influences human well-being and feeds back into decision-making and direct and indirect drivers of change (TEEB, 2010). Ecosystem services are grouped into provisioning, regulating, cultural, and supporting services (MA, 2005). Relating to marine ecosystem services, provisioning services include the supply of fish, seafood, and medicinal plants. Regulating services include climate regulation, and water purification. Cultural services include spiritual, aesthetic, and recreational values, and supporting services include habitat provision and primary production (see also Table 1).

Table 1. Marine ecosystem goods and services.

Provisioning services	**Regulating services**
• Provision of food • Provision of genetic resources/medicine • Provision of energy (wind, wave, tide) • Provision of other renewable resources for other purposes (jewelry, souvenirs, etc.) • Provision of non-renewable resources • Provision of space and transport routes	• Gas and climate regulation • Storm and flood protection • Erosion control • Bioremediation of waste • Water purification and detoxification
Cultural services	**Supporting services**
• Recreation and leisure • Aesthetics and inspiration • Cultural heritage and identity • Spiritual and religious values • Science and education	• Primary production • Biogeochemical cycling • Ecosystem stability and resilience • Habitats • Food web dynamics • Biodiversity

Classification based on Arcadis Belgium (2010).

The TEV approach tries to capture all components that contribute to the value of ecosystem goods and services for humans. It divides the total value into use values and non-use values. Use values can further be divided into direct use values, indirect use values and option values. Non-use values can further be divided into existence values, bequest values and altruistic values (Pearce and Turner, 1990; see Figure 1 for examples in the marine context). The two concepts are interrelated. Regulating services mostly contribute to indirect use values, while provisioning and cultural services mostly create direct use values, which may be consumptive or non-consumptive. Cultural values according to the MA also create non-use values. All three ecosystem service categories can also provide option values. Supporting services are valued through the other categories of ecosystem services to avoid double counting (TEEB, 2010).

Ecosystem goods and services thus provide benefits to humans but their protection is costly. Consequently, measures that aim at protecting the marine environment may carry opportunity costs, and there will always be a need to choose between different conservation measures or to weigh conservation against other investment opportunities. Choosing between different measures or policies requires a thorough analysis of the pros and cons, the benefits and costs related to each of them. There are different forms of appraisal that use different sets of decision criteria. Box 1 provides a short overview of important appraisal methods.

An assessment of the costs and benefits related to a measure to protect the marine environment needs to distinguish between a financial and an economic analysis and thus between prices and values. Price, which is mostly used in financial analysis, is only that portion of value which is realized in markets. If markets are competitive and function without further distortions, prices may be a good approximation for value, i.e., for the relative scarcity of a good or service. If public goods are concerned or external effects exist, prices are biased and do not reveal the value attached to an ecosystem good or service. For most environmental goods and services, markets and thus prices do not exist at all. Economic analysis aims at unveiling the value of a change in the provision of such goods and services, incorporating as many constituents of value as possible (Turner et al., 2010; Bateman et al., 2011).

While it is often relatively easy to determine the costs of conservation measures, e.g., through foregone revenues, it is much more difficult to elicit the associated benefits of these measures. Environmental valuation provides a way to make explicit in monetary terms the benefit flows generated by natural capital stocks and the effects of human decisions on these benefit flows.

Box 1: Methods for project appraisal

One method, which is often used for project appraisal, is cost-benefit analysis (CBA). It aims at eliciting the welfare gain or loss for society related to a certain policy or project. Therefore, it involves identifying and measuring in monetary terms the costs and benefits associated with this policy or project. In this context, costs relate to welfare losses and benefits relate to welfare gains. Benefits or costs that cannot be monetized are often left out of the analysis. However, they can and should be integrated in qualitative terms.

A second method for project appraisal is cost-effectiveness analysis (CEA). It aims at finding a policy which can reach a predefined target at least cost. At this point, marginal costs are equal among policy options. Compared to CBA, the benefits of the policy do not have to be elicited as they are now held fix via the predefined target. This way of appraisal only refers to cost minimization, not to finding a policy with the most favorable relationship between benefits and costs.

A third method is multi-criteria-analysis (MCA). It offers a framework to rank different policy options according to well-specified evaluation criteria. Compared to CBA and CEA, these criteria do not have to be expressed in monetary terms, they only have to be measurable in some way. Moreover, MCA allows for stakeholder involvement and deliberation.

(See Turner et al. (2010) and references cited therein for more details.)

Environmental valuation takes an anthropocentric view and is based on people's preferences for ecosystem goods and services. This implies that values can only be assigned to ecosystem services in so far as they fulfill human needs or bring about satisfaction for humans, thus contributing directly or indirectly to human well-being. Several methods have been developed that aim at eliciting the value people attach to ecosystem goods and services (see section 3.2). All methods have in common that they investigate how people's preferences are affected if there is a marginal change in the provision of a certain ecosystem good or service. Therefore, environmental valuation is not suited for the valuation of whole ecosystems. Moreover, environmental valuation is subjective and context-dependent (TEEB, 2010; Turner et al., 2010).

3.2 Valuation methods

The key question in environmental valuation is what is the maximum that a household would be willing to pay (WTP) for an improvement in environmental conditions, or alternatively, what is the minimum that the household would be willing to accept (WTA) as compensation for a move to an inferior situation. The existing environmental valuation methods can be classified into direct market valuation methods, revealed preference methods, and stated preference methods.[8] Direct market valuation methods use market data which is directly available for ecosystem goods that are traded on markets. Revealed preference methods also assume that consumer preferences can be revealed by their purchasing habits. They use the relationship between a non-market ecosystem service and a market good or service to estimate the WTP or WTA for a change in the ecosystem service. Stated preference methods, by contrast, use structured questionnaires to elicit people's preferences for a change in a certain ecosystem service. See Figure 1 for an overview of existing valuation methods and their applicability in the context of the TEV.

3.2.1 Direct market valuation methods

The *market price method* estimates economic values for ecosystem goods or services that are bought and sold in commercial markets, e.g., the market for fish and fish products.[9] Direct and indirect use values can be captured but non-use values cannot.

[8]For an overview on the theory of the individual methods see Freeman (2003). See TEEB (2010) for a discussion of their applicability, advantages, disadvantages, and limitations.
[9]If markets are distorted, prices may need to be adjusted.

The *production function method* estimates how much a certain ecosystem service contributes to the provision of another ecosystem good or service, which is typically traded on commercial markets. This method is able to capture indirect use values.

3.2.2 Revealed preference methods

Individuals can buy market goods and services to defend against negative environmental impacts (*averting behavior*). In the marine context, an example could be special shoes that are bought because a beach is littered. This approach can capture direct and indirect use values.

The *hedonic method* assumes that property prices are determined by the characteristics of the property, including environmental characteristics such as a pleasant view. The value of ecosystem goods and services would thus be capitalized into property prices. Hedonic pricing can measure direct and indirect use values but its applicability in the marine context is limited.

The *travel cost (TC) method* is a survey-based method used to estimate recreational values associated with ecosystems or sites. Today, studies are mostly based on random utility models (RUM) to value changes in the quality or the quantity of an environmental characteristic at a particular site. The approach captures direct use values.

3.2.3 Stated preference methods

The *contingent valuation (CV) method* uses questionnaires to create a hypothetical market and to ask people for their WTA or their WTP for a change in a certain ecosystem service. The approach can, in principal, capture all elements of the TEV. However, surveys need to be explicit about the type of value that is to be elicited.

In *choice experiments (CE)*, people are asked to choose among sets of ecosystem services or environmental characteristics. Unlike CV, people are not directly asked for their WTP or WTA. This information is inferred from the trade-offs they make. For example, people can choose between different scenarios of water quality, characterized by different attributes such as water clarity or species abundance and the price that would have to be paid to achieve this state. Choice modeling can, again, capture all elements of the TEV.

Stated preference methods are very flexible and can be applied to a wide range of contexts. Also, they are the only methods that can estimate non-use values. It seems plausible to assume that in the marine context, where ecosystem goods and services are less visible than on land, non-use values are particularly important to consider.

3.2.4 Benefit transfer

Benefit transfer consists of an analysis of information provided by one single valuation study or a group of studies from the existing literature to value similar goods or services in another context. For this reason, it can only cover those elements of the TEV that were included in the original studies. Benefit transfer comprises point estimate transfer, functional transfer, and, more recently, meta-analysis.

Each of the valuation methods presented in this section has characteristic advantages and disadvantages and may be suited only for the valuation of certain ecosystem goods and services (DEFRA, 2007), but a comprehensive review of these specific advantages and disadvantages is beyond the scope of this paper. For an overview see TEEB (2010), Turner et al. (2010), and Bateman et al. (2011).

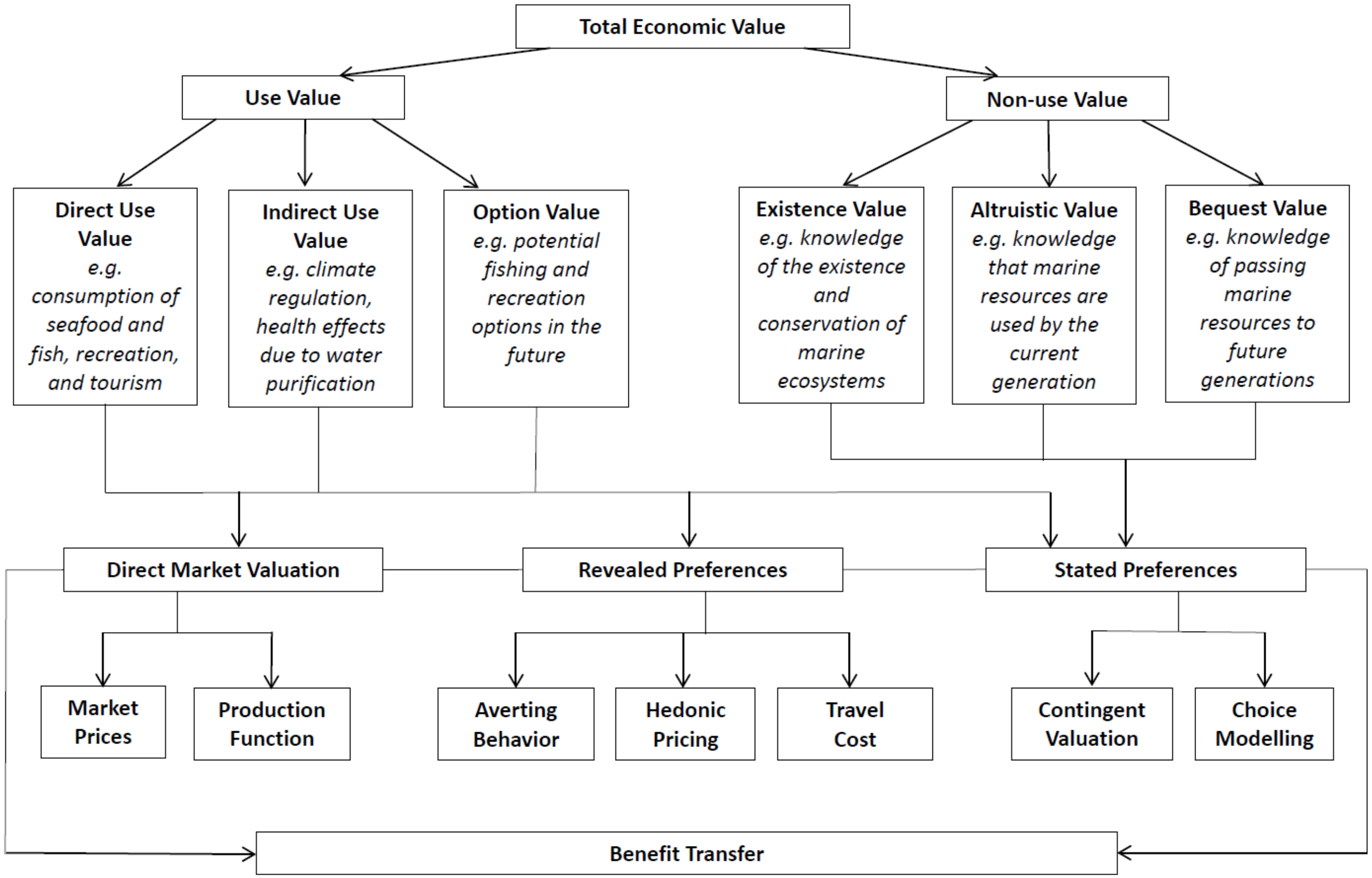

Figure 1. The concept of total economic value (TEV) and existing valuation methods.

Adapted from: Review of Technical Guidance on Environmental Appraisal (DETR/eftec, 1999), in DEFRA (2007), p. 34. Additional information from Nunes et al. (2009), TEEB (2010), and Remoundou et al. (2009).

4 Eutrophication in European marine and coastal ecosystems

4.1 Interrelation between the ecological and the human dimension

Eutrophication remains a major problem in all enclosed seas and sheltered marine waters across the pan-European region (EEA, 2007).[10] The effects of eutrophication are most pronounced in regional seas which have a combination of a high population density in the catchment area and physiographic characteristics predisposing the sea to nutrient enrichment (HELCOM, 2009), such as the Baltic Sea or the Mediterranean Sea. Eutrophication causes complex changes within ecosystems. These changes in the biophysical sphere influence the extent to which marine environments are able to provide ecosystem goods and services to humans. Consequently, also human activities and benefits will be influenced by changes in the environmental state of the seas. Figure 2 provides a detailed overview of ecological eutrophication effects and their interaction with human activities and benefits via an alteration of the provision of ecosystem services. The complex interactions sketched in the figure also illustrate the implications for CBA required by the MSFD if an ecosystem-based approach is to be followed.

4.1.1 The ecological dimension

The starting point of the assessment is a decrease of the pressure "nutrient and organic matter enrichment" (Annex III, Table 2 MSFD).[11] This is shown at the top of Figure 2. One of the most prominent and direct effects of a reduction of nutrient inputs would be a decrease in phytoplankton productivity and biomass as well as a

[10]The term eutrophication describes water conditions in which excessive amounts of nutrients such as nitrogen (N) and phosphorus (P) lead to a series of undesirable effects. In Europe, nutrients are transported to seas via rivers, direct discharges from sources along the coast and atmospheric deposition (HELCOM, 2009). The main human sources for eutrophication in the Baltic Sea can be divided into point sources such as industrial or municipal wastewater plants and diffuse sources such as agriculture and airborne loads, e.g., from road traffic (HELCOM, 2009). In the Mediterranean Sea, urban wastewater discharges are important nutrient sources, particularly when they are untreated (EEA, 2006). In the Black Sea, the two major sources for eutrophication are riverine nutrient transport and atmospheric deposition, followed by direct discharges from large wastewater plants (BSC, 2009).

[11]The focus of this paper is on pressure reductions because the MSFD requires CEA and CBA to be carried out specifically to analyze improvement measures, which aim at maintaining or restoring a GES.

decline of short-lived macroalgae stocks. Subsequently, the pressure reduction would induce complex changes in the structure and functioning of the entire marine ecosystem and an increase in ecosystem stability. These changes are described in more detail below and illustrated in the upper part of Figure 2.

The solid, green arrows in Figure 2 indicate a positive relationship between the two states in the two neighboring boxes. For example, higher water transparency induces a higher stock of seagrass due to better light penetration. A dashed, red arrow in Figure 2 indicates a negative relationship between the two states in the two neighboring boxes. For example, higher production of phytoplankton induces less water transparency. Thus, the arrows represent the direct effect of a change between two boxes. The sign in the upper right edge of each box indicates the total expected net change of a state following the initial reduction of the pressure. For example, a reduction in nitrate and phosphate inputs would lead to a decrease in hydrogen sulphide (H_2S) emissions and toxic algal blooms.

Reduced nutrient enrichment would induce less murky water owing to blooms of planktonic algae, fewer mats of macroalgae at shores, increased distribution of benthic habitats such as eelgrass meadows due to enhanced light penetration, and less oxygen depletion resulting in fewer deaths of benthic animals and fish as well as decreasing occurrences of toxic algal blooms. Moreover, the decrease in primary production induces a decrease in sedimentation of organic matter to the seafloor (HELCOM, 2009; Claussen et al., 2009). Additional effects include enhanced carbon dioxide (CO_2) capture capacity due to increased kelp forests and lesser production of toxic H_2S, which can induce death of fish and benthic invertebrates (OSPAR, 2010).

4.1.2 The human dimension

The ecosystem services impacted by reduced eutrophication (sketched in the middle of Figure 2) constitute the link between the ecological and the human dimension, which refers to the benefits and values humans derive from marine ecosystem services. Less oxygen deficiency in less eutrophicated waters would, for example, avoid killings of fish, which would increase valuable fish stocks. Thus, direct use

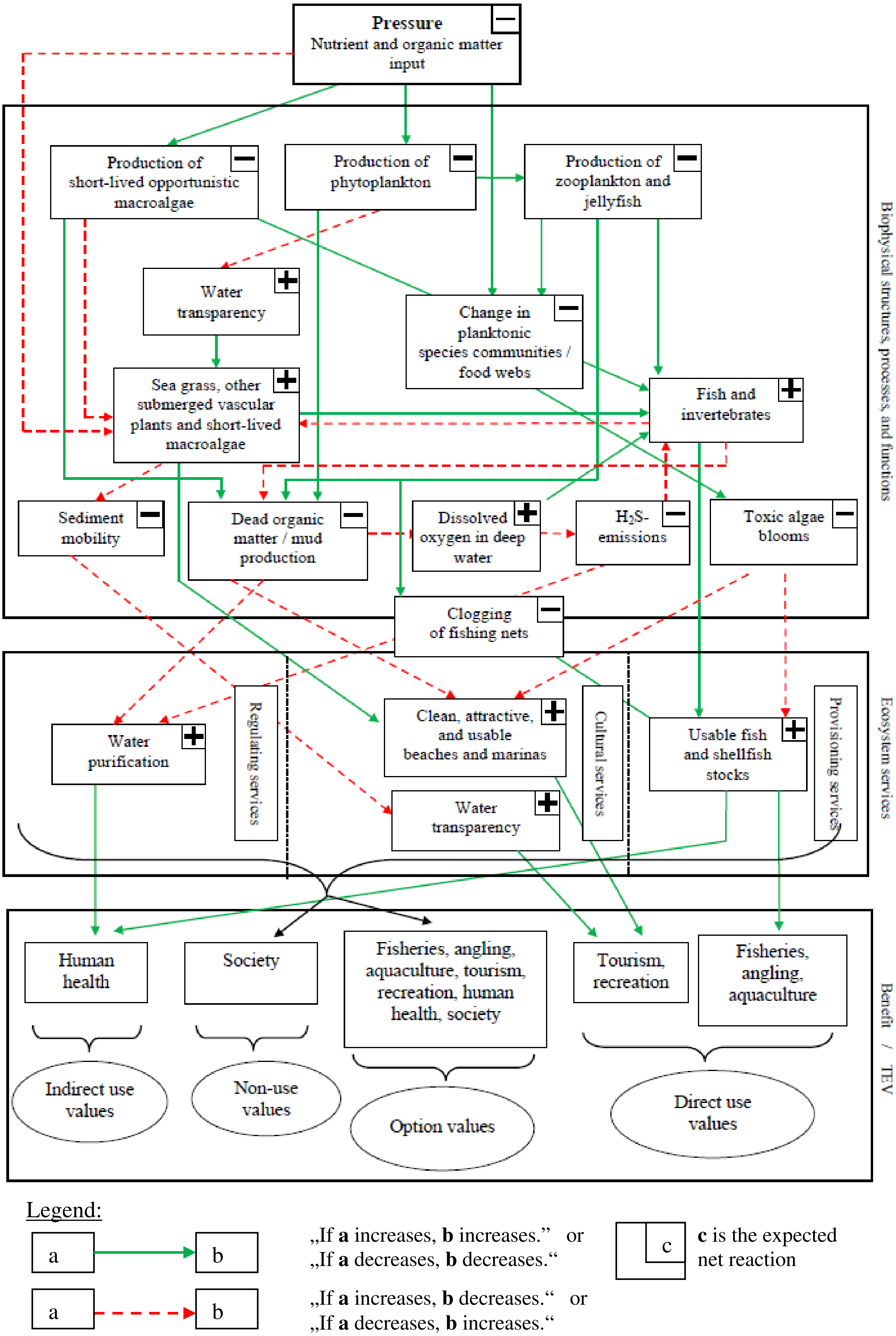

Figure 2. Effects of eutrophication on marine ecosystem services and relationship to uses and benefits. Own presentation.

values derived from harvesting and consuming fish would increase. Moreover, less algal blooms would reduce the extent of unsightly foam masses and unpleasant smells. This would increase direct use values derived from recreational and aesthetic uses of the sea. Recreation and tourism are further affected by increased water transparency and by fewer blooms of toxic blue-green algae. These toxic algal blooms would otherwise impede the possibility to swim safely in the sea. Moreover, toxins that are produced by some algae may harm humans through the consumption of contaminated shellfish, though the exact link to nutrient enrichment is not yet established (HELCOM, 2009). Reduced eutrophication would alleviate such health effects, which would imply an increase in indirect use values.

In addition to these changes in use values, also non-use values and option values are positively influenced by a reduction in eutrophication. Lesser degrees of eutrophication would increase the ecosystem`s ability to react to future disturbances and thus the option to provide a stable flow of ecosystem services in the future. Moreover, non-use values would be increased because of the increase in some species stocks or the amelioration of the ecosystem as a whole.

4.2 Economic valuation of eutrophication effects in Europe

As has become evident in the previous section, eutrophication causes complex changes within ecosystems and has been recognized as a major pressure for the European marine environment. Moreover, it has considerable impacts on the provision of ecosystem goods and services and human well-being. Despite the relatively large literature on natural science aspects of eutrophication, the economic valuation literature on eutrophication is relatively small and information is rather fragmented. Table 2 summarizes the findings of the valuation literature on eutrophication in European marine and coastal ecosystems.[12] Short summaries of the valuation studies are provided in Appendix A of this paper.

[12]In the context of the WFD, a couple of economic studies have been carried out to value the benefit of reduced eutrophication in freshwater ecosystems. However, this literature is not considered further since significant differences exist between eutrophication occurring in the sea and in freshwater. Moreover, the MSFD specifically refers to marine and coastal ecosystems.

Table 2. Overview of studies which value/estimate the benefits of reduced eutrophication in the European seas.

	Author(s), date, and type of publication	Year of survey data	Region	Country	Benefit[c]	Method	Quality Indicator	WTP/WTA	Remarks
Baltic Sea	Kosenius (2010) (J)	2006	Gulf of Finland	Finland	Not specified	CE	Water clarity, abundance of coarse fish, state of bladder wrack & occurrence of blue-green algae blooms	Annual household WTP: 149-611 € to achieve most modest scenario, 210-666 € to achieve most ambitious scenario	Multinomial logit, random parameters logit, latent class model
	Vesterinen et al. (2010) (J)	1998-2000	Finnish coastal waters and lakes	Finland	Recreation (swimming, fishing, and boating)	TC	Water clarity	WTP for one water recreation day; increase in water clarity by 1 m would increase consumer surplus by 6% for swimmers, by 15% for fishermen, and by 0% for boating	Study uses national recreation inventory data
	Ahtiainen (2009) (J); Huhtala et al. (2009) (PR)	1994-2008	Baltic Sea	Whole Baltic region and US	Recreation, fisheries	Meta-analysis	n.a.	WTP per month: ~3.30-10 € for a 50% water quality improvement	Estimates the effect of, e.g., income or the type of elicitation method on WTP
	Hyytiäinen et al. (2009)[a] (WP)	-	Finnish coastal waters	Finland	Recreation	CBA/TC, Meta-analysis	Water clarity/Secchi depth	WTP (no per unit values available)	Integrated simulation model for assessing nutrient abatement policies
	Atkins and Burdon (2006)[a] (J); Atkins, Burdon & Allen (2007)[a] (J)	2003	Randers Fjord	Denmark	Recreation	CBA/CV	Secchi depth	WTP per month over ten years: ~12 € for increasing Secchi depth by 2.5-3 m (~7.60 € without outliers)	-
	Soutukorva (2005) (WP)	1998-1999	Stockholm archipelago	Sweden	Recreation	TC	Secchi depth	Aggregate consumer surplus: ~9.60-31 million € per year for a 1 m increase in Secchi depth	Random utility model with conditional logit specification
	Kosenius (2004) (TH)	2003	Hanko, Gulf of Finland	Finland	Tourism, recreation, shellfish consumption/ health	CV	Water quality: Reduction of harmful algal blooms	WTP per person per year for a 25% reduction in algae blooms and a 50% reduction in the risk of shellfish poisoning: ~24.90 €	Focuses on benefits for tourism
	Olsson (2004) (WP)	2001	Swedish West Coast, Skagerrak, Kattegatt	Sweden	Recreational fishing	CV	Cod stock	Median WTP for increasing the catch of cod per hour from 2 kg to 100 kg: ~17.30-28.80 €	Comparison between open-ended questions and dichotomous choice and between tax and license fee

Baltic Sea	Eggert and Olsson (2003) (WP)	2002	Swedish West Coast, Skagerrak, Kattegatt	Sweden	Recreation	CE	Fish stocks, bathing, water quality & biodiversity level	WTP per month: ~13 € for avoiding reduction of biodiversity, ~5.60 € for improving biodiversity, ~5.60 € for improving water quality, and ~12.10 € for improving fish stocks	Multinomial logit, mixed multinomial logit model
	Hökby & Söderqvist (2003) (J)	1995-1999	Baltic Sea	Sweden	Not specified	Meta-analysis of CV studies	n.a.	WTP per month: ~5.75-66 € (range from different studies)	Estimates income and price elasticities of demand for reduced eutrophication
	Söderqvist & Scharin (2000) (WP)	1998	Stockholm archipelago	Sweden	Recreation	CV	Secchi depth	WTP per month: 4.10-6.80 € for 10 years to increase Secchi depth by 1m	-
	Markowska & Zylicz (1999)[a] (J)	1994	Baltic Sea	Sweden, Poland, Lithuania; BT to whole Baltic region	Not specified	CBA/CV & BT	Overall state of Baltic Sea	WTP for reaching a GEcS comparable to that of the 1960s (BDBP): 252 US$ in Sweden, 56 US$ in Poland, and 28 US$ in Lithuania	Use WTP and costs estimates to investigate cost-sharing for a public good in the case of the Baltic Sea (Chander-Tulkens model)
	Turner et al. (1999) (J)	1994	Baltic Sea	Sweden, Poland, Lithuania; BT to whole Baltic region	Not specified	CBA/CV & BT	Overall state of Baltic Sea	WTP per month for reaching a GEcS comparable to that of the 1960s (BDBP): 31-55.60 € in Sweden and 4-7.90 € in Poland	Interdisciplinary simulation study
	Frykblom (1998)[b] (TH)	-	Laholm Bay, Swedish West Coast	Sweden	Recreation	CV	Overall state of Laholm Bay	WTP per month: ~86.10 € for a 50% reduction in nutrient emissions	-
	Gren, Söderqvist & Wulff (1997) (J)	1994	Baltic Sea	Sweden and Poland; BT to whole Baltic region	Not specified	CBA/CV & BT	Overall state of Baltic Sea	WTP for reaching a GEcS comparable to that of the 1960s; WTP per month over 20 years: ~30 € in Sweden, ~3 € in Poland	Benefit transfer from Sweden to market economies and from Poland to formerly centrally planned economies
	Sandström (1996) (WP)	1990-1994	Laholm Bay, Swedish West Coast	Sweden	Recreation	TC	Secchi depth	Aggregate consumer surplus: 27-61 million € for a 50% reduction in nutrient load along the Swedish coastline	Random utility model with nested multinomial logit and conditional logit specifications
	Zylicz et al. (1995) (WP)	1994	Polish coastal waters	Poland	Recreation	CV	Dirty beaches & oxygen deficiency/ abundance of marine life	WTP per year for reaching a GEcS (BDBP): ~84 US$ per year	Scenario descriptions were adapted to Polish respondents because they were not familiar with the effects of eutrophication

North Sea	Longo et al. (2007) (PR)	2006	Belgian Coast	Belgium	Recreation	CE	Water quality: Amount and duration of algal blooms and foam	WTP: 16.39 € (8.40 €) for a low (middle) level of foam, WTA: 24.79 € for a high quantity of foam	-
	Le Goffe (1995) (J)	1993	Brest Natural Harbor	France	Recreation, health/shell fish consumption	CV	Water quality	WTP per month: ~2.70 € for reducing eutrophication, ~2 € for risk-free bathing and shell fish consumption	Reduced eutrophication and contamination from other pollutants are considered
	Taylor & Longo (2010) (J)	2006	Varna Bay	Bulgaria	Recreation	CE	Water quality: Visibility and duration of algal blooms	WTP per person: ~9.73 € for a program that entails no algal bloom	WTP decreases with duration of algal blooms and with decreasing visibility
Black Sea	Knowler, Barbier & Strand (1997) (WP)	-	Black Sea	Black Sea littoral countries	Fishing	Production function	Anchovy stocks and catch	Annual increase in steady state harvest revenues: 2.25 million US$	Bio-economic model with nutrients as input in natural production function
	Torres, Riera & Garcia (2009) (J)	2006	Santa Ponça Bay, Mallorca	Spain	Recreation	CE	Water quality: clarity and duration of algal blooms	Bimonthly WTP per person (2[nd] home residents): 35.42 € (26.05 €) for a low (medium) water transparency loss, 16.04 € (2.13 €) for a low (medium) duration of the bloom	Conditional logit specification, results hint at a non-linear relationship between attribute levels and WTP, comparison between 1[st] and 2[nd] home residents
Mediterranean Sea	Alberini, Zanatta & Rosato (2007) (J)	2002	Lagoon of Venice	Italy	Recreational fishing	TC (actual and contingent behavior)	Catch rate	Consumer surplus per person per year for a 50% increase in catch rates: 1,379 € for Venice residents, 745 € for others	Find that responses to contingent behavior questions are consistent with actual behavior
	Kontogianni et al. (2003) (J)	1999	Thermaikos Bay	Greece	Recreation/ not further specified	CV	Water quality	Mean WTP per month for five years (for operation of a wastewater treatment plant): 3.81 €	Eutrophication and other pollution effects are considered together; open-ended questions

Own presentation. The table contains information from publications that look at the value of reduced eutrophication effects in European coastal and marine waters from 1990 to 2011. Only publications in English are considered. [a] Information given only refers to the benefit part of the CBA. [b] Information taken from a summary in SEPA (2008).
[c] Recreation includes activities such as sunbathing, swimming, boating, recreational fishing, and enjoying the outside. However, this varies from study to study.
Abbreviations: J: Peer-reviewed journal publication, WP: Working or Discussion Paper, PR: Project Report, TH: PhD or Master`s Thesis, BT: Benefit transfer, CBA: Cost-Benefit-Analysis, CE: Choice Experiment, CV: Contingent Valuation, GEcS: Good ecological status, n.a.: not applicable, TC: Travel Cost, WTA: Willingness to accept, WTP: Willingness to pay. Monetary values are given in current terms in euros or in US$. Values reported in the studies have been converted to euros if necessary using the following exchange rates: SEK 100 = EUR 11.35 and FRF 100 = EUR 15.24.

The literature overview demonstrates that there are still considerable gaps in knowledge, particularly if one takes into account the ambitious provisions of the MSFD concerning the application of economic CBA and CEA based on an ecosystem-based approach. These gaps refer to i.) the regional focus of the valuation studies, ii.) the relation of the benefit to the initial reduction in nutrient inputs, iii.) the category of ecosystem services that is considered, and iv.) the category of values and benefits that is covered. In the following, these individual gaps are discussed in more detail.

The first gap relates to the regional focus of the studies. All studies have a clear regional focus, with the majority of them having been carried out in Scandinavian countries. However, the last systematic and coordinated research effort to value the benefits of water quality improvements for the Baltic Sea, the Baltic Drainage Basin Project (BDBP), dates back to the 1990s (Turner et al., 1990) and may be considered outdated. Since then, mostly isolated valuation studies with a local or regional focus have been carried out.[13] In particular, there are only very few studies that value eutrophication effects for the other European seas (see Table 2). The isolated nature of most existing studies hinders a straightforward comparison between the estimated values.

The second gap, which is mentioned in virtually all of the studies, is the missing link between nutrient loads and resulting effects on benefits. A viable CBA that analyzes the effects of reduced eutrophication would require the relationship between drivers and benefits to be established. So far, in the case of eutrophication, costs have mostly been expressed as cost per ton of nutrient reduction; and these costs depend on the kind of measures taken. Benefits, on the other hand, are expressed in terms of benefit

[13] A recent and still ongoing attempt for a new internationally coordinated evaluation of the Baltic Sea, including eutrophication effects, is the so-called BalticStern project. This project will encompass valuation studies of benefits but also estimates of cost functions for measures to mitigate eutrophication. So far, the published information on links between the costs of pressure reductions and related benefits are at best indicative (Huhtala et al., 2009).

for a certain quality increase.[14] Consequently, costs and benefits cannot be linked directly to the same improvement measures and are thus not directly comparable.

Since the work of the BDBP, many studies have assumed that a certain reduction of nitrogen (N) and/or phosphorus (P) discharges, mostly by 50%, will induce a certain good ecological status (GEcS) of the Baltic Sea, e.g., the one that persisted during the 1960s.[15] In these studies, people are asked for their maximal WTP to achieve this GEcS compared to the current condition. A viable comparison between costs and benefits would only be possible if a measure or a bundle of measures to achieve this GEcS could be defined. This would require the usage of detailed ecological models.

However, the linkages between pressure reduction and benefit effects can be complex, and there may be interactions and feedback effects. Some work has been carried out to advance interdisciplinary research and to extend the degree of understanding of these issues (e.g., in Hyytiäinen et al., 2009). But Huhtala et al. (2009) acknowledge that there are still gaps in the "understanding of key physical, chemical, and biological processes governing nutrient cycling in the Baltic Sea" and that knowledge is lacking to forecast the response of the environment to changes in nutrient loading. In addition, there is even less knowledge about eutrophication effects and links to benefits for the other European seas. However, exactly this type of knowledge is needed to fulfill the requirements of the MSFD to follow an ecosystem-based approach in the appraisal of improvement measures.

The third identified gap regards the types of benefits that are analyzed in the valuation studies. Apparently, most of them focus on recreational benefits. However, the activities subsumed under recreation vary across studies. Most valuation studies for Sweden, for example, ask respondents for their recreational activities including sunbathing, swimming, enjoying the outdoors and surfing as well as, e.g., recreational fishing. Other studies, only consider recreational fishing on its own

[14]In addition, the assessment of the WTP for reduced eutrophication is based on the change of one attribute, namely water clarity. The influence of other attributes is neglected unless these attributes are clearly mentioned and described and unless the corresponding scenarios are presented with the survey.
[15]This reduction target is in line with HELCOM regulations.

(Olsson, 2004). This complicates the comparability of elicited values between studies. Other effects on benefits, like health effects or effects on fisheries are not considered in most of the studies. In particular, there are no comprehensive studies that look at the effects of a certain change on all benefit categories.

The fourth identified gap concerns the categories of values (direct use values, indirect use values, option values, etc.) that are investigated. Many valuation studies mention the different value categories that are affected by reducing eutrophication. However, in the actual valuation exercise, they focus on direct non-consumptive use values by estimating recreational benefits. Direct use values related to fisheries and aquaculture or indirect use values related to health and climate effects are often neglected or only implicitly contained in people's valuation of the water quality change (see Figure 2). Moreover, non-use values are mostly not mentioned explicitly in the studies, though these values might be included in the results, depending on what the respondents thought of, when they answered the survey questions. The scope of benefits included in the valuation depends crucially on the scenario description provided to respondents.

In principle, the CV method is able to capture the TEV in the sense that people may express their WTP for a certain change in environmental quality taking into consideration a whole range of reasons. Söderqvist (1998) describes such reasons uttered by respondents taking part in the Swedish CV study that was part of the BDBP. The results indicate that the motive of about one third of respondents was related to the direct use of the Baltic Sea, either their own use, other people`s use or other people`s use in the future. Moreover, about 20% of respondents refer to human survival or human health, though this had not been mentioned in the scenario description of the questionnaire. This seems to indicate that most people attach a positive value to indirect use values and option values provided by the Baltic Sea.

5 Implications for the environmental effectiveness of the MSFD

5.1 Issues related to the economic requirements of the MSFD

5.1.1 The role of economics for the initial assessment

The EU WG ESA published a guidance document in December 2010 to clarify the role of economic analysis for the initial assessment (EC, 2010). This guidance document suggests two tools for the initial ESA, the Marine Water Accounts Approach and the Ecosystem Services Approach, without precluding further approaches. While the former approach focuses on financial costs and benefits accruing in economic sectors that directly use marine environments, the latter focuses on identifying ecosystem services provided by marine environments and the related benefits humans derive from these services, including non-use values. It is open to MSs to choose one of these or any other approach. The Marine Water Accounts Approach does not meet the requirements of the MSFD to follow an ecosystem-based approach. It is much too narrow and precludes important constituents of the TEV of marine ecosystem goods and services from the analysis. This in turn could undermine the environmental effectiveness of the MSFD.

5.1.2 The role of economics for determining GES

One important part of the MSFD is the definition of a GES based on scientific criteria such as physical and chemical features, habitat types, biological features, and hydro-morphology. In addition, social and economic concerns should be taken into account (Art. 10.1 in connection with Annex IV, no. 9 MSFD). So far, however, socio-economic criteria have not been discussed in detail in the process of defining GES but rather as a separate issue, relevant above all for the initial assessment required by the MSFD. As a consequence of this separation, the definition of the GES will be based on expert knowledge and findings from natural sciences only. Thus, the environmental targets of the MSFD would be defined without taking into account optimality and efficiency criteria regarding the trade-off between environmental and socio-economic effects. Instead, the MSFD's intent to reach the

GES by 2020 can be considered a political objective, based on insights from natural sciences irrespective of social and economic consequences. This would not necessarily lead to wrong results. Still, it decreases the possibility to find efficient targets in the sense of a reasonable weighting of the related social costs and benefits.

5.1.3 The role of economics for the development of improvement measures

The overall aim of Art. 13 MSFD is to ensure that the chosen program of measures allows reaching the GES at least costs. CEA is a suitable tool to choose between a variety of proposed measures designed to achieve the same pre-defined target. This would be the case if the targets have been determined by GES indicators before selecting the measures. Only cost-effective measures or bundles of measures should then be considered for implementation. CBA, on the contrary, is a tool that allows prioritizing measures with different targets and different costs. It would thus be more suited to discuss measures and targets simultaneously. Therefore, more clarity of Art. 13 MSFD regarding the policy-decisions, which are to be informed by the economic considerations, is needed to choose the correct methodology (COWI, 2010).

However, even if targets are determined, e.g., by GES indicators, CBA might still offer the opportunity to prioritize measures among regions and over time. It is, for example, possible to determine where and when welfare gains of measures will be highest. This is closely related to the economic analysis of the cost of degradation carried out during the initial assessment (COWI, 2010).

In addition, even if targets are determined before measures are chosen, so that CEA will be the main tool to choose among measures, each (cost-effective) measure that is considered for implementation would also have to be evaluated with the help of CBA if Art. 13.3 MSFD was interpreted literally. Measures would only have to be taken as long as benefits exceed costs by a certain amount. This also implies that the results of the CBA will be of particular importance to defend situations in which a MS intends to take no action to maintain or restore the GES.

5.1.4 The role of economics for the justification of exceptions

Another issue that needs further clarification is the role of economic analysis for the justification of exceptions due to disproportionate costs of measures – a problem that has been and still is prominent in the context of the WFD. Disproportionate costs as mentioned in Art. 14.4 MSFD can be verified by looking at the cost-benefit ratio (CBR) of measures or by comparing their net present values (NPVs). According to WFD guidelines, the CBR should significantly exceed the value one for granting exceptions. In the context of the WFD, use values were often sufficient to show that costs of measures were not disproportionate. In these cases, it was not necessary to calculate non-use values to demonstrate that it was favorable to implement the measure under investigation. However, it is still unclear what a sufficient CBR is in the context of the MSFD to grant exceptions. Compared to the implementation of the WFD, this question gains importance in the context of the MSFD.

The reason for this is that information on costs and benefits related to measures to reach a GES of marine waters is scarce, and its inference is connected to large uncertainties. Particularly, this holds true for non-use values and indirect use values, which is important to consider, as indirect benefits from regulating services often constitute the largest share of the TEV (TEEB, 2009). Moreover, use values might even be less important in the context of the MSFD than in the context of the WFD, particularly for offshore areas. This implies that the valuation of non-use values may become necessary, which poses a far greater challenge for economic valuation exercises (Eftec/Enveco, 2010).

As a consequence, special attention should be given to the question if a valuation approach is able to capture the TEV and thus the total benefit of a certain improvement measure. In many cases, eliciting mechanisms tend to underestimate total benefits. This would favor the justification of exceptions and hinder environmental effectiveness of the MSFD. Consequently, qualitative data on benefits should be included in the decision-making process in order not to neglect the major components of the benefit. Moreover, this would call for an ecosystem service approach rather than just focusing on financial benefits in order to capture the whole value of marine protection measures.

It can be expected that this question will be discussed more intensely in the future during the implementation phase of the MSFD. In particular, it will be necessary to define an appropriate CBR during the political process. For cases where monetization of benefits does not seem sensible, other measures to weigh costs and benefits need to be developed and applied.

5.1.5 International cooperation

International cooperation will be much more important for the implementation of the MSFD than for the implementation of the WFD due to its regional coverage. The provisions of the WFD refer to river basins, which are mostly located within one country, though they may be shared by two or more countries. The MSFD, however, implies a substantially higher effort to account for cross-border effects as it refers to marine regions or subregions that are shared by a number of littoral countries (Eftec/Enveco, 2010).

The literature review in section 4.2 on eutrophication showed that valuation studies have mostly been carried out for single countries, predominantly in Scandinavian and Baltic countries. However, these studies often assume that eutrophication effects are to be alleviated by internationally coordinated action because action in one country would not be sufficient to reach a GEcS. Naturally, the studies do not provide details on how internationally concerted action is to be achieved and granted. But particularly the fact that the management of marine resources has to take into account transboundary effects and requires international cooperation increases the challenges posed by the MSFD.

Referring to the analysis of cost-effectiveness, for example, the question arises whether cost-effectiveness should only be assessed within one country or also across European countries. As has been demonstrated by empirical studies, for international environmental problems the same abatement goal can be achieved with considerably lower costs if cost-effectiveness is analyzed across countries (see, e.g., Neumann and

Schernewski, 2001). Moreover, measures taken in one country may be more efficient than the same measures taken in other countries. However, the spatial distribution and heterogeneity of costs and benefits related to improvement measures adds an additional dimension to the policy problem, calling for more intense international co-operation. In some cases this might also have to include international compensation schemes.

5.2 Issues related to the nature of environmental valuation

5.2.1 Incomplete representation of the TEV

This issue is touched upon in section 5.1 and underlined by the literature review in section 4.2. In particular, the review revealed that the existing valuation studies on eutrophication mostly focus on one category of benefits, namely the benefits generated by the cultural service recreation. Other possible effects of reducing eutrophication, e.g., those on fisheries and recreational fisheries, health, climate and transportation, are neglected. Moreover, most studies claim to follow the approach of TEV, yet the difficulties in identifying the effects on different value categories and in determining option and non-use values are only mentioned vaguely. Consequently, it is often not clear what people value when they answer questions in a stated preference survey (see, e.g., Söderqvist, 1998).

This issue should be kept in mind also when measures to mitigate other pressures listed in the MSFD are analyzed. It would be important to investigate what would happen if one included hints on the different motives in the scenario descriptions of valuation studies. The question is whether people's WTP would change if they were reminded of other people or future generations being able to use and enjoy the marine environment. This would shed more light on the question whether stated preference approaches really capture the whole TEV of pressure reductions. Moreover, it would thus affect the way in which the results of such studies could be used for CBA within the framework of the MSFD.

In this context, particular attention needs to be drawn to the concept of option value. Increasing economic activities coupled, e.g., with higher nutrient emissions and pollution throughout the drainage basin of the Baltic Sea has led to higher vulnerability of the ecosystem (Turner et al., 1999). The question is how the option value of maintaining or restoring the GES of an intact marine environment should be elicited. In the study by Söderqvist (1998), 7% of the respondents stated that reducing eutrophication would be important for the future. Still, it is questionable whether this is sufficient to estimate an option value. Instead, the valuation of option values and indirect use values resulting from reducing the pressures listed in the MSFD should be subject to more scientific investigation from the natural science perspective.

5.2.2 Preference Uncertainty

Valuation studies are based on the assumption that people have well-defined preferences for the provision of ecosystem services, which exist independently of the experiment or survey being carried out. Empirical evidence however suggests that people are uncertain about their preferences (TEEB, 2010). Moreover, it is possible that preferences are formed only during the experiment or survey if people have not been aware of the problem at hand before.

Consequently, the question arises whether, e.g., the mentioning of other people or future generations using the sea would elicit existing preferences or whether this would induce preferences that did not formerly exist. This issue is also important for determining the benefit of improving environmental conditions in open waters. The question is whether preference-related elicitation measures are appropriate to define the benefit of changes that are not experienced directly by people (Nunes et al., 2009). Eutrophication, for example, can lead to a wide area of "seafloor deserts" in open waters, where marine life is killed by oxygen depletion, lack of light and sedimentation. The question is whether people really value an amelioration of such conditions and, in addition, how economists should deal with the problem that people are mostly unaware of such issues until they are confronted with them during the surveys.

On the other hand, there is evidence that people actually do value the existence of undisturbed ecosystems, particularly marine ecosystems. This becomes obvious, e.g., via the large number of TV documentaries that is produced and watched by people. Consequently, at least part of the population has preferences regarding the importance of marine ecosystems and seems to attach positive values to their current and continuing existence.

5.2.3 Marginality, non-linearities, thresholds, and irreversebility

Decision-making in terms of CBA for project appraisal requires information on marginal changes of ecosystems. In the context of marine ecosystem services, this could be a small change in the area affected by eutrophication or a relatively small change in the water quality. Marginal analysis also requires information on the transition path the ecosystem might take if the current state is disturbed. In the case of a full coral reef system, for example, this transition path may be stepped, while it may be relatively smooth for the invasion of alien species into an area. Consequently, the impacts of human actions on ecosystem functioning might not be linear. For example, an ecosystem might seem unaffected by a human perturbation until a certain point is reached, which induces a sudden and drastic change in the state of an ecosystem. The assumption of linear behavior in economic analysis could thus lead to biased policy decisions if underlying ecological processes are in fact non-linear (Turner et al., 2010).

The possible existence of non-linearities is particularly important in the context of the initial assessment required by the MSFD, which shall also include the analysis of the possible costs of degradation if no action is taken to improve the conditions of the European seas. In this case, the costs of inaction could increase substantially if non-linear effects occurred in the behavior of marine ecosystems. The ecosystem-based approach mentioned in the MSFD would thus require taking such effects into account.

Moreover, it has become obvious in the study of ecosystems that thresholds may exist beyond which a drastic change in the state of an ecosystem occurs. Such a behavior is not compatible with marginal economic analysis, which assumes continuity of the benefit provision. Crossing these thresholds may in addition be irreversible if it is not possible to restore the initial state of the ecosystem. The possibility of triggering irreversible changes in ecosystems could support the demand for safe minimum standards. This would imply that a conservation option should be taken if an irreversible effect on the ecosystem is probable unless the related costs of this option are regarded as unacceptable. The principle of safe minimum standards is thus based on minimizing the maximum possible loss, not on maximizing expected gains, as in CBA and risk analysis. Of course, it is open to discussion in which cases costs of conservation are unacceptable, particularly if one faces large uncertainties regarding future impacts of human uses on complex ecosystems. However, the imposition of safe minimum standards may provide one way of incorporating the precautionary principle into decision-making by choosing conservation measures even if there is no certainty about future damages (Turner et al., 2010; Ledoux and Turner, 2002).

The MSFD mentions the precautionary principle and states that the programs of measures and the actions of the MSs should be based on it (Preamble, no. 26 and 44 MSFD). Still, the precautionary principle is only mentioned in the preamble of the MSFD and not in its main part, and there are no specific provisions that regulate its application.

5.3 Issues related to the knowledge about the natural science background and the interrelation with human well-being

Though natural science is starting to shed light on the functioning of ecosystems and the creation of ecosystem services, important links between ecosystem functioning, ecosystem services, and human benefits are still poorly understood, which makes a robust CBA even more difficult (Bateman et al., 2011). One example is the role of biodiversity for ecosystem functioning and the provision of ecosystem services (TEEB, 2010). Uncertainty is even more prevalent in the context of marine ecosystem services, particularly those services which are not so visible and removed from people's direct experience, e.g., climate regulation (Remoundou et al., 2009).

This lack in knowledge complicates the implementation of the MSFD and the required economic valuation exercises. The design of CEs, for example, requires intense collaboration with natural scientists and a careful pilot phase to create realistic scenarios (Kosenius, 2010). Gren, Söderqvist and Wulff (1997) describe the integrated tools and steps that would be necessary to obtain complete information and acknowledge that even for eutrophication there is no complete picture. So far, only some work has been carried out to advance interdisciplinary research on eutrophication and to extend the degree of understanding of these issues (Hyytiäinen et al., 2009). Moreover, the lack of comparable data across all seas still presents a major obstacle for pan-European marine assessments, even of well-known problems such as eutrophication. More and better data are needed to develop a pan-European marine protection framework that addresses environmental issues in a cost-effective way (EEA, 2007).

For the example of eutrophication, the literature review in section 4.2 revealed that most of the studies on eutrophication are relatively old and that information is rather fragmented in geographical but also in methodological terms. New data is needed on the status of the European seas, on necessary nutrient load reductions and on the costs and benefits of these reductions to inform decision-making regarding the measures that need to be taken to reach GES. However, the literature on eutrophication is even further developed than the literature on waste, pollution, noise or other threats to the marine environment, which are also covered by the MSFD. Consequently, the MSFD poses a huge challenge for policy-makers and researchers.

In addition, there are complex interactions between the different pressures and target indicators listed in the MSFD. More research is needed to account for interrelations and feedback effects between them. Consequently, a detailed analysis is needed in order to determine the effect of a reduction of a certain pressure on the probability to reach an ecological target (Borja et al., 2010). Moreover, the measures taken to achieve a GES also need time to take effect. Such time lags have to be accounted for if a GES is to be achieved by 2020, as requested by the MSFD.

6 Concluding remarks

The aim of this paper is to present the economic requirements of the MSFD and to analyze which effects these requirements could have on the environmental effectiveness of the MSFD. To this end, the existing valuation literature is analyzed, focusing on one of the most important threats to European marine and coastal waters: eutrophication. The approaches and applications of environmental valuation are assessed and reconsidered taking into account background knowledge from natural sciences and the principle of an ecosystem-based approach, which is required by the MSFD and built on the suggestions of MA and TEEB.

To conclude, the analysis demonstrates that the implementation of the MSFD requires more coordinated research, so that studies to evaluate benefits can be carried out across countries using comparable, state-of-the-art valuation methods. This could also include the combination of different valuation methods, e.g., of stated and revealed preference methods, to gain more reliable benefit estimates. Moreover, integrated modeling will be of utmost importance to link bio-geophysical and socio-economic systems and to trace the effects of changes in the marine environment to their impact on benefits.

Moreover, the analysis reveals a considerable risk that the MSFD might fail to achieve its environmental targets. In particular, the problems related to capturing all benefits related to pressure reductions in the marine context might induce an underestimation of the related benefits and a relative overestimation of the related costs. Consequently, the CBR defined to represent disproportionate costs should be high enough, i.e., at least higher than in the context of the WFD, to reduce the number of situations in which exceptions to implement improvement measures are granted even though benefits are underestimated. This becomes even more severe if one takes the possible but uncertain existence of non-linearities and threshold effects into account. This calls for a conservative approach when benefits and costs are weighted against each other. Where benefits cannot be monetized, economic analyses should be complemented by qualitative assessments.

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Appendix A. Short summaries of the valuation studies contained in Table 2.

A.1 Studies that value eutrophication effects in the Baltic Sea region

Besides the work of the BalticStern project, the results of which have not been published yet, the work by Kosenius (2010), Vesterinen et al. (2010), Ahtiainen (2009), and Hyytiäinen et al. (2009) constitute the most recent approaches to evaluating eutrophication effects in the Baltic Sea.[16] A special focus of these studies is on the transboundary nature of eutrophication and on the benefits and costs of water quality improvements likely to occur in Finland.

Kosenius (2010) estimates the magnitude of benefits from three selected nutrient reductions in the Gulf of Finland for the Finnish people by applying a CE. The data were analyzed using three different econometric approaches, namely the multinomial logit (MNL), the random parameters logit (RPL) and the latent class (LCM) model. The paper incorporates natural science knowledge by using results from an ecological simulation model. Moreover, it takes into account that necessary reductions in nutrient loads will also have to take place in the neighboring countries, e.g., Estonia and Russia. However, the paper also acknowledges that there are still considerable knowledge gaps regarding the link between objective improvement of quality indices and the quality improvements as perceived by people as well as the actual link between quality attributes and actual nutrient reductions necessary to achieve certain quality improvements.

Vesterinen et al. (2010) utilize Finnish recreation inventory data combined with water quality data to model recreation participation and estimate the benefits of water quality improvements for the Finnish coast of the Baltic Sea as well as for Finnish lakes. The methods used are designed to account for the fact that water recreation

[16] Huhtala et al. (2009) provides a recent meta-analysis of studies that value the impact of water quality changes on recreational activities related to the Baltic Sea. They also categorize and analyze the ecosystem services provided by the Baltic Sea and assess the feasibility of CBA in the context of selected examples. Moreover, they present a prototype stochastic simulation model for projecting the development of nutrient budgets, damages from eutrophication, and the costs of abatement activities in the Baltic Sea.

activities in Finland mostly take place close to home. The smallest benefit estimates per trip per person ranged from approximately 6.30 to 8.30 € based on respondents` reported travel costs. Calculated travel costs for people traveling by car provided higher estimates, in the range of 18.90 to 19.00 € per visit per person. In both cases, the higher figures result from taking the opportunity cost of time into account. The work of Hyytiäinen et al. (2009) is described in more detail below.

Atkins and Burdon (2006) examine the costs and benefits of reduced eutrophication in the Randers Fjord in Denmark.[17] Their work is based, *inter alia*, on a study by Nielsen et al. (2003), which provides the natural science foundation to determine reference conditions of the Fjord to define its GEcS according to the WFD. The costs of achieving the GEcS are borne predominantly by Danish farmers. The study presents some cost estimates for reducing nutrient loads from the implementation of former action plans as well as cost estimates from a study by Gren (2000). The focus of the study is on assessing individual preferences for water quality improvements in the Fjord by carrying out a CV study. The paper only evaluates the benefit for recreationalists derived from higher water transparency. Benefits for recreational anglers from possibly increased catches are mentioned but not evaluated.

Like Kosenius (2010), Eggert and Olsson (2003) employ a CE to value changes in the state of the Baltic Sea. They consider the waters along the Swedish West Coast and use the attributes biodiversity, fish stocks and bathing water quality. The WTP for improving fish stocks refers to an increase in per hour catch from 2 kg to 100 kg of cod. The WTP for improving water quality refers to reducing the number of beaches that fail to pass standards from 12% to 5%. In particular, they note that the WTP to avoid the reduction of biodiversity from a medium to a low level (~160 €) is higher than the WTP to improve biodiversity from a medium to a high level (~68 €). Olsson (2004) carries out a CV study to evaluate the benefits of improved cod stocks along the Swedish West coast. The WTP for improving cod stocks refers to an

[17] Updated results are presented in Atkins et al. (2007).

increase in per hour catch from 2 kg to 100 kg of cod, as in Eggert and Olsson (2003).

Söderqvist and Scharin (2000) estimate recreational benefits of reduced eutrophication in the Stockholm archipelago by applying the CV method. Sight depth was used as an indicator for water quality. Soutukorva (2005) examines how improved water quality affects the demand for recreation in the same region, also using sight depth as an indicator for water quality. Benefits from reduced eutrophication are elicited using the TC method combined with estimating a RUM. Sandström (1996) also uses the TC method to elicit the benefits from reduced eutrophication along the Swedish coast and applies a RUM based on data gathered from the Swedish tourism and travel data base (TDB). The latter addresses the link between sight depth and nutrient loads by running a simple regression of sight depth on water temperature as well as P and N concentrations. However, he acknowledges that this relationship should rather be established by natural scientists to account more accurately for the effects of changing nutrient concentrations on sight depth.

The remaining primary studies date back to the year 2000 or earlier and were carried out mostly in the context of the Baltic Drainage Basin Project (BDBP). The BDBP followed an interdisciplinary approach that incorporates natural sciences and socio-economic aspects to evaluate the cost and benefits of reducing nutrient loads and thus eutrophication in the Baltic Sea. Nutrient loads are modeled using geographical information systems (GIS) for the whole drainage basin of the Baltic Sea. The link to nutrient concentrations in the Baltic Sea is established empirically by analyzing historical data. Cost-effective bundles of measures are defined for nutrient-reduction policies. Benefits were estimated using CV and TC methods in Poland and Sweden. These estimates were then transferred to other countries within the drainage basin to estimate basin-wide benefits. These were compared to basin-wide costs. The results are based on the assumption that a 50% reduction in N and P loads will restore a GEcS of the Baltic Sea comparable to that during the 1960s (Markowska and Zylicz, 1999; Turner et al., 1999; Gren et al., 1997; Turner et al., 1995).

Zylicz et al. (1995) present the CV studies carried out in Poland. They use the number of dirty beaches as well as the abundance of marine life due to oxygen supply in the water as quality indicators to describe the state of the Baltic Sea. The reason for this is that they found that Polish people are not very familiar with eutrophication effects. However, this somehow biases results, as beach closures may also be due to other causes besides eutrophication. Markowska and Zylicz (1999) use the results of the CV studies carried out in the course of the BDBP to investigate how costs should be shared optimally between littoral states if the Baltic Sea was considered a public good, based on national abatement cost curves for reducing N input and national WTP to reduce eutrophication. Subsequently, theoretical transfers between countries are compared to actual transfers. The study compares annual costs of reaching a 50% reduction in N discharges to the annual WTP for international clean-up action.

Some studies have reviewed the economic valuation literature on marine and coastal ecosystem services and carried out meta-analyses and meta-regressions. Ledoux and Turner (2002), for example, present the concept of TEV as a basis for valuing environmental goods and services as well as valuation methods and problems related to valuation. Moreover, they provide a broad overview of valuation studies dealing with marine and coastal ecosystem goods and services and exemplify this by a couple of case studies including the results from the BDBP. Ahtiainen (2009) presents a meta-analysis covering studies on water quality changes in the Baltic Sea and the adjacent drainage basin as well as in the United States to estimate, e.g., the effects of income or the type of elicitation method on the WTP for enhanced water quality. The final data set consists of 32 studies and 54 observations. Hökby and Söderqvist (2003) carry out a meta-analysis, estimating particularly income and price elasticities of the demand for reduced eutrophication in Sweden. They state that “none of the [single] CV studies […] is advanced enough in itself to make an estimation of a demand function possible”. The reason for this is that CV settings mostly do not allow for a choice between different combinations of price and quantity. They assume (based on Gren et al., 1997) that a 50% reduction in nutrient loads is consistent with the scenarios described by the five valuation studies on which they base their meta-analysis. Furthermore they assume that such a reduction leads to concentration levels similar to those prevailing during the 1950s. However, there are considerable uncertainties related to this, including the possibility of non-linearities (Hökby and Söderqvist, 2003).

The work by Hyytiäinen et al. (2009) is a recent approach to integrating knowledge from natural and social sciences. They present an integrated simulation model that incorporates the stochastic development of water quality, the underlying ecological processes as well as the relevant economic activities in the area, and the possible economic benefits to be gained from water quality improvements in Finland and neighboring countries. Concerning drivers, the model focuses on nutrient inputs from agriculture. The paper presents the structure of the model as well as an application with preliminary parameters. Nutrient emissions in neighboring countries are included in the model. Benefits of reducing eutrophication are obtained from other studies, which use the TC method and meta-analysis. Travel cost data is based on the work of Vesterinen et al. (2010). This information is used to construct functions that connect benefits derived from reduced eutrophication to water clarity. Results indicate that the benefits of engaging in activities to decrease eutrophication would only exceed costs for Finland if neighboring countries also engaged in such abatement activities.

A.2 Studies that value eutrophication effects in the North Sea region

Le Goffe (1995) carried out one of the few studies that value eutrophication effects in the North Sea. He considers reduced eutrophication and microbial contamination in Brest Natural Harbor in France and reports WTP for reducing eutrophication (effects on recreation) and WTP for risk-free bathing and shellfish consumption (health effects), respectively. Thus he captures direct and indirect use values; however, they result from different pressure reductions. He used a CV approach with open-ended WTP questions and payment cards.

Longo et al. (2007) carry out a CE to value the effects of eutrophication on recreational activities along the Belgian coast. This study is part of the Thresholds project, which also foresees similar valuation studies in the Black Sea and in the Mediterranean Sea (see below).[18] The attributes used in the Belgian CE are i.) the extent of algal blooms and the quantities of foam on the beach, ii.) the duration of algal blooms, iii.) and the congestion of the beaches. Longo et al. (2007) also present several sources from which threshold effects could arise when eutrophication is considered. However, these thresholds are not explicitly mentioned in the valuation study.

A.3 Studies that value eutrophication effects in the Mediterranean Sea region

Torres et al. (1997) carry out a CE to value the effects of eutrophication on recreational activities in Santa Ponça Bay, Mallorca, Spain. The attributes used in the Spanish CE are similar to those used in Longo et al. (2007) but specifically adapted to conditions in Santa Ponça Bay. The attributes used are i.) water transparency, ii.) the duration of algal blooms, iii.) and the congestion of the beaches. There is no direct link to the reduction in nutrient inputs needed to achieve the water quality improvements described in the CE.

Alberini et al. (2007) consider recreational fishing in the Lagoon of Venice in the Mediterranean Sea. They use the TC method to estimate the increase in consumer surplus resulting from a 50% increase in catch rates, achieved by reduced pollution. In particular, they use actual data and compare them to contingent behavior data, which they elicited via questionnaires. They do not find a significant difference between actual and contingent data.

Kontogianni et al. (2003) consider the case of a wastewater treatment plant in Thessaloniki, Greece. They elicit the people`s WTP for maintaining this plant, which would induce water quality improvements in the adjacent Thermaikos Bay. To this end, they use the CV method with open-ended elicitation questions.

[18] Within the Thresholds project, it is also planned to estimate the costs for reducing nutrient emissions both from agriculture and wastewater treatment (Longo et al., 2007).

A.4 Studies that value eutrophication effects in the Black Sea region

Talyor and Longo (2010) carry out a CE to value the effects of eutrophication on recreational activities in Varna Bay, Bulgaria. The attributes used in the Bulgarian CE are similar to those used in Longo et al. (2007) but specifically adapted to conditions in Varna Bay. The attributes used are i.) water clarity and visibility, ii.) the duration of algal blooms, iii.) and the congestion of the beaches. There is no direct link to the reduction in nutrient inputs needed to achieve the water quality improvements described in the CE. They state that there is a lack of scientific models that accurately predict algal blooms.

Knowler et al. (1997) construct a bio-economic model to link nutrient concentrations in the Black Sea with anchovy stocks via the prevalence of an exotic predatory Jellyfish species. They consider the impact of changing nutrient concentrations on steady state solutions in an open access regime, in particular the effect on anchovy harvest, which represents the direct use value generated by the anchovy stocks.

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Paper 4: Biodiversity and optimal multi-species ecosystem management

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Abstract:

We analyze optimal multi-species management in a dynamic bio-economic model taking into account both harvesting profit and biodiversity value. Within an analytical model, we show that extinction is never optimal when a global biodiversity value is taken into account. Moreover, a stronger preference for species diversity leads to a more even distribution of stock sizes in the optimal steady state, and a higher value of biodiversity increases steady state stock sizes for all species when species are ecologically independent or symbiotic. For a predator-prey ecosystem, the effects may be positive or negative depending on relative prices and the strength of species interaction. The analytical results are illustrated and extended using an age-structured three-species predator-prey model for the Baltic cod, sprat, and herring fisheries. In this quantitative application, we find that using stock biomass or stock numbers as abundance indicators in the biodiversity index may lead to opposite results.

Keywords: marine biodiversity, fisheries economics, Baltic Sea

JEL classification: Q22, Q57

1 Introduction

Ecosystems, in particular the oceans, provide a wide range of goods and services directly supporting human societies and economies. But in spite of heightened awareness, "the ocean remains chronically undervalued, poorly managed and inadequately governed" (GOC, 2014). Similar concerns apply to other types of ecosystems. Conflicting interests between short-term economic uses and conservation continue to cause their over-use and degradation (MA, 2005; Stavins, 2011; TEEB, 2010). In particular, current ocean governance arrangements do not ensure sufficient protection of marine biodiversity, and they do not foster the sustainable use of marine living resources (Visbeck et al., 2014).

Fisheries management has to some extent reacted to these developments by adopting the goal of employing ecosystem-based approaches. This implies that not only economic profits should be maximized but that conservation goals also need to be taken into account (Pikitch et al., 2004). Consequently, bio-economic models, which can be used to derive recommendations for fisheries management, should not only include multiple species but also multiple values. Against this background, we reconsider optimal multi-species management in a bio-economic model taking into account both harvesting profit and biodiversity value. More specifically, we analyze how optimal management changes when a biodiversity index is introduced in the objective function of a bio-economic dynamic optimization model to capture the value of biodiversity.

The rationale for introducing a biodiversity value into such a model is twofold: First, people can attach a value to in-situ stocks of species simply for their existence (Bulte and van Kooten, 2000). The shadow price of biodiversity could thus be interpreted as a social willingness-to-pay (WTP) for species conservation. In our application, this value would not be attributed to a single species but to the entire state of the multi-species ecosystem reflected by the biodiversity index. Second, a biodiversity value can serve as a place holder for ecosystem services that are not explicitly modeled via ecological interactions in a bio-economic model. The shadow price of biodiversity could thus also be interpreted as a proxy for the value of some external effects that are too complex to be integrated explicitly in the model.

The literature on renewable resources and bio-economic modeling has dealt with the inclusion of existence values before but mostly considers single-species models. For example, Alexander (2000) includes existence values in a one-species model and derives implications for the potential optimality of extinction. More recent papers also include existence values in multi-species models. Kellner et al. (2010) introduce an existence value for each single species in a multi-species predator-prey model but they do not aggregate the values into one index. Voss et al. (2014a) also consider a multi-species predator-prey model, but the existence value they introduce only applies to one stock. With a slightly different focus, Quaas and Requate (2013) include a constant-elasticity-of-substitution (CES)-function in a bio-economic model, but preferences for diversity are attached to the consumption of fish, not to the biodiversity of the ecosystem. Finally, Noack et al. (2010) use a biodiversity index similar to the one we are using in this paper, but they do not consider the context of fisheries management and, more importantly, do not investigate the effects of the introduction of this index on the single species stocks captured by the index.

Since multi-species applications are increasingly prominent in bio-economic modeling and biodiversity conservation is high on the international political agenda, it is important to investigate the properties of a bio-economic model when an aggregate biodiversity index based on species abundances is included to capture biodiversity values. Buckland et al. (2005) state axioms which an index based on species abundances should fulfill if it was used for monitoring biodiversity developments over time. These axioms include the requirement that the index value should decrease if overall abundance is decreasing while the number of species as well as species evenness stay constant. Prominent ecological indices such as the Simpson-Index or the Shannon-Index do not fulfill this axiom. Here, we consider the class of CES-functions to aggregate single stocks into one index. CES-functions fulfill the axioms stated in Buckland et al.

(2005)[1] and thus seem well-suited to track biodiversity developments over time. Moreover, the class of CES-functions is a frequently used and well-studied specification to describe production processes or consumer preferences in economics (Arrow et al., 1961; Dixit and Stiglitz, 1977).

We explore the effects of introducing such an index in a multi-species model of a harvested ecosystem on the optimal steady-state, which has, to our knowledge, not been investigated before. In addition, we exemplify the effects in a more complex age-structured model applied to the example of a predator-prey system of three Baltic Sea fish species (cod, sprat, and herring). The age-structured framework enables us to study the effects of switching from an index calculated using biomasses to an index calculated using the number of individuals. We also illustrate the role of the elasticity of substitution for optimal management. We find that both aspects of the biodiversity index crucially influence optimal management, which has important implications for actual management decisions when biodiversity indices are applied.

The remainder of the paper is structured as follows: Section 2 describes the biodiversity index, presents a dynamic biomass model and compares optimal multi-species management without biodiversity value to optimal multi-species management with biodiversity value. We analytically show the effects of changes in the shadow price of biodiversity, in market prices, and in the elasticity of substitution between species in the biodiversity index on optimal steady state stocks for different kinds of ecological interactions. Section 3 introduces a state-of-the-art age-structured model and simulates the effects of changing the shadow price of biodiversity and of changing the elasticity of substitution on steady state stocks, profits and biodiversity levels for the example of a three-species predator-prey ecosystem in the Baltic Sea. We also compare the effects when switching from a biodiversity index with biomass to an index using the number of individuals. Section 4 discusses the results and concludes.

[1]More specifically, a CES-function fulfills the first four axioms mentioned in Buckland et al. (2005). Axiom 5 and 6 refer to characteristics of the biodiversity measures related to their empirical estimation, which is not relevant in the theoretical context of this paper.

2 Dynamic model of optimal multi-species ecosystem management

2.1 Measuring biodiversity

One of the most established ways to measure species-level biodiversity is to calculate a diversity index based on individual species abundances. The elements that influence such a biodiversity index are the number of different species (species richness), and the evenness in the distribution of species abundances. A large number of such indices exist, and they are widely used in ecology to measure species-level biodiversity (Magurran, 2004). Buckland et al. (2005) explore the characteristics of such indices when measuring changes in biodiversity over time. They note that species abundance can be measured either in terms of biomass or in terms of the number of individuals to compute these indices. Different weightings for different species are also possible. They also state axioms which a biodiversity index should fulfill if it was used for monitoring biodiversity over time. These axioms are:[2]

> 1.) For a system that has a constant number of species, overall abundance and species evenness, but with varying abundance of individual species, the index should show no trend.
>
> 2.) If overall abundance is decreasing, but number of species and species evenness are constant, the index should decrease.
>
> 3.) If species evenness is decreasing, but number of species and overall abundance are constant, the index should decrease.
>
> 4.) If number of species is decreasing, but overall abundance and species evenness are constant, the index should decrease.

Biodiversity indices such as the Simpson-Index (Simpson, 1949) or the Shannon-Index (Shannon, 1948), which are common in ecology, are constructed using relative abundances. These indices, however, do not allow a consistent comparison of biodiversity

[2]These are the first four out of six axioms in Buckland et al. (2005). Their last two axioms refer to the sampling and empirical estimation of the indices and are thus not relevant in the theoretical context of this paper.

levels over time because they do not satisfy Axiom 2 (Buckland et al., 2005). They remain constant if overall abundance is decreasing while the number of species and species evenness are constant.

In this paper, we use absolute abundances, x_{it}, of the in-situ stocks of species $i = 1, \ldots, n$ at time t for calculating the biodiversity of a multi-species ecosystem with n in-situ stocks. Specifically, we measure biodiversity by using the following class of functions:

$$B_t = B_t(x_{1t}, ..., x_{nt}) = \left(\frac{1}{n} \sum_{i=1}^{n} x_{it}^{\frac{\omega-1}{\omega}} \right)^{\frac{\omega}{\omega-1}} \tag{1}$$

The index (1) satisfies the following conditions (using subscripts to denote partial derivatives with respect to the corresponding variables):

$$B_{x_i} > 0 \ \text{ and } \ B_{x_i x_i} < 0 \ \text{ and } \ B_{x_i x_j} > 0 \ \ \forall\, i, j \ \text{ with } \ j \neq i \tag{2}$$

The functional form (1) with a constant elasticity of substitution, ω, does not only fulfill the axiomatic conditions specified in Buckland et al. (2005) for functions that can be used to measure and compare biodiversity levels over time; it is also a common form for production functions (Arrow et al., 1961) or utility functions (Dixit and Stiglitz, 1977) in economics and resource economics (Quaas and Requate, 2013). In economic terms, the parameter $\omega > 0$ measures the elasticity of substitution between the stocks of the n species. For $\omega \to 0$, the elasticity of substitution would be zero and the stocks would be perfect complements. For $\omega \to \infty$, the elasticity of substitution would be infinitely large and the stocks would be perfect substitutes. The index (1) may also be interpreted as a generalized mean of species abundances. For $\omega \to \infty$, the index (1) simply gives the arithmetic mean of species. For $\omega \to 1$, the index (1) becomes the geometric mean of species abundances.

The axioms stated in Buckland et al. (2005) do not restrict the functional forms of potential biodiversity indices regarding the structure of the exponent outside the parentheses in (1). Any monotone transformation of the CES-function in (1) would fulfill the axioms as well. Here, we assume a structure of the exponent such that the resulting index is linear homogeneous. This is different from the structural form of the exponent used in the ecological indices that are based on relative abundances. Our specification implies that the index value is measured in terms of species abundances, just as the abundances for the individual species, which alleviates its interpretation.

Regarding the role of the elasticity of substitution, ω, note that the value of the biodiversity index, B_t, *ceteris paribus* increases as ω increases. Only for equal abundances of all stocks, the value of B_t is not affected by a change in ω. For the case of complements, $\omega < 1$, the biodiversity index, B_t, approaches zero if the stock x_i of any species i, is driven towards zero. The marginal biodiversity values, B_{x_j}, of the other species $j \neq i$ also go to zero if $x_i \to 0$. Substituting a species stock, x_i, by another stock, x_j, thus becomes less acceptable, the smaller the stock of species i already is. For the case $\omega > 1$, substitution between the species is more easily possible, i.e., the biodiversity index would continuously increase with an increasing stock of at least one species even if all other species are driven towards zero or go extinct. More formally, for $x_i \to 0$ both B_t and B_{x_j}, $j \neq i$ with $x_j > 0$, are finite and stay positive.

Note, however, that for the case $\omega \leq 1$, Axiom 4 cannot be applied because in this case the index is defined only for a constant number of species and not defined if the abundance of any one species becomes zero. Considering the different species as complements, $\omega \leq 1$, is thus only reasonable for issues where species extinction is out of scope, such as the three-species fishery in the Baltic Sea, considered in section 3.3 below. Consequently, for biodiversity considerations where species richness can vary, an index of the form (1) can be applied under the restriction that different species are substitutes, i.e., under the assumption $\omega > 1$. As a varying species diversity is of particular relevance at the global level, we refer to this case as "global" biodiversity value.

Also note that in all cases, the weight of the abundances of the single species in the CES-function, $(1/n)$, needs to be determined once, based on the initial number of species, and may not be changed afterwards, even if the number of species changes, e.g., due to extinction. Changing the weight when measuring biodiversity over time would violate Axiom 4 from Buckland et al. (2005).

2.2 Model framework and optimal management

In this section, we introduce an analytical biomass model and compare optimal management solutions with biodiversity value to optimal management solutions without biodiversity value. Section 2.3 presents comparative static effects of parameter changes on optimal steady states, differentiating between cases with and without ecological interactions. We use a general set-up with n species and possible ecological interactions. In addition, we use a CES-function (1) aggregating the biomasses of the different species to capture the value of biodiversity in the objective function.

2.2.1 Species dynamics

There are n species ($i = 1, ..., n$). The dynamics of each stock x_{it} are determined by its natural growth, G_{it}, and the biomass harvested, h_{it}, at time t:

$$\dot{x}_{it} = G_{it}(x_{1t}, ..., x_{nt}) - h_{it} \quad \text{for} \quad \forall\ i, t \tag{3}$$

We assume perfectly selective harvesting, i.e., harvest of species i does not directly affect the dynamics of any other resource stock $j \neq i$.[3] Species may interact ecologically, which is captured by the dependency of species i's growth function $G_{it}(\cdot)$ on the other species' stock sizes x_j. For some parts of the analysis, however, we will reduce the complexity of the model by assuming that species are ecologically independent, i.e. we impose the assumption that all species, $i = 1, ..., n$, are ecologically independent:

$$\text{Assumption:} \qquad G_{ix_j} \equiv 0 \ \ \forall \ \ j \neq i. \tag{A.1}$$

[3]This is a reasonable assumption for the Baltic Sea as different species are caught by different fleets (Voss et al., 2014b). We thus use this assumption in the analytical part of this paper as well as in the application to Baltic Sea fisheries.

Time subscripts are dropped for notational clarity from now on unless needed to avoid confusion. Furthermore, we use the vector notation $\mathbf{x} \equiv (x_1, ..., x_n)$ and $\mathbf{h} \equiv (h_1, ..., h_n)$.

2.2.2 Objective function

We now consider socially optimal multi-species management. An ecosystem manager simultaneously chooses harvest quantities for all species over time such that the present value of benefits, i.e., the sum of net benefits from harvest plus biodiversity value, is maximized:

$$\max_{\mathbf{h}} \int_0^\infty e^{-\rho t} \Big[\Pi(\mathbf{h}, \mathbf{x}) + v\, B(\mathbf{x}) \Big] dt \tag{4}$$

subject to the stock growth equations (3) and given the initial stock sizes x_{i0} for all species i. Here, we use ρ to denote the social discount rate and $\Pi(\mathbf{h}, \mathbf{x})$ to denote the economic net benefit derived from harvesting the multi-species ecosystem at time t. The net benefit per species, Π_i, is composed of a fixed price, p_i, multiplied with harvested biomass, h_i, minus possibly stock-dependent harvesting costs, $C_i(x_i, h_i)$ with $C_{ih_i} > 0$ and $C_{ix_i} < 0$. The parameter v measures the marginal value of biodiversity relative to harvest benefits in the objective function.

A couple of remarks are in place to discuss the meaning of using the biodiversity index (1) in the objective function (4). First, we interprete the parameter v as the shadow price of biodiversity. It converts biodiversity into monetary units. As equation (1) is linear homogeneous, biodiversity is measured in units of species abundance. Thus, v is measured in monetary units per species abundance (euros per ton of biomass, for example). As discussed above, however, the value of the biodiversity index (1) changes with ω. Thus, when a biodiversity index (1) is applied, it has to be kept in mind that v has to be adjusted when using different values of ω in the objective function.

Second, the assumption of linear homogeneity of the biodiversity index (1) carries economic meaning. Specifically, the particular specification of the exponent outside the parentheses in (1) becomes important because it determines (a) the elasticity of intertemporal substitution between the abundances of species, and (b) the elasticity of substitution between biodiversity and (monetary) income. In both respects, more general specifications of biodiversity are conceivable, but we think that the linear homogeneous index (1) is a simple and appealing specification. Thus, we focus on analyzing the implications of using this particular index in the objective function (4).

Third, using a CES-function to aggregate the stocks implies that the marginal biodiversity value of one species depends not only on the own stock but also on the stocks of the other species. This introduces an interdependency between the species on the management side. Hence, species have to be managed jointly even in a case where species are modeled as ecologically independent, i.e., when Assumption A.1 holds. This is in contrast to Kellner et al. (2010), where marginal biodiversity or non-fishing values are non-linear but only depend on the own resource stock, corresponding to the case of perfect substitutes, i.e., $\omega \to \infty$, in our more general set-up. See sections 2.2.4 and 2.2.5 for a detailed discussion of the resulting implications.

2.2.3 Necessary first order conditions and optimal steady state

To derive the conditions for optimal multi-species management, we consider the current-value Hamiltonian

$$H^c = \Pi(\mathbf{h}, \mathbf{x}) + v\, B(\mathbf{x}) + \sum_{i=1}^{n} \lambda_i (G_i(\mathbf{x}) - h_i). \tag{5}$$

Applying the maximum principle, the necessary conditions for optimal management are

$$\frac{\partial H^c}{\partial h_i} = 0 \qquad \Leftrightarrow \qquad \Pi_{h_i} = \lambda_i \tag{6a}$$

$$-\frac{\partial H^c}{\partial x_i} = \dot{\lambda}_i - \rho\lambda_i \qquad \Leftrightarrow \qquad \rho = \frac{\dot{\lambda}_i}{\lambda_i} + G_{ix_i} + \sum_{j\neq i} \frac{\lambda_j}{\lambda_i} G_{jx_i} + \frac{\Pi_{x_i}}{\lambda_i} + v\,\frac{B_{x_i}}{\lambda_i} \tag{6b}$$

with $\lambda_i, h_i, x_i \geq 0$, initial stock sizes given, and transversality conditions for all $i = 1, \ldots, n$. The first condition states that the marginal net benefit of harvesting species i

should equal the marginal opportunity costs of reducing stock i, captured by the shadow price of this stock, λ_i. This shadow price is determined by condition (6b). We will have a closer look at the different terms in that condition when discussing the optimal steady state, i.e., the long-run optimal stock sizes and harvest levels for the multi-species ecosystem in section 2.2.4. The steady state conditions are obtained by using (6a) and the conditions $\dot{x}_i = 0$ in (3) and $\dot{\lambda}_i = 0$ in (6b):

$$G_i(\bar{\mathbf{x}}) = \bar{h}_i \tag{7a}$$

$$\rho = G_{i\bar{x}_i} + \sum_{j \neq i} \frac{\Pi_{\bar{h}_j}}{\Pi_{\bar{h}_i}} G_{j\bar{x}_i} + \frac{\Pi_{\bar{x}_i}}{\Pi_{\bar{h}_i}} + \frac{v\, B_{\bar{x}_i}}{\Pi_{\bar{h}_i}} \tag{7b}$$

both for all $i = 1, \ldots, n$.

When taking species interactions into account, corner solutions to the dynamic optimization problem (4) may become possible. Here we focus on interior solutions described by the necessary conditions (6). To this end, we assume that species interactions are such that the maximized Hamiltonian is concave in the stock variables, i.e., the sufficiency conditions for the dynamic optimization problem are fulfilled (Arrow and Kurz, 1970).[4]

Condition (7b) states that in the optimal steady state the social discount rate, ρ, has to equal the interest rate earned on a marginal increase of each stock x_i. In the general case considered here, this own interest rate is determined by the marginal stock growth of the species itself, the marginal increase of the stock growth of all other species $j \neq i$, the value of the marginal stock effect reducing future harvesting costs, and the marginal contribution to biodiversity value.

[4]It is straightforward to verify that the sufficiency conditions are always fulfilled in the absence of species interactions, i.e., if Assumption A.1 holds, and if the biomass growth functions are concave.

2.2.4 Optimal steady state solutions for different cases of ecological interaction and fishery structure

To shed some light on the implications of (7b) for optimal steady state solutions, consider the case of ecologically independent species, i.e., impose Assumption A.1. In this case, the second term on the right-hand side (RHS) of (7b) would vanish. Note that even in this case the RHS of equation (7b) does not only depend on the stock of species i but also on the stocks of the other species, $j \neq i$, if biodiversity values are considered in addition to harvesting values, i.e., if $v > 0$. Thus, the biodiversity value modeled as a CES-function introduces an interdependency in steady state conditions even if the species are modeled as ecologically independent. It tends to balance steady state stocks and to reduce one-sided stock concentrations.

We argue that this interdependency on the management side is reasonable given that so far only the most important, direct interactions between marine species can be explicitly accounted for in bio-economic models such as predator-prey relationships or competition for food, and not all interrelationships are known. This holds particularly for the role of species for ecosystem functioning and regulation (Rockström et al., 2009). It thus seems reasonable to consider the whole ecosystem also when determining optimal stock levels for single species.

If, in addition to ecological independence, the economic benefits of harvest were independent of stock size, $\Pi_{x_i} = 0$, and if there was no biodiversity value, $v = 0$, the optimal steady-state stock sizes would be below the maximum-sustainable-yield (MSY) stock sizes, $x_{i,\mathrm{MSY}}$, defined by $G_{ix_i}(x_{i,\mathrm{MSY}}) = 0$ for all species i. This is a well-known result of discounting at a positive rate, $\rho > 0$. The two effects of stock-dependent harvesting costs, $\Pi_{x_i} > 0$, and of biodiversity value, $v > 0$, both tend to increase optimal steady-state stock sizes. If these effects are strong enough, the optimal steady-state stock sizes in absence of biological interactions will all be larger than $x_{i,\mathrm{MSY}}$. Thus, taking into account biodiversity values with a positive weight, $v > 0$, has a similar effect to introducing stock-dependent harvesting costs and implies a positive differential between ρ and $G_{i\bar{x}_i}$, i.e., $\rho > G_{i\bar{x}_i}$ in optimal steady state with $v > 0$.

This result still holds in the case of ecological interactions, i.e., if Assumption A.1 does no longer hold, as long as all other species j depend positively on the stock of species i, i.e., $G_{j\bar{x}_i} > 0$ for all $j \neq i$. This would imply that there is an additional positive external effect of the stock x_i. Negative ecological interactions, i.e., $G_{j\bar{x}_i} < 0$, can induce a negative differential between ρ and $G_{i\bar{x}_i}$, which would imply that steady state stocks of species i are lower. Brown et al. (2005) analyze this effect in a predator-prey model without biodiversity values. For all cases of ecological interactions, steady state stocks with biodiversity value *ceteris paribus* are larger than steady state stocks without considering biodiversity value.

Slightly rearranging (7b), multiplying with x_i and summing over all species i leads to the following condition:

$$\sum_{i=1}^{n} \left((\rho - G_{i\bar{x}_i})\, \Pi_{\bar{h}_i} - \Pi_{\bar{x}_i} - \sum_{j \neq i} G_{j\bar{x}_i}\, \Pi_{\bar{h}_j} \right) \bar{x}_i = v \sum_{i=1}^{n} \bar{x}_i B_{\bar{x}_i} = v\, B \tag{8}$$

The RHS of equation (8) is the weighted value of biodiversity in steady state, i.e., the total biodiversity value attached to aggregate steady state stock levels. The LHS of equation (8) represents the opportunity costs connected to introducing biodiversity values and increasing steady state stock levels such that a higher share of fish remains unfished. These opportunity costs are given by the sum of the economic net benefits of each steady state stock multiplied with the differential between the social discount rate and the marginal growth rate, but they are reduced by the positive effect of larger stocks on harvesting costs and further affected by the effect of increased stock sizes of species i on the other species due to ecological interdependencies.

2.2.5 Implications for the optimality of extinction

To proceed with the theoretical analysis, we focus on the effect of the biodiversity value and simplify the analysis by neglecting harvesting costs, i.e., we impose the following assumption:

$$\text{Assumption:} \quad \Pi(\mathbf{h}, \mathbf{x}) = \mathbf{p}'\,\mathbf{h} = \sum_{i=1}^{n} p_i\, h_i. \tag{A.2}$$

It has been shown in the literature that extinction can be optimal under certain circumstances. More specifically, for the case without biodiversity value, $v = 0$, and in the case of ecologically independent species (Assumption A.1 holds), extinction may be optimal for species with $G_{ix_i}(0) \leq \rho$ (Clark, 1973).

Proposition 1

Under Assumptions A.1 and A.2, and if $1 < \omega < \infty$ and $v > 0$ hold, extinction is never optimal, i.e., $x_i > 0$ for all $i = 1, \ldots, n$.

Proof. See Appendix A. □

Thus, extinction is never optimal when a biodiversity value specified as above is part of the objective function, and for $1 < \omega < \infty$, i.e., the specification of the biodiversity index that is applicable with a changing number of species. The reason for this result is that the marginal biodiversity value, B_{x_i}, diverges to infinity in this case if the stock x_i approaches zero.

Extinction can, however, be optimal if the substitution elasticity is infinitely large, i.e., in the case of perfect substitutes, $\omega = \infty$, but only for a species with $G_{ix_i}(0) < \rho - \frac{v\,n}{p_i}$. This implies that extinction can only be optimal in the case of perfect substitutes when the intrinsic growth rate is sufficiently small. Note that given that v, n, and p_i are all positive, extinction in the case with biodiversity value is *ceteris paribus* optimal at lower levels of intrinsic growth rates than in the case without biodiversity value.

2.3 Effects of parameter changes on optimal management

In this section, we analyze how changes in the shadow price of biodiversity, v, the market prices of the different species, p_i, and the elasticity of substitution, ω, influence optimal steady state solutions. We discuss the results for different cases of ecological interactions, i.e., case i) for ecologically independent species (Assumption A.1 holds), case ii) for a symbiotic system, $G_{ix_j} > 0$ for $i \neq j$, case iii) for a competitive system, $G_{ix_j} < 0$ for $i \neq j$, and case iv) for a predator-prey system $G_{1x_2} > 0$ and $G_{2x_1} < 0$ or the other way around. In all cases, we assume that there are no harvesting costs, i.e., Assumption A.2 holds, and we reduce complexity by considering a two-species ecosystem.

2.3.1 Comparative statics with respect to the biodiversity shadow price

We now analyze how changes in the shadow price of biodiversity, v, influence optimal steady state stocks in a two-species ecosystem. Under Assumption A.2, but with ecological interactions, we derive the following conditions:

$$\frac{d\bar{x}_1}{dv} = \frac{1}{\Delta}\left(B_{\bar{x}_2}\left(p_1 G_{1\bar{x}_1\bar{x}_2} + p_2 G_{2\bar{x}_1\bar{x}_2} + v B_{\bar{x}_1\bar{x}_2}\right) - B_{\bar{x}_1}\left(p_1 G_{1\bar{x}_2\bar{x}_2} + p_2 G_{2\bar{x}_2\bar{x}_2} + v B_{\bar{x}_2\bar{x}_2}\right)\right) \tag{9}$$

$$\frac{d\bar{x}_2}{dv} = \frac{1}{\Delta}\left(B_{\bar{x}_1}\left(p_1 G_{1\bar{x}_2\bar{x}_1} + p_2 G_{2\bar{x}_2\bar{x}_1} + v B_{\bar{x}_2\bar{x}_1}\right) - B_{\bar{x}_2}\left(p_1 G_{1\bar{x}_1\bar{x}_1} + p_2 G_{2\bar{x}_1\bar{x}_1} + v B_{\bar{x}_1\bar{x}_1}\right)\right) \tag{10}$$

The derivation and the definition of Δ is contained in Appendix B.As, by assumption, the sufficient conditions for optimality are met, it follows that $\Delta > 0$. The signs of the comparative static effects of a change in the biodiversity shadow price v on the optimal steady-state stock sizes depend on the types of ecological interactions (cases i-iv) and, in case iii) of competition, or case iv) of a predator-prey relationship, also on output prices and the biodiversity shadow price. Only for case i) of ecologically independent species or case ii) of a symbiotic relationship, the sign is unambiguous, as stated in the following proposition.

Proposition 2

The optimal steady-state stocks $\bar{x}_i$ of species $i = 1, 2$, increase with v,

$$\frac{d\bar{x}_i}{dv} > 0 \tag{11}$$

if i) species j and i are ecologically independent, i.e., if $G_{i\bar{x}_i\bar{x}_j} = 0$, or if ii) species i and j have a symbiotic relationship, i.e., if $G_{i\bar{x}_j\bar{x}_i} > 0 \;\; \forall \;\; i, j \;$ with $\; i \neq \; j$.

Proof. See Appendix C. □

We shall briefly discuss all possible cases of ecological relationships between the two species.

Case i) In the case of ecologically independent species, the term $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2}$ in equations (9) and (10) vanishes such that the positive effect of v on steady state stocks can be directly and unambiguously determined.

Case ii) In the case of symbiosis, the term $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2}$ is unambiguously positive such that the RHS of equations (9) and (10) are positive, and the positive effect of v on steady state stocks can be unambiguously determined.

Case iii) In a competitive ecosystem, the effect of v on both steady state stocks is ambiguous as $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} < 0$.

Case iv.) In a predator-prey system, the sign of $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2}$ is ambiguous and depends on the relative prices and the predation relationship between the species. Consider the example of a Lotka-Volterra predator-prey relationship such that the term modifies to $p_1\, \alpha_1 + p_2\, \alpha_2$, with constants α_1 and α_2. Assume without loss of generality that species 1 is the predator ($\alpha_1 > 0$) and species 2 is the prey ($\alpha_2 < 0$). Now, the term $p_1\, \alpha_1 + p_2\, \alpha_2$ would only be positive if the predator is sufficiently more valuable than the prey, $p_1 > (-\alpha_2/\alpha_1)\, p_2$, or if the predation coefficient α_1 is sufficiently larger than α_2 in absolute terms, $\alpha_1 > (-p_2/p_1)\, \alpha_2$, In these cases, again, the effect of v on both predator and prey stocks would be positive in steady state.

2.3.2 Comparative statics with respect to output prices

We now analyze how changes in the market prices of the two species, p_i, influence optimal steady state stocks in a two-species ecosystem. Under Assumption A.2, but with ecological interactions, we derive the following conditions (see Appendix D):

$$\frac{d\bar{x}_1}{dp_1} = \frac{1}{\Delta}\left(-G_{1\bar{x}_2}\ (p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_2}) + \right.$$
$$\left. + (\rho - G_{1\bar{x}_1})\ (p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2})\right) \quad (12)$$

$$\frac{d\bar{x}_2}{dp_1} = \frac{1}{\Delta}\left(-(\rho - G_{1\bar{x}_1})\ (p_1\, G_{1\bar{x}_2\bar{x}_1} + p_2\, G_{2\bar{x}_2\bar{x}_1} + v\, B_{\bar{x}_2\bar{x}_1}) + \right.$$
$$\left. + G_{1\bar{x}_2}\ (p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1})\right) \quad (13)$$

with $\Delta > 0$ as above. From these conditions, we derive the following result.

Proposition 3

The optimal steady state stocks $\bar{x}_j$ of species $j = 1, 2$ decrease with p_i,

$$\frac{d\bar{x}_j}{dp_i} < 0 \quad (14)$$

if i) species j and i are ecologically independent, i.e., if $G_{i\bar{x}_j} = 0$ and $G_{i\bar{x}_i\bar{x}_j} = 0$, or if ii) species j and i have a symbiotic relationship, i.e., if $G_{i\bar{x}_j} > 0$ and $G_{i\bar{x}_j\bar{x}_i} > 0 \ \forall \ i, j$ with $i \neq j$.

Proof. See Appendix E. □

The sign of the comparative-static effect of output prices on optimal steady-state stock sizes is unambiguous only for case i) of ecologically independent species or case ii) of a symbiotic relationship. In the other cases, the sign also depends on output prices and the biodiversity shadow price. Again, we shall briefly discuss all possible cases of ecological relationships between the two species.

Case i) In the case of ecologically independent species, equations (12) and (13) simplify such that the RHS of both are unambiguously negative. The negative effect of an increase in the price of species i on the steady state stock of species j is in contrast to a model of independent species without biodiversity value. For the case $v = 0$, a change in the price of either species would not affect the optimal steady-state stock sizes, as then $\rho = G_{i\bar{x}_i}$ for $i = 1, 2$. The negative cross price effect follows from the consideration of the biodiversity value which tends to balance steady state stock sizes among species.

Case ii) In the case of symbiosis, the term $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2}$ is unambiguously positive such that the effect of p_i on steady state stocks can be unambiguously determined as for case i).

Case iii) In a competitive system, the effect of p on both steady state stocks is ambiguous as $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} < 0$.

Case iv) For a predator-prey system, the effect depends on the relative prices and the predation relationship between the two species. Consider, again, the example of a Lotka-Volterra model with species 1 being the predator and species 2 being the prey, such that $p_1\, \alpha_1 + p_2\, \alpha_2$ is the decisive term. If the predator is sufficiently more valuable than the prey or if the predation coefficient α_1 is sufficiently larger than α_2 in absolute terms, the effect of changes in the price of the predator species is negative on both predator and prey. The effect of changes in the price of the prey species is still ambiguous in this case.

2.3.3 Comparative statics with respect to the elasticity of substitution ω between species in the biodiversity index

We now analyze how changes in the elasticity of substitution between the two species, ω, influence optimal steady-state stock sizes. We focus on a two-species ecosystem $n = 2$. Unambiguous conclusions for the effect of ω on steady state stocks are, however, only possible for ecologically independent species, i.e., for case i). Under Assumptions A.1 and A.2, we derive the following conditions:

$$\frac{d\bar{x}_1}{d\omega} = \frac{v}{\Delta}\left(\frac{\partial B_{\bar{x}_2}}{\partial \omega}\, v\, B_{\bar{x}_1\bar{x}_2} - \frac{\partial B_{\bar{x}_1}}{\partial \omega}\, (p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2})\right) \tag{15}$$

$$\frac{d\bar{x}_2}{d\omega} = \frac{v}{\Delta}\left(\frac{\partial B_{\bar{x}_1}}{\partial \omega}\, v\, B_{\bar{x}_2\bar{x}_1} - \frac{\partial B_{\bar{x}_2}}{\partial \omega}\, (p_1\, G_{1\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1})\right) \tag{16}$$

We have (using the notation $\hat{x} = x_1/x_2$):

$$\frac{\partial B_{\bar{x}_1}}{\partial \omega} = \frac{B_{\bar{x}_1}}{\omega^2\,(1-\omega)\left(1+\hat{x}^{\frac{1-\omega}{\omega}}\right)}\left(\omega^2\left(1+\hat{x}^{\frac{1-\omega}{\omega}}\right)\ln\left(2\,B_{\bar{x}_1}\right) + \hat{x}^{\frac{1-\omega}{\omega}}\,\ln\hat{x}\right). \tag{17}$$

It turns out that the comparative static effect of a change in the elasticity of substitution, ω, on the optimal steady-state stock sizes, is rather complicated. The reason is that a change in ω has two effects on the biodiversity index. One effect is that with a higher value of ω the decision maker cares somewhat less for the evenness in species abundances. In addition, for a given biodiversity shadow price v the biodiversity value decreases with ω, as pointed out in Section 2.1. The following proposition states conditions under which the former effect dominates the latter. We use $\hat{\omega}$ to denote the solution of $\hat{\omega} + \ln(\hat{\omega} - 1) = 0$, which is $\hat{\omega} \approx 1.28$.

Proposition 4

(a) If $\omega \geq \hat{\omega}$, the smaller steady state stock decreases with ω. (b) If $\omega < \hat{\omega}$, it exists a $0 \leq \underline{\hat{x}} < 1$ such that the smaller steady state stock decreases with ω for all $\hat{x} \in (\underline{\hat{x}}, 1)$.

Proof. See Appendix F. □

The sign of the comparative static effect of a change in the elasticity of substitution, ω, on the optimal steady-state stock sizes is ambiguous for the larger steady state stock and negative for the smaller one if ω is large enough. The intuition behind this result is that a larger elasticity of substitution allows a larger divergence between optimal stock sizes in steady state. Consequently, the smaller stock decreases with ω in steady state. Again, this result only holds for case i) and cannot be unambiguously derived for the other three cases of possible ecological interactions.

3 Application to Baltic Sea fisheries

3.1 Baltic Sea fisheries

The marine ecosystem in the central Baltic Sea is dominated by three fish species: cod, sprat, and herring. These species also form the basis of the economically most important fisheries in the Baltic Sea. In addition, their stocks are closely connected by strong ecological inter-connections among species (Köster and Möllmann, 2000), as cod preys on both sprat and herring. In economic terms, the cod fishery used to be the most important of the three. Overfishing, however, caused a decline in the cod stock during the last decades, and only recently the introduction of a long-term management plan has led to some signs of stock recovery again.

The upper panel in Figure 1 shows the development of the stock sizes of cod, sprat, and herring from 1974 to 2012 measured in units of spawning stock biomass, i.e., the biomass of all fish in spawning age. The lower panel shows the corresponding levels of the biodiversity index (1) using spawning stock biomasses as abundance indicators for a relatively large elasticity of substitution ($\omega = 2$) and for a relatively small elasticity of substitution ($\omega = 0.5$). It becomes obvious that the biodiversity indices for both elasticities of substitution follow the same trend in general. Biodiversity measured using the smaller elasticity of substitution, however, always is below biodiversity levels measured using a larger elasticity of substitution. This reflects the influence of ω discussed in section 2.1. Both measures are the closer together the more even the distribution of

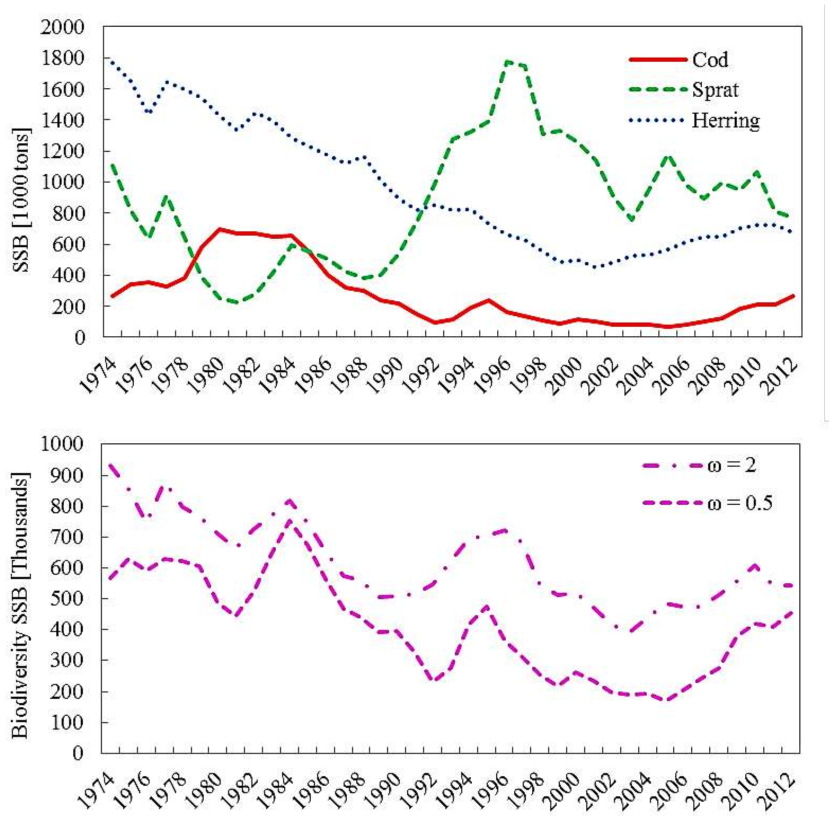

Figure 1. Historic development of spawning stock biomasses and levels of the biodiversity index using spawning stock biomasses as abundance indicators for cod, sprat, and herring in the Baltic Sea.

species abundances in the ecosystem is. Overall, the smaller the elasticity of substitution, the relatively more important the smallest stock in the ecosystem becomes for the biodiversity index.

3.2 Description of the age-structured model

For the application to the Baltic sea fisheries, we replace the biomass model of resource dynamics (equation 3) by a state-of-the-art age-structured population model. Considering the more complex age-structured model allows comparing the effects of using biodiversity indices calculated in terms of biomasses and in terms of the number of individuals as measures of species abundance. The model we employ here builds on Voss et al. (2014a,b), who provide a bio-economic fishery model for the Baltic cod, sprat, and herring fisheries, taking the predator-prey relationship between cod and the two other species into account. This model is an extension of a single-species age-structured fishery model (Tahvonen, 2009; Tahvonen et al., 2013). A second deviation from the continuous-time model used in section 2 is that we consider a discrete-time, discrete age-structured setting for the quantitative application.

In the following, we use x_{ist} to denote the number of fish of species $i \in \{C, S, H\}$, where C stands for cod, S for sprat, and H for herring, in age group $s = 1, \ldots, S$ and at the beginning of period $t = 0, 1, \ldots$. Using the indices $i \in \{C, S, H\}$ for the species, and $s = 1, \ldots, S$ for the age group, where $S > 1$ is the oldest age group considered in the model, we use $\alpha_{is} > 0$ to denote age-specific survival rates, $\gamma_{is} > 0$ to denote age-specific proportions of mature individuals, and w_{is} to denote the mean weights (in kilograms). For cod, all of these parameters are assumed to be constant (Tahvonen, 2009) as in the standard biological stock assessments for the Eastern Baltic cod (ICES, 2012). For sprat and herring, we assume that proportions of mature individuals and weights are constant, but the survival rates of both depend on cod spawning stock biomass. For the age-specific survival rates, we use the specification

$$\alpha_{is} = \exp\left(-M_{2is} - \delta_{is}\, x_{C0t}\right) \quad \text{for} \quad i = S, H, \tag{18}$$

where M_{2is} is instantaneous natural mortality of sprat ($i = S$) and herring ($i = H$) cohort s in the absence of cod, and $\delta_{is} > 0$ is a parameter that measures the dependency of instantaneous natural mortality of sprat ($i = S$) and herring ($i = H$) cohort s on cod spawning stock biomass, x_{C0t}.

Denoting the recruitment function for species i by $\varphi_i(\cdot)$ and the spawning biomass by x_{i0t}, the age-structured population model with harvesting activity can be summarized as follows:

$$x_{i0t} = \sum_{s=1}^{S} \gamma_{is}\, w_{is}\, x_{ist} \tag{19a}$$

$$x_{i1,t+1} = \varphi_i(x_{i0t}) \tag{19b}$$

$$x_{i,s+1,t+1} = \alpha_{is}\, (x_{ist} - h_{ist}) \qquad \text{for } s = 1, \ldots, S-2 \tag{19c}$$

$$x_{i,S,t+1} = \alpha_{i,S-1}\, \left(x_{i,S-1,t} - h_{i,S-1,t}\right) + \alpha_{iS}\, \left(x_{iSt} - h_{iSt}\right) \tag{19d}$$

Here, we use h_{ist} to denote the number of fish harvested from cohort s of species i in period t. We maintain the assumption of perfect selectivity of harvest with respect to the species, which is a reasonable assumption for the Baltic Sea, as different species are caught by different fleets. Aggregate instantaneous fishing mortality F_{it} for species i in year t translates into age-specific fishing mortalities, captured by the constant, age-specific catchability coefficients $q_{is} \geq 0$, such that

$$h_{ist} = q_{is}\, x_{ist}\, \left(1 - \exp\left(-F_{it}\right)\right). \tag{20}$$

For cod and herring we assume stock-recruitment functions of the Ricker (1954) type (Voss et al., 2014a), i.e., we assume

$$\varphi_i(x_{i0t}) = \phi_{i1}\, x_{i0t}\, \exp\left(-\frac{x_{i0t}}{\phi_{i2}}\right) \tag{21}$$

with $\phi_{i1}, \phi_{i2} > 0$. For sprat we assume a Beverton-Holt type (Tahvonen et al., 2013), i.e., we assume

$$\varphi_S(x_{S0t}) = \frac{\phi_{S1}\, x_{S0t}}{1 + \frac{x_{S0t}}{\phi_{S2}}} \tag{22}$$

with $\phi_{S1}, \phi_{S2} > 0$.

For modeling the profits of the cod fishery, we use the specification from Quaas et al. (2012) with age-specific prices and a cost function of the Spence type (Spence, 1974). Thus, profits of the cod fishery in year t are

$$\pi_{Ct} = \sum_{s=1}^{S} p_{Cs}\, w_{Cs}\, h_{Cst} - c_C\, F_{Ct}, \tag{23}$$

where p_{Cs} are prices for cod in age group s, instantaneous fishing mortality F_{Ct} equals instantaneous effort, and c_C is the unit effort cost for the cod fishery. Sprat and herring are modeled as schooling fisheries (Tahvonen et al., 2013), where the market price p_i is assumed to be independent of age. The profits in the sprat and herring fisheries thus are

$$\pi_{it} = (p_i - c_i) \sum_{s=1}^{S} p_{is}\, w_{is}\, h_{ist} \tag{24}$$

with analogous interpretations for the symbols as for the cod fishery. The values for the parameters of the population model and for prices and harvesting costs are taken from Voss et al. (2014b) and can be found in Appendix G.

For the harvesting benefits $\Pi(\mathbf{h}, \mathbf{x})$, we assume that the fishery manager has some aversion against income inequality across fisheries, and specify

$$\Pi(\mathbf{h}, \mathbf{x}) = \left(\frac{1}{3}\pi_{Ct}^{1-\eta} + \frac{1}{3}\pi_{St}^{1-\eta} + \frac{1}{3}\pi_{Ht}^{1-\eta} \right)^{\frac{1}{1-\eta}} . \tag{25}$$

In our simulations, we use $\eta = 0.25$.

We assume that the aim is to maximize (4) with harvesting benefits (25) and taking a biodiversity value (1) into account. Here, we use two versions of the biodiversity index: In one version, we use the spawning stock numbers, $\sum_{s=1}^{S} \gamma_{is}\, x_{ist}$, as abundance indicators for species i, in the other one, we use the spawning stock biomasses, x_{i0t}, as abundance indicators. We vary the marginal WTP for biodiversity, v, and use a discount rate of zero, $\rho = 0$, in the numerical optimization.

The numerical optimization is performed using Knitro (version 8.1) with AMPL. Programming codes are available upon request.

3.3 Numerical optimization results

The results of our numerical optimization show the effects of changes in the shadow price of biodiversity, v, on the optimal steady state stocks of cod, sprat, and herring as well as on the optimal profits of the three fisheries and on optimal biodiversity levels. We show the sensitivity of the results to the shadow price of biodiversity, v, for a relatively large elasticity of substitution ($\omega = 2$) and for a relatively small elasticity of substitution ($\omega = 0.5$), and we do so for formulating the biodiversity index in the objective function in terms of biomass and in terms of number of individuals.

Main effects of stock changes on biodiversity

Before interpreting the simulation results in more detail, we would like to point out that a change in the steady state stock of one species has two main effects on biodiversity that drive the results to be presented. First, there is the stock or abundance effect: Increasing overall abundance, in terms of biomass or numbers, *ceteris paribus* increases biodiversity. Second, there is the diversity or scarcity effect: If the stock of a relatively scarce species increases, this increases the evenness of stock sizes in the ecosystem, which *ceteris paribus* increases biodiversity. If the stock of a relatively abundant species increases, however, the evenness of the stock sizes in the ecosystem is reduced, which is negative for biodiversity.

In our application, the relatively scarce predator species cod crucially influences the biodiversity of the ecosystem. First, there is a positive scarcity or diversity effect: Cod is a relatively scarce species; in terms of number of individuals, it even is the scarcest species among the three species in the Baltic ecosystem considered here. Thus, increasing cod stocks is positive for biodiversity as it increases the evenness of the species distribution. Second, however, there is a negative effect on the stocks of the prey species: An increase in the cod stock leads to decreasing stocks of sprat and herring. If this leads to a reduction of total stock biomass or number of individuals, this tends to reduce the biodiversity index.

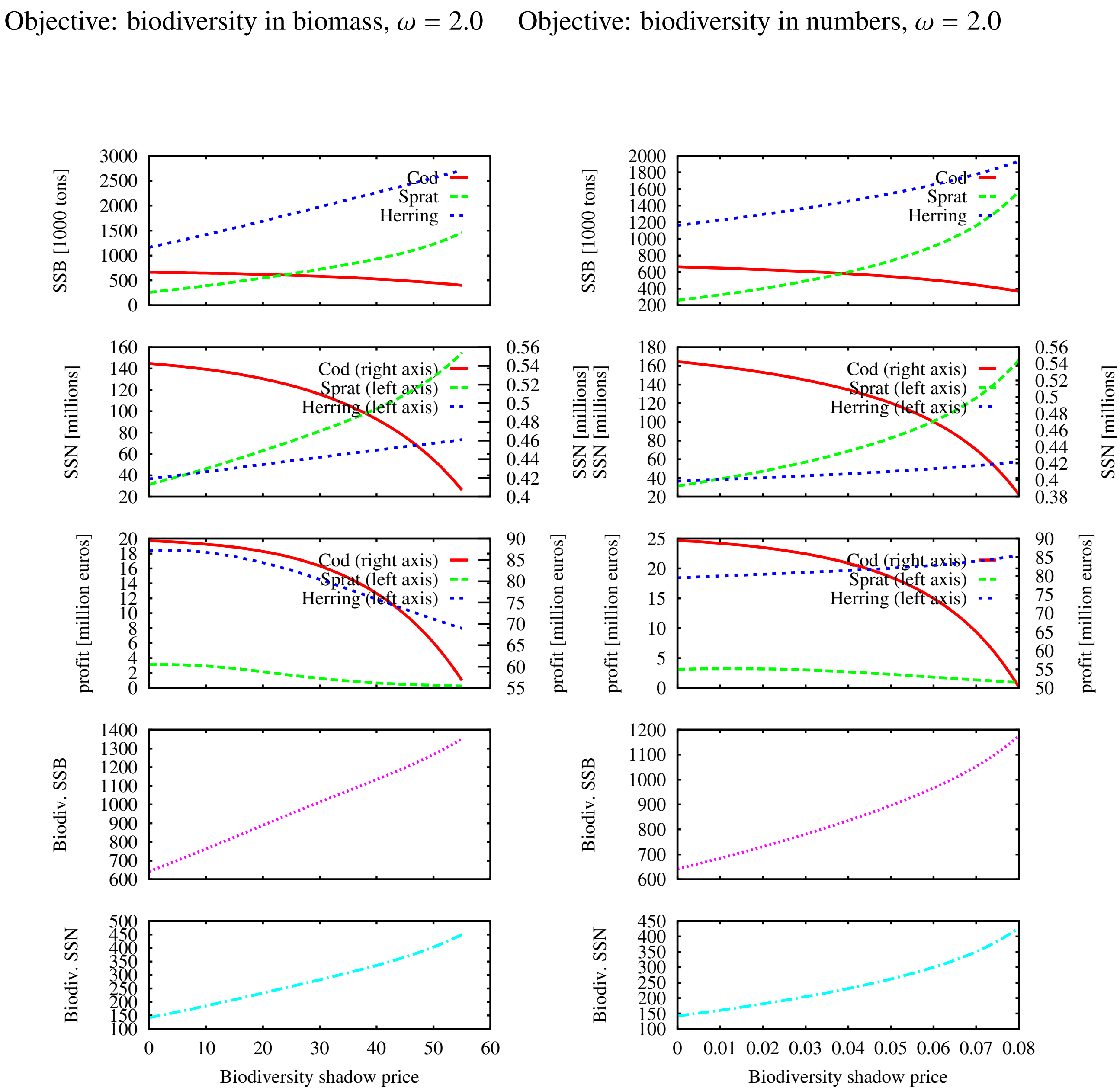

Figure 2. Effects of varying biodiversity shadow price v for a biodiversity objective with relatively large elasticity of substitution, $\omega = 2$. Panels on the left-hand side show results when the biodiversity objective is formulated with spawning stock biomasses, SSB, as abundance measures, panels on the right-hand side show results for a biodiversity objective with spawning stock numbers, SSN, as abundance measures. The panels in the rows show, from top to bottom, optimal steady-state stock sizes in terms of SSB and SSN, profit and biodiversity indices with abundance measures SSB and SSN.

Objective: biodiversity in biomass, ω = 0.5 Objective: biodiversity in numbers, ω = 0.5

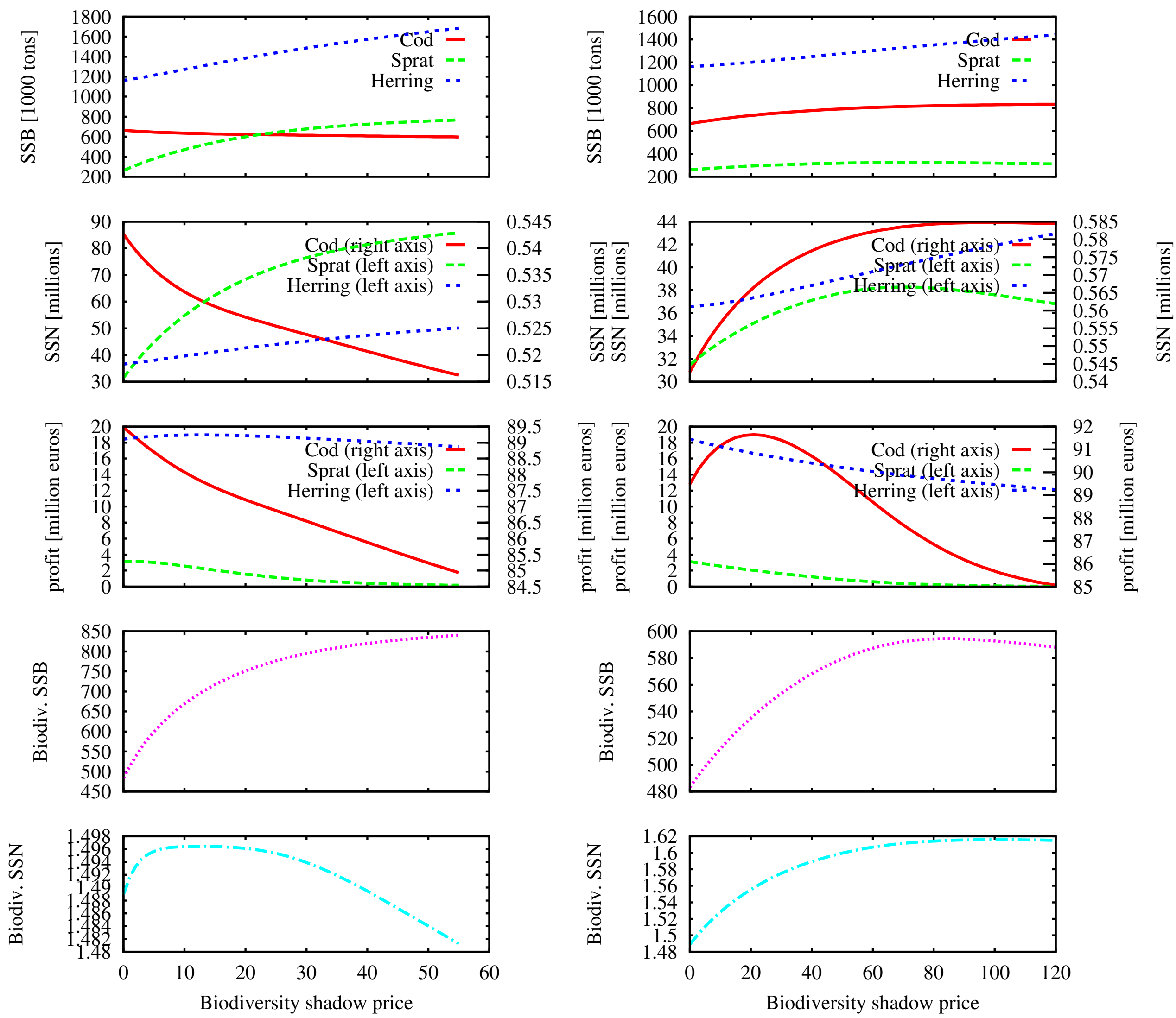

Figure 3. Effects of varying biodiversity shadow price v for a biodiversity objective with relatively small elasticity of substitution, ω = 0.5. Panels are as in Figure 2.

Effects on optimal steady state stocks

Figure 2 shows the results of a change in the shadow price of biodiversity, v, for a relatively large elasticity of substitution, $\omega = 2$. The optimal stocks of the prey species sprat and herring increase with v while the optimal stock of the predator species cod decreases with v. This holds for optimal stocks measured in biomass (first row) and for optimal stocks measured in numbers (second row). This shows that the species interaction makes a qualitative difference compared to an ecosystem with ecologically independent species. As our theoretical results derived in section 2.3.1 have shown, the effect of v on optimal steady state stocks would be unambiguously positive if there was no ecological interactions between the species.

The results for $\omega = 2$ are quite similar when comparing whether the objective is to maximize biodiversity in terms of biomass (left panel in Figure 2) or in terms of numbers (right panel in Figure 2), although the evenness of species is very different when abundances are measured in terms of biomasses or in terms of numbers. With abundances measured in terms of numbers, the stock size of cod is about two orders of magnitude smaller than the stock sizes of sprat and herring, while with biomasses as abundance measures the difference is much smaller. The relatively large elasticity of substitution, however, implies that an even distribution of species abundances is relatively less important than the absolute aggregate biomass or number of individuals such that the main aim is to increase overall abundances.

Figure 3 shows the results for a relatively low elasticity of substitution, $\omega = 0.5$. In this case, the evenness of species abundances plays a relatively large role. The figure shows that this leads to interesting differences in the effect of a change in v on optimal stocks between the objective to maximize biodiversity in terms of biomass and in terms of numbers. For the biodiversity objective measured in terms of numbers, the optimal stock of cod now increases while the optimal stock of sprat first increases and then decreases with v. The reason is that cod is particularly scarce when abundance is measured in numbers of individuals. This scarcity effect dominates the overall abundance effect such that the cod stock increases with v, although this causes increased predation on the more numerous sprat and herring stocks.

Effects on optimal levels of biodiversity

For $\omega = 2$, we observe the expected effect that biodiversity levels both measured in terms of biomass and number of individuals increase with v. For $\omega = 0.5$, in contrast, we find a trade-off between the two types of biodiversity measures for large values of the biodiversity shadow price. The left panel of Figure 3, where the biodiversity objective is formulated in biomass, shows that the biodiversity index in terms of biomass unambiguously increases with the shadow price, as expected. If we measure the biodiversity outcome in terms of numbers of individuals, however, we observe a *decline* in biodiversity when the shadow price of biodiversity increases beyond a level of about 20 euros per ton of spawning stock biomass. Looking at the right-hand panel of Figure 3, we find the reverse pattern for biodiversity shadow prices beyond about 100 euros per million fish: While the biodiversity index in terms of numbers (the one included in the objective function in this case) continues to slightly increase with v, biodiversity measured in terms of biomass decreases with the shadow price of (number) biodiversity.

The reason for this trade-off is that in terms of biomass the unevenness between the three stock sizes is by far not as pronounced as in terms of numbers of individual fish. Thus, when caring for biodiversity in terms of biomass, the desire for an overall larger abundance of fish dominates the desire for evenness, and one tends to slightly decrease the cod stock in order to build up the other two stocks. Measuring biodiversity in terms of numbers, however, implies that the unevenness increases so strongly that the biodiversity index decreases despite an overall increasing abundance of fish. This effect is reversed if the fishery manager cares for biodiversity measured in terms of numbers of individual fish (right-hand panel of Figure 3), which in this case leads to a decrease in the biomass-biodiversity index when the shadow price for biodiversity in numbers increases beyond a certain value.

As discussed in section 2.2, the shadow price of biodiversity, v, is not independent of the elasticity of substitution, ω, and the index value, B, which is also reflected in the numerical optimization results. Comparing Figure 2 and Figure 3, it becomes obvious that for the same objective and the same stock sizes, the value of B is lower for $\omega = 0.5$ than for $\omega = 2.0$. This is particularly pronounced for biodiversity in terms of numbers for the case $\omega = 0.5$, for which the unevenness is highest and (number) biodiversity is two orders of magnitudes smaller than for $\omega = 2$.

Comparison between optimal and historic stocks and biodiversity levels

Comparing optimal biodiversity levels and stock sizes to historic ones (Figure 1), we observe that current biodiversity levels measured in biomass are below the optimal levels, even if one considers $v = 0$. The same applies to current stocks of herring and cod, which are also below optimal levels. Both results hold for $\omega = 0.5$ and for $\omega = 2.0$. Current stocks of sprat, in contrast, are above optimal levels for low values of v. In the case of $\omega = 0.5$ and when biodiversity in terms of numbers is the objective, current sprat stocks are higher than optimal levels even for a larger range of values for v. Comparing optimal biodiversity levels to historically high levels that prevailed for example during the 1980s, we observe that these historically high levels are in the range of optimal levels for all cases except the one in which $\omega = 0.5$ and the objective is to maximize biodiversity in terms of numbers.

Effects on optimal profits

The effects of v on fishing profit are mostly negative, and aggregate profits always fall with the introduction of biodiversity values. Increasing the shadow price of biodiversity v has this negative effect on profits because harvested amounts of fish decrease in order to increase the standing stocks. For $\omega = 2$, a positive effect of v on profits only occurs for the herring fishery when the objective is to maximize biodiversity in numbers. The reason is that the concern for biodiversity leads to a decreasing cod stock, thus alleviating predation pressure on herring. The resulting larger steady-state stock size of herring enables a more profitable fishery.

For $\omega = 0.5$ and small values of the biodiversity shadow price, we also see an increase in cod profits with v. The reason is that without biodiversity value, it is optimal to reduce the cod stock below the single-species optimal steady-state stock size (i.e., the steady state stock size that would result when neglecting the predation effect on the two other species), in order to reduce fishing pressure on the two prey species. As cod is the scarcest species among the three in terms of stock numbers, increasing the biodiversity value leads to an increased cod stock size in steady state. Thus, the cod stock approaches its single-species optimal stock size and profit increases. For still higher biodiversity shadow price, also the cod stock is built up beyond the single-species profit-maximizing stock level and profits decrease again.

4 Discussion and conclusions

In this paper, we have studied how the consideration of a biodiversity value in the objective function of a dynamic bio-economic model affects the optimal management of a multi-species ecosystem with and without ecological interactions. The biodiversity index used in this paper is a CES-function of in-situ species abundances. This index fulfills all axioms of Buckland et al. (2005) if the species are assumed to be (imperfect) substitutes, $\omega > 1$. Such an index thus seems well-suited to monitor developments of biodiversity over time, in particular if the number of species might change over time as it is the case for global biodiversity. Specifying the substitution elasticity to a value below or equal to one, $\omega \leq 1$, however, is reasonable only for situations where extinction is out of scope because the biodiversity index would be undefined in that case.

We have shown analytically that species extinction is never optimal when biodiversity, measured as a CES-function with $1 < \omega < \infty$, plays a role in the objective function. This is in contrast to prior findings such as by Alexander (2000), where existence values made extinction less likely but not impossible in optimal steady states. The reason for our result is that for the case of imperfect substitutes, the marginal contribution of a species to the biodiversity index goes to infinity if its stock size approaches zero.

Our analysis has also revealed the implications of varying the shadow price of biodiversity, v, in the objective function. This shadow price controls how strongly conservation goals are weighted in the objective function. Without ecological interactions, increasing the shadow price of biodiversity, v, unequivocally increases the steady state stock sizes of all species in the system. With predator-prey interactions, optimal steady state stock sizes may increase or decrease with v, depending on the strength of species interaction and the relative market price of predator and prey species. A quantitative application to the three-species fishery in the Baltic Sea has shown that the steady state stock of a relatively scarce predator species may decrease with the biodiversity shadow price. This is the case if substitutability between the species is relatively high such that the objective to increase the overall abundance dominates the objective to increase species evenness.

We have further shown how the specification of the substitution elasticity, ω, between species, and the choice of abundance indicators in the biodiversity index influence optimal management solutions. The larger the elasticity of substitution, ω, between the species, the more valuable it is to increase aggregate abundance compared to ensuring an even distribution of species abundances. In the case of the Baltic fisheries, cod is significantly less abundant than sprat and herring. This relative scarcity of cod is particularly pronounced when species abundance is measured in terms of numbers, as individual cod are much larger than individual sprat or herring. Consequently, optimal cod stocks increase with the shadow price of biodiversity, v, when the objective is to maximize biodiversity in numbers and ω is relatively small. In all other cases, the optimal cod stocks decrease with v, in order to reduce cod predation on herring and sprat stocks and thus increase the overall abundance of fish in the ecosystem.

These results show that the exact specification of the biodiversity index has important implications for optimal management. Results can change qualitatively when the indicator of species abundance or the value for the elasticity of substitution is changed. One conclusion, however, seems to be robust: as long as species diversity (as measured by a CES-function) plays a role in the objective function, and species are imperfect substitutes, species extinction is never optimal.

Appendix A. Proof of proposition 1.

Under the given assumptions, the condition for the optimal steady state simplifies to

$$(\rho - G_{ix_i})\, p_i = v\, B_{x_i} \tag{A.1}$$

for all $i = 1, \ldots, n$. Here,

$$B_{x_i} = \frac{1}{n}\, x_i^{-\frac{1}{\omega}} \left(\frac{1}{n} \sum_{j=1}^{n} x_j^{\frac{\omega-1}{\omega}} \right)^{\frac{\omega}{\omega-1}-1} = n^{\frac{\omega}{1-\omega}} \left(1 + \sum_{j \neq i}^{n} \left(\frac{x_i}{x_j} \right)^{\frac{1-\omega}{\omega}} \right)^{\frac{1}{\omega-1}}$$

For $x_i \to 0$ and $1 < \omega < \infty$, B_{x_i} diverges to infinity. Thus, the RHS of equation (A.1) diverges to infinity while the LHS of equation (A.1) is positive but finite. Equation (A.1) will thus never be balanced for $x_i \to 0$.

Appendix B. Comparative statics w.r.t. v.

Under Assumption A.2, but with ecological interactions, condition (7b) simplifies to

$$\left(\rho - \sum_{j=1}^{n} \frac{p_j}{p_i} G_{jx_i}\right) p_i = v\, B_{x_i} \tag{B.2}$$

Differentiating (B.2) with respect to v, we obtain

$$-\left(p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1}\right) \frac{d\bar{x}_1}{dv} - \left(p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_2}\right) \frac{d\bar{x}_2}{dv} = B_{\bar{x}_1} \tag{B.3}$$

$$-\left(p_1\, G_{1\bar{x}_2\bar{x}_1} + p_2\, G_{2\bar{x}_2\bar{x}_1} + v\, B_{\bar{x}_2\bar{x}_1}\right) \frac{d\bar{x}_1}{dv} - \left(p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2}\right) \frac{d\bar{x}_2}{dv} = B_{\bar{x}_2} \tag{B.4}$$

Solving yields

$$\frac{d\bar{x}_1}{dv} = \frac{1}{\Delta}\left(B_{\bar{x}_2}\left(p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_2}\right) - B_{\bar{x}_1}\left(p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2}\right)\right) \tag{B.5}$$

$$\frac{d\bar{x}_2}{dv} = \frac{1}{\Delta}\left(B_{\bar{x}_1}\left(p_1\, G_{1\bar{x}_2\bar{x}_1} + p_2\, G_{2\bar{x}_2\bar{x}_1} + v\, B_{\bar{x}_2\bar{x}_1}\right) - B_{\bar{x}_2}\left(p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1}\right)\right) \tag{B.6}$$

with

$$\begin{aligned}
\Delta = {} & p_1^2\left(G_{1\bar{x}_1\bar{x}_1}\, G_{1\bar{x}_2\bar{x}_2} - G^2_{1\bar{x}_1\bar{x}_2}\right) + p_2^2\left(G_{2\bar{x}_1\bar{x}_1}\, G_{2\bar{x}_2\bar{x}_2} - G^2_{2\bar{x}_1\bar{x}_2}\right) \\
& + p_1\, p_2\left(G_{1\bar{x}_1\bar{x}_1}\, G_{2\bar{x}_2\bar{x}_2} + G_{2\bar{x}_1\bar{x}_1}\, G_{1\bar{x}_2\bar{x}_2} + 2\, G_{1\bar{x}_1\bar{x}_2}\, G_{2\bar{x}_1\bar{x}_2}\right) \\
& + v\Big(B_{\bar{x}_1\bar{x}_1}\left(p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2}\right) + B_{\bar{x}_2\bar{x}_2}\left(p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1}\right) \\
& \qquad - 2\, B_{\bar{x}_1\bar{x}_2}\left(p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2}\right)\Big) \\
& + v^2\left(B_{\bar{x}_1\bar{x}_1}\, B_{\bar{x}_2\bar{x}_2} - B^2_{\bar{x}_1\bar{x}_2}\right) > 0 \\
= {} & p_1^2\left(G_{1\bar{x}_1\bar{x}_1}\, G_{1\bar{x}_2\bar{x}_2} - G^2_{1\bar{x}_1\bar{x}_2}\right) + p_2^2\left(G_{2\bar{x}_1\bar{x}_1}\, G_{2\bar{x}_2\bar{x}_2} - G^2_{2\bar{x}_1\bar{x}_2}\right) \\
& + p_1\, p_2\left(G_{1\bar{x}_1\bar{x}_1}\, G_{2\bar{x}_2\bar{x}_2} + G_{2\bar{x}_1\bar{x}_1}\, G_{1\bar{x}_2\bar{x}_2} + 2\, G_{1\bar{x}_1\bar{x}_2}\, G_{2\bar{x}_1\bar{x}_2}\right) \\
& + v\Big(B_{\bar{x}_1\bar{x}_1}\left(p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2}\right) + B_{\bar{x}_2\bar{x}_2}\left(p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1}\right) \\
& \qquad - 2\, B_{\bar{x}_1\bar{x}_2}\left(p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2}\right)\Big) > 0
\end{aligned} \tag{B.7}$$

Appendix C. Proof of proposition 2.

For $G_{i\bar{x}_i\bar{x}_j} = 0$, the expressions (9) and (10) simplify to

$$\frac{d\bar{x}_1}{dv} = \frac{1}{\Delta}\left(B_{\bar{x}_2}\, v\, B_{\bar{x}_1\bar{x}_2} - B_{\bar{x}_1}\,\left(p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2}\right)\right) > 0 \tag{C.8}$$

$$\frac{d\bar{x}_2}{dv} = \frac{1}{\Delta}\left(B_{\bar{x}_1}\, v\, B_{\bar{x}_2\bar{x}_1} - B_{\bar{x}_2}\,\left(p_1\, G_{1\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1}\right)\right) > 0 \tag{C.9}$$

with

$$\Delta = p_1\, p_2\, G_{1\bar{x}_1\bar{x}_1}\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_1}\, p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2}\, p_1\, G_{1\bar{x}_1\bar{x}_1} > 0. \tag{C.10}$$

This concludes the proof of part (i) of the proposition.

For $G_{i\bar{x}_i\bar{x}_j} > 0$, $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} > 0$ such that the RHS of (9) and (10) are positive. This concludes the proof of part (ii) of the proposition.

Appendix D. Comparative statics w.r.t. p.

Differentiating (B.2) with respect to p_1 for $i = 1, 2$ we obtain

$$(\rho - G_{1\bar{x}_1}) - (p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1})\, \frac{d\bar{x}_1}{dp_1} - (p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_2})\, \frac{d\bar{x}_2}{dp_1} = 0 \quad \text{(D.11)}$$

$$- G_{1\bar{x}_2} - (p_1\, G_{1\bar{x}_2\bar{x}_1} + p_2\, G_{2\bar{x}_2\bar{x}_1} + v\, B_{\bar{x}_2\bar{x}_1})\, \frac{d\bar{x}_1}{dp_1} - (p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2})\, \frac{d\bar{x}_2}{dp_1} = 0 \quad \text{(D.12)}$$

Solving yields

$$\frac{d\bar{x}_1}{dp_1} = \frac{1}{\Delta}\, ((-G_{1\bar{x}_2}\, (p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_2}) + (\rho - G_{1\bar{x}_1})\, (p_1\, G_{1\bar{x}_2\bar{x}_2} + p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2})) \quad \text{(D.13)}$$

$$\frac{d\bar{x}_2}{dp_1} = \frac{1}{\Delta}\, (-(\rho - G_{1\bar{x}_1})\, (p_1\, G_{1\bar{x}_2\bar{x}_1} + p_2\, G_{2\bar{x}_2\bar{x}_1} + v\, B_{\bar{x}_2\bar{x}_1}) + G_{1\bar{x}_2}\, (p_1\, G_{1\bar{x}_1\bar{x}_1} + p_2\, G_{2\bar{x}_1\bar{x}_1} + v\, B_{\bar{x}_1\bar{x}_1})) \quad \text{(D.14)}$$

with Δ as above.

Appendix E. Proof of proposition 3.

For $G_{i\bar{x}_j} = 0$ and $G_{i\bar{x}_i\bar{x}_j} = 0$, the expressions (12) and (13) simplify to

$$\frac{d\bar{x}_1}{dp_1} = \frac{1}{\Delta}(\rho - G_{1\bar{x}_1})(p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2}) < 0 \tag{E.15}$$

$$\frac{d\bar{x}_2}{dp_1} = \frac{-1}{\Delta}(\rho - G_{1\bar{x}_1}) v\, B_{\bar{x}_2\bar{x}_1} < 0 \tag{E.16}$$

with

$$\Delta = p_1\, p_2\, G_{1\bar{x}_1\bar{x}_1}\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_1\bar{x}_1}\, p_2\, G_{2\bar{x}_2\bar{x}_2} + v\, B_{\bar{x}_2\bar{x}_2}\, p_1\, G_{1\bar{x}_1\bar{x}_1} > 0. \tag{E.17}$$

This concludes the proof of part (i) of the proposition.

For $G_{i\bar{x}_j} > 0$ and $G_{i\bar{x}_j\bar{x}_i} > 0$, $p_1\, G_{1\bar{x}_1\bar{x}_2} + p_2\, G_{2\bar{x}_1\bar{x}_2} > 0$ and $(\rho - G_{1\bar{x}_1}) > 0$ such that the RHS of (12) and (13) are negative. This concludes the proof of part (ii) of the proposition.

Appendix F. Proof of Proposition 4.

We first determine the sign of the following term in (15)

$$\frac{\partial B_{\bar{x}_2}}{\partial \omega} B_{\bar{x}_1 \bar{x}_2} - \frac{\partial B_{\bar{x}_1}}{\partial \omega} B_{\bar{x}_2 \bar{x}_2}$$
$$= -2^{-2\,\frac{\omega}{\omega-1}} \frac{\hat{x}^{\frac{1}{\omega}} \left(1 + \hat{x}^{\frac{1-\omega}{\omega}}\right)^{\frac{2}{\omega-1}}}{x_2\,\omega^2} \underbrace{\frac{1}{1-\omega} \left(\frac{\hat{x}^{\frac{1-\omega}{\omega}}}{1 + \hat{x}^{\frac{1-\omega}{\omega}}} \ln(\hat{x}) + \ln\left(\frac{1 + \hat{x}^{\frac{1-\omega}{\omega}}}{2}\right)^{\frac{\omega}{\omega-1}} \right)}_{\equiv \Omega} \leq 0 \quad \text{(F.18)}$$

Lemma 1

$\Omega \geq 0$ with $\Omega = 0$ only for $\hat{x} = 1$.

Proof. Ω has a global minimum $\Omega = 0$ at $\hat{x} = 1$, as

$$\frac{d\Omega}{d\hat{x}} = \frac{\hat{x}^{\frac{1-\omega}{\omega}} \ln(\hat{x})}{\omega\,\hat{x}\left(1 + \hat{x}^{\frac{1-\omega}{\omega}}\right)^2} \quad \text{(F.19)}$$

is zero if and only if $\hat{x} = 1$; negative for all $\hat{x} < 1$ and positive for all $\hat{x} > 1$. □

Lemma 1 implies that $d\bar{x}_1/d\omega < 0$ if $\frac{\partial B_{x_1}}{\partial \omega} < 0$. To determine the sign of this last expression, we define

$$\Gamma \equiv \frac{1}{1-\omega} \left(\frac{\hat{x}^{\frac{1-\omega}{\omega}}}{1 + \hat{x}^{\frac{1-\omega}{\omega}}} \ln(\hat{x}) + \omega \ln\left(\frac{1 + \hat{x}^{\frac{1-\omega}{\omega}}}{2}\right)^{\frac{\omega}{\omega-1}} \right). \quad \text{(F.20)}$$

Note that $\frac{\partial B_{x_1}}{\partial \omega} \lesseqqgtr 0$ if and only if $\Gamma \lesseqqgtr 0$.

We have $\Gamma = 0$ for $\hat{x} = 1$, and furthermore

$$\frac{d\Gamma}{d\hat{x}} = \frac{d\Omega}{d\hat{x}} + \frac{\hat{x}^{\frac{1-\omega}{\omega}}}{\hat{x}\left(1 + \hat{x}^{\frac{1-\omega}{\omega}}\right)} = \frac{\hat{x}^{\frac{1-\omega}{\omega}}}{\omega\,\hat{x}\left(1 + \hat{x}^{\frac{1-\omega}{\omega}}\right)^2} \left(\ln(\hat{x}) + \omega\left(1 + \hat{x}^{\frac{1-\omega}{\omega}}\right)\right) \quad \text{(F.21)}$$

Thus, for $\hat{x} \geq 1$, $d\Gamma/d\hat{x} > 0$. Thus, $\Gamma > 0$ for all $\hat{x} > 1$. This shows that for the larger stock, the effect of ω on $\bar{x}$ is ambiguous.

If x_1 is the smaller stock, i.e. $\hat{x} < 1$, the situation is more complicated. For the following lemma, we use $\hat{\omega}$ to denote the solution of $\hat{\omega} + \ln(\hat{\omega} - 1) = 0$, which is $\hat{\omega} \approx 1.28$.

Lemma 2

(a) If $\omega \geq \hat{\omega}$, $\Gamma < 0$ for all $0 < \hat{x} < 1$.

(b) If $\omega < \hat{\omega}$, it exists an $0 \leq \underline{\hat{x}} < 1$ such that $\Gamma < 0$ for all $\hat{x} \in (\underline{\hat{x}}, 1)$.

Proof. In the following we show that for $\omega \geq \hat{\omega}$, $d\Gamma/d\hat{x} > 0$ for all $\hat{x} > 0$, and that for $\omega < \hat{\omega}$, Γ has a unique minimum where it assumes some negative value.

To this end, we consider the expression $\Sigma \equiv \ln(\hat{x}) + \omega \left(1 + \hat{x}^{\frac{1-\omega}{\omega}}\right)$, which determines the sign of $d\Gamma/d\hat{x}$, i.e. $d\Gamma/d\hat{x} \gtreqless 0$ if and only if $\Sigma \gtreqless 0$. We show that the equation $\Sigma = 0$ has no solution (i.e. Γ is monotonic) if $\omega > \hat{\omega}$, and (at least) one solution if $\omega \leq \hat{\omega}$.

Note that for $\hat{x} = 1$, $\Sigma = 2\,\omega > 0$, and $\lim_{\hat{x}\to 0} \Sigma = +\infty$ for $\omega > 1$, as

$$\lim_{\hat{x}\to 0} \frac{\ln(\hat{x})}{1 + \hat{x}^{\frac{1-\omega}{\omega}}} = \lim_{\hat{x}\to 0} \frac{\frac{1}{\hat{x}}}{\frac{1-\omega}{\omega}\,\hat{x}^{\frac{1-\omega}{\omega}-1}} = \lim_{\hat{x}\to 0} \frac{\omega}{1-\omega}\,\hat{x}^{\frac{\omega-1}{\omega}} = 0. \tag{F.22}$$

Furthermore,

$$\frac{d\Sigma}{d\hat{x}} = \frac{1}{\hat{x}}\left(1 - (\omega - 1)\,\hat{x}^{\frac{1-\omega}{\omega}}\right) \tag{F.23}$$

Case $\omega \geq \hat{\omega}$: Σ has a minimum $\Sigma^* = \frac{\omega}{\omega-1}\,(\omega + \ln(\omega - 1))$ at $\hat{x}^\star = (\omega - 1)^{\frac{\omega}{\omega-1}}$. This minimum is non-negative if $\omega \geq \hat{\omega}$ and thus $\Sigma \geq 0$ for all $\hat{x} < 1$ in this case. This implies that Γ monotonically increases with $\hat{x}$ if $\omega \geq \hat{\omega}$. Since $\Gamma = 0$ for $\hat{x} = 1$, this implies that $\Gamma < 0$ for all $0 < \hat{x} < 1$. This concludes the proof of part (a) of the lemma.

Case $\omega < \hat{\omega}$: The minimum of Σ is negative, $\Sigma^\star < 0$. As $\Sigma = 2\omega > 0$ for $\hat{x} = 1$, this implies that there exist a value $\underline{\underline{\hat{x}}} > \hat{x}^\star$ where $\Sigma = 0$. At $\underline{\underline{\hat{x}}}$, Γ assumes a minimum, i.e. for values of $\hat{x} < \underline{\underline{\hat{x}}}$, Γ decreases with $\hat{x}$. Depending on the value of $\omega < \hat{\omega}$, Γ may or may not intersect with zero for some value $\hat{x} < \underline{\underline{\hat{x}}}$. Let $\underline{x}$ be the maximum of that value of $\hat{x}$ where $\Gamma = 0$ and zero. This concludes the proof of part (b) of the lemma. □

Appendix G. Parameter values for the age-structured bio-economic model.

Table G.1. Age-dependent parameters for the age-structured bio-economic model.

Age	Maturity γ_{is}			Weight w_{is} [g]			Catchability q_{is}			Mortality M_2			δ_{is} $[10^{-4}]$		Price p_{Cs}
	C	H	S	C	H	S	C	H	S	C	H	S	H	S	C
1	0.00	0.0	0.17	80	11	52	0.00	0.28	0.27	0.0	0.170	0.132	3.32	8.74	0.00
2	0.13	0.7	0.93	179	20	84	0.11	0.44	0.49	0.2	0.173	0.137	2.31	7.08	0.35
3	0.36	0.9	1.0	511	25	96	0.42	0.66	0.79	0.2	0.178	0.132	0.45	6.74	0.35
4	0.83	1.0	1.0	838	31	105	0.81	0.82	0.85	0.2	0.188	0.132	0.45	6.74	0.35
5	0.94	1.0	1.0	1204	37	111	1.00	0.97	1.00	0.2	0.188	0.132	0.45	6.74	0.48
6	0.96	1.0	1.0	1796	43	113	1.00	0.96	1.00	0.2	0.188	0.132	0.45	6.74	0.48
7	0.96	1.0	1.0	2596	48	111	1.00	1.00	1.00	0.2	0.188	0.132	0.45	6.74	0.64
8	0.98	1.0	1.0	4068	53	113	1.00	1.00	1.00	0.2	0.188	0.132	0.45	6.74	0.73

C: Cod; H: Herring; S: Sprat; δ_{is}: Predation coefficient.

Table G.2. Age-independent parameters for the age-structured bio-economic model.

	Recruitment function	ϕ_{i1}	ϕ_{i2}	Prices and harvesting costs	
Cod	Ricker	1.7	549	c_C	55.2 million euros
Herring	Ricker	30.33	2156	$p_H - c_H$	0.100 euros/kg
Sprat	Beverton-Holt	104.2	503.2	$p_S - c_S$	0.039 euros/kg

ϕ_{i1}, ϕ_{i2}: Parameters of the recruitment function of species $i = C, S, H$; c_C: Unit effort costs for the cod fishery; $p_H - c_H$, $p_S - c_S$: Net benefit per kg for the herring and sprat fisheries.

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Eidesstattliche Erklärung

Ich erkläre hiermit an Eides statt, dass ich meine Doktorarbeit „Essays on the Economics of Ecosystems and Biodiversity" selbstständig und ohne fremde Hilfe angefertigt habe und dass ich alle von anderen Autoren wörtlich übernommenen Stellen, wie auch die sich an die Gedanken anderer Autoren eng anlehnenden Ausführungen meiner Arbeit, besonders gekennzeichnet und die Quellen nach den mir angegebenen Richtlinien zitiert habe.

Christine Bertram

Kiel, November 2014

Curriculum Vitae

Professional experience

	Kiel Institute for the World Economy
Since 01/2009	Researcher in the research area „The Environment and Natural Resources“
11/2011 - 09/2014	Assistant to the head of the research area and management of (third party-funded) projects
	NORD/LB Norddeutsche Landesbank Girozentrale Hannover
08/2003 - 12/2004	Credit risk analyst for foreign financial institutions
07/2002 - 07/2003	Internal training in various credit departments
08/2000 - 06/2002	Apprenticeship as a bank business management assistant

Academic education and schooling

	University of Kiel
01/2009 - 02/2015	Participant in the PhD program „Quantitative Economics“ of the Faculty of Business, Economics, and Social Sciences
04/2005 - 12/2008	Studies of International Economics (Diplom)
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	Leibniz-Akademie Hannover
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