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Review

How to value biodiversity in environmental management?

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ABSTRACT

Biodiversity is globally recognised as a cornerstone of healthy ecosystems, and biodiversity conservation is increasingly becoming one of the important aims of environmental management. Evaluating the trade-offs of alternative management strategies requires quantitative estimates of the costs and benefits of their outcomes, including the value of biodiversity lost or preserved. This paper takes a decision-analytic standpoint, and reviews and discusses the alternative aspects of biodiversity valuation by dividing them into three categories: socio-cultural, economic, and ecological indicator approaches. We discuss the interplay between these three perspectives and suggest integrating them into an ecosystem-based management (EBM) framework, which permits us to acknowledge ecological systems as a rich mixture of interactive elements along with their social and economic aspects. In this holistic framework, socio-cultural preferences can serve as a tool to identify the ecosystem services most relevant to society, whereas monetary valuation offers more globally comparative and understandable values. Biodiversity indicators provide clear quantitative measures and information about the role of biodiversity in the functioning and health of ecosystems. In the multi-objective EBM approach proposed in the paper, biodiversity indicators serve to define threshold values (i.e., the minimum level required to maintain a healthy environment). An appropriate set of decision-making criteria and the best method for conducting the decision analysis depend on the context and the management problem in question. Therefore, we propose a sequence of steps to follow when quantitatively evaluating environmental management against biodiversity.

Contents

1. Introduction

Biodiversity is increasingly recognised as one of the cornerstones of healthy ecosystems (Kremen, 2005; Worm et al., 2006; Duffy et al., 2007; Hector and Bagchi, 2007; Pinto et al., 2014). The loss of biodiversity due to human action has the potential to reduce multitrophic-level interactions (Costanza et al., 1997; Schneiders et al., 2012) and cause trophic cascade repercussions (Lindberg et al., 1998; Österblom et al., 2007; Tylianakis et al., 2008). Legislatures and international treaties increasingly reflect this need to protect biodiversity, with the convention of biological diversity (CBD; UNEP, 1992) as the first treaty in international law to

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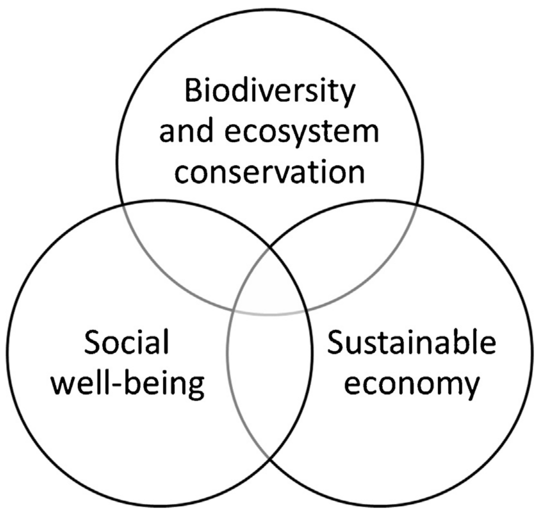

Fig. 1. The concept of ecosystem-based management covers the ecological, economic, and social aspects of environmental issues, aiming for sustainable development by acknowledging their interplay.

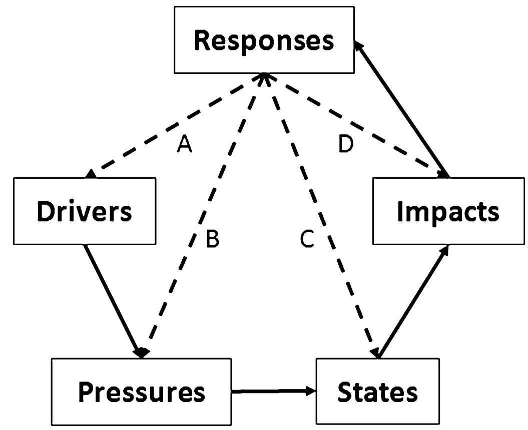

Fig. 2. The DPSIR problem-structuring framework for environmental management analysis. The various ways to manage the system appear as links A–D with descriptions in the text. The diagram is modified from the doctoral thesis of Lehikoinen (2014).

emphasise the vital importance of biodiversity conservation. More recently, the European Union (EU) has also begun to emphasise the importance of biodiversity, as is evident in the EU Biodiversity Strategy, an important policy driver; biodiversity is also one of the descriptors of Good Environmental Status in the Marine Strategy Framework Directive (MSFD; European Commission, 2008).

The main idea of environmental management is to safeguard and enhance the environmental state as well as to sustain economic and social benefits from the ecosystems (Elliott, 2011, 2013). Ecosystem-based management (EBM) (Fig. 1), required by both the CBD and MSFD, is shifting the focus towards more comprehensive decision-making processes by recognising ecological systems as a rich mixture of interacting elements and by acknowledging their social and economic features (e.g., Christensen et al., 1996; Ruckelshaus et al., 2008; Gregory et al., 2013). Because preventing the loss of biodiversity is increasingly becoming one of the important aims of environmental management, biodiversity must be defined in an operational way in order to facilitate setting management targets and evaluating management's performance. As stated in Section 2, biodiversity is inherently a multi-dimensional subject, spanning genes and species, functional forms, adaptations, habitats and ecosystems, as well as the variability within and between them. All these dimensions of biodiversity are tightly interconnected, affecting the state, stability, and productivity of the ecosystem as well as ecosystem services (Schneiders et al., 2012), thereby making biodiversity not only an ecological, but also a social and economic issue. This article therefore analyses the value of biodiversity from these three perspectives.

Some see ecosystem services as a means to quantify biodiversity in economic terms, usually defined as the benefits people can extract from ecosystems (Lamarque et al., 2011; Mace et al., 2012). The Millennium Ecosystem Assessment (MA) classifies benefits into four groups: provisioning, regulating, cultural, and support services (MA, 2005). Biodiversity may play three different roles in ecosystem services: as a regulator of ecosystem processes, as a final ecosystem service or as a good (Mace et al., 2012). However, because a description of biodiversity is complicated, accounting for the role of biodiversity or for the impacts of its decline on ecosystem services in general is not straightforward (TEEB, 2010a).

Environmental management problems are typically complex and multidisciplinary, involving various unavoidable trade-offs and uncertainties (Uusitalo et al., 2015) in informed decision-making. Decision analysis can help to structure the problem, to integrate knowledge and any prevailing uncertainty, and to visualise the results (Cooper, 2012; Lehikoinen et al., 2014; Rahikainen et al., 2014). The ultimate goal of decision analysis is to successfully select the management alternative that minimises risks and costs while maximising benefits and public acceptance (Keeney, 1982; Burgman, 2005; Kiker et al., 2005). However, using decision analysis requires that management targets, including biodiversity, have a quantitative value as to make them comparable.

To illustrate the aim of this paper, we use the Driving forces–Pressures–States–Impacts–Responses (DPSIR) framework for structuring problems (Fig. 2), a framework commonly used in the field of environmental management analysis (e.g., Borja et al., 2006; Maxim et al., 2009; Atkins et al., 2011; Gregory et al., 2013). This framework strives to systematically capture and represent the causes and consequences of environmental change as well as human responses to it. Response links A–D in Fig. 2 describe the different ways to manage the system. Links A and B generally relate to managing the principal and secondary causes (Drivers and Pressures) of environmental change, whereas link C represents the actions that strive to control or mitigate the consequences for the ecosystem (State). An example of drivers might include divergent economic or political trends affecting the volume of oil transportations within a certain sea area (see Lehikoinen, 2014). One pressure factor fuelling these drivers that causes or has the potential to cause harmful changes in the state of the ecosystem is a possible oil accident. The likely impact of such an accident on biodiversity would in this case be represented by the DPSIR-element State. After all, the best management alternative depends on the objectives that the society chooses (Impact). In the example provided, this could mean how the people actually value biodiversity. Modifying this decision-making criterion (link D) could therefore change the ranking order of the alternatives (Lehikoinen, 2014).

This review aims to discuss the use of biodiversity as a criterion against which to evaluate the impacts of human activities on the ecosystem and to review the alternative methods applicable for decision-analytical purposes. First, we provide an overview of biodiversity-related terminology and then focus on different approaches that purport to quantify the value of biodiversity. The aim is to provide a comprehensive analysis of the different evaluation techniques for measuring the value of biodiversity in terms of its ecological, economic, and social aspects. Further, we analyse these techniques to propose a suitable protocol for identifying the best decisions for alternative environmental management.

2. Biodiversity terminology

The term 'biological diversity' has been widely used since the 1980s (e.g., Lovejoy, 1980; Norse et al., 1986), whereas the use of the term 'biodiversity' began increasing towards the end of that decade (Harper and Hawksworth, 1995). These two terms, 'biological diversity' and 'biodiversity', are frequently used interchangeably (Harper and Hawksworth, 1995; Magurran, 2004). The division of biodiversity into three spheres – genetic diversity (within-species diversity), species diversity (number of species), and ecosystem diversity (diversity of communities) – has seen wide use since its launch during the Convention of Biological Diversity at 'The Earth Summit' in 1992.

In the Convention, the word 'biodiversity' meant "the variability among living organisms from all sources including, inter alia, terrestrial, marine, and other aquatic ecosystems as well as the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UNEP, 1992). The Convention states that the bedrock of biodiversity is genetic variability (intra-specific diversity), which refers to the genetic variation within a population and among populations of a species (Féral, 2002). Genetic variation is vital to ensuring that populations evolve in response to environmental changes (Reed and Frankham, 2003; Laikre et al., 2008). Species variation is the level of biodiversity that takes into account the number of species (species richness) and their proportional abundances (heterogeneity diversity) (Gray, 2000). This type of biodiversity offers valuable information about the structure of groups of organisms in the ecosystem. Ecosystem diversity encompasses the variety of habitats, various biotic communities and ecological processes in the biosphere, and refers to the variety of ecosystems in a given location (Pearce and Moran, 1994). Ecosystem diversity also encompasses the patchiness of a system, which shows the spatial distribution of communities, as well as the resilience, productivity, and stability of the system (Folke et al., 1996). In addition to the divisions mentioned above, the most frequently proposed division occurs at the level of molecular biodiversity, which represents the molecular richness of life (Campbell, 2003). The preservation of molecular diversity is vital, since evolution cannot occur without it.

Another important aspect of biodiversity is functional diversity, which represents the richness of functionally different types of organisms (e.g., with different feeding niches, habitats, or positions in the food webs) (Pearce and Moran, 1994). Functionally diverse communities are resilient against stress or shock and are less likely to change their behaviour (Folke et al., 1996; Nunes and van den Bergh, 2001). In addition, Tilman et al. (1997) discovered that species differ in their ability to modify ecosystem processes, but some species with certain functional traits have greater influence than others do.

One can also study biodiversity in the different spatial levels of alpha, beta, and gamma, corresponding respectively to within-habitat diversity, differentiation among habitats and total species diversity in a landscape (Whittaker, 1960; Magurran, 2004). In environmental management, the spatial aspect is of utmost importance, since spatial planning or land-use management can conserve biodiversity (Forman and Collinge, 1997; Theobald et al., 2000; Geneletti, 2008). However, this requires sufficient spatial data on biodiversity (i.e., data on species and habitats).

3. Value of biodiversity

Researchers across the globe have extensively studied recent unprecedented rates of biodiversity loss, which are to the direct result of increased human activities (e.g., climate change, pollution, deforestation, overexploitation of natural resources, habitat loss and the introduction of exotic species) (e.g., Pandolfi et al., 2003; Lotze et al., 2006; Butchart et al., 2010; Butt et al., 2013). When striving to minimise the negative impact of human activities on the environment, decision makers should be able to compare quantitatively the anticipated results of implementing alternative management actions. It is therefore necessary to measure the level of harm caused to biodiversity and to assign a value to the current state of biodiversity as well as the altered state.

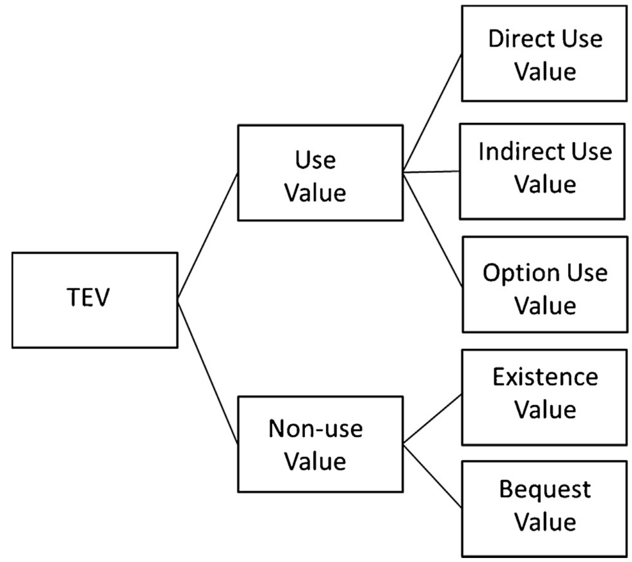

Fig. 3. The concept of total economic value (TEV); explanations of the terms appear in the text.

Our review of the published literature has suggested itemising three broad perspectives on valuing biodiversity, perspectives that cover the economic, socio-cultural, and ecological benefits of biodiversity as distinguished in the Millennium Ecosystem Assessment (MA, 2005) and the TEEB (2010a). The first approach is to value biodiversity in terms of the services provided for society, whereas the second approach is to assess socio-cultural values; the last approach adopts a biological viewpoint. However, integrative approaches that take into account all three perspectives of the sustainability are lacking (Nieto-Romero et al., 2014). Researchers have long discussed this division between standpoints both within scientific society and publicly. The central issue has been which perspective should determine how we value biodiversity; in other words, should we value all elements of biodiversity (e.g., the existence of a species, the resilience of communities, etc.) in monetary terms or should they possess an intrinsic value regardless of anthropogenic benefit (Nunes and van den Bergh, 2001; Bräuer, 2003; Nijkamp et al., 2008; Justus et al., 2009; Salles, 2011).

In the following sections, we review the focal literature with the above-mentioned three perspectives on valuing biodiversity in mind. The first section focuses on an economic perspective, followed by biodiversity's socio-cultural and ecological aspects.

3.1. Methods of economic valuation

This utilitarian approach aims to quantify the impact of a change in biodiversity on our economy or human welfare. Total economic value (TEV) is the main framework for valuing biodiversity in monetary terms (Fig. 3, Pearce and Moran, 1994; Adger et al., 1995; Fromm, 2000; Turpie et al., 2003; Nijkamp et al., 2008; Oxford Economics, 2009; Rolfe and Windle, 2010). The total value of environmental assets includes both use and non-use values (Fig. 3, Pearce and Moran, 1994; Pagiola et al., 2004). The use value is further divided into direct (e.g., food, timber, and medicine), indirect

Table 1
Sample studies that use monetary valuation in environmental management.

Valuation technique	Studies
Contingent valuation	Loomis and Larson (1994), Adger et al. (1995), Loomis and White (1996), Costanza et al. (1997), Stevens et al. (1997), Loomis et al. (2000), Appelblad (2001), Navrud (2001), Cardoso de Mendonça et al. (2003), Turpie et al. (2003), Toivonen et al. (2004), Paulrud (2004), Parkkila (2005), Christie et al. (2006), Beaumont et al. (2008), Oxford Economics (2009), Barbier et al. (2011), Ressurreição et al. (2012)
Market price	Pimentel et al. (1997), Turpie et al. (2003), Oxford Economics (2009), McClanahan (2010)
Travel cost	Brown and Mendelsohn (1984), Turpie et al. (2003), Oxford Economics (2009), Barbier et al. (2011)
Production function	Turpie et al. (2003)
Choice modelling	Paulrud (2004), Christie et al. (2006), Rolfe and Windle (2010), Christie and Rayment (2012), Jobstvogt et al. (2014)
Benefit transfer	Beaumont et al. (2008)
Replacement cost	Beaumont et al. (2008), Oxford Economics (2009), Gren (2013)

(e.g., natural water filtration, storm protection, and carbon sequestration) and optional values (the option to use ecosystem goods and services in future), whereas the non-use value is divided into a bequest value (referring to benefits from ensuring that biodiversity or ecosystem services will be preserved for future generations) and an existence or ‘passive’ use value (individuals do not actually use these resources, but would feel their loss if they disappeared) (Pearce and Moran, 1994; Pagiola et al., 2004).

The environment can be valued monetarily with the following three distinct groups of techniques: revealed preference, stated preference, and direct market valuation techniques. Direct market valuation techniques are divided into the market price (the monetary value of goods and services that can be bought and sold in commercial markets) and their production function (an estimate of the contribution of a certain ecosystem service to the production of other marketable good) (Bertram and Rehdanz, 2013).

Without direct market prices for environmental goods such as biodiversity, their value can be inferred using consumer preferences (Nijkamp et al., 2008; Remoundou et al., 2009). Revealed preference techniques (i.e., indirect valuation approaches) are based on observed consumer behaviour and include hedonic pricing, travel cost and replacement cost methods for analysing individuals’ actual choices (Haab and McConnell, 2002; Pagiola et al., 2004). The hedonic pricing method serves to calculate the value of environmental goods such as landscape, air quality, and noise (Turner et al., 2010). This method evaluates the implicit price that individuals are willing to pay for the relevant environmental characteristics based on house prices, the time, and money spent on recreational trips or other expenses (Turner et al., 2010). The replacement cost method, on the other hand, quantifies the cost of replacing or restoring an ecosystem service (Pearce and Moran, 1994; Balmford et al., 2002).

While revealed preference techniques are useful only for use values, stated preference techniques can serve to assess the TEV (i.e., use and non-use values) (Wardman, 1988; Nijkamp et al., 2008). Stated preference techniques (e.g., Haab and McConnell, 2002; Pagiola et al., 2004; Hajkowicz, 2007) derive from respondents’ answers to questions about how much they would be willing to pay to maintain/improve the quality of the environment (the contingent valuation method; Turner et al., 2010) or after presenting them with choices between goods and expenses (the choice modelling approach; Hanley et al., 2001; Turner et al., 2010).

Table 1 summarises examples of published studies that employ monetary valuations of biodiversity or ecosystem services. The list is based on the search results in the Scopus database (October 2014). Keyword searches for ‘monetary value’, ‘biodiversity’, and ‘ecosystem service’ yielded 342 studies between 1971 and 2014. Without citing all of the search results, the list in Table 1 shows variability among the results in terms of the valuation method employed. The selected examples represent various geographical and subject matter areas.

Of all the monetary valuation methods presented in Table 1, the contingent valuation approach is the one most commonly used to measure the extent of gain or loss in biodiversity (Mitchell and Carson, 1989; Nijkamp et al., 2008). These works focus mostly on individual species and habitats, but do not value the diversity itself (Pearce, 2001; Cardoso de Mendonça et al., 2003; Christie et al., 2006; Beaumont et al., 2008). Even though accurate estimates of people’s willingness to pay (WTP) (Hanemann, 1994) for a number of non-marketed ecosystem services are available, we still know little about the value of biodiversity per se (i.e., the value associated with changes in the variation of genes, species, and functional traits) (Cardinale et al., 2012). It is noteworthy that, in order to know what to quantify, we also need to know more about the uncertainty between measures of biodiversity loss and their impact on certain ecosystem services (Balvanera et al., 2014). The review by Nunes and van den Bergh (2001) provides an overview of how much households are willing to pay to preserve either single or multiple species in terrestrial or marine habitats. Loomis and White (1996) and Martín-López et al. (2007) conducted similar surveys of the species-contingent valuation study by studying people’s varied attitudes towards particular species. These studies found that people were willing to pay more to preserve more familiar or interesting species than less attractive ones (Loomis and White, 1996; Martín-López et al., 2007). Ressurreição et al. (2012) conducted a study that estimates the public’s WTP to preserve five specific marine taxa (mammals, birds, fish, invertebrates, and algae) as a representation of marine biodiversity. The study used a multi-site perspective across three different locales, namely Portugal (the Azores), the United Kingdom (the Isles of Scilly), and Poland (the Gulf of Gdansk), which provided a comprehensive view of cultural differences across public preferences (Ressurreição et al., 2012). One disadvantage related to these contingent valuation surveys is that they pose a hypothetical question of people’s WTP, which leads to the broadly studied problem of ‘hypothetical bias’ (e.g., Venkatachalam, 2004; Murphy et al., 2005; Loomis, 2011; Hausman, 2012). Contingent valuation experiments have found that answers to hypothetical questions about respondents’ WTP exceed their actual WTP (i.e., what people say differs from what they are actually willing to do).

All of the above-mentioned methods face multiple challenges, namely the reliability of their results. One example that highlights the unreliability of the results relates to the hedonic pricing method, which assumes that people can buy the exact property and associated characteristics they desire (Opaluch et al., 1999; OECD, 2002). However, outside influences (e.g., taxes, interest rates) that can skew the valuation results may influence the housing market (Turner et al., 2010). Another problematic issue in indirect valuation arises with the travel cost method, which requires significant resources to produce a reliable analysis (Turner et al., 2010). The travel cost method requires large sample sizes, making it very labour and finance intensive. Additionally, assessing the value of time poses difficulties because the method always assumes that a trip is for a single attraction and cannot separate the travel cost for multiple sites (Dwyer, 2006; Tisdell, 2010; Graves, 2013). The direct valuation approach uses questionnaires, which researchers must carefully design and pre-test to avoid biased results. As with the travel cost method, the sample sizes should be large enough to produce reliable results (Turner et al., 2010). Despite these limitations, the methods are widely used to assess the value of particular ecosystem benefits (Pagiola et al., 2004).

The existing literature points out the importance of strong and reliable biological information when using the TEV approach (Pearce and Moran, 1994; Costanza et al., 1997; Bulte and Van Kooten, 2000; Brito, 2005). Economists, in contrast, have stated that, with the TEV approach, they cannot capture the whole value, but only the monetary value (Pearce and Moran, 1994; Nijkamp et al., 2008). Another critical remark is that monetary valuation takes into account only the direct human benefits of ecosystem services and not an ecosystem's resilience (Admiraal et al., 2013). Psychological motivations, driven by impure altruistic forces, are also important factors influencing 'willingness to pay' questionnaires, a point which researchers should bear in mind when analysing the results and specifically when using them to draft policy (Nunes, 2002; Nunes and Onofri, 2004; Nunes and Schokkaert, 2003). These impure altruistic forces are related to the particular respondents, or warm glowers, who find satisfaction in contributing to conservation efforts (Nunes, 2002; Nunes and Onofri, 2004). Desvousges et al. (1993) criticise the contingent valuation technique because participants always lack information about issues in the questionnaires, thus skewing the resultant value. On the other hand, a value measured in monetary terms can make the values for biodiversity more visible to a larger audience (TEEB, 2010b) and, more importantly, promote comparability between biodiversity conservation and the economic world, thereby facilitating the integration of environmental management into political decisions (Bräuer, 2003). Consequently, there is growing discussion about the precise definition and classifications of ecosystem services, which provide a solid foundation for future work (Boyd and Banzhaf, 2007; Fisher and Turner, 2008; TEEB, 2010a; Böhnke-Henrichs et al., 2013).

3.2. Socio-cultural perspective of biodiversity valuation

When assessing the value of biodiversity, it is seldom necessary or even possible to assign it a monetary value because different human societies and communities place different values on species, ecosystems, and biodiversity in general. For example, the cultural or spiritual values of local people in certain regions may be sufficient to ensure sustainable use and protection (TEEB, 2010b). In other words, assessing the socio-cultural value of biodiversity, which in this case provides society with benefits such as mental well-being and ethical, spiritual, and cultural values, is necessary (Posey, 1999; Christie et al., 2012).

In their review of socio-cultural valuation techniques, Christie et al. (2012) provide a comprehensive list of methods, including quantitative and qualitative techniques (i.e., surveys, interviews), participatory and deliberative tools, and methods for expressing preferences in non-monetary yet quantifiable terms. The number of studies utilising geographic information system (GIS) applications to map the spatial distribution of stakeholders' social or recreational values has also grown (Rees et al., 2010; Sherrouse et al., 2011).

In some cases, socio-cultural perspectives can serve as the main factors in determining the success or failure of environmental management (Mascia et al., 2003). Even so, environmental studies have thus far focused mostly on approaches to ecological and monetary valuation (Vihervaara et al., 2010). However, it is important to note that, in some cases, biodiversity may have a heavier cultural and spiritual value than other standpoints.

3.3. Ecological approach to the value of biodiversity

3.3.1. Classical biodiversity indices

One central weakness of the economic valuation approaches is that the prices of some benefits or services provided by a diverse ecosystem can be difficult to evaluate. Even the scientific understanding of the role of biodiversity in the functioning and health of ecosystems, and in provisioning ecosystem services, remains incomplete. In public discussion, biodiversity is commonly represented by charismatic, often endangered, macrofauna, such as giant pandas, white-tailed eagles, or whales (Mikkelsen and Cracraft, 2001). However, many sensitive or threatened species remain invisible or unknown to the majority of people and are thus difficult to value. Further, biodiversity protection often emerges from the promise of unrevealed but potential ecosystem services, such as the possibility of finding new medicines, which may seem too uncertain an investment. Therefore, there is a need to complement the monetary and socio-cultural valuation approaches of biodiversity with one based on the prevailing natural scientific knowledge and understanding about how ecosystems function. A natural approach would be to identify the minimum level of biodiversity to maintain. This will require researchers to measure biodiversity needs, which usually takes place through indices that reduce multifaceted issues to a few key variables that describe a certain aspect of the phenomenon (Heip et al., 1998).

The classical biodiversity indices (or functions that take into account the relative frequencies of species present at the site), which describe the richness and distribution of species (Heip et al., 1998) weighted in different ways, include the Shannon–Weiner diversity index (Hill, 1973; Heip et al., 1998), Simpson's index (Simpson, 1949; Hill, 1973; Heip et al., 1998), the Berger–Parker index (Hill, 1973; Magurran, 2004), and Pielou's evenness index (Pielou, 1969; Van Dyke, 2008). The classical indices mostly describe alpha diversity (i.e., the diversity within a site or sample). However, Czekanowski's similarity index studies the similarities between samples representing beta diversity (Czekanowski, 1909; Schubert, 2013). Beta diversity is the variation in species composition along an environmental gradient and thus describes he rate of change, or turnover, in species composition (Whittaker, 1960, 1972). Whittaker (1960, 1972) first proposed computing the ratio of two diversity indices: beta diversity = γ/α, where γ (gamma) diversity is the total species diversity of a landscape, and α (alpha) diversity is the mean species diversity per habitat. Gamma diversity is usually calculated using alpha diversity samples from several communities or lists of species (Whittaker, 1972; Legendre et al., 2005).

Later, Petchey and Gaston (2002) proposed the functional diversity index, which measures the total branch length of the functional dendrogram built on the regional pool of species. Another, more recent study examined the functional diversity of the marine diatom *Skeletonema marinoi* by observing the potential effects of grazing pressure (reflected by different grazer levels) (Sjöqvist et al., 2013). The study confirmed that genetically distinct individuals of *S. marinoi* are functionally more diverse.

3.3.2. Towards the eco-social approach

In addition to the classical biodiversity indices, numerous other specific measures of biodiversity have been developed and are used to measure the biodiversity of specific ecosystem components or habitats. For all of the above-mentioned classical indicators, threshold levels can be set which dictate the minimum level of biodiversity to maintain. These thresholds serve as the minimum level of biodiversity that society seeks to preserve. Therefore, defining such a threshold represents the first social aspects of the analysis. The Baltic Marine Environment Protection Commission (HELCOM) has recently published a core set of biodiversity indicators for evaluating the overall state of the Baltic Sea (HELCOM, 2013), and similar work is underway in the other seas as well. Böhnke-Henrichs et al. (2013) provide guidance for selecting appropriate indicators for all relevant marine-focused ecosystem services that reflect changes in the state of the ecosystem. Moreover, the work of Hattam et al. (2015) presents a practical approach to the use of indicators to quantify marine ecosystem services. These types of indicators have served to simplify aspects of the environment that lead to management decisions and policy guidelines (Gubbay, 2004).

Several proposals define ecosystem health in terms of functional and structural status, and involve the human perspective with different grades in forms of objective-formulation and weighting. Table 2 summarises some of these approaches. The Marine Trophic Index (MTI) demonstrates the decline in the mean trophic level of fishery landings (Pauly et al., 1998; Pauly and Watson, 2005). Biological valuation maps (BVM) help to determine the total biological value, together with ecological information from subareas, by using valuation criteria that take into account rarity, consequences of fitness, aggregation, naturalness, and proportional importance in a given study area (Derous et al., 2007; Pascual et al., 2011). The marine BVM represents a baseline showing the holistic biological and ecological values from the genetic to the ecosystem level while integrating data on seabirds, macrobenthos, demersal fish, and epibenthos (Derous et al., 2007).

The Biodiversity Benefits Index (BBI, Oliver and Parkes, 2003), a modification of the 'habitat hectares' index of Parkes et al. (2003), aims to assess the current biodiversity value of a habitat based on biodiversity measures such as vegetation condition, conservation significance, and landscape context. The Biodiversity Intactness Index (BII) approach by Scholes and Biggs (2005) calculates the overall state of biodiversity in a given area. The BII requires baseline information (before value) on the species richness in a specific area after calculating the weighted impacts of anthropogenic activities (e.g., acute pollution events) (after value) on the population of a group of organisms, which are then compared in order to evaluate the harm caused. This technique is largely applied when studying large terrestrial areas. One specific disadvantage of the BII is that the impacts of pollution or climate change on biodiversity emerge slowly over long periods (Scholes and Biggs, 2005).

Aubry and Elliott (2006) proposed an integrative indicator that combines an appropriated set of indicators (including physico-chemical and biological elements) and uses expert judgement to weigh and rank those indicators based on their perceived relative importance in assessing the seabed disturbance in estuaries and coastal waters. Tett et al. (2013) have proposed a state space approach to track changes in an ecosystem state as well as to estimate system resilience by selecting state variables. The first requirement is to identify the state variables that represent the condition of the ecosystem (i.e., biodiversity and production of the study area). Another requirement is to use an extended series for detecting inter-annual variability, which reveals the resilience of the system (Tett et al., 2013).

The biodiversity indicator approach for valuation shares similar disadvantages with the monetary valuation approach, since it also requires large amounts of data. Because data abundance and quality typically vary both overtime and space, comparisons of different areas or scenarios or both are inevitably somewhat biased (e.g., Collen et al., 2008). Another issue to take into account is the need to clarify the transition of a system from a normal state to an impacted one. Determining the baseline state of the environment is problematic but important to assess if one is to define the change in biodiversity (Parr et al., 2003; Borja et al., 2012). Specifically, the difficulty lies in finding an adequate historical dataset or an un-impacted control area to detect the 'shifting baseline' phenomenon (Duarte et al., 2009; Carstensen et al., 2011).

When selecting specific indicators for use in a certain area, the analyst must decide which and how many taxonomic or functional groups as well as which habitats to include. The EU MSFD guidelines (EU, 2010) for evaluating the biodiversity descriptor provide a recent example of how this complex issue can be compressed into indicators. The MSFD requires a biodiversity assessment at the species, habitat, and ecosystem levels. On the species level, assessment should account for distribution, population size, and population condition, but include subspecies and populations separately if they are under threat. Population distribution is related to the availability and quality of habitats, which also need safeguarding. The condition of the population refers to age and sex structure, survival and reproduction, and the genetic structure of the population. Habitats, defined as both abiotic characteristics and the associated biological community, as well as habitat complexes and functional habitats (such as spawning or feeding areas) must be evaluated for their distribution, extent, and condition (with a particular focus on the condition of typical species and communities). The ecosystem levels then view the composition and relative proportions of the habitats and species.

4. Discussion

In this paper, we recognise three approaches to valuing biodiversity: the economic, the socio-cultural, and the ecological. They provide different and complementary perspectives, each with its own advantages and limitations. Overall, these ecological biodiversity indicators are useful, quantitative tools for assessing the state of biodiversity, as well as for communicating complex, environmental issues in order to integrate them more thoroughly into policy decisions (UNEP, 2003; TEEB, 2010a). We noticed that many developed biodiversity indicators already include some social aspects, which shows that attempts to totally separate human beings from the ecosystem are artificial – if not impossible.

When it comes to assessing and managing the anthropogenic use of the environment, we unavoidably head for a situation in which pure biological information alone is insufficient. This is why, from the decision-analytic viewpoint, the three above-mentioned aspects of valuing biodiversity cannot be fully separated from each other. Decisions cannot be evaluated or ranked without first defining the objectives (i.e., the decision-making criteria) (e.g., Keeney, 1982). Selecting the criteria, defining a sufficiently good state of the environment, as well as the acceptable risk for failing to achieve the goals are social choices that people made. Moreover, monetary resources nearly always limit management in some sense, it is therefore useful to try to describe the value of the objectives – in this case, biodiversity – in monetary terms also. In addition to the basis for communication, monetary resources allow us to carry out cost-efficiency analyses for alternative management strategies. Sometimes, because aspects of socio-cultural valuation can override other arguments (Mascia et al., 2003), acknowledging them is also of the utmost importance. So, as a basis for decision-making, which seeks the sustainable use of the environment, we suggest the multi-criteria valuing of biodiversity, covering all three aspects of the ecosystem-based management (EBM) framework.

Some researchers have proposed using biodiversity indicators as a basis for monetary valuation (Nunes and van den Bergh, 2001). The monetary value of biodiversity, or specifically, the value it provides for supporting the human population and its lifestyle, is increasingly under evaluation by assessing the value of ecosystem services (e.g., Costanza et al., 1997). These kinds of all-encompassing, monetary estimates for biodiversity could help larger audiences understand the importance of protecting biodiversity, even if the diversity index value is misunderstood (Polasky, 2008). The challenge with this approach lies in correctly and exhaustively identifying and measuring the ecosystem services provided (Vihervaara et al., 2010; Seppelt et al., 2011).

Our analysis agrees with that of Bräuer (2003), that monetary value can still serve as a useful link between environmental problems and political decision-making processes, although the future challenge is to identify common ground for comparing monetary

Table 2
Example studies that used biodiversity indicator valuation methods.

References	Subjects	Valuation techniques
Baillie et al. (1996)	To assess global changes in biodiversity by defining the conservation status of major species groups and their extinction risk	The IUCN red list of threatened species
Borja et al. (2000)	The index serves to observe the response of soft-bottom communities to natural and anthropogenic changes in water quality	The marine biotic index (BI)
Ribaudo et al. (2001)	The index sums up the soil erosion risk, water quality risk, and wildlife habitat quality to estimate ecological benefits in the area	The environmental benefits index (EBI)
Oliver and Parkes (2003)	The index serves to predict the change after land use activity	The biodiversity benefits index
Scholes and Biggs (2005)	The index calculates the impacts of a set of activities on a group of organisms by using relative changes in species richness	The biodiversity intactness index (BII)
Aubry and Elliott (2006)	The integrative indicator measures the state of and pressures on coastal and estuarine environments by integrating knowledge of physico-chemical and biological elements	The environmental integrative indicator
Loh et al. (2005), Collen et al. (2008)	The index based on abundance trends in populations of vertebrates from around the world	The living planet index
Derous et al. (2007), Pascual et al. (2011)	The area-specific weight is estimated by the following criteria: rarity, consequences of fitness, aggregation, naturalness, and proportional importance	Biological valuation maps
Ihaksi et al. (2011), Kokkonen et al. (2010), Jolma et al. (2014)	An index-based evaluation method links the weighting of threatened species (based on several criteria, including legislation and certain ecological features) in the decision-making process for combatting oil spills	The OILECO index
HELCOM (2013)	To assess anthropogenic pressures on the state of biodiversity in the Baltic Sea	The HELCOM core set of biodiversity indicators
Altartouri et al. (2013)	The index takes into account the conservation value, legislative status, oil-induced loss and recovery potential of species and habitats, as well as the efficiency of combatting methods	The OILRISK index

and intrinsic values. Social and cultural factors affect not only how people appreciate nature, but also how they value their money or how risk aversive they are (Pratt, 1964; Chow and Sarin, 2002; Burgman, 2005). Consequently, the socio-cultural perspective is an inseparable part of the ecosystem-based biodiversity valuing approach. Social preferences can serve as a tool to identify the most relevant ecosystem services for people (Martín-López et al., 2012; Martínez et al., 2013).

The management of ecosystem services should not always be equated with the management of biodiversity and vice versa. The most desirable approach would be to optimise the management so that it could achieve many goals simultaneously while recognising that biodiversity alone would provide some ecosystem services (Mace et al., 2012). Furthermore, there is a need to define the roles of biodiversity and ecosystem services in environmental management and conservation (Geijzendorffer and Roche, 2013). Does biodiversity indeed have an intrinsic value, irrespective of any usefulness or function, or is biodiversity valuable only to the extent that it can provide ecosystem services or support their provision? Alternatively, should biodiversity be considered separate from the ecosystem services, but equal in terms of environmental management? The answer to these questions dictates whether we indeed need valuation or indicators for biodiversity, or whether these serve only as proxies for indicators and the valuation of ecosystem services.

Although it seems that integrating the socio-cultural, monetary, and ecological biodiversity indicator approaches together could provide some useful insights, one should use careful consideration when combining them. Monetary valuation often yields its results on a continuous scale, while biodiversity indicators often yield results on a binary pass/fail scale. Consequently, the latter approach offers no preference for a management option when biodiversity values are barely or far below the threshold value. Care must therefore be taken when developing the decision analysis models, especially in the cases where finding management options that would lead to the achievement of good biodiversity status is unlikely. A probabilistic approach, revealing the probability that an indicator remains in a certain state, can offer one possible solution to the problem (Lehikoinen et al., 2014). This would provide us a biological margin of safety for the minimum level of biodiversity to be achieved – the width of which would depend on the risk-aversion of the decision-makers or of society. After achieving the defined biological minimum with an acceptable level of certainty, the benefits acquired per each extra unit could be expressed in monetary terms and the cost-effectiveness of the management options could be evaluated in light of that information. In future, appropriate platforms for this kind of decision-making tool, taking into account the uncertainties and allowing for the definition of optimisation rules in different phases of the process, merit investigation. Bayesian Networks (Jensen and Nielsen, 2007) could be one method with which to explore this idea.

This review reveals a lack of studies that use the quantitative values of biodiversity as a tool for predicting the impacts of alternative management decisions, with the praiseworthy exceptions of Nicholson et al. (2012), Ressurreição et al. (2011), and Ressurreição et al. (2012). Otherwise, the value of biodiversity has served to provide information about single species or the natural habitat of the species in question. Therefore, the scopes of most of these valuation studies take into consideration only a fraction of the existing biodiversity (e.g., the grey whale by Loomis and Larson, 1994; Atlantic salmon populations by Stevens et al., 1997; recreational sites and fisheries by Paulrud, 2004), which fails to reveal the truth about the total biodiversity. This may be insufficient to make reliable management decisions.

A single explicit indicator alone, providing the total value of biodiversity, clearly does not exist, but a selection of a balanced suite of indicators (see, e.g., the marine biodiversity indicators of the MSFD, discussed in Section 3.1) is necessary, the best selection depending on the context and aim of the environmental management case in question (Nunes and van den Bergh, 2001). We therefore recommend taking the following steps when quantitatively evaluating environmental management against biodiversity:

(1) Clarify the environmental management problem to be analysed. For example, “How to minimise the environmental impact of increasing oil shipping in the Gulf of Finland?” would be further defined as two separate questions: “How to minimise

the probability of oil spills in the Gulf of Finland and, in the event of a spill, the ecological effect the spill is likely to have?" (Helle et al., 2011; Lehikoinen et al., 2013; Jolma et al., 2014).

(2) Identify the alternative solutions/management actions to be compared. In the example above, identified management measures are also split into two categories: those that increase the safety of oil shipping, and those that optimising the oil recovery and prevent the pollution of the most important locations in the event of a spill. The first category includes technical and naval changes such as double-hulls, piloting obligations, winter navigation training for captains, and changes in fairways to avoid the most dangerous of fragile areas (Soomere et al., 2011). The second category focuses on one's readiness to respond to accidents in a timely and optimised manner, such as choosing the optimal distribution of the oil combatting vessels along the coast (Lehikoinen et al., 2013), prioritising the locations of oil booms to protect the most vulnerable species and areas (Helle et al., 2011), and choosing whether to use oil dispersants, among other strategies. One must define the selection of management measures to include in the assessment precisely and at all possible levels (e.g., double hull obligation implemented/unimplemented; booms placed according to plans A, B, or C; etc.).

(3) Expressing the potential gains and losses in terms of biodiversity. In the present example, the ecosystem components and areas to be taken into consideration could include the potential mortality of bird or seal populations, the amount of oiled shoreline and the affected flora and fauna, fish populations that may be affected by dispersed oil, and specific endangered species or populations that may be affected by the stranded oil (e.g., Ihaksi et al., 2011; Lecklin et al., 2011). Choosing valuation approaches to identify the best management decisions depends on the type of biodiversity to be analysed and the abundance and quality of available data. Are economic valuation data available or, if not, can they be easily acquired? Are enough data available to evaluate biodiversity indicator values reliably? Some biodiversity components can be valued economically based on their ecosystem service value or perceived existence or bequest value, whereas others may be unknown to society at large and therefore be better evaluated with biodiversity indicators. In addition, the existence of economic valuation results and biodiversity indicators relevant to the case should be ascertained and used when necessary. In the oil shipping example, the optimal suite of valuation methods might include the ecological indicator approach for endangered species and vulnerable habitats, the direct economic valuation of damage to the fishing industry, and the indirect economic valuation of the perceived value of charismatic species or popular recreation areas. In addition, the calculations should reflect the direct costs of implementing each of these management measures.

(4) Decide an appropriate method for quantitative analysis. The choice of the best model for evaluating management options depends on many factors, including
   (a) The time frame of the evaluation, as well as the required precision of the results; a precise result that is too late for the decision-making process has no value.
   (b) The abundance and quality of existing models. Can existing models serve as input or can parts of the decision support model? Do the models provided information about the variables we are interested in and in the relevant spatial and temporal scales?
   (c) The existing research/literature. Can the literature serve to find additional information to support and supplement the data?
   (d) The area(s) of analysis. This aspect should be taken into account in relation to the previous points (a–c). How much and what kind of data do we already have from the area and how many resources are available to conduct further sampling? What kinds of models or other results describing the area are available? Can some data, models or results from corresponding areas be exchanged, updated or extrapolated and thus serve in the analysis at hand (see e.g., Pulkkinen et al., 2011)?

A thorough analysis using the suggested framework requires considerable multi-disciplinary data or modelling results from both the ecological responses and the economic value of biodiversity as well as the costs of implementing the management measures. Because the decision support models must be able to evaluate the expected results of the various combinations of management measures, many of which have not yet been implemented and about which no data yet exists, the model must therefore be able to extrapolate such data. Here, rendering the extrapolated results useful will require careful analysis of the assumptions related to this extrapolation.

The proposed approach can, in principle, serve not only to value biodiversity, but to evaluate the full-scale of environmental management also. In practise, however, evaluating the full-scale of various environmental management measures and other activities affecting the environment and all its components could lead to a restrictively complex model. Evaluating large environmental management programmes that affect several ecosystems or ecosystem components, such as the ambitious MSFD Programme of Measures, will likely require piecemeal evaluation, first by identifying the main paths of effect of each management measure, and then by creating models for each cluster of measures and effects separately.

The aim of environmental management is to achieve and maintain a healthy and sustainable ecosystem. This paper proposes an environmental management framework that recognises the importance of biodiversity. Realising the aim of environmental management requires one to consider the comprehensive ecological status as well as the economic importance of a healthy ecosystem. The common yardstick must be drawn in order to establish more transparent and solid grounds for acceptable environmental management practices.

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