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Protected Area management: Fusion and confusion with the ecosystem services approach

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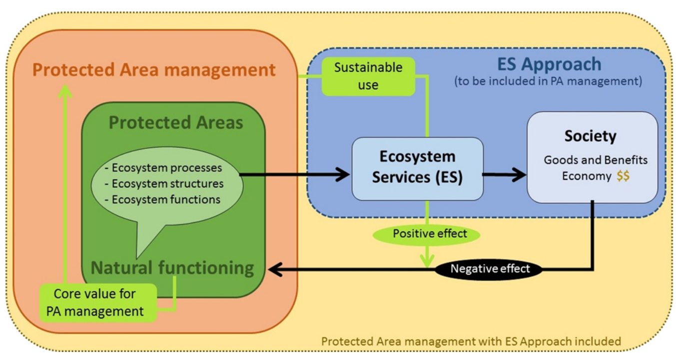
HIGHLIGHTS

- Protected Areas are important in conservation strategies.
- Protected Areas maintain (species) diversity, landscapes and Ecosystem Services.
- The Ecosystem Services approach is scarcely used in Protected Area management.
- Operationalising the Ecosystem Services approach in Protected Area management may prove difficult.
- The Ecosystem Services approach could be used in Protected Area management with some changes.

GRAPHICAL ABSTRACT



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ABSTRACT

For many years, Protected Areas (PA) have been an important tool for conserving nature. Recently, also societal aspects have been introduced into PA management via the introduction of the Ecosystem Services (ES) approach. This review discusses the historical background of PAs, PA management, and the ES approach. We then discuss the relevance and applicability of the ES approach for PA management, including the different definitions of ES, different classification methods, and the ways in which ES are measured. We conclude that there are still major challenges ahead in using the ES approach in PA management and so recommendations are given on the way in which the ES approach should be integrated into PA management.

1. Introduction

Protected Areas (PAs) are one of the most important tools in conservation science and management (Chape et al., 2005). They have long been regarded as important for maintaining species and habitat

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diversity, as well as protecting specific landscapes or sacred areas (Brooks et al., 2004; Rodrigues et al., 2004; Coad et al., 2008; Wild and McLeod, 2008; Butchart et al., 2010). Conservation strategies have traditionally taken the view that biodiversity should be protected because species have both a functional and an inherent value (Wilson, 1988; Kareiva, 2012). More recently, there has been a transformation towards considering Ecosystem Services (ES) and human well-being in the design and management of PAs (Doak et al., 2015). As such, there is a transition from focusing on the protection of (threatened) species towards the sustainable use and protection of landscapes and ecosystem complexes against various anthropogenic pressures. However, an ongoing theoretical debate (although with profound applied implications) raises doubts on the role of humans in natural systems and in particular questions whether humans are primarily a threat to biodiversity, or whether they can be integrated into a PA as managers of biodiversity conservation (Mantell et al., 1986). In that sense the use of ES in PA management has raised concerns that economic valuation of nature would lead to "selling out on nature" (McCauley, 2006) and commodification (Turnhout et al., 2013). Focussing only on the outcomes of the system that are important or of value to mankind, often by trying to classify them, would lead to a potential loss of the view on the whole (eco)system, and of the understanding of its importance in sustaining delivery of ES (Kremen, 2005). Therefore, there is an urgent need for a balanced and inclusive combination of the societal-focussed ES approach and the traditional view of conservation, protecting nature, and biodiversity, in order to become adopted in current management strategies.

It is contended here that the ES approach could become a central facet in PA management, when using a holistic assessment of the ecosystem. This means including ES, the ecosystem features (biodiversity, structure and functioning), and the natural and socio-economic pressures that act on them. An understanding of this holistic system is then inherent in communication with different stakeholders when designing new PAs (Reid et al., 2006; Cowling et al., 2008; Menzel and Teng, 2010; de Groot et al., 2010).

This review presents concepts and approaches used in, and for, PA and ES, and their use in environmental management. Firstly, we describe the history and evolution of PAs and their designation and management. Secondly, we review the evolution of the ES concept and the way in which ES are defined, classified, measured and assessed. The synthesis of both concepts will show the advantages and disadvantages of using an ES approach in PA management. The analysis aims to be relevant to terrestrial, freshwater and marine systems although certain aspects are more applicable to only one of these systems.

2. History of Protected Areas & Protected Area management

The most frequently used definition of a PA is that of the CBD (Convention on Biological Diversity): "A geographically defined area, which is designated or regulated and managed to achieve specific conservation objectives" (CBD, 1992). As such, in 2017, PAs covered in total about 15% of the land surface of the planet and about 7% of the marine environment (ProtectedPlanet, 2017).

Protecting places that are special or of societal use, with the purpose of conserving them, has been a tradition for many centuries. The Mauryan kings in Northern India, from 322 BCE to 187 BCE, had a system to protect forests in order to maintain and manage wildlife stocks, such as tigers and elephants, including laws and penalties for offenders (Rangajaran, 2005). The Mauryas sought to preserve supplies of elephants since it was cheaper and took less time to catch, tame and train wild elephants than to raise them. The tigers were protected for their skins. In 134 BCE, Roman Emperor Hadrian staked his claim to the mountains of Lebanon to protect its trees, because of their importance for ship-building. Over 200 stones were engraved to delineate his imperial forest with: "IMP(eratoris) HAD(riani) AUG(usti) D(e)F (initio) S(ilvarum) A(rborum) G(enera) IV C(etera) P(rivata)" meaning "Boundary of the forests of the Emperor Hadrian Augustus; four species of trees were reserved under the imperial privilege" (McNeill, 2007; Rich, 2013). Hence, those places were already protected for the ES they delivered to society.

PAs, or a network of PAs, can have many purposes including maintenance of healthy functioning ecosystems (Dudley, 2008), acting as a sanctuary (Liu et al., 2001), saving specific habitats, preserving ecological processes unable to survive in intensely managed land- or seascapes, providing space for assuring normal ecological functions, and preventing ecosystem fragmentation (Parrish et al., 2003; Chape et al., 2005).

PAs can also be managed to promote and preserve valuable cultural ES such as tourism, recreation, research, education and scenery or religious sanctuaries (Campos and Nepstad, 2006; Coad et al., 2008; Cardelús et al., 2013; Scull et al., 2017), providing the base for sustainable development.

PAs can also be used as a benchmark to assess the effects of human interactions with the environment. PAs are well-known for acting as refuges for species and ecological processes that would not persist in intensely managed landscapes and seascapes, and for their ability to provide space for natural evolution and potential ecological restoration (Dudley et al., 2010). This implies that the quality of nature and dependent services is higher in PAs than in the surrounding areas where human influence is present. In this way, the protected and unprotected areas can be compared to determine anthropogenic influence. Dudley (2008) even suggested that they can prevent threatened species (often endemic) from becoming extinct.

The first 'modern' PA was Yellowstone National Park, founded in 1872 and protected under United States law as "a public park or pleasuring ground for the benefit and enjoyment of the people". Similar types of PAs have been set up worldwide during the past 150 years (Bishop, 2004) although for different reasons. In North America, PAs were set up to protect dramatic and sublime scenery, in Africa parks were set up to protect game and their habitats in order to maintain elite hunting traditions, and in Europe PAs were established to protect the landscape and seascape (Adams and McShane, 1996; Draper et al., 2004; Phillips, 2007).

PAs are now considered essential in most national and international conservation strategies. Many public, private, community and voluntary organisations are active in promoting the conservation and sustainable management of particular areas with relevant environmental value. International networks of PAs have been established under global regulations, for example UNESCO World Heritage Sites, UNESCO Global Geoparks, Biosphere Reserves and Ramsar Conventions (Matthews, 1993; Jungmeier et al., 2008). Increasingly, regional agreements create networks of PAs, such as the Natura 2000 network in Europe (EU, 2000; Maiorano et al., 2007). In total, more than 200,000 sites meet the definition of a PA (Deguignet et al., 2014).

This broad variety of international and national conservation and management strategies, conventions, directives, networks and ownerships leads to a wide-ranging nomenclature for PAs, at different levels, and by many different bodies (IUCN, 2004). However, all these initiatives have in common that they are set up to achieve similar goals, as is shown in Table 1 (McNeely, 1993).

Just as the goals for most PAs have changed with time, their management practices have likewise changed since the establishment of the very first PA, such as most importantly the management of indigenous peoples. Local people living in the territory of the PA were often moved and excluded, with enforcement often carried out through either fences or fines, thus creating so-called "fortress conservation" (Brockington, 2002; Hutton et al., 2005; Buscher and Whande, 2007). This top-down "fortress conservation" has been the preferred way of conservation for most of the twentieth century (Hutton et al., 2005), especially in game reserves in Africa, such as the Mkomazi game reserve in Tanzania and the Kruger National Park in South Africa (Brockington, 1999). De-colonialisation in Africa emphasised that new ways of managing PAs without excluding (native) people were needed. Further, it

Table 1
Goals to be achieved by Protected Areas.

Protection type	Goal
Preserving nature	Safeguard outstanding areas of living richness, natural beauty and cultural significance. Maintain the diversity of ecosystems, species, genetic varieties, and ecological processes. Protect genetic variation and species which are needed to meet human needs.
Preserving the interaction between nature and humans	Provide homes to human communities with traditional cultures and knowledge of nature. Protect landscapes reflecting the history of human interaction with the environment.
To protect societal assets in nature	Provide for scientific, educational, recreational and spiritual needs of societies. Provide benefits to local and national economies.

became clear that a top-down approach for PA management as was used prior to the 1970's, did not only unjustly disempower local residents, it did not always provide the appropriate protection for biodiversity (Pimbert and Pretty, 1997). Consequently, since then there has been a more bottom-up inclusive, participatory and sustainable way of managing PAs (Buscher and Whande, 2007). Despite this, “fortress conservation” has remained one of the important ways of managing PAs worldwide (Oates, 1999; Terborgh, 1999; Cernea and Schmidt-Soltau, 2006; Buscher and Whande, 2007).

The new way of managing PAs has arisen through public awareness of academic ecology. Prior to the 1970's, ecology was largely viewed as a sub-discipline of biology, but since then has been regarded as an integrative discipline that links both the physical and biological processes and natural and social sciences (Odum, 1977). In the 1990s, a further academic paradigm shift was taking place in which humans and their activities were considered increasingly integral to the ecological research agenda. Such changes were reflected in the European school of landscape ecology (Naveh and Lieberman, 1994; Naveh, 2000) and in the disciplinary evolution of socio-ecology (Collins et al., 2011; Haberl et al., 2006; Singh et al., 2013). In the marine field, the socio-ecological system has become a driving factor in environmental management (Turner and Schaafsma, 2015; Elliott et al., 2017).

This also means that the assessment of the success of PA or networks of PAs should be evaluated in a more multidisciplinary way. This evaluation should use measures such as coverage of endemic and threatened species or representativeness in terms of their species diversity, genetic diversity and connectivity, but also should include socioeconomic metrics (Rodrigues et al., 2004; Júnior et al., 2016; Corrigan et al., 2017), assuming that PAs provide effective protection once established (Geldmann et al., 2013). Alternatively, PAs can be evaluated by means of their management measures, i.e. the presence of management plans, boundaries, staffing, and other management systems and processes (Jachmann, 2008). It is assumed that increased levels of management lead to a more successful protection (Geldmann et al., 2013). However, these managerial-directed analyses may not describe how conditions inside PAs change over time (Craigie et al., 2010). The business adages that you cannot manage anything without measuring it and that management and monitoring need precise goals against which their success is judged become relevant here (Roberts et al., 2003; Leemans, 2017; Pieraccini et al., 2017). Proper monitoring and evaluation of the goals set by the PA, are required to evaluate the effectiveness of management measures. Inadequate management linked to poor monitoring of outcomes will lead to what are critically called ‘paper parks’.

3. The concept of ES

The concept of ES was firstly described as “Environmental Services” in SCEP (1970). Then Westman (1977) suggested that the social value of the benefits that ecosystems provide could potentially be quantified so that society can make more informed policy and management decisions, and introduced the term “Nature's Services” (Fisher et al., 2009). In the 1980's the term “Ecosystem Services” (ES) was firstly used by Ehrlich and Ehrlich (1981) (also Mooney et al., 1997). The term ES became more accepted in scientific research in the 1990's, mainly as an important way to communicate societal dependence on nature as the most important life support system for humans (Costanza et al., 1992; Perrings et al., 1992; Daily, 1997; De Groot et al., 2002). To show this importance, different methods were developed to value ES economically (Costanza et al., 1997). An important step in introducing ES into policy was made by the Millennium Ecosystem Assessment (MEA, 2003) and since then the ES literature has increased exponentially (Fisher et al., 2009; Gómez-Baggethun et al., 2010).

The original definition of ES as indicated in the MEA is: “The benefits that people obtain from ecosystems”. These include *provisioning services* such as food, water, timber and fibre; *regulating services* such as those attenuating climate related impacts, floods, diseases, wastes and water quality; *cultural services* that provide recreational, aesthetic, and spiritual benefits, and *supporting services* such as soil formation, photosynthesis and nutrient cycling (MEA, 2005). This, however, is just one of many definitions that lead to some ambiguity (see Table 2). This ambiguity becomes particularly relevant when practitioners attempt to identify, characterise and value ES for a given area. Although there are different interpretations of what are ES, central in most of the definitions is the delivery by ecosystems of usable products and benefits to satisfy the needs of society (Daily, 1997; Costanza et al., 1997; De Groot et al., 2002; MEA, 2005; Boyd and Banzhaf, 2007; Fisher and Turner, 2008; Haines-Young and Potschin, 2010; TEEB, 2010; Harrison et al., 2010; Staub et al., 2011; Landers and Nahlik, 2013; Haines-Young and Potschin, 2013).

More recently, at least in the marine field, there has been an attempt to reduce this confusion by separating ES from Societal Goods and Benefits (SG&B) (Turner and Schaafsma, 2015; Scharin et al., 2016; Elliott et al., 2017; Burdon et al., 2018). This takes the view that there is a continuum whereby ecosystem structure leads to ecosystem functioning which in turn produces ES. Obtaining SG&B from those ES requires an input of complementary assets or human capital, such as time, energy, money and skills. For example, while a fully functioning sea can produce

Table 2
Overview of definitions of ES.

Authors	Year	Definition of ES
Daily	1997	A wide range of conditions and processes through which natural ecosystems, and the species that are a part of them, help sustain and fulfil human life
Constanza et al.	1997	The benefits human populations derive, directly or indirectly, from ecosystem functions
De Groot et al	2002	The capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly
MEA	2005	The benefits people obtain from ecosystems
Boyd & Banzhaf	2007	Components of nature, directly enjoyed, consumed, or used to yield human well-being
Fisher & Turner	2008	The aspects of ecosystems utilized (actively or passively) to produce human well-being
Haines-Young & Potschin	2010 + 2013	The contributions that ecosystems make to human well-being, and arise from the interaction of biotic and abiotic processes
TEEB	2010	The direct and indirect contributions of ecosystems to human well-being
Harrington et al.	2010	The benefits that humans recognise as obtained from ecosystems that support, directly or indirectly, their survival and quality of life
Staub et al.	2011	Aspects of ecosystems that have a recognisable connection to human welfare and that are used or valued in some form or other by the human population
Landers & Nahlik	2013	The components of nature, directly enjoyed, consumed or used to yield human wellbeing

fish, it is necessary for society to learn how to catch and use those fish. It is contended that such a separation of ES from SG&B helps to prevent definitions such as those in Table 2 where the two concepts are conflated.

4. Classifying ES

There have been many approaches to classification systems for ES (De Groot et al., 2002; MEA, 2005; Wallace, 2007; Costanza and van den Belt, 2011; Haines-Young and Potschin, 2011; Liquete et al., 2013; Landers and Nahlik, 2013; Turner et al., 2014; Rhodes, 2015; Pascual et al., 2017; Haines-Young and Potschin, 2018), but none of these classification systems has been universally accepted. Several definitions (Table 2) include ecosystem functions and processes but many of these classification systems conflate ES and societal benefits. Due to this conflation of ecosystem functions, ES and benefits, it may be difficult to distinguish between the actual services, related (economic) benefits, and the ecological processes that provide these services and benefits (see Table 3).

De Groot et al. (2002) attempted to classify the ES based on the Ecosystem Functions delivering them. The Millennium Ecosystem Assessment (2005) partly continued this, but was the first to group ES, and added the category provisioning services, making a fourth category next to regulating, supporting and cultural services. Wallace (2007) classified ES based on the human values they support, but omits the provisioning services. TEEB (2002) created a new classification to couple ES to the (economic) benefits they provide. CICES is gaining acceptance by scientists and policy makers globally but particularly in Europe. CICES does not include the MEA (2005) "supporting services" (La Notte et al., 2017; Czúcz et al., 2018). More recently, there are proposals in the marine field to separate ES from SG&B given the need to insert human complementary assets between the two (e.g. Scharin et al., 2016). The latest IPBES classification system of Nature's Contributions to People (NCP) proposes renaming ES and classifying these NCP on the contribution of nature to a good quality of life (Pascual et al., 2017). However, the classification of NCP largely resembles the ES classification by CICES (see also Table 3).

As indicated here, there is an increasing "taxonomisation" (hierarchical subdivisions) of ES, i.e. it becomes more important to assign a certain service to a certain category, instead of making it easier practically to measure certain ES. This "taxonomisation" may eventually lead to an incomprehensible categorisation of ordinary attributes as for example "maple syrup collectors" (Landers and Nahlik, 2013). In the American classification systems of Final Goods and ES (FEGS-CS), the Maple Syrup Collectors should firstly be categorised under Food Extractors in Forests and coded 21.0201, being regarded as a service. However, it is suggested here that the collectors are not a service but rather those who benefit from a certain service.

Moreover, most classification systems miss the connection with, on one hand, ecological attributes (e.g. ecosystem functions and structures) that give rise to the ES, and, on the other hand, the socio-economic attributes (e.g. the resulting SG&B and the factors influencing the ES). Knowing the causal connections encompasses the full range of interactions and dependencies from biophysical structures to socio-economy which are important for the existence and sustainable delivery of ES.

The Cascade model for ES (Haines-Young and Potschin, 2010) is a step forward in connecting ecosystem structures, functions, services and economic benefits. It allows linking natural systems to elements of human wellbeing, following a pattern similar to a production chain: from ecological structures and ecosystem functioning (processes), to the ES and SG&B. The advantage of this continuum is to effectively communicate societal dependence on ecosystems. Yet, as a cascade can be considered to be a continuum, there are no direct feedback loops, whereas in nature these feedback loops do exist, some of which may be adverse (Odum and Barrett, 1971). For example, the over-extraction of SG&B such as fish from the sea would adversely affect the ecosystem structure and functioning.

Feedback loops have already since long been used for adaptive risk assessment and risk management as encapsulated in the DPSIR (Driver-Pressure-State-Impact-Response) approach (Patrício et al., 2016). Haines-Young and Potschin (2010) combine DPSIR and ES thus possibly providing a feedback loop (Rounsevell et al., 2010; Kandziora et al., 2013). However, the proposed cause-effect chain described by Kandziora et al. (2013) with a feedback loop from human well-being to ecosystems and biodiversity still lacks the direct feedback between the various components of the system, i.e. ecosystem structures and functions, ES, threats and socio-economy. Rounsevell et al. (2010) overcomes this anomaly with Ecosystem Service Providers (structures and functions that deliver ES), that are dependent on Ecosystem Service Beneficiaries (socio-economy), so there is a possibility for direct feedback between both. In addition, a Supporting System (structures and functions that do not contribute to ES delivery) is proposed, but it remains unclear what are the effects on this part of the feedback loop. More recently, anomalies in the DPSIR framework have been corrected using the DAPSI(W)R(M) cycle (Patrício et al., 2016; Elliott et al., 2017) in which **D**rivers of basic human needs (such as for food) require **A**ctivities (such as fishing) which cause **P**ressures (as mechanisms of change to the ecosystem, e.g. scraping a trawl over the seabed). The **P**ressures then can cause a **S**tate Change on the natural system (a loss of fish) leading to **I**mpacts (on human **W**elfare, e.g. no fish for consumption, i.e. a loss of SG&B). Those adverse consequences then require **R**esponses (using management **M**easures; e.g. fish stock management plans). A feedback loop allows the management to respond to and control the drivers, activities and pressures and, in turn, to prevent negative consequences. Therefore, we emphasise the need to adopt, both for practitioners and for communication, the linkages of the cyclical adaptive management framework coupled to and encompassing feedback to the ES and SG&B analysis.

5. Assessment of ES

There are many techniques and approaches for valuing ES and SG&B (see Table 1 in Cooper et al., 2013). As an example here, ES can be assessed in monetary terms (for example using direct market, indirect market, contingent and group valuation terms) and non-monetary terms (Turner and Schaafsma, 2015).

5.1. Monetary assessment of ES

5.1.1. Direct market valuation

The trade value of ES on the open market (De Groot et al., 2002) is used to assess the economic value of SG&B provided by ES. This method is useful to measure provisioning services and some cultural services that can be traded, for example tourism or seagrass meadows and their value for fisheries (Vassallo et al., 2013; Jackson et al., 2015). An example of direct market valuation is the value of trees for firewood or construction wood which can be priced on the open market.

5.1.2. Indirect market valuation

When no explicit markets for certain services exist, a more indirect way of assessing the value of ES can be used, see Table 4.

5.1.3. Contingent valuation

Here, demand for ES is elicited by hypothetical scenarios that involve describing alternatives in a survey or a questionnaire. For example, respondents may be asked to express their preference of increasing the level of water quality in a stream, lake or river so that they might enjoy activities such as swimming, boating, or fishing (Wilson and Carpenter, 1999). In order to obtain realistic values, the respondents must have a good understanding of the ES or environmental quality

Table 3
Comparison of several different ways of classifying ES, and their links, *italic* headings are the terms used by the authors, green columns refer to Ecosystem Functions and structure, blue columns refer to Regulating ES, yellow columns to Provisioning ES, orange columns refer to Social and Cultural ES. If ES are not divided into different categories the colour grey is used.

Authors	Year	Way of classifying	Classification categories						Remarks:
			Ecosystem functions and structures		**Ecosystem elements of societal use**				
De Groot et al.	2002	ES are classified on the basis of related Ecosystem Functions	*Habitat functions:* refuge and reproduction habitat to wild plants and animals	*Production functions:* Photosynthesis and nutrient uptake used by secondary producers to create living biomass	*Regulation functions:* the capacity of ecosystems to regulate essential ecological processes and life support systems			*Information functions:* reflection, spiritual enrichment, cognitive development, recreation and aesthetic experience	Services are not explicitly named as such in the classification system, the ES are classified by the functions they depend upon. In this way the ecosystem delivering certain services is of big importance to the classification of these services.
Millennium Ecosystem Assessment	2005	ES are classified on the basis of the type of benefits humans can obtain from nature	*Supporting services:* those that are necessary for the production of all other ES		*Regulating services:* benefits obtained from the regulatory role of ecosystem processes		*Provisioning services:* products obtained from ecosystems	*Cultural services:* nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences	ES are classified in a more or less similar way as De Groot et al. (2002) classified functions. Ecosystem functions and structures are named *Supporting services*, because they support the delivery of all other services. *Provisioning services* are added to describe explicitly the products humans can get from ecosystems.
Wallace	2007	ES are classified on the basis of the specific human values they support	*Adequate resources:* basic needs that support the life of individuals		*Benign physical and chemical environment:* ES that keep the human physical and chemical environment within the tolerance levels of humans	*Protection from predators, disease and parasites:* the abundance and distribution of harmful organisms is sufficiently low that human well-being is not threatened		*Socio-cultural fulfilment:* ethical positions including those related to intrinsic values	The whole classification is more human oriented then previous systems.
Fisher & Turner,	2008	ES are categorised by the degree of connection to human welfare	*Intermediate services:* ecosystem structures and processes	*Final services:* services that humans utilise, but indirectly	*Benefits:* end products of the ecosystem utilised by humans				Supporting services are split into two categories. All other services are grouped and named "*benefits*"
TEEB (the Economics of Ecosystems and Biodiversity)	2010	ES are classified based on the direct or indirect benefits they provide to humans	*Habitat services:* the way in which ecosystems provide habitats, and gene pool protection		*Regulating services:* the services that ecosystems provide by acting as regulators		*Provisioning services:* ecosystem services that describe the material or energy outputs from ecosystems	*Cultural Services:* benefits people obtain from ecosystems through spiritual experience, recreation, mental and physical health, tourism and aesthetic appreciation	Largely follows the Millennium Ecosystem Assessment, Supporting services are now called Habitat services to highlight the importance of ecosystems to provide habitats.
Haines-Young & Potschin (CICES – the Common International Classification of Ecosystem Services)	2011, 2018	ES are classified by ecosystem outputs that directly affect human well-being			*Regulating and Maintenance services:* ways in which ecosystems control or modify biotic or abiotic parameters that define the environment of people		*Provisioning services:* tangible things that can be exchanged or traded, as well as consumed or used directly by people in manufacture	*Cultural and Social services:* non-material ecosystem outputs that have symbolic, cultural or intellectual significance	Supporting services are not considered to be services, and left out of the classification. Regulating services are renamed in *Regulating and maintenance services*, Provisioning services category is expanded from goods to tangible things.
Liquete et al.	2013	Classification largely follows CICES			*Regulating and Maintenance services*		*Provisioning services*	*Cultural and Social services*	No major changes
Rhodes (NESCS)	2015	ES are classified by the way they affect human welfare	*Environment:* spatial units with similar biophysical characteristics that are located on or near the Earth's surface and that contain or produce "end-products"		*End-Products:* biophysical components of nature that are directly used or appreciated by humans				Core ecosystem processes and beneficial ecosystem processes are grouped to *Environment*, Benefits are renamed to *End Products*, explicitly being biophysical components that are appreciated
Pascual et al. (IPBES)	2017	ES are called Nature's Contributions to People and classified accordingly			*Regulating contributions:* Functional and structural aspects of organisms and ecosystems that modify environmental conditions experienced by people		*Material contributions:* Material elements from nature that sustain people's physical existence and infrastructure	*Non-material contributions:* Contribution to people's subjective or psychological quality of life	Although using different nomenclature, largely resembles CICES

changes about which they will be asked, of the hypothetical method of payment, and of the social context of the payment.

5.1.4. Group valuation

This brings together stakeholders to discuss the values of ES which are regarded as public goods, and decisions regarding them affect many people. Therefore, their valuation should not come from values based on the opinion of individuals, but on public discussion (Jacobs, 1997; Sagoff, 1998; Wilson and Howarth, 2002; De Groot et al., 2002).

5.2. Non-monetary assessment of ES

Some of the ES values are difficult to assess directly, as described by Boerema et al. (2017), "no measures were found for the ES part of the

Table 4
Methods for indirect market valuation (from Bishop and Heberlein, 1979; Adamowicz, 1991; Hoevenagel, 1994; Toman, 1997; De Groot et al., 2002; Freeman, 2003).

Method:	Explanation:	Example:
Willingness to pay	Willingness to pay for the availability of certain ES.	The amount of money you would like to pay to have a nature reserve nearby your house.
Willingness to accept	Willingness to Accept compensation for the loss of certain ES.	The amount of money you would expect to receive if the wetland near your house would be lost due to bad management.
Avoided cost	ES allowing society to avoid certain costs that would have been there if these services would have been absent.	Flood control by e.g. dunes, it avoids the costs of property damage or damage to field crops.
Replacement cost	The value of an ES is related to the costs of replacing it by a man-made system.	Coastal defence by dunes can partly be replaced by building costly dikes or walls.
Factor income	ES can enhance incomes.	The way in which natural water quality increases commercial fisheries and income of fishermen.
Travel cost	The use of some ES may require travel to get to them. The travel costs can be seen as a reflection of the value of the service.	The travel costs made to travel by car to the nearest forest for a walk, to enjoy the scenery.
Hedonic pricing	Demand for ES can be reflected in the prices people will pay for associated goods.	A house near the beach is more expensive than a similar house near less attractive scenery.
Production costs	Costs to get back certain ES that have been lost due to human behaviour.	The costs of cleaning or repair due to pollution.
Dose-response	To what extent changing an ES affects the production costs of a product.	If lumber gets more expensive, because of declining forests.
Averting behaviour	The expenditures to defend against negative impacts of a certain ES.	Sunscreen sales on a beach.

cascade". In such case, indicators are used as proxies (Layke, 2009; Layke et al., 2012; Müller and Burkhard, 2012; Kandziora et al., 2013).

There is a plethora of indicators of ES, each of which may differ with ecosystem and be relevant for certain areas, habitats and ecosystems (Dobbs et al., 2011; van Oudenhoven et al., 2012; Atkins et al., 2015). Boerema et al. (2017) give proxies for measuring ES:

- Ecosystem properties: For example, often simple measures or indicators of biodiversity and population size are used for all ES that depend on biodiversity, such as Genetic Resources, Biological Control, Pollination and Life Cycle Maintenance,.
- Ecosystem functions: The functions and processes underpinning each ES are diverse and often composed of different components (Smith et al., 2013). Proxies for pollination may, for example, be intraspecific diversity, pollination effectiveness, visit rate, plant growth rate and infestation rate.

Non-monetary assessments of ES are particularly important given the persistent criticism of the ES assessment process that it could treat nature as a commodity, increasing economic discrepancies and question underlying philosophies that ecosystems and their biodiversity should be protected for their intrinsic value (Kosoy and Corbera, 2010; Spangenberg and Settele, 2010). This can be countered by jointly using both economic and ecological valuation (e.g. Pascual et al., 2012). Non-monetary assessments can aim to define a more aesthetic and less-tangible view of nature, ecosystems and biodiversity and their influence on social relationships, cultural evolution and spirituality (Chan et al., 2012; Raymond et al., 2013).

There are many social research methodologies for carrying out such assessments (Christie et al., 2008; Cooper et al., 2013). These range from spatially-oriented participatory GIS (Fagerholm et al., 2012; Brown and Fagerholm, 2015), to traditional social methods – both qualitative and quantitative – including interviews, surveys, observational studies and focal group discussions (Tengberg et al., 2012; Orenstein et al., 2015; Eizenberg et al., 2017). While the findings from such studies can be more challenging to convey to policy makers who may prefer economic valuations and quantitative data, these studies can help to characterise the intensity to which ES contribute to human well-being in both tangible and intangible ways, and compensate for the shortcomings of economic assessments.

6. Discussion

The confusion regarding the delineation, classification and categorisation of ES shows an inconsistency between approaches, and has blurred borders between ecosystem functioning, the services and the goods and benefits. Despite increasing approaches and a wealth of literature in the last 15 years, there has not yet been any agreement. Moreover, PA management is lagging behind in the introduction and use of the ES concept.[1] This discussion aims to link the different aspects of the ES approach in PA management, the confusion, commonalities and differences, connected to the classification and use of the ES approaches.

6.1. The inclusion of ES in international management frameworks

Since the Millennium Ecosystem Assessment (2003, 2005) there has been an exponential growth of literature on ES (Fisher et al., 2009; Gómez-Baggethun et al., 2010). This should have increased international interest in the use of ES in management and decision-making in general. In addition, an increased interest in an ES approach should be observed within PAs, given that PAs can be more effective in supplying ES, in comparison to exploited areas. People and society can benefit from an array of goods and services, including basic life-support goods, such as drinking water, or processes that regulate water and air quality, prevent natural hazards such as flooding, or mitigate climate change by storing carbon. PAs may even deliver sustainably produced crops or timber (Costanza et al., 1997; Daily, 1997; Dudley and Stolton, 2003; Sohngen and Brown, 2006; Campbell et al., 2008). Moreover, PAs provide cultural services such as recreation, tourism, research opportunities and maintaining cultural identity (Butcher, 2005; Eagles and Hillel, 2008).

A proper delivery of ES in PAs is dependent on a healthy and resilient ecosystem, since ES delivered by PAs are fundamentally supported by, and have to be in balance with, key ecosystem processes (Stolton et al., 2008; Dudley et al., 2010). These healthy and natural resources in PAs may, because of connectivity between ecosystems, positively reflect on the bordering areas thereby increasing the importance and spill-over of ES delivery within PAs to a much larger area (Di Lorenzo et al., 2016). Therefore again, it might have been expected that the ES concept is included in the various aspects and strategies of PA management and nature conservation. However, the use of an ES approach in biodiversity conservation is not explicitly mentioned in the Convention on Biological Diversity (CBD, 1992). Only later was it included explicitly in the "2020 Aichi targets", thereby complementing the CBD (Maes et al., 2012). A similar policy has been followed in the past 2 decades by the European Union as part of its commitment to the CBD. Protecting, valuing and appropriately restoring natural resources will help not only

[1] For instance, at a recent academic seminar in Israel, where the nearly-concluded Israel National Ecosystem Assessment was presented, several of a panel of stakeholders (representatives of land management agencies and conservation organisations) suggested that the ES concept was administratively too complicated, irrelevant, or in contradiction to the values the organisation promotes (in reference to the NGO).

to conserve biodiversity for its intrinsic value, but also for its essential contribution to human wellbeing and economic prosperity (European Commission, 2011). Consequently, the strategies and directives commissioned by the European Commission (EC), such as the Natura2000 framework (incorporating the Wild Birds and Habitats Directives, aiming for Favourable Conservation Status), the Water Framework Directive (for Good Ecological and Chemical Status), and the Marine Strategy Framework Directive (MSFD, for Good Environmental Status) aim not only to halt the loss of biodiversity, but also to halt the degradation of ES in the EU by 2020.

PAs are therefore key to achieving sustainability, maintaining biodiversity, ecosystem health, and delivering ES. As PA management currently is still strongly oriented towards sustainability, maintaining biodiversity, and maintaining ecosystem health, the ES framework should enhance current conservation strategies and management approaches in PA (Chan et al., 2006; Daily and Matson, 2008; Nelson et al., 2009; Egoh et al., 2009). However, the ES frameworks remain poorly explored across Europe (Haslett et al., 2010; Harrison et al., 2010), let alone implemented (Cowling et al., 2008; Daily and Matson, 2008). Similarly, although appealing to decision makers and implicitly included in top-level EC, UN and UNESCO documents, such as the Sustainable Development Goals (e.g. Cormier and Elliott, 2017), ES are not yet anchored in environmental legislation (Maes et al., 2012). Hence there is no legislative instrument requiring the ES framework in practice. This absence of legislation of course does not alleviate, and may indeed be the cause of the mismatch between the advanced theoretical outline of ES, increasing uptake in (inter)national directives and the lack of practical implementation and operationality of the ES framework in PAs.

6.2. Definition and classification systems of ES and use in PA management

A commonality in all ES definitions is that ES are the elements delivered by ecosystems that satisfy societal needs (Table 2). However, most of the definitions of ES tend to be broad which can cause confusion (Fu et al., 2011), even with the separation of ES from SG&B (Elliott et al., 2017). Hence the emphasis in this review on the increase of separating and making a distinction between the several elements in the continuum from ecosystem structure via ecosystem functions and ES to SG&B. For example, where the Millennium Ecosystem Assessment includes ecosystem functions and structures as supporting services, the CICES classification of ES omits them (see also Table 2, section comments).

It is acknowledged in this review that several functions are simultaneously services, hence the use of the term 'continuum'. Ecosystem functions refer to the physical, chemical, and biological rate processes that maintain an ecosystem—including material circulation, energy flow, information connections, and their dynamic evolution, and are considered intrinsic properties of ecosystems (Odum and Barrett, 1971; Wallace, 2007).

Turner (1999) originally considered ecosystem structures as a service, because the "infrastructure" is of value that its prior existence and maintenance is necessary for service provision. More recently, Turner and Schaafsma (2015) acknowledge the 4-step continuum from ecosystem structure to SG&B described above.

The confusion in the definition of services versus functions or benefits may even lead to contradictive constructions as observed by Fisher et al. (2009) who pondered the role of ES if there were no humans to benefit from them. If ES are defined as the benefits to humans then the planet could have ecosystem structures and processes, but no services. Fisher et al. (2009) and later collaborators (Turner and Schaafsma, 2015; Scharin et al., 2016) use the term "intermediate services", as pollination, primary productivity, water regulation and soil formation, and "final services" such as clean water, storm protection and constant stream flow are mentioned on the 'humanless planet'. Yet, following Fisher's initial reasoning in their thought experiment, it would have been logical not to call any of the mentioned structures and processes as intermediate or final services, since these processes would also occur on the planet without humans, and thus are according to Fisher's own definition normal functions and processes. What Fisher and co-workers further describe as benefits (drinking water, property protection, recreation, etc.) would have been ES only in case of human presence. The transition between the natural system and anthropogenic system, i.e. where ecosystem structures and functions become ecosystem services, is not clearly defined and causes confusion. This confusion is only to be overcome if we accept humans on the planet as an integral part of ecology, and then by separating ES from SG&B (Elliott et al., 2017) whereby the planet can produce ES as long as the ecosystem structure and functions are maintained, but that SG&B can thus only be achieved after the introduction of humans and complementary assets and human capital.

More confusion in classifying ES comes from the use of their economic valuation. Many environmental economists deem the MEA classification not fit-for-purpose as including supporting services may increase the risk of double counting of services (Boyd and Banzhaf, 2007; Wallace, 2007; Fisher et al., 2009; Fu et al., 2011). With ES, this usually occurs when processes ('means') and benefits (so-called 'ends') are mixed. For example, "nutrient cycling" is a supporting service, "water flow regulation" is a regulating service, and "recreation" is a cultural service, depending on "surface water for non-drinking purposes", a provisioning service. If a PA manager contemplates creating a wetland using a cost-benefit analysis, including these three services, there would be double counting, as "nutrient cycling" and "water flow regulation" help to provide the same service "surface water for non-drinking purposes" on which the service "recreation" depends. A solution to this problem can be found in Turner and Schaafsma (2015), by looking at the final SG&B, i.e. what is valued by society in financial or non-financial terms. Despite this, another solution can be not to classify each ES to fit into one of the possible categories, but to acquire its value by summing its different values to society.

6.3. Assessment methods for ES and use in PA management

Maintaining ES is becoming an ever-growing priority in sustainability science, and conservation plans increasingly emphasise joint protection or improvement of ES and biodiversity (Graves et al., 2017). To achieve this requires a harmonised and consistent monitoring scheme linked to pre-defined indicators and with an agreed action plan of measures if the indicator is breached or not reached (Borja et al., 2017). The monitoring has to be harmonised and quality controlled especially if the data from different areas are to be combined for a holistic assessment.

Direct market valuation is a straightforward method of valuation which relates to the trade value on the open market. Its major disadvantage is that many ES are not traded directly on markets (Koetse et al., 2015). The extraction of materials is usually forbidden in PA, this means that any ES that would normally be traded on the open market has no direct monetary value at all. For example, you cannot put a direct monetary value on trees in a PA, because you are not allowed to cut those trees, so you cannot trade them on the open market. Also if markets for ES are highly distorted, for example by taxes, subsidies, or government control, this method does not yield a proper value (Koetse et al., 2015). In addition, the market value of a certain service does not reflect the real capacity of a system to deliver this service, for example, if a good is in high demand, the market price goes up, without the stocks of the service going up.

Other valuation techniques such as non-market methods are necessary to evaluate societal appreciation of a certain ES. The downside is that the values are often subjective. Willingness to pay (WTP) values may be sensitive to context, or task given (Mitchell and Carson, 1989; Boyle et al., 1994), or how the method handles non-compliance, refusal to value or protest bids (Spash and Hanley, 1995). There are also questions regarding whether WTP values for non-economic, non-traded goods are valued by participants as consumers, or as citizens (Keat, 1994; Blamey et al., 1995; Sagoff, 1998). A good example of the questionable character of these methods is the contingent valuation approach on UK nature conservation policy, where some participants emphasised the difficulty of putting a monetary value on nature (Clark et al., 2000). There are also doubts over these methods as shown by questioning the difference between ‘willingness to pay’, which can be dependent on household income, social setting, etc., from ‘willingness to accept’ (Hanemann, 1991; Bateman and Turner, 1993). Some methods even discard responses such as zero or infinite values on the grounds that they are unreasonable, without making clear why they are unreasonable (Diamond and Hausman, 1994; Ludwig, 2000).

Both market and non-market valuation of ES have the difficulty that some in society tend to regard nature as something that should exist in its own right, without having an explicit value (Costanza et al., 1997; Ludwig, 2000). Nevertheless, if we keep the above-mentioned limitations in mind, market and non-market valuation methods can be valuable as they yield straightforward, easy to interpret values, until more reliable, less biased methods of measuring ES are developed.

The interest in non-monetary ES quantification has led to numerous ecological assessments of ES. These assessments typically identify indicators or proxies for ES, attempt their quantification, or try to spatially map them (Burkhard et al., 2012; Crossman et al., 2013; Hattam et al., 2015). However, in spite of growing policy and scientific interest, non-monetary valuation of ES still does not have a formalised methodology (Nieto-Romero et al., 2014). Despite the latter, it is still useful to perform these assessments, as they often provide supplementary information that cannot be captured through monetarisation, even if sometimes more coarse or arbitrary indicators/proxies are used. The added information can often address values that are not captured, or cannot be captured using monetary approaches (Layke, 2009; Seppelt et al., 2011). It has, however, to be taken into account that when using different measures per PA, the outcomes will be less comparable and therefore for the management of networks of PA (such as Natura 2000 sites in Europe) less accurate. Establishing at international, e.g. EU, scale a standardised set of harmonised practical measures for ES, both monetary and non-monetary, is therefore necessary for a proper implementation of the ES framework in PA management (see below).

Assessing cultural services is difficult as monetary methods often cannot be used, thereby creating a challenge for both scientists and PA management. PAs may exist for their biota (biological components) or just their land- or seascape (physical and habitat components) – for the latter, cultural services can be translated into societal benefits irrespective of the organisms present, i.e. natural landscapes that are particularly enjoyed by onlookers, e.g. water bodies, green and diverse vegetation, or orderly nature (e.g. Nassauer, 1995; Dramstad et al., 2006). Charismatic species, such as whales or giant pandas, may be the focus of ecotourism (Small, 2011) although recreational, inspirational or spiritual enjoyment of the landscape (cultural ES) is seldom attributed to a single organism or species (Sagie et al., 2013; Orenstein and Groner, 2015), and the latter can be even immaterial; hence, this feature is difficult to capture in a concrete attribute. Landscape preference is more often the result of the sum of many biological (and geological) parts. As such, the “value” of a particular organism or species cannot be considered independently of its ecosystem and surroundings. One potential solution to this quandary is the adoption of the concept of landscape services instead of ES (Termorshuizen and Opdam, 2009).

A complication in properly connecting ecological indicators/proxies to services is that ecosystem functions that provide ES often rely on a minimum level of ecosystem health, whereby the decline of ES delivery in a degrading system and their recovery in a recovering system are often not linear (Layke, 2009; Tett et al., 2013). This makes it difficult to couple ecosystem functioning to the delivery of ES. Determining the best indicators to represent ES remains then a crucial challenge (Anderson et al., 2009; Feld et al., 2009; Eigenbrod et al., 2010; Müller and Burkhard, 2012; Graves et al., 2017). Nevertheless for the future, establishing a harmonised and standardised set of valid ecological indicators (e.g. see Cormier and Elliott, 2017) will be the way forward to increase the perception of the direct connection between an ES and its underlying ecological processes, and thus giving insight in the functioning and potential impacts of/on(changes in) the ecosystem. It will also deliver the right tools to manage the environmental quality and simultaneously the sustainable use of (potential) ES.

A further complication is that the valuation is often benefit or beneficiary dependent (Boyd and Banzhaf, 2007), which means that the benefits of interest will change your appreciation of what is an ES, and what is not. For example, when used in PA management, water regulation services can be seen as an input to the final service of clean water provision, for example for recreational swimmers, and a benefit may be higher water quality. From the point of view of a recreational fisherman, however, clean water provision would no longer be a final service, but an intermediate one, leading to fish production as a final service. This means that whether a service is considered final or intermediate, and even what its economic value might be, will change depending on what is being valued, monitored or measured, as well as on who are the beneficiaries (Fisher et al., 2009).

The complication with benefit or beneficiary dependency is in fact an artificial problem, because to be able to use an ES approach it is necessary to measure or value a certain service, and whether this service is a primary function or an intermediate or final service is immaterial. This complication results from the scientist's tendency to deconstruct systems as a way to understand and manage them. However, it is often forgotten that these pieces or categories are artificial, and were only created for the better and easier understanding of a system. The appreciation and understanding of the whole system reconstructed from these artificial pieces is, however, often lacking or forgotten (Tansley, 1935). This occurs in many ES studies and although the classification of ES could be valuable in understanding a system, it is only the starting point for measuring the ES attributes themselves. The remaining confusion therefore prevents the use of the ES concept in the management and conservation strategy. It is therefore recommended that time should be spent on measuring the ES (or their proxies) relevant for stakeholders rather than on further classification.

6.4. The use of the ES approach in PA management

The ES concept is the route towards delivering the significant goods and benefits that ecosystems (natural, semi-natural, and human-dominated systems) supply to human society. This may emphasise the value of nature to different stakeholders, and in this way may assist the management of PAs. It has, however, to be taken into account that monetary assessments of ES must be used with considerable caution as many in society may regard nature of intrinsic value irrespective of its explicit human value (Costanza et al., 1997; Ludwig, 2000). In addition, many PA were founded to protect nature, so putting a value on something that should exist in its own right may seem odd. Therefore, a monetary assessment of ES might be less useable for the PA management practice. Non-monetary valuation methods might give a better fit to the aims of the PA management, but there are still no widely adoptable and standardised methods (Nieto-Romero et al., 2014). Because of this, there is a danger of using different measures in each PA, making the outcomes between sites non-comparable, and unsuitable for the management of networks of PA. Moreover, the decline of services delivery in a degrading system is often not linear (Layke, 2009; Tett et al., 2013). This makes it difficult to couple ecosystem functioning to the delivery of ES. Determining the best indicators to represent ES remains a crucial challenge (Eigenbrod et al., 2010; Anderson et al., 2009; Feld et al., 2009; Müller and Burkhard, 2012; Graves et al., 2017), but will eventually yield a way of looking at ES that is compatible with PA management.

The absence of one agreed, clear and harmonised classification system for ES in PA management (Daily, 1997; De Groot et al., 2002;

MEA, 2005; Liquete et al., 2013) makes the ES approach difficult to use in management, as a lack of methodological standardisation could hamper (cross-site) comparability and scalability across different spatial and temporal scales (Pereira and Cooper, 2006; Haase et al., 2018). As Nahlik et al. (2012) concluded, to be able to move the concept of ES into practice, there is a need for a (more) unified approach.

A test of suitability of the ES approach in a decision-making context has been done for the Millennium Ecosystem Assessment by Wallace (2007). He concluded that the classification of ES as described in the MEA cannot be used in decision making as it mixes 'means' (how to achieve a goal) with 'ends' (the goal that needs to be achieved). For providing an effective decision making context, as is needed in PA management, the classification of ES must show a PA manager the planning implications of certain decisions, through the diverse interactions between functions, structures, services, and pressures.

Moreover, the current ES concept has a high reliance on economic value, partly caused by the absence of fully operational non-monetary measurement systems for ES, making monetary valuation currently the best method available. This, however, may lead to, or even force, the management of PAs to drift away from the intrinsic value of protection of, and thus to a reduced focus on, nature and/or biodiversity per se (Sagoff, 2004; McAfee and Shapiro, 2010; Redford and Adams, 2009).

During interviews with PA managers, Fisher and Brown (2014) found that the ES concept was used in PA management already, but far from whole-heartedly, and some of the respondents even replied with *"I wouldn't say there has been any change in the central mission… nor how it looks on the ground, but there has been a lot of change in how we package it, promote it…the biggest change in that has been the ES stuff …."* or *"… I view with horror the idea that the way you protect nature is through communicating about it just in terms of services …."*. Continuing this, interviews with managers of 26 different European PAs within the EcoPotential project (Hummel et al., 2018) showed that only 2 out of 26 used the concept of ES in the management of their PA, although both managers did not know which framework (CICES, TEEB, etc.) was used. Nine of the managers replied that they are still considering whether to use the ES framework, and 15 respondents did not use the ES framework in their management at all; one of them even replied: "*The ES framework is a capitalistic way of preserving nature, how can you put a value on nature?*"

This indicates that the usage of the ES framework in PA management is not yet common practice, and when used, it is not always greeted with enthusiasm. This might be overcome by a bottom-up ES approach understood by the management, given that conflicts emerge when conservation strategies for PAs follow a top-down approach that excludes local practices or interests (West et al., 2006).

7. Conclusions and recommendations

The discussion here suggests that using an ES approach could possibly lead to commodification of nature, enlarging economic inequalities, or undermine the protection of nature and biodiversity for its intrinsic value (Kosoy and Corbera, 2010; Spangenberg and Settele, 2010). The strong focus in ES literature on the ES categorisation ('taxonomy') instead of measuring ES in practical terms, the lack of a harmonised system, and the lack of systematic insight in the relation of ES with underlying ecosystem functions and structures or socio-economic pressures, makes the ES framework not yet suitable for use in PA management.

As suggested by Tansley (1935), it is not useful to look at single elements of a system other than for simplifying the system for research purposes. It is important to consider the whole ecosystem including its attributes and interactions. Hence, it is argued that following the continuum from ecosystem structure and functioning, through ES to SG&B, is central for successful and sustainable PA management.

The ES framework is by definition highly anthropocentric (McCauley, 2006; Sagoff, 2008; Redford and Adams, 2009), and is nowadays mostly used as a tool to consider how to maximise profit and benefits from nature. Hence it has resonance with policy makers and implementers concerned with economic benefits, but the challenge is to ensure that such benefits can be accrued while also protecting the natural system.

The ES framework could also make clear to society that people are highly dependent on nature, not only for tangible goods, but also for (spiritual) well-being. As PAs are considered to be the "building blocks" of healthy land and seascapes and are central to achieve several important global targets (Juffe-Bignoli et al., 2014), it is beneficial to incorporate the ES approach into their management. This incorporation of the ES approach in PA management would need ES to become one of the central objectives in adaptive management, next to protecting and maintaining natural structures and functions, and at the same time deliver ES from which SG&B can be obtained (Elliott, 2011; Elliott et al., 2017). As such, there are several recommendations for the way forward in the application of the ES approach in PA management:

- *Reduce taxonomisation of ES:* Less emphasis on classifying and categorising ES, and more emphasis on developing (ecologically or socio-economically-based) methods to measure the proxies for a core-set of ES in a standardised way is needed.
- *Focus on a bottom-up approach on implementing the ES concept in PA management:* A stronger bottom-up way of implementing an ES approach that is understood by PA management is needed. Conflicts will emerge when conservation strategies for PAs follow a top-down approach that excludes local practices or interests. The PA management community should be incorporated in implementing the ES approach in a way that is practical and suitable for their purposes.
- *Avoid too much distinction between ecosystem functions and services*: There should be less emphasis on trying to find a distinction between ecosystem functions and services, as several functions are simultaneously services. Healthy ecosystem functions refer to a good status of the physical, chemical, and biological processes, and they all together contribute to the proper functioning of PAs and the sustainable maintenance of ES.
- *Develop a standardised set of indicators for ES - assessment in PA:* A standardised set of indicators for ES should be developed, established at international and transboundary scale, using monetary and non-monetary ES assessment methods, together with measures of the related ecosystem functions and structures and relevant pressures in and on the system, that are for practitioners easy to measure and understandable to use.

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