



The challenge of valuing ecosystem services that have no material benefits



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ABSTRACT

Since the Millennium Ecosystem Assessment, ecosystem service science has made much progress in framing core concepts and approaches, but there is still debate around the notion of cultural services, and a growing consensus that ecosystem use and ecosystem service use should be clearly differentiated. Part of the debate resides in the fact that the most significant sources of conflict around natural resource management arise from the multiple managements (uses) of ecosystems, rather than from the multiple uses of ecosystem services.

If the ecosystem approach or the ecosystem service paradigm are to be implemented at national levels, there is an urgent need to disentangle what are often semantic issues, revise the notion of cultural services, and more broadly, practically define the less tangible ecosystem services on which we depend. This is a critical step to identifying suitable ways to manage trade-offs and promote adaptive management.

Here we briefly review the problems associated with defining and quantifying cultural ecosystem services and suggest there could be merit in discarding this term for the simpler *non-material ecosystem services*. We also discuss the challenges in valuing the invaluable, and suggest that if we are to keep ecosystem service definition focused on the beneficiary, we need to further classify these challenging services, for example by differentiating services to individuals from services to communities. Also, we suggest that focussing on ecosystem service change rather than simply service delivery, and identifying common boundaries relevant for both people and ecosystems, would help meet some of these challenges.

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1. Introduction

1. a Rationale and scope

Since the Millennium Ecosystem Assessment, ecosystem service science has made considerable progress in framing core concepts and approaches that are relevant to both academics and practitioners (see es-partnership.org for a broad overview). As a consequence, our understanding of the ways in which ES support human wellbeing has improved dramatically. Lately much research and debate has gone into defining exactly what constitutes an ecosystem service. This is a bid to better quantify the extent to which ecosystems and 'natural capital' contributes to human

wellbeing (Hails and Ormerod, 2013). One persistent issue in this debate is the idea of cultural ecosystem services, and those services with no obvious material benefits. Closely linked with our emotional perceptions of the world, valuing these services remains a real challenge.

Yet, if the ecosystem approach or the ecosystem service paradigm are to be implemented at national levels it is necessary to disentangle what are often semantic issues, and practically revisit the notion of cultural services. This is a critical step to identifying suitable ways to manage trade-offs and promote adaptive management. It is thus timely to synthesise what we know about ecosystem services including those that have no material benefits, assess remaining challenges in the implementation of this paradigm, and propose novel perspectives to ensure wider benefits from the knowledge acquired.

This paper starts by providing an introduction to the notion of ecosystems and ecosystem services. It presents current

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representations of the links between ecosystems and their beneficiaries. Section 2 underlines the semantic challenges in valuing ecosystem services, and particularly those with non-material benefits. Section 3 investigates how valuation frameworks have tackled the case of non-material services, and what challenges have been overcome so far. Key lessons are identified here. Section 4 builds on these lessons to propose a framework that is adaptive to the scale of analysis, and takes into account the plurality of ways that ecosystem services are valued. The conclusions discuss why such a framework is critical to identifying suitable ways to manage trade-offs and promote adaptive management.

1. b Ecosystems and ecosystem services

Functioning ecosystems provide a range of services that are essential to support economic performance and human welfare (Costanza et al., 2014; EFTEC, 2006; Pascual and Muradian, 2010). 'Ecosystem services', are generated when ecosystems directly or indirectly contribute towards meeting human needs. This means the services that an ecosystem delivers is defined by society. In consequence, they are particular to a given human requirement or activity (Banzhaf and Boyd, 2005; Haines-Young and Potschin, 2010). Some are essential for human survival (for example food), while others are more desirable services for human enjoyment (for example recreation). The Millennium Ecosystem Assessment (MEA, 2005) was pivotal in promoting the ecosystem service concept. Not only did it reveal how ecosystem degradation jeopardised human wellbeing, but it also provided the basis on which to describe the diverse services that ecosystems provide to people. This paved the way for further assessments including the UK's National Ecosystem Assessment (UKNEA, 2011), and led to

ecosystem service thinking being integrated into decision making, planning and evaluation processes (Satz et al., 2013).

Over the past two decades, the concept of ecosystem services has been continually refined, moving from a descriptive concept focused on status and trends (Balvanera et al., 2014), to a more analytical science focused on understanding the processes by which ecosystems provide services. Interest in the complexity of the causal links has also led to models that describe the sequence of processes that connect the biophysical world with human wellbeing. These are typically described with cascading models (see Haines-Young and Potschin (2010) and Potschin and Haines-Young (2011)), which show the interdependence between an ecosystems' functions and processes (for example Bastian et al., 2013; Bateman et al., 2011; De Groot et al., 2010, 2002; Müller et al., 2010; Spangenberg et al., 2014b). These models have played a key role in formalising ecosystem service science (Balvanera et al., 2014).

The underlying principle of these cascading models (Fig. 1) is that functioning ecosystems provide a range of services that have many potential uses with different values attached. For example, in the case of fish production (an ecosystem service identified in the MEA, 2005), the ecosystem service cascade would investigate how primary production (which depends on the 'ecosystems organisation') supports the fish populations ('ecosystem functioning' or 'potential use') that are harvested to provide food ('ecosystem service' or 'actual use') with high nutritional content ('benefit') to the child that eats it. At this point the ecosystem service has a 'value' to society by contributing to the child's growth and survival. It is therefore only when people, individually or collectively (society), harvest an ecosystem service (in this example fish) that it becomes apparent what the contributions to human wellbeing (also called benefits) are; whether these be of financial nature, in

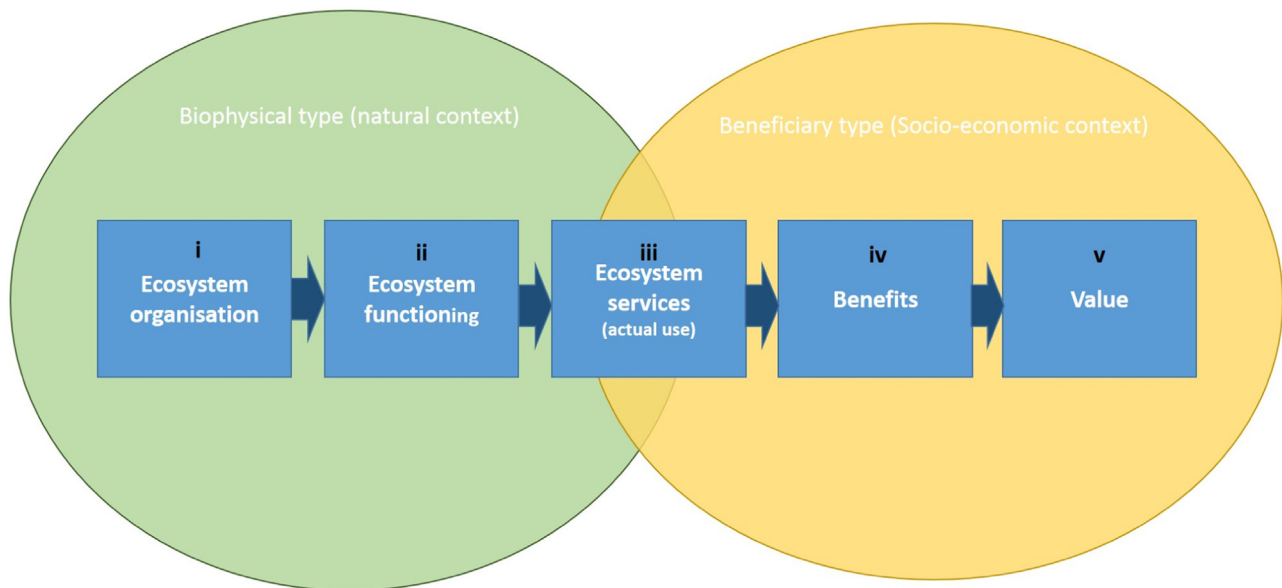


Fig. 1. Key aspects to understand when examining the links between the ecosystem and the benefits derived (cascade model adapted from Bastian et al. (2013), De Groot et al. (2010); Potschin and Haines-Young (2011), Haines-Young and Potschin (2010), Müller and Burkhard (2012), Müller et al. (2010), Spangenberg et al. (2014a)). This model highlights the elements necessary to understand and assess the links between the ecosystem and the benefits that we, as humans, derive from them. While many versions of this model have been proposed in both the social and natural sciences. In those reviewed there are slight differences in terminology. However, most concur that the main steps include assessing the links in the following ways. Firstly, the way the ecosystem is organised and works, referred to here as 'Ecosystem organisation', but referred to as 'biophysical structure or processes' by De Groot et al. (2010); Müller and Burkhard (2012); Müller et al. (2010); Potschin and Haines-Young (2011), or as 'ecosystem functions and elements' by Spangenberg et al. (2014a) and 'properties' by Bastian et al. (2013). Secondly, how the ecosystem functions i.e. its ecological integrity (De Groot et al., 2010; Müller et al., 2010); Potschin and Haines-Young (2011) or its potential to provide an ecosystem service, outlined in models by Bastian et al. (2013) as 'potentials' or 'ecosystem service potential' Spangenberg et al. (2014a). Thirdly, the way the ecosystem is actually used, all studies refer to 'ecosystem service' to describe the final services utilised. This is where the biophysical and socio-economic contexts overlap. Fourthly, the way an ecosystem benefits people and contributes to wellbeing (referred to as 'benefit(s)' or human benefit) although in frameworks by Bastian et al. (2013) benefits have been grouped with values. Finally, the "value" placed on the ecosystems, which depends on the character of the beneficiary.

the form of natural resources, or simply enhancing and sustaining human wellbeing (Bastian et al., 2013; Spangenberg et al., 2014a). Using and managing ecosystems for our own needs creates feedback loops that consequently modify the ecosystems organisation, and thus its potential to provide future ES (Balvanera et al., 2014).

While these cascading models have been widely recognised as a logical route to advancing our current understanding of the effects of ecosystem change on human wellbeing, the terminology around the notions of 'value' or 'use' of nature has met with criticism. Since the term 'value' is most commonly employed in a monetary sense, this term is indeed confusing when accounting for non-monetary 'values' such as the aesthetic value of a landscape, the historical or even sentimental value of a place (James, 2015). In fact, the terminology employed in the models seems to convey the sole idea that ecosystems deliver tangible benefits that we utilise and give value to, such as crops. This is despite the fact that many 'values' perceived by beneficiaries are non-monetary. Moreover there is often no direct 'use' of services (i.e. pollination), and there may be non-consumptive 'use' (i.e. recreation) as much as consumptive 'use' (i.e. eating fish) (Pascual and Muradin, 2010; James, 2015). While semantic in origin, the strong focus of these models on the benefits humans obtain from ecosystems has the potential to inadvertently advocate a materialistic and solely economic approach to ecosystem management, taking non-material valuation of ecosystem services altogether out of the equation (see James, 2015 for a more detailed critique).

It is important to note also that while many of these models seem to have a utilitarian perspective on ecosystems, these models refer to 'ecosystem service use' and not 'ecosystem use'. This has been the subject of confusion, with some arguing, for example, that using rivers to produce electricity or as conduits for waste disposal, could be considered as an ecosystem service. In both these cases, whether or not these rivers are lifeless, they can be used. Since no ecological processes are involved to produce the service – only physical processes, it is inexact to refer in this case to an ecosystem service. Here the term geosystem-service would possibly be more appropriate. Since the confusion has also stemmed from the misuse of the term 'ecosystem', often used to mean 'nature', many ecologists now use the term 'eco-services' (Mulder et al., 2015). We retain here this ecological perspective, that differentiates ecosystem from 'natural capital' (Ekins et al., 2003), and implies that ESs need to stem either from the organisation (i.e. biodiversity) or functioning of the ecosystem (i.e. fish biomass production) (see also models Fig. 1).

It is clear from the cascade model that ecosystem services might have many different beneficiaries. It is therefore likely that conflict will arise in sharing these services. For example, river micro-organisms growing on river rocks (which form part of the 'ecosystem organisation') play a key role in regulating water quality, thus delivering clean water (potential service use) that could be used by different beneficiaries (for example water companies, farmers or bathers). Conflicts may arise between these potential beneficiaries that may require trade-offs between users' needs when water supply is low during drought conditions. However, perhaps the most significant source of conflict arises from the multiple managements (uses) of ecosystems, rather than from the multiple uses of ecosystem services. For example, a river ecosystem could be managed to deliver eco-services such as clean water stemming from purification by the river ecosystem, but other beneficiaries within the same community may want to manage the river ecosystem as a conduit for waste disposal. In this case, the 'ecosystem use' to produce a geoservice (waste disposal) and the 'ecosystem service' (water purification) are incompatible since waste pollution would destroy purifying organisms in the river.

2. Semantic challenges in valuing ecosystem services

Valuation of ecosystem use and the services delivered by that ecosystem are an important step to resolve potential conflict between beneficiaries, establish trade-offs and potentially manage ecosystems in a more sustainable way (Costanza et al., 2011; Farber et al., 2002). This valuation process is fraught with difficulties, and we show here how semantic issues play a key part of the challenge. We first investigate how the complex and subjective nature of 'value' combined with the plurality of beneficiaries, work to obscure what the 'value' of an ecosystem service means. We then discuss how these challenges are exacerbated in the case of services with no material benefits, and explore the semantic barriers to valuing non-material services.

2. a Dealing with the complex and subjective nature of 'value'

Firstly, valuation of benefits from ecosystem services is complex and subjective. The volume *The Economics of Ecosystems and Biodiversity* (see TEEB, 2010, 2016) distinguishes ecological, social and economic benefits and values, highlighting that valuing ecosystem services and associated benefits is not straightforward. For example, some people will value their income higher than their cultural identity, and may be willing to give up this identity for wealth (TEEB, 2010). Also, different values can be attached to a particular benefit. These different values commonly fall within three broad types of benefits and associated values: ecological, socio-cultural or economic value (MEA, 2005; TEEB, 2010).

Ecological values include functional integrity, health or resilience of an ecosystem to sustain life (De Groot et al., 2010). These are important indicators to determine critical thresholds and minimum requirements for ecosystem service provision (TEEB, 2010). Although these measures contribute to welfare in their current form, they cannot readily be taken into account in the expression of individual preferences as they are too indirect and complex. Yet they are crucial for human survival since ecosystems play key roles in the maintenance of essential life-support processes (MEA, 2005).

Socio-cultural values describe the way all ecosystem service values are culturally constructed and contextualised (Brondizio et al., 2010), and can broadly be declined in three value domains. First, the 'intrinsic value' reflects the value of an ecosystem regardless of people, or in other words 'the sense of value that exists independently of human valuations' (O'Neill, 1993, p. 8). This value stems either from a 'deontological' view that ecosystems should not be harmed for the greater good of others, or from a 'consequentialist' view that nature itself does not have intrinsic value but nature's wellbeing does (Davidson 2013). The 'instrumental value' reflects how an ecosystem and its services directly contribute to the beneficiaries wellbeing (Chan et al., 2016; De Groot et al., 2002; Kenter et al., 2015; Scholte et al., 2015). For example, going in a river for a swim and seeing salmon in the water. Last, 'relational values' refer to the way people relate to nature and others (Chan et al., 2016), for example the way traditional farmers care for their land. It is then the combination of intrinsic, instrumental and relational values we hold, that influences our behaviour towards managing and using ecosystems (Brondizio et al., 2010). Indeed many of these 'underlying values' are those that shape people's perception of the world and thus guide their decisions (Ives and Kendal, 2014).

Economic values reflect the importance of an ecosystem and its services expressed in monetary terms. Economic value comprises both use values (i.e. direct value of using an ecosystem for recreational opportunities, or food provision; as well as indirectly using the ecosystem for regulating services for example carbon regulation) and non-use values. An example is the value people place on protecting and preserving the survival of charismatic

species like the Red Kite (*Milvus Milvu*) of the Otter (*Lutra lutra*) for future generations (option value) or for moral reasons (bequest and existence values) (De Groot et al., 2010). More generally economic value influences the notion of ownership and property applied to biodiversity (Brondizio et al., 2010). Over the long term, the 'value that people assign to things' may also change the way beneficiaries relate and manage the environment (Ives and Kendal, 2014).

It is important however, to bear in mind that despite the apparent plurality of values, recent studies have shown that there seems to be a consensus across societies on what values are important (e.g. transcendental values, Raymond and Kenter, 2016), and this has been shown to be stable over time (Bardi and Schwartz, 2003; Schwartz and Bardi, 2001) and across generations (Manfredo et al. in press). Schwartz and colleagues found that *benevolence*, *universalism*, and *self-direction* values to be most important and *power* and *simulation* values to be least important (Schwartz, 2012). This seems thus to indicate a natural hierarchy of values, that would play in favour of ecosystem services with non-material benefits.

2. b Coping with the plurality of beneficiaries

Secondly, the diversity of beneficiaries contributes to add an extra layer of complexity and plurality to the idea of ecosystem service 'value'. At the individual level, notable differences in valuation stem from geographic, inter-generational or cultural differences. For example, some plants currently considered as pests by some could be regarded as life-saving medicinal plants by future generations; invertebrates in rivers could be regarded as a threat by bathers but as an important asset by fishermen. Perhaps more importantly, local benefits to individuals may not coincide with benefits to societies at the global scale. For example, while it might benefit farmers to harvest crops from poor upland soils, it might not be beneficial for downstream fisheries that are affected by subsequent fertilizer runoff. Managing these environmental changes is thus about balancing the diverse competing perspectives for ES and how different types of beneficiaries prioritise these services (Hicks et al., 2013).

Differing ideas of the value of an ecosystem service, whether at individual or community level, give rise to the potential for conflicting values. One of the key challenges thus resides in developing a valuation approach that explicitly considers ecological, socio-cultural and economic values of a service, for a range of different beneficiaries. Indeed, rarely is the full range of potential beneficiaries taken into account, and most often, only ecological or economic values seem to prevail (Martín-López et al., 2014). Yet, the consideration of socio-cultural and ecological values is key if the ecosystem service paradigm is to support conservation policies and guide the environmental management (De Groot et al., 2002, 2010). Moreover, the ability to identify values for different beneficiaries, and thus different needs and perceptions, is the key to informing trade-offs and resolving potential social conflicts (Martín-López et al., 2012). This is particularly true since we know that valuation methods, and particularly those with an economic focus, not only elicit values but also tend to shape the value elicited (Martín-López et al., 2014).

2. c The specific case of services with non-material benefits

Lastly, perhaps the biggest challenge in ecosystem service valuation lies in giving a value to services that have no direct or indirect material benefits, referring here to benefits that are conceptual rather than physical (Chan et al., 2011; Oleson et al., 2015). Examples include spiritual enrichment, cognitive development, recreation and aesthetic experiences (MEA, 2005). Note that this does not include regulating services since these have indirect

material benefits (e.g. pollination increases crop yield). Non-material benefits are by definition intangible and subjective, and since the demand for such services is not obvious, quantifying the supply of these services is difficult (Daniel et al., 2012; De Groot et al., 2002; MEA, 2005; Milcu et al., 2013; Oleson et al., 2015; Plieninger et al., 2013; Weyland and Larterra, 2014). Efforts to value the non-material benefits of ecosystems have focused on services with tangible elements such as benefits arising from aesthetical and recreational opportunities (Milcu et al., 2013; Baulcomb et al., 2015). Since valuation processes in many modern societies are often achieved through monetary measurement, less tangible elements are frequently left out of traditional valuation frameworks (Atkinson et al., 2012; Chan et al., 2012b, 2011; Hernández-Morcillo et al., 2013; Milcu et al., 2013; Raymond et al., 2014; Scholte et al., 2015). Yet, it is often these less tangible ecosystem services that shape societies, cultures, welfare, and often drive environmental change. These services require a consistent accounting and valuation if the environment is to be managed in a sustainable way. It is likely that implementing an ecosystems approach to valuation could encourage the development of management actions that take into account both tangible and less tangible services (DEFRA, 2007). With governments across the world looking increasingly to adopt the ES approach, this agenda is becoming a clear priority.

Non-material benefits obtained from ecosystems have traditionally been grouped under the term 'cultural services' by the MEA (2005). This term has been widely used in subsequent international studies such as *The Economics of Ecosystems and Biodiversity* (TEEB, www.teebweb.org), which focuses on making the value of biodiversity and ES visible, and the *Common International Classification of Ecosystem Services* (CICES). The term 'cultural', here, was used to integrate the immaterial, mental and experiential values of ES. However, this terminology has met with much criticism (see for example, Chan et al., 2011, 2012a,b). A summarised overview of some of the key criticisms given to the term cultural services follows.

Firstly, combining all these notions of cultural value (for example moral, religious, aesthetic) under one term has led to an unclear definition of what culture means in this context (Pröpper and Hupps, 2014). Generally, culture is recognised as being the characteristics and knowledge of a particular group of people. Culture is classically defined by language, social habitats and structures, viewpoints, traditions and religion, which link to an individual's basic beliefs (Limburg et al., 2002; Spangenberg et al., 2014; Villamagna et al., 2013). Yet economic approaches treat 'culture' as a type of commodity for stimulating our minds e.g. paintings and films (Winthrop, 2014). To further muddy the waters, the terms 'socio-cultural value' and 'cultural ecosystem services' have often been fused in ecosystem service literature despite their conceptual differences (Costanza et al., 1997; De Groot et al., 2002; Scholte et al., 2015). For example, De Groot et al. (2002) refers to socio-cultural value as 'non-material wellbeing relating to information functions', whereas the MEA (2005) incorporated these information functions into the ecosystem services framework as cultural services (Scholte et al., 2015). In fact, as Pröpper and Hupps (2014) underline, these classifications have created a 'miscellaneous' type category where all contradictions and uncertainties relating to non-material values are stored away and treated as though they do not exist. They suggest that the term 'intangible/immaterial ecosystem services' is a more appropriate definition to use when dealing with the category of culture in the existing ecosystem service frameworks.

Secondly, both cultural services and associated values are prevalent across and within other ES categories (for example regulating and provisioning) (Scholte et al., 2015; MEA, 2005). For instance, fish can be considered both a cultural service through

recreational angling and a provisioning service in terms of food production. This has led to concerns of double-counting. Chan et al. (2012b) underline that this problem has been encouraged by the layout and definition of major global ecosystem service classifications. In fact, there is confusion on whether 'cultural' services can or should be integrated into the current ecosystem service framework. Chan et al. (2012a) suggest that the current ecosystem service framework is designed for material values and that including non-material services would require a new vision and methods. Indeed the necessity for altogether separate frameworks for *non-material ecosystem services* has been suggested (for example Kirchhoff, 2012; Pröpper and Hautps, 2014; Winthrop, 2014).

Thirdly, since culture is underpinned by a series of social, communicative and productive processes which determine how an individual (or community) interact with their environment (Pröpper and Hautps, 2014), the values people attach to cultural services are by definition fluid, spatially varied, scale and context dependent. It is clear that valuation frameworks need further refinement to be more sensitive to the many manifestations of culture arising from human interactions with ecosystems (Church et al., 2014).

3. Current knowledge on valuing services with non-material benefits

To understand these complex non-material services and associated socio-cultural values requires an exploration of the relationship between places, people, and the values they reflect and sustain at a range of organisation levels from the individual behaviour to societal governance. A way forward in addressing these challenges is to implement methodologies that can take into account the links between environmental state, associated non-material services and manifestations of culture and value that arise (Baulcomb et al., 2015; Martín-López et al., 2014). Here we review how valuation frameworks have tackled the case of non-material services, and what solutions have been achieved and lessons learned so far. We conclude in the last section by highlighting key characteristics for a valuation framework that is more adapted to take non-material services into account.

3. a Valuing the invaluable

The valuation of provisioning and regulating ES has progressed faster than the valuation of non-material ES (Abson and Termansen, 2011). Indeed, it is relatively straightforward to value material benefits (for an overview of monetary valuation methods see Chee, 2004; Christie et al., 2012; EFTEC, 2006). Initially, attempts to value non-material services were also performed using traditional monetary approaches (e.g. MEA, 2005), although most studies focused on the more accountable services – such as recreation or tourism (Milcu et al., 2013). A range of methods were tested, either based on market prices where possible, but also based on surveys of revealed preferences (e.g. experiences are associated with a cost, either direct cost or costs of time), stated preference (e.g. willingness to pay to increase or enhance the benefit from a service), and cost-based approaches (e.g. cost to replace the service or to avoid damage from non-existing service, see Milcu et al., 2013).

These approaches are widely utilised, but one of their main issues lies in that each service is given only 'one value' (Kenter et al., 2015). In other words, the tendency is to consider that 'the value' to society of a service is simply the aggregation of individual values provided by the surveys (Kenter et al., 2015; Klammer, 2003; Raymond et al., 2014). However, the 'value' that groups of individuals, as a society, attribute to a given ecosystem service

can be significantly different from the simple aggregated individual 'values', simply because the factors that influence communities can be different to what drives individual behaviours.

It has been frequently suggested for moving forward, that the best way to understand these individual values is to disaggregate the beneficiaries (Reyers et al., 2013; Daws et al., 2011; Hicks et al., 2013; Mastrangelo et al., 2015; Horcea-Milcu et al., 2016; Raymond et al., 2014) as it helps to ensure that subtle differences between beneficiary groups (for example land managers, farmers, local residents) are maintained and their views represented. To examine attitudes and preferences expressed by people about nature clearly requires alternative approaches that move away from monetisation and the instrumental paradigm and instead reflect more on the plurality of values (Raymond et al., 2014).

This knowledge has led to a series of valuation approaches which include: (i) the more classical suite of 'Preference based' methods (for example contingent valuation) that start to incorporate the relative importance (or instrumental value) people attribute to a given service, and (ii) more innovative approaches that aim to better capture the non-monetary value attached to these preferences by engaging stakeholders (Scholte et al., 2015; Raymond et al., 2014). The latter use both qualitative and quantitative research methods such as: surveys or interviews, deliberative and participatory tools (focus groups, in depth discussion groups, public participation GIS), observation approaches (participant, structured and unstructured) and expert based approaches (for example Delphi surveys).

3. b A question of scale

A further factor that makes the valuation of *non-material ecosystem services* challenging is that governance and social levels of organisation (local, community, region, nation, intergovernmental) are at scales that rarely match the scales that ecosystems are organised at (organism, ecosystem, landscape, biome, global) (Tzanopoulos et al., 2013; Folke et al., 1996; Cash et al., 2006; Galaz et al., 2007). This phenomenon is sometimes referred to as the 'Problem of Fit' (Folke et al., 2007; Young, 2002; Brown, 2003; Cumming et al., 2006). This is a challenge when we need to understand the links between ecosystems and people in these local, regional, national and global scale contexts.

The *deliberative and participatory methods* mentioned above are increasingly being advocated as a way to encompass the multi-scalar determinants that influence the value of non-material services (Kenter et al., 2015; Milcu et al., 2013). Deliberative and participatory methods can be further divided into three distinct groups based on the level of involvement and interaction of individuals within the group surveyed: *survey based*, *deliberative*, *analytic-deliberative* (Fish, 2011). *Survey based* methods include structured questionnaires, focus groups and interviews (semi, structured, unstructured). They can be used to gain insight into people's attitudes, preferences and behaviours and to explore the way people think about an issue. *Deliberative* methods include citizen's juries, in-depth discussion groups and deliberative opinion polls. It allows beneficiaries to ponder, debate, reflect and negotiate on matters of mutual interest. Kenter et al. (2011) noted that deliberative choice experiments elicited deeper held values from respondents, which highlights that socio-cultural values assessed collectively through deliberative settings are as important as those assessed individually using instruments (Raymond et al., 2014; Kenter et al., 2016). This resulted in respondents becoming more aware of the consequences of peoples actions on the environment and in their unwillingness to trade them off regardless of financial cost. *Analytic-deliberative* methods include an analytical step, which then leads to participatory modelling, deliberative monetary valuation and deliberative

multi-criteria analysis (Van Berkel and Verburg, 2014). This process can provide a means to assessing service benefits under different scenarios. *Deliberative* and *analytic-deliberative* tend to allow participants more time to reflect, construct or modify their preferences (Christie et al., 2006).

3. c The power and risks of maps

One of the important characteristics of these deliberative and participatory methods is that they are often associated to mapping techniques that spatially represent the distribution of ESs together with the values and preferences expressed for them by the beneficiaries (Darvill and Lindo, 2015; Plieninger et al., 2013; Raymond et al., 2009, 2014; Sherrouse et al., 2011; Villamagna et al., 2014). Spatial representation of the sources of ES has been an important characteristic of ecosystem service assessments at both global (for example MEA) and local level (e.g. UKNEA). Spatial representation in the form of maps, are indeed key tools for decision makers (Medcalf et al., 2014), but can also facilitate discussions among stakeholders at the local level (Church et al., 2014; Medcalf et al., 2014). Participatory initiatives of this sort have been very successful in helping beneficiaries and decision makers to visualise these less tangible services alongside other more tangibles one, and even facilitating trade-offs where these arise (Gould et al., 2014).

However, there are significant issues when trying to map the less tangible ecosystem services. There are effectively two main ways to map such services: i) through a consultation that identifies locations where selected non-material ecosystem service benefits can be found and experienced, or ii) without consulting the beneficiaries but based on assumptions instead, for example by selecting preferred landscape features, such as beautiful lakes and old woodlands (Scholte et al., 2015). In the first case, the issue is that participatory spatial valuations do not account for services present beyond the study area which respondents are asked to comment on. As a consequence, well known places tend to be over-valued and participants unfamiliar with the geography of an area are unlikely to comment on those less familiar yet potentially valuable areas (Van Berkel and Verburg, 2014). In the second case, values are based on literature that is not necessarily relevant to the concerned community nor to the context of the location. Environmental experiences are socially constructed, hold symbolic dimensions and are multi-dimensional (Winthrop, 2014). Current ecosystem service assessment are thus often limited in terms of highlighting all the social-ecological interactions that can take place. In fact, James (2015) raises the point that value may stem from how we respond to a place rather than from recognising actual service delivery.

3. d Key lessons

Following from the above a series of challenges remain: (i) methods that are rarely comparable because they were developed to address specific problems (Hernández-Morcillo et al., 2013; Winthrop, 2014); (ii) quality control and repeatability are rarely feasible since most of these methods rely on transient accounts (Winthrop, 2014); (iii) a lack of data at the resolution needed to recognise who in society is benefitting from the services, and where these individuals are located (Ambrose-Oji and Pagella, 2012); (iv) some perceived barriers for scientists to cross disciplinary boundaries (Norton et al., 2012), and (v) time and financial costs to collecting detailed data. However, the various trials to address the multi-faceted nature of these elusive non-material services have each provided some key tenants for a valuation framework capable of dealing with ecosystem services

that have no material benefits. Here we discuss the importance of two major lessons.

- 1 The relationships formed between ecosystems and their beneficiaries are multi-scaled and as a consequence, the 'full' value of a service needs to encompass the individual and community values across these scales.

Beneficiaries, attribute values to an ES, both on the basis of their own specific character (their natural character) and on the basis of their background (their nurtured character). These individual providers of values, aggregate to form groups, communities and societies at larger scales which determine their own set of values. Importantly, the sum of the aggregated individual values does not necessarily correspond to the community value. Take a valley with its river as an example. At the individual level of organisation, value may be placed on different recreational benefits for example canoeing or cycling depending on what the value holder prefers. The aggregation of values begins with groups where more than one individual begins to share similar values, for example, it could include different farmer groups and decisions over food production on the valley lands (e.g. intensive crops or extensive livestock). At the community level of organisation, the main community might be a fishing community that will value the river fish stocks that provide food and income for the community. At societal level (national and international scales), shared values may just stem from the perceived social benefit that trees contribute towards carbon sequestration and mitigate the social impacts of global climate change. In other words, beneficiaries at a national or international scale may also perceive benefit from knowing that particular ecosystem services exist and are being provided in localities that are not necessarily theirs.

- Socio-ecological systems are dynamic systems in space and time, because social and environmental drivers often operate and interact at different scales and at different speeds (Fig. 3). As a consequence, valuation frameworks need to account for the fact that ecosystem service values will also vary in space and time. There are two corollaries to this.

First, socio-ecological systems are already very complex, but understanding how socio-economic drivers of change interact among themselves, and with environmental drivers of change, at different scales, is not insignificant. Fig. 3 illustrates this complexity, revealing how different social and environmental drivers interact. It underlines how these are likely to operate at different scales in space and time. For example, if a small geomorphic process like a local landslide suddenly covers the river bed with sediments, this could temporarily disturb local water supply, but could also alter fish stocks downstream in the longer term if fish eggs are smothered. Thus this change could temporarily affect local communities dependent on clean water provision from the river, but also potentially affecting whole fishing communities at a catchment level for a long time. Somehow, valuation frameworks will need to take in account these dynamic linkages.

Secondly, it is also likely that the value beneficiaries attribute to a service will be more sensitive to change at local scales since it is at this scale that individuals interact most with their natural environment. To take an example, even though individuals are likely to appreciate, or at least know of, the value of woodlands for climate regulation at the global scale, most individuals are more likely to interact with woodlands in close proximity for recreation activities. However, as one then considers higher levels of social and environmental organisation, values attributed to the ES are increasingly likely to stem from shared values (for example Fig. 2).

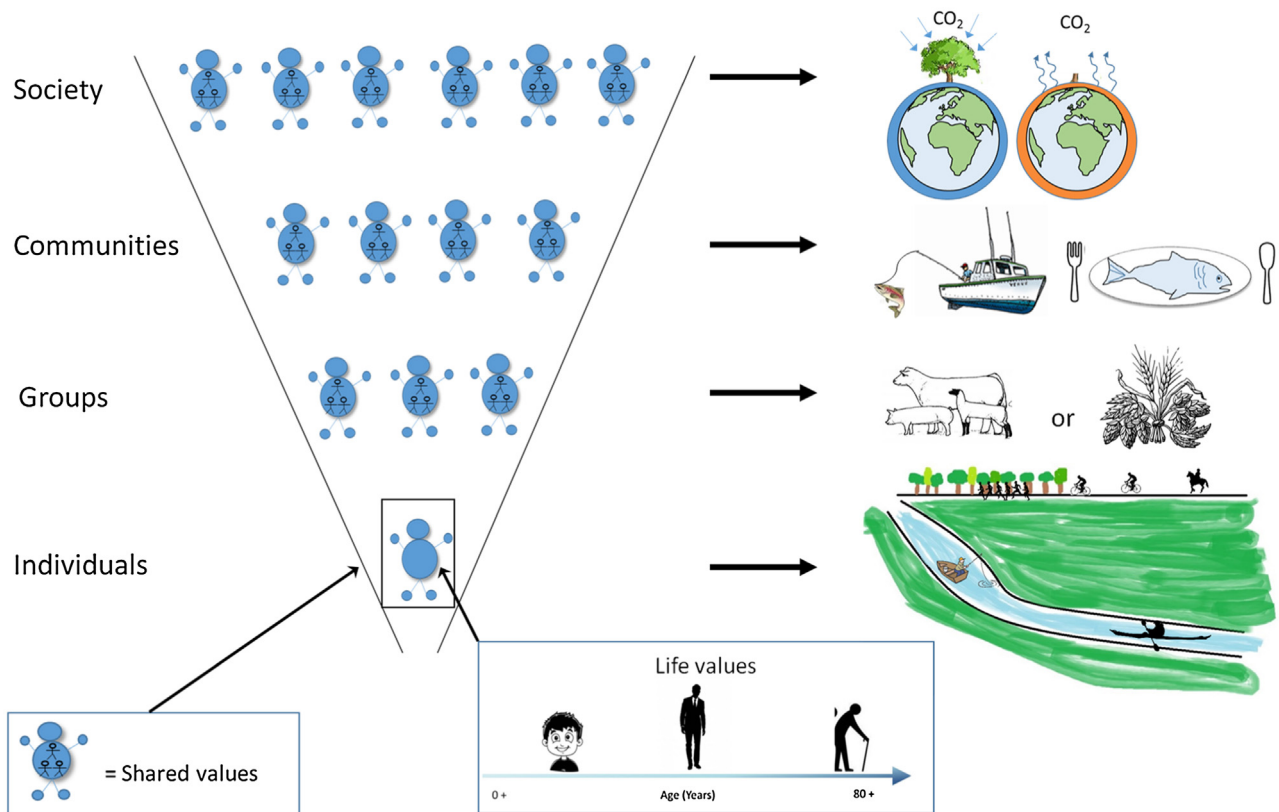


Fig. 2. Individual and shared values at different levels of organisation.

Left-hand side shows the different levels of social organisation. Starting with individuals, and as more people interact and share similar characteristics and interests, they begin to form groups, for example, 'livestock farmers' or 'crop farmers'. Communities are a type of social unit where groups of people are connected and share common values. The highest level, society, is the aggregation of people living together in an organised manner. The right-hand side illustrates different examples of the types of values held at each of these levels. At the individual level, each person has his own view or preference. One beneficiary of a wooded catchment may prefer fishing but a different individual may prefer and value the opportunity to cycle. At the community level, although each individual has their preference, collectively they share similar values, for instance a fishing community values fishing for the income it brings in but also because it is a part of who they are as a community. The temporal diagram also highlights that, for each individual, the values they attribute to an ecosystem service can change through their lifetime time

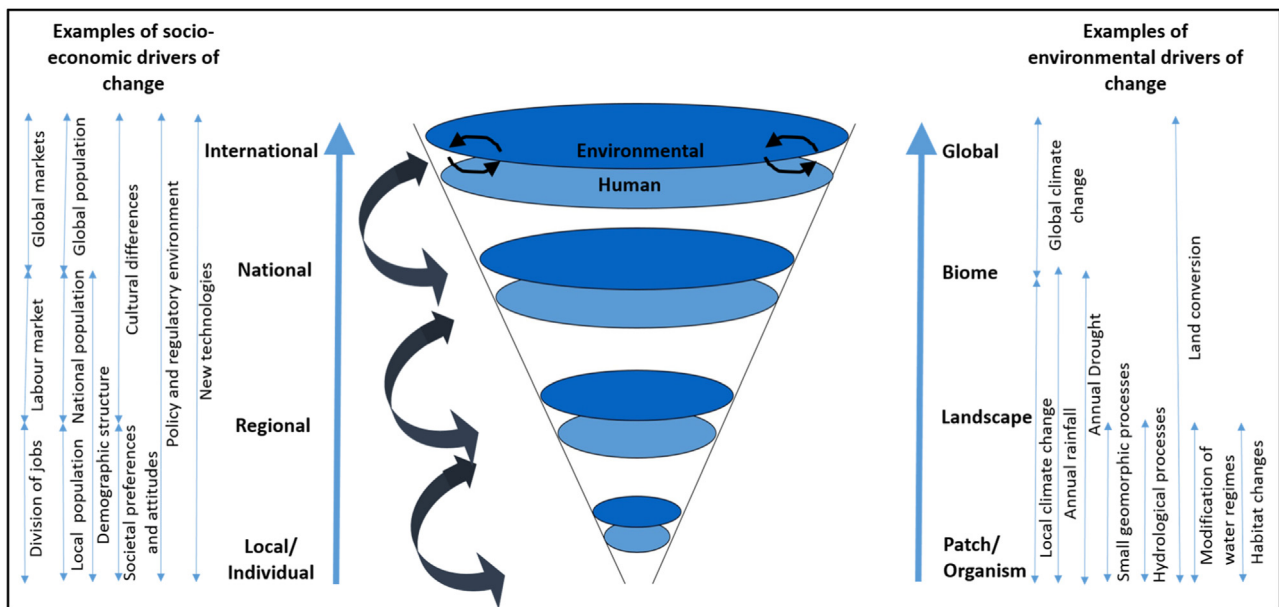


Fig. 3. An illustration of how socio-economic (on the left) and environmental drivers (on the right) of change interact at different scales.

4. A multi-scalar perspective on the link between ecosystems, ecosystem services and people

While there already seems to be a proliferation of approaches to assess non-material ES, there is clearly scope for a framework (e.g. Fig. 4) that simplifies and addresses the challenges associated with non-material valuation (see Sections 2 and 3). Building on the lessons learned in the previous section, we propose a framework that integrates a) beneficiary diversity, b) a changing world, and c) issues of scale. To illustrate the framework, we use the example of freshwater ecosystem services, since many of the services provided by rivers and lakes are non-material: recreational angling, water recreation activities such as swimming and canoeing, enjoyment of water landscapes and ecosystems stemming from spiritual or artistic feelings

4. a Addressing beneficiary diversity

If we assume that ecosystems have mixed groups of beneficiaries, and that the values they place on ES are influenced by their cultural background (Fig. 2), creating a simple classification that is relevant to the providers of value and how these relate to service benefits (Fig. 4a) could provide a useful solution. Here the focus is on disaggregating beneficiaries as a way forward, instead of simply aggregating individuals, communities and societies as one entity. For example, when valuing upland freshwater ecosystem services, beneficiary groups could include: local communities, farmers, tourism facing businesses, forestry, conservation groups, statutory organisations, and water industry.

Achieving disaggregation over a small area is relatively straightforward, for example through a place-based approach. Taking such an approach acknowledges the importance of local context and local conditions. At this scale, the meaning of a place – that is the way beneficiaries invest them with symbolic meanings, ideas and attachments (Hausmann et al., 2016) – can also be taken in account. For example, when valuing upland recreational fishing, the feelings of belonging or other symbolic meanings attached to a places' appropriateness to conduct fishing (e.g. Kirchhoff, 2012) can be incorporated, and the values that depend on an areas unique context (see Daniel et al., 2012) can be captured. It is also where problems can be identified, and solutions for that particular context can be articulated. During this process the different beneficiary groups and specific cultures can be identified, the

values they hold can be expressed and captured, and any conflicts arising can be resolved, with solutions found in a deliberative manner (e.g. Potschin and Haines-Young, 2012; Hausmann et al., 2016).

To achieve this level of disaggregation on much larger areas, for example at national level, requires a slightly different approach. First, the different demographic profiles and preferences that exist in the area of study need to be assessed. One option here is to use readily available data like Census data, or composite data such as the UK developed Index of Multiple Deprivation (Department for Communities and Local Government, 2015) that provides synoptic information on how deprived an area is in terms of income, employment, health, education, access to services, crime, physical environment, housing or services. Datasets like these could be used to develop a broad classification of beneficiary type that reflects some aspects of cultural background. However, at this high level of assessment, there is a possibility of over simplifying and suggesting that a particular group of beneficiaries all hold the same preferences. To reduce the likelihood of this oversimplification, a finer-scale analysis on social understanding of how different beneficiaries use and value ecosystems would still need to be conducted.

To gather this much more complex data, particularly at the local scale, would require tools that can help tease out the plurality of ways different groups of people value ES. Such information could be collected by using a combination of research methods, for example, focus groups, citizen's juries, participatory modelling, and deliberative choice modelling. Yet the practicalities (for example cost, resources, repeatability, comparability etc.) of implementing these methods would have to be considered carefully on a case basis to ensure they are suitable. The development of a fine level classification of potential beneficiaries would contribute towards a clearer understanding of how beneficiary interests vary (Hicks et al., 2013) and how each individual or group value and prioritise different aspects of the same system, thus providing further insight to how trade-offs amongst ecosystem services can occur.

4. b Taking into account a changing world

Our analysis of existing valuation frameworks (Section 3) has underlined the benefits of a place based approach, while also highlighting that the value of ecosystem services, and particularly

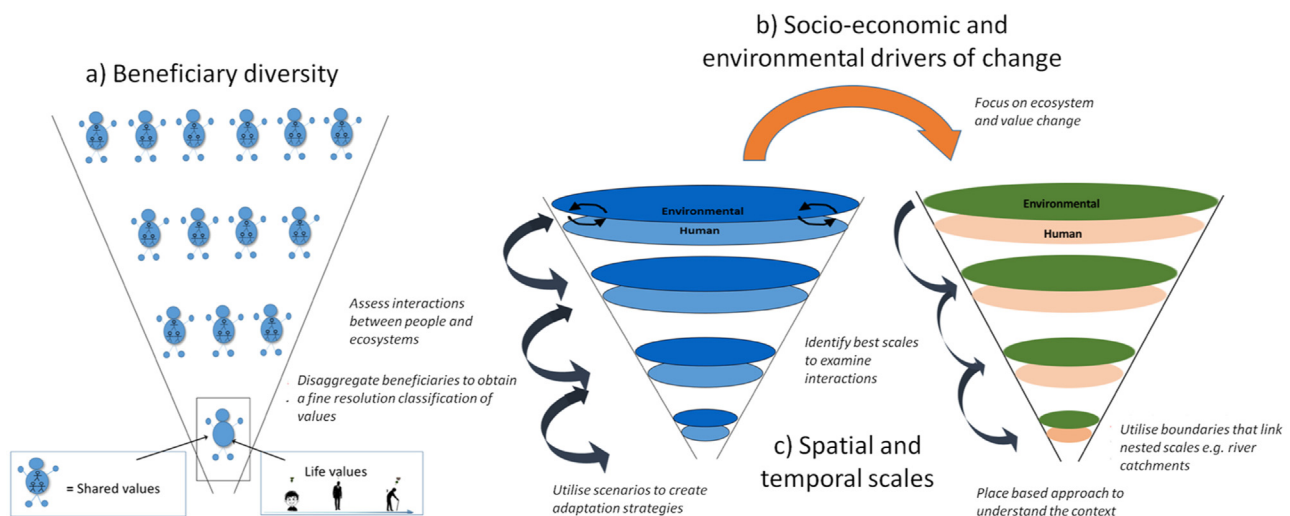


Fig. 4. A simple framework designed to deal with non-material services would address beneficiary diversity (a), take in account a changing world (b) and the different spatial and temporal scales relevant to both people and ecosystems (c). Some of the proposed solutions are provided in italics to illustrate how the framework could be practically implemented.

non-material ecosystem services, is influenced by a complex and dynamic set of social and environmental drivers (Fig. 3). Without necessarily treating socio-ecological systems as a black box, one solution might be to focus on how a specific landscape change might affect ecosystem service delivery and value. To take into account the complexity associated with social and environmental change, non-material valuation could consider taking a dynamic perspective as opposed to a traditional static one. Taking forward a dynamic approach thus helps to focus on ecosystem flows and is likely to provide a more adaptable framework that better reflects real situations (Fig. 4b).

To achieve this in practice, one solution is to adopt a place-based approach that uses our knowledge of the past to explore the future. Indeed, a promising avenue for considering the potential effects of global and local socio-ecological drivers of change on an ecosystem lies in scenario approaches where plausible changes are investigated through models linking landscape change to ecosystem delivery (Durance et al., 2016; Mulder et al., 2015). Scenarios provide alternative images of how the future may play out, and are a popular technique for thinking innovatively about dynamic, uncertain and alternative complex futures (Reed et al., 2013; Rickebusch et al., 2011). When combined to models linking landscape change to ecosystem and ecosystem service change, scenario narratives can aid the development of adaptable land management strategies where they can be used as a testbed to evaluate responses to the identified visions of the future. For example, the impact of an upland intensification scenario on dipper (*Cinclus cinclus*) birdwatching can start to be investigated using models showing how bird populations have declined where there has been intensification in the past. This also provides an excellent way of avoiding the challenge of fully understanding the dynamic socio-economic systems involved, since models relating ecosystems to ecosystem services are derived from our understanding of the past, and do not necessarily require an understanding of each individual link. Despite the fact that any depiction of the future is unlikely to be perfect, and what is important to a particular collection of beneficiaries today may not be tomorrow, such an approach could still be extremely valuable for planning and exploring potential future options (Durance et al., 2016).

4. c Addressing issues of Spatial and Temporal scale

Section 3 has revealed how, traditionally, researchers have assessed how a given land use (e.g. a plot of arable land) provides a given ecosystem service (e.g. rice production) at a given scale (e.g. across a whole region). Clearly, as drivers of change operate at and across different scales (Fig. 3), it might also be useful to move away from this scale dependent approach towards a more multi-scalar approach that investigates how changes in landscape might affect ecosystem service provision and values across different scales (Fig. 4c). This is all the more important because the relationship between individuals and services is in fact multidimensional – the value individuals attribute to services does not necessarily stem from direct interaction with the service (e.g. an individual may highly value dippers despite never have seen them).

Since valuation assessments are resource demanding, an interesting solution is to investigate how different drivers operate across different scales of organisation to identify the most efficient scales to assess ecosystems – that is the scales where most interactions between people and the ecosystem take place. For example, rather than considering just local authority boundaries when assessing ES, catchment boundaries, which are also historically natural human boundaries, can offer a better alternative (Durance et al., 2016). Catchments also provide opportunities to establish new forms of joined up thinking between upstream

land managers and downstream beneficiaries (Poppy et al., 2014). A step towards identifying ‘best scales’ of assessment is to use Geographical Information Systems (GIS). These platforms can deal with various scales and facilitate the harmonisation of socio-economic and ecological datasets in a spatial framework. In fact, it would also be possible to take this one step further by considering using other spatial tools like network-based tools that have proved successful wherever interactions between multiple entities are important (Rathwell and Peterson, 2012). Network tools have facilitated considerable advances across the divide between the social sciences and ecology (Mulder et al., 2015) and could easily be incorporated in our framework.

5. Conclusion

This review highlights that there is still much work needed on the ecosystem service paradigm to facilitate its broader implementation. More specifically, significant efforts are needed to enable a better valuation of ecosystem services that have no material benefits or obvious market value as these services are key to identifying sustainable ways to manage our natural resources for people and ecosystems in the future. Since current ES frameworks struggle to include non-material services, there is increasing recognition that either the ES frameworks should be amended to better account for the non-financial motivations and commitments beneficiaries have for the environment, or a completely new position should be taken to examine ‘culture’ and the many manifestations of value that arise from our interactions with nature. We have identified in this paper three promising avenues to address current barriers to the appropriate valuation of what we have termed *non-material ecosystem services*.

First, there is a need to disaggregate beneficiaries of ecosystem services to better reflect how societies across the globe are composed of different communities, groups and individuals, all with diverse and competing needs, at different times of life. This is of relatively straightforward application when completed in a place-based context, and the data to achieve these categorisations are mostly available, although it may be difficult to obtain at the fine local level and in developing countries.

Second, since both social and environmental drivers of change play a key role in determining *non-material ecosystem services*, we propose that ES assessments focus on *changes* in the ES delivery, thereby reducing the need for a full understanding of what are otherwise extremely complex dynamic socio-ecological systems. Scenarios, which provide narrated and mapped understanding of the future, coupled with dynamic landscape – ecosystem models, offer straightforward means within geographic information systems to achieve this goal. However, while we think this is a promising approach, already trialled in some national scale studies (e.g. Durance et al., 2016), there is considerable research needed to fully document the landscape change–ecosystem change models that this approach relies on (Balvanera et al., 2015; Mulder et al., 2015). To achieve this requires thoughtful curation of (and open access) to the large social and ecological data sets that scientists across the globe have been building.

Third, there is also a need to identify common boundaries relevant for both social and environmental systems. While this may be achieved by taking advantage of natural nested scales like river catchments, or by being creative with current spatial analysis tools, this still represents a significant barrier to implementing the ecosystem services paradigm, foremost when it comes to *non-material ecosystem services*. Efforts to collate and then harmonise social and environmental datasets at relevant scales will be key to implementing more broadly the ecosystem services paradigm, particularly where non-material benefits are concerned. As with ecosystem science more generally, the maintenance of long term

observatories that can bring together social and environmental data will also be central to understanding and potentially deciding on the best options for our landscapes and natural resources.

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References

- Abson, D.J., Termansen, M., 2011. Valuing ecosystem services in terms of ecological risks and returns. *Conserv. Biol.* 25, 250–258.
- Ambrose-Oji, B., Pagella, T., 2012. Spatial Analysis and Prioritisation of Cultural Ecosystem Services: A Review of Methods. Forest Research, Alice Holt Lodge Farnham, Surrey.
- Atkinson, G., Bateman, I., Mourato, S., 2012. Recent advances in the valuation of ecosystem services and biodiversity. *Oxford Rev. Econ. Policy* 28, 22–47.
- Balvanera, P., Siddique, I., Dee, L., Paquette, A., Isbell, F., Gonzalez, A., Byrnes, J., et al., 2014. Linking biodiversity and ecosystem services: current uncertainties and the necessary next steps. *Bioscience* 64, 49–57.
- Balvanera, P., Qujas, S., Martin-Lopez, B., Barrios, E., Dee, L., Forest, I., Durance, I., et al., 2015. The links between biodiversity and ecosystem services. In: Potschin, M., Haines-Young, R., Fish, R., Turner, K.R. (Eds.), *Handbook of Ecosystem Services*. Routledge Eds., pp. 45–61.
- Banzhaf, S., Boyd, J., 2005. The architecture and measurement of an ecosystem service index. *Discuss. Pap. Resour. Future DP* 54.
- Bardi, A., Schwartz, S.H., 2003. Values and behavior: strength and structure of relations. *Pers. Soc. Psychol. Bull.* 29, 1207–1220.
- Bastian, O., Syrbe, R.-U., Rosenberg, M., Rahe, D., Grunewald, K., 2013. The five pillar EPPS framework for quantifying, mapping and managing ecosystem services. *Ecosyst. Serv.* 4, 15–24.
- Bateman, I., Abson, D., Beaumont, N., Darnell, A., Fezzi, C., Hanley, N., Kontoleon, A., Maddison, D., Morling, P., Morris, J., Mourato, S., Pascual, U., Perino, G., Sen, A., Tinch, D., Turner, K., Valatin, G., Andrews, B., Asara, V., Askew, T., Aslam, U., Atkinson, G., Beharry-Borg, N., Bolt, K., Cole, M., Collins, M., Comerford, E., Coombes, E., Crowe, A., Dugdale, S., Dunn, H., Foden, J., Gibbons, S., Haines-Young, R., Hattam, C., Hulme, M., Ishwaran, M., Lovett, A., Luisetti, T., MacKerron, G., Mangi, S., Moran, D., Munday, P., Paterson, J., Resende, G., Siriwardena, G., Skea, J., van Daan, Soest, Termansen, M., 2011. Economic values from ecosystems. The UK National Ecosystem Assessment Technical Report. United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC), Cambridge.
- Baulcomb, C., Fletcher, R., Lewis, A., Akoglu, E., Robinson, L., von Almen, A., Hussain, S., et al., 2015. A pathway to identifying and valuing cultural ecosystem services: an application to marine food webs. *Ecosyst. Serv.* 11, 128–139.
- Brondizio, E.S., W., G. F. Zografos, C., Kumar, M., 2010. Socio-cultural context of ecosystem and biodiversity valuation. In: Kumar, P. (Ed.), *The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations*. Earthscan, London and Washington.
- Brown, K., 2003. Integrating conservation and development: a case of institutional misfit. *Front. Ecol. Environ.* 1, 479–487.
- Cash, D.W., Adger, W.N., Berkes, F., Garden, P., Lebel, L., Olsson, P., Pritchard, L., et al., 2006. Scale and cross-scale dynamics: governance and information in a multilevel world. *Ecol. Soc.* 11.
- Chan, K.M.A.G., Satterfield, J., Hannahs, T., Kikiloi, N., Naidoo, K., R., Vadeboncoeur, U., Woodside, 2011. Cultural services and non-use values. In: Kareiva, P.M., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S. (Eds.), *Natural Capital: Theory and Practice of Mapping Ecosystem Services*. Oxford University Press, Oxford.
- Chan, K.M.A., Guerry, A.D., Balvanera, P., Klain, S., Satterfield, T., Basurto, X., Bostrom, A., et al., 2012a. Where are cultural and social in ecosystem services? A framework for constructive engagement. *Bioscience* 62, 744–756.
- Chan, K.M.A., Satterfield, T., Goldstein, J., 2012b. Rethinking ecosystem services to better address and navigate cultural values. *Ecol. Econ.* 74, 8–18.
- Chan, K.M.A., Balvanera, P., Benessaiah, K., Chapman, M., Díaz, S., Gómez-Baggethun, E., Gould, R., et al., 2016. Opinion: why protect nature? Rethinking values and the environment. *Proc. Natl. Acad. Sci.* 113, 1462–1465.
- Chee, Y.E., 2004. An ecological perspective on the valuation of ecosystem services. *Biol. Conserv.* 120, 549–565.
- Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R., Hyde, T., 2006. Valuing the diversity of biodiversity. *Ecol. Econ.* 58, 304–317.
- Christie, M., Fazey, I., Cooper, R., Hyde, T., Kenter, J.O., 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. *Ecol. Econ.* 83, 67–78.
- Church, A., Fish, R., Haines-Young, R., Mourato, S., Tratalos, J., Stapleton, L., Willis, C., et al., 2014. UK National Ecosystem Assessment Follow-on. Work Package Report 5: Cultural Ecosystem Services and Indicators. UNEP-WCMC, LWEC, UK.
- Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., et al., 1997. The value of the world's ecosystem services and natural capital. *Nature* 387, 253–260.
- Costanza, R., Kubiszewski, I., Ervin, D., Bluffstone, R., Boyd, J., Brown, D., Chang, H., et al., 2011. Valuing ecological systems and services. *F1000. Biol. Rep.* 3.
- Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., et al., 2014. Changes in the global value of ecosystem services. *Glob. Environ. Change* 26, 152–158.
- Cumming, G.S., Cumming, D.H.M., Redman, C.L., 2006. Scale mismatches in social-ecological systems: causes, consequences, and solutions. *Ecol. Soc.* 11.
- Defra, 2007. Securing a Healthy Natural Environment: An Action Plan for Embedding the Ecosystem Approach. Department for Environment, Food and Rural Affairs, London.
- Daniel, T.C., Muhar, A., Arnberger, A., Aznar, O., Boyd, J.W., Chan, K.M.A., Costanza, R., et al., 2012. Contributions of cultural services to the ecosystem services agenda. *Proc. Natl. Acad. Sci. U. S. A.* 109, 8812–8819.
- Darvill, R., Lindo, Z., 2015. The inclusion of stakeholders and cultural ecosystem services in land management trade-off decisions using an ecosystem services approach. *Landsc. Ecol.* 31, 533–545.
- De Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* 41, 393–408.
- De Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol. Complex.* 7, 260–272.
- Department for Communities and Local Government, 2015. https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/465791/English_Indices_of_Deprivation_2015_-_Statistical_Release.pdf.
- Durance, I., Bruford, M.W., Chalmers, R., Chappell, N.A., Christie, M., Cosby, B.J., Noble, D., et al., 2016. The challenges of linking ecosystem services to biodiversity: lessons from a large-scale freshwater study. *Adv. Ecol. Res.* 54, 87–134.
- EFTEC, 2006. Valuing Our Natural Environment ? Final Report. EFTEC, London.
- Ekins, P., Simon, S., Deutsch, L., Folke, C., De Groot, R., 2003. A framework for the practical application of the concepts of critical natural capital and strong sustainability. *Ecol. Econ.* 44, 165–185.
- Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. *Ecol. Econ.* 41, 375–392.
- Fish, R.D., 2011. Environmental decision making and an ecosystems approach: some challenges from the perspective of social science. *Prog. Phys. Geogr.* 35, 671–680.
- Folke, C., Holling, C.S., Perrings, C., 1996. Biological diversity, ecosystems, and the human scale. *Ecol. Appl.* 6, 1018–1024.
- Folke, C., Pritchard Jr, L., Berkes, F., Colding, J., Svedin, U., 2007. The problem of fit between ecosystems and institutions: ten years later. *Ecol. Soc.* 12.
- Galaz, V., Hahn, T., Olsson, P., Folke, C., Svedin, U., 2007. The problem of fit between ecosystems and governance systems: insights and emerging challenges. In: Young, O., King, L.A., Schroeder, H. (Eds.), *The Institutional Dimensions of Global Environmental Change: Principal Findings and Future Directions*. MIT Press, Boston Massachusetts, USA.
- Gould, R.K., Ardo, N.M., Woodside, U., Satterfield, T., Hannahs, N., Daily, G.C., 2014. The forest has a story: cultural ecosystem services in Kona, Hawaii. *Ecol. Soc.* 19.
- Hails, R.S., Ormerod, S.J., 2013. Ecological science for ecosystem services and the stewardship of Natural Capital. *J. Appl. Ecol.* 50, 807–810.
- Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D., Frid, C. (Eds.), *Ecosystem Ecology: A New Synthesis*. BES Ecological Reviews Series. University Press, Cambridge: Cambridge.
- Hausmann, A., Slotow, R., Burns, J.K., Di Minin, E., 2016. 'The ecosystem service of sense of place: benefits for human well-being and biodiversity conservation'. *Environ. Conserv.* 43, 117–127.
- Hernández-Morcillo, M., Plieninger, T., Bieling, C., 2013. An empirical review of cultural ecosystem service indicators. *Ecol. Indic.* 29, 434–444.
- Hicks, C.C., Graham, N.A.J., Cinner, J.E., 2013. Synergies and tradeoffs in how managers, scientists, and fishers value coral reef ecosystem services. *Glob. Environ. Change* 23, 1444–1453.
- Horcea-Milcu, A.-I., Leventon, J., Hanspach, J., Fischer, J., 2016. Disaggregated contributions of ecosystem services to human well-being: a case study from Eastern Europe. *Reg. Environ. Change* 16, 1779–1791.
- James, S.P., 2015. Cultural ecosystem services: a critical assessment. *Ethics Policy Environ.* 18, 338–350.
- Kenter, J.O., Hyde, T., Christie, M., Fazey, I., 2011. The importance of deliberation in valuing ecosystem services in developing countries—evidence from the Solomon Islands. *Glob. Environ. Change* 21, 505–521.
- Kenter, J.O., O'Brien, L., Hockley, N., Ravenscroft, N., Fazey, I., Irvine, K.N., Reed, M.S., et al., 2015. What are shared and social values of ecosystems? *Ecol. Econ.* 111, 86–99.
- Kenter, J.O., Reed, M., Fazey, I., 2016. The deliberative value formation model. *Ecosyst. Serv.* 21, 194–207.
- Kirchhoff, T., 2012. Pivotal cultural values of nature cannot be integrated into the ecosystem services framework. *Proc. Natl. Acad. Sci. U. S. A.* 109.
- Klamer, A., 2003. A pragmatic view on values in economics. *J. Econ. Methodol.* 10, 191–212.
- Limburg, K.E., O'Neill, R.V., Costanza, R., Farber, S.v., 2002. Complex systems and valuation. *Ecol. Econ.* 41, 409–420.
- Müller, F., Burkhard, B., 2012. The indicator side of ecosystem services. *Ecosyst. Serv.* 1, 26–30.
- Müller, F., de Groot, R., Willemen, L., 2010. Ecosystem services at the landscape scale: the need for integrative approaches. *Landsc. Online* 23, 1–11.
- MEA, 2005. Ecosystems and Human Well-Being: Synthesis. MEA, Washington D.C.

- Manfredo, M., Bruskotter, J.T., Teel, T.L., Fulton, D., Schwartz, S.H., Arlinghaus, R., Oishi, S., Uskul, F., Redford, K., Kitayama, S., Sullivan, L., 2017. Why social values cannot be changed for the sake of conservation. *Conserv. Biol.* doi:http://dx.doi.org/10.1111/cobi.12855 (in press).
- Martín-López, B., Iniesta-Arandia, I., García-Llorente, M., Palomo, I., Casado-Arzuaga, I., Del Amo, D.G., Gómez-Baggethun, E., et al., 2012. Uncovering ecosystem service bundles through social preferences. *PLoS One* 7.
- Martín-López, B., Gómez-Baggethun, E., García-Llorente, M., Montes, C., 2014. Trade-offs across value-domains in ecosystem services assessment. *Ecol. Indic.* 37, 220–228.
- Mastrangelo, M.E., Weyland, F., Herrera, L.P., Villarino, S.H., Barral, M.P., Auer, A.D., 2015. Ecosystem services research in contrasting socio-ecological contexts of Argentina: critical assessment and future directions. *Ecosyst. Serv.* 16, 63–73.
- Medcalf, K., Small, N., Finch, C., Williams, J., Blair, T., Haines-Young, R., Potschin, M., et al., 2014. Further Development of a Spatial Framework for Mapping Ecosystem Services. JNCC, Peterborough.
- Milcu, A.I., Hanspach, J., Abson, D., Fischer, J., 2013. Cultural ecosystem services: a literature review and prospects for future research. *Ecol. Soc.* 18.
- Mulder, C., Bennett, E.M., Bohan, D.A., Bonkowski, M., Carpenter, S.R., Chalmers, R., Cramer, W., et al., 2015. 10 years later: revisiting priorities for science and society a decade after the millennium ecosystem assessment. In: Woodward, G., Bohan, D.A. (Eds.), *Ecosystem Services: From Biodiversity to Society*, Pt 1, vol. 53. Elsevier Academic Press Inc., San Diego, pp. 1–53.
- Norton, L.R., Inwood, H., Crowe, A., Baker, A., 2012. Trialling a method to quantify the 'cultural services' of the English landscape using Countryside Survey data. *Land Use Policy* 29, 449–455.
- O'Neill, J., 1993. *Ecology, Policy and Politics: Human Well-Being and the Natural World*. Routledge, pp. 181–182.
- Oleson, K.L.L., Barnes, M., Brander, L.M., Oliver, T.A., van Beek, I., Zafindrasilivonona, B., van Beukering, P., 2015. Cultural bequest values for ecosystem service flows among indigenous fishers: a discrete choice experiment validated with mixed methods. *Ecol. Econ.* 114, 104–116.
- Pascual, U., Muradian, R., 2010. The economics of valuing ecosystem services and biodiversity. In: Kumar, P. (Ed.), *Ecological and Economic Foundations*. Earthscan, Abingdon.
- Plieninger, T., Dijks, S., Oteros-Rozas, E., Bieling, C., 2013. Assessing, mapping, and quantifying cultural ecosystem services at community level. *Land Use Policy* 33, 118–129.
- Poppy, G.M., Chiotha, S., Eigenbrod, F., Harvey, C.A., Honzák, M., Hudson, M.D., Jarvis, A., et al., 2014. Food security in a perfect storm: using the ecosystem services framework to increase understanding. *Philos. Trans. R. Soc. Lond. B: Biol. Sci.* 369.
- Potschin, M.B., Haines-Young, R.H., 2011. Ecosystem services: exploring a geographical perspective. *Prog. Phys. Geogr.* 35, 575–594.
- Pröpper, M., Haupts, F., 2014. The culturality of ecosystem services. Emphasizing process and transformation. *Ecol. Econ.* 108, 28–35.
- Rathwell, K.J., Peterson, G.D., 2012. Connecting social networks with ecosystem services for watershed governance: a social-ecological network perspective highlights the critical role of bridging organizations. *Ecol. Soc.* 17.
- Raymond, C.M., Kenter, J.O., 2016. Transcendental values and the valuation and management of ecosystem services. *Ecosyst. Serv.* 21, 241–257.
- Raymond, C.M., Bryan, B.A., MacDonald, D.H., Cast, A., Strathearn, S., Grandgirard, A., Kalivas, T., 2009. Mapping community values for natural capital and ecosystem services. *Ecol. Econ.* 68, 1301–1315.
- Raymond, C.M., Kenter, J.O., Plieninger, T., Turner, N.J., Alexander, K.A., 2014. Comparing instrumental and deliberative paradigms underpinning the assessment of social values for cultural ecosystem services. *Ecol. Econ.* 107, 145–156.
- Reed, M.S., Kenter, J., Bonn, A., Broad, K., Burt, T.P., Fazey, I.R., Fraser, E.D.G., et al., 2013. Participatory scenario development for environmental management: a methodological framework illustrated with experience from the UK uplands. *J. Environ. Manage.* 128, 345–362.
- Reyers, B., Biggs, R., Cumming, G.S., Elmqvist, T., Hejnowicz, A.P., Polasky, S., 2013. Getting the measure of ecosystem services: a social-ecological approach. *Front. Ecol. Environ.* 11, 268–273.
- Rickebusch, S., Metzger, M.J., Xu, G., Vogiatzakis, I.N., Potts, S.G., Stirpe, M.T., Rounsevell, M.D.A., 2011. A qualitative method for the spatial and thematic downscaling of land-use change scenarios. *Environ. Sci. Policy* 14, 268–278.
- Satz, D., Gould, R.K., Chan, K.M.A., Guerry, A., Norton, B., Satterfield, T., Halpern, B.S., et al., 2013. The challenges of incorporating cultural ecosystem services into environmental assessment. *Ambio* 42, 675–684.
- Scholte, S.S.K., van Teeffelen, A.J.A., Verburg, P.H., 2015. Integrating socio-cultural perspectives into ecosystem service valuation: a review of concepts and methods. *Ecol. Econ.* 114, 67–78.
- Schwartz, S.H., Bardi, A., 2001. Value hierarchies across cultures: taking a similarities perspective. *J. Cross-Cult. Psychol.* 32, 268–290.
- Schwartz, S.H., 2012. An overview of the Schwartz theory of basic values. *Online Read. Psychol. Cult.* 2.
- Sherrouse, B.C., Clement, J.M., Semmens, D.J., 2011. A GIS application for assessing, mapping, and quantifying the social values of ecosystem services. *Appl. Geogr.* 31, 748–760.
- Spangenberg, J.H., Görg, C., Truong, D.T., Tekken, V., Bustamante, J.V., Settele, J., 2014a. Provision of ecosystem services is determined by human agency, not ecosystem functions. Four case studies. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manage.* 10, 40–53.
- Spangenberg, J.H., von Haaren, C., Settele, J., 2014b. The ecosystem service cascade: further developing the metaphor. Integrating societal processes to accommodate social processes and planning, and the case of bioenergy. *Ecol. Econ.* 104, 22–32.
- TEEB, 2010. *Ecological and Economic Foundations*. Earthscan, Abingdon.
- TEEB, 2016. *The Initiative* [Online]. Available at: <http://www.teebweb.org/about/the-initiative/> (Accessed: 26/04/2016).
- Tzanopoulos, J., Mouttet, R., Letourneau, A., Vogiatzakis, I.N., Potts, S.G., Henle, K., Mathevet, R., et al., 2013. Scale sensitivity of drivers of environmental change across Europe. *Glob. Environ. Change* 23, 167–178.
- UKNEA, 2011. *The UK National Ecosystem Assessment: Synthesis of the Key Findings*. UNEP-WCMC, Cambridge.
- Van Berkel, D.B., Verburg, P.H., 2014. Spatial quantification and valuation of cultural ecosystem services in an agricultural landscape. *Ecol. Indic.* 37, 163–174.
- Villamagna, A.M., Angermeier, P.L., Bennett, E.M., 2013. Capacity, pressure, demand, and flow: a conceptual framework for analyzing ecosystem service provision and delivery. *Ecol. Complex.* 15, 114–121.
- Villamagna, A.M., Mogollón, B., Angermeier, P.L., 2014. A multi-indicator framework for mapping cultural ecosystem services: the case of freshwater recreational fishing. *Ecol. Indic.* 45, 255–265.
- Weyland, F., Laterra, P., 2014. Recreation potential assessment at large spatial scales: a method based in the ecosystem services approach and landscape metrics. *Ecol. Indic.* 39, 34–43.
- Winthrop, R.H., 2014. The strange case of cultural services: limits of the ecosystem services paradigm. *Ecol. Econ.* 108, 208–214.
- Young, O.R., 2002. *The Institutional Dimensions of Environment Change: Fit, Interplay, and Scale*. Cambridge University Press, Cambridge, UK.