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RESEARCH ARTICLE

The need for integrated spatial assessments in ecosystem service mapping

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Abstract Publications on the modelling and mapping of ecosystem services (ESs) have increased exponentially in recent years. In this literature, a call for integrated environmental assessment is increasingly made, even if, in the ES context, the concept of integration remains fuzzy and can refer to different elements. First, this review paper attempts to clarify to which elements the word 'integration' is attributable in the ES literature. Integrated assessment can refer to the consideration of pluralistic values, the attention paid to several ESs and ecosystems, the accounting for multiple spatial and temporal scales, the implication of different stakeholders or the combination of techniques stemming from different disciplines. Second, this paper provides a review of the latest advances in the literature on mapping ESs, from the ecological to the economic perspective, in order to illustrate what can be done and what progress remains to be made to perform integrated and spatially explicit assessments of ESs. Third, this paper reviews examples of studies performing integrated assessments using the different meanings integration can take. Finally, it concludes by presenting the remaining challenges that research on this topic faces to perform fully integrated spatial assessments.

Keywords Economic valuation · Ecosystem service assessment · Ecosystem service mapping · Integrated assessment · Integrated valuation

1. Introduction

Ecosystem services (ESs) are situated at the interface between two spheres (Fig. 1). On the one hand, there is an ecological sphere represented by the structure and the functioning of the ecosystems that produce potentially used flows (supply side) that

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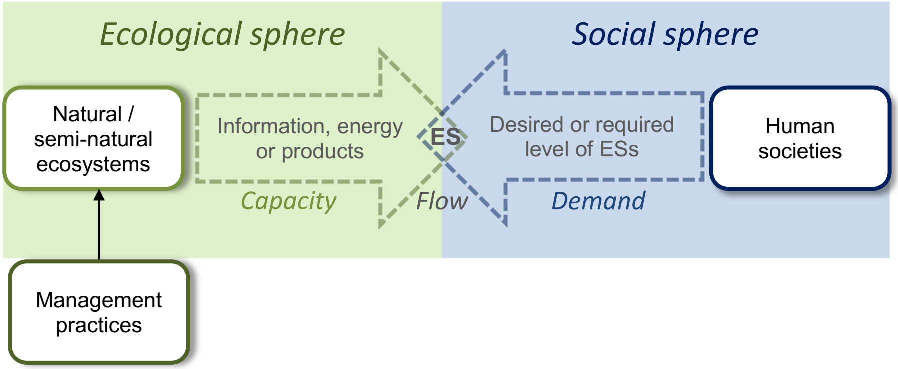

Fig. 1 Supply and demand sides of ecosystem services

may be influenced by management practices. This supply side can also be referred to as the ecosystem capacity to deliver ESs (Bastian et al. 2013; Villamagna et al. 2013; Baró et al. 2016). On the other hand, there is a social sphere represented by human societies and components likely to benefit from it (demand side). The demand is the level of ESs required by society (Villamagna et al. 2013), while the realised flow of satisfied demand represents the ESs.

The mapping ES literature aims to assess, in a spatially explicit way, the different processes involved in the ES flow delivery. Rapid scientific advances are underway in this field. Researchers from the involved disciplines (such as hydrology, ecology, agronomy, geography, sociology and economics) are developing models, mapping tools, open-source data sets, international or national projects and working groups (e.g. the Ecosystem Services Partnership,[1] the Natural Capital Project,[2] and Esmeralda [3]). Publications on mapping ESs have exponentially increased in recent years. According to Scopus, the subject now comprises approximately 769 peer-reviewed papers and books referenced in ScienceDirect and published between 1996 and 2015, containing 'Mapping ecosystem service' in their titles, abstracts or keywords. The majority of the papers are classified as life science (60%), mainly in environmental sciences, and a smaller part is classified as social science (31%). From a decision-making viewpoint, since the Millennium Ecosystem Assessment, the subject is also increasingly supported by public decision makers and government administrations (IPBES,[4] EFESE in France, [5] MAES,[6] UKNEA[7] and the WAVES[8] Partnership) since Action 5 of the EU Biodiversity Strategy to 2020 calls on member states to assess and map the state of ecosystems and their services in their national territory. Private businesses and industries also have an increasing interest in this type of assessment, which could be useful to

[1] http://www.fsd.nl/esp/79222/5/0/5
[2] http://www.naturalcapitalproject.org/
[3] http://www.esmeralda-project.eu/
[4] http://www.ipbes.net/
[5] Evaluation Française des Ecosystèmes et Services Ecosystémiques http://www.fondationbiodiversite.fr/fr/societe/avec-la-societe/appui-a-la-decision/appui-a-l-evaluation/efese.html
[6] EU initiative on Mapping and Assessment of Ecosystems and their Services http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/index_en.htm
[7] United Kingdom National Ecosystem Assessment: http://uknea.unep-wcmc.org/
[8] Wealth Accounting and the Valuation of Ecosystem Services: https://www.wavespartnership.org/

evaluate business opportunities and reduce risks for companies whose operations rely on ESs (TEEB 2012; Tardieu and Crossman 2017).

The need for integrated assessment in ES mapping is increasingly acknowledged (Bockstael et al. 2000; Carpenter et al. 2009; Eigenbrod et al. 2010). Indeed, it is now widely recognised that a narrow perspective can yield misleading information/ recommendations for management planning about issues with broad systemic effects such as ESs (Martín-López et al. 2014; Wicke et al. 2015; Pascual et al. 2017). Integration means combining one element with another to form a coherent whole (Gómez-Baggethun et al. 2014). As Jakeman and Letcher (2003) argue, discussing integration may lead to many different issues since elements may be different. Integration in the case of ES modelling and mapping may refer to the following:

– Integrated treatment of *values*: a comprehensive methodological approach in which biophysical, sociocultural and monetary value domains are explicitly considered and their relationship to each other is analysed
– Integration across *ESs* and *ecosystems*: when multiple ESs supplied by multiple ecosystems are considered and can be produced in a synergistic or antagonistic way
– Integration with *stakeholders*: implication of stakeholders in the assessment process
– Integration of *disciplines*: combining modelling techniques from different disciplines, e.g. combining supply-side and demand-side assessments
– Integration of *scales*: considering ESs at various spatial and temporal scales

Obviously, different types of integration are not mutually exclusive. For instance, integrated assessment of environmental, social and economic values may require the integration of modelling techniques from different disciplines that may function on different spatial scales (Kelly et al. 2013).

Different literature reviews have been made available in scientific books or series of special issues concerning ES mapping (Kareiva et al. 2011; Burkhard et al. 2013; Crossman et al. 2013; Willemen et al. 2015; Maes et al. 2016; Burkhard and Maes 2017). However, reviews are mainly focused on 'one discipline' assessment techniques, and none of them to date has focused on advances or research needs for integrated assessments of ES changes. This review paper attempts to fill this gap by presenting more recent practices in ES mapping, from the ecological to the economic perspective, and the state of the art of integrated assessments. The paper particularly focuses on the following questions: (1) What are the methodological approaches currently used? (2) What are the advances in integrated ES assessments? (3) What are remaining gaps to be filled that would allow us to comprehend the multiple dimensions of ESs?

The remainder of this paper is organised as follows: Section 2 presents the spatial characteristics of ESs and particular issues reviewed in mapping ES literature. Section 3 presents the current practices and methodologies in the spatial assessment of ESs in order to illustrate what can be done and what progress remains to be made to perform integrated and spatially explicit assessments of ESs. Section 4 presents selected examples of the integrated assessment of ESs. Finally, Section 5 concludes by presenting the remaining research avenues needed for integrated assessments that better support land management decisions.

2. Why map ecosystem services?

2.1. Spatial variation of ecosystem services

Services depend not only on whether or not they are potentially supplied in a specific location but also on whether people are benefiting from these ES flows (Mandle and Tallis 2016). The levels of ES supply, flow and demand are influenced by several geographical components, and their extent is affected by the different scales at which they occur. The characterisation of the supply side and the demand side based on the cascade framework (Haines-Young and Potschin 2010) is the conceptual framework generally used in the mapping and modelling literature (for a review, see Maes et al. 2012 and Maes et al. 2016). In this framework, natural or managed ecosystems (e.g. agroecosystems) provide the necessary structures and processes that underpin ecosystem functions. ESs are derived from ecosystem functions and represent the realised flow of services in relation to the benefits and values of the people who use them.

On the supply side, biophysical structures and ecosystem functions can vary due to ecosystem type, land use, land configuration, climate variables, hydrology, soil conditions, fauna and topography (Nelson et al. 2009; Burkhard et al. 2012). In managed ecosystems, they also vary by land use practices (agro-ecosystems, forests or urban parks).[9] On the demand side, ES demand depends on the number of beneficiaries and their location (e.g. urban proximity or the presence of infrastructures), socio-economic context (income, gross domestic product per capita), preferences and social practices. Socio-economic characteristics of beneficiaries are not explicitly spatial variables per se, but differences between beneficiaries can be defined from a spatial point of view (Brander et al. 2012).

Benefits derived from beneficiaries, thus, vary with all these factors, making willingness to pay (WTP) spatially heterogeneous (Schaafsma et al. 2013; Czajkowski et al. 2016). Issues of scarcity may also be underlined. Microeconomic foundations state that the incremental value granted to an abundant good or service decreases when quantity increases (as marginal utility declines). Conversely, a scarce demanded good may hold a high marginal value, and this value will increase when the quantity decreases. In our case, an abundance of ESs occurs when the ecosystem is large or when surrounding ecosystems supply the same service, constituting substitutes. This dimension implies that the reduction in the area of a large ecosystem (or one that is abundant in the area) that provides a particular service induces a lower loss than the reduction in the area of a small ecosystem (or one that is relatively scarce in the area).

When changes are modelled by scenarios, they are observed in an overlapping way for each process, function, service or demand and in terms of social welfare. Assessing the changes in a spatially explicit way requires the definition of biophysical structures and process flows in the initial conditions in order to model ecological/biophysical changes in ecosystem functions and in service provision in terms of quantity or quality after the policy implementation, to identify the changes for potential beneficiaries

[9] In agro-ecosystems for instance, farmers may benefit from ESs (ESs are, thus, production factors like other anthropogenic inputs), but they may also co-produce services to the society (Swinton et al. 2007; Zhang et al. 2007). ESs flowing *to* and flowing *from* managed ecosystems then have to be distinguished even if they are both influenced by land use practices. A typology of ES benefits derived by farmers and ESs co-produced by farmers to the society can be found in Zhang et al. (2007) or Garbach et al. (2014).

according to the spatial context and socio-economic characteristics (travel distance, substitutes) and to assess the welfare change in accordance with the biophysical and social contexts. This process requires a substantial amount of data at every step. Moreover, as Carpenter et al. (2009) argue, it is rare to find a linear causal path from processes to human well-being or from feedbacks to drivers. As a result and as Boyd (2008) argues, to determine ES values, three things really matter: location, location, location. Similar to the value of a house, the value derived from ESs will depend on its quality and on its neighbourhood, even though the importance of location clearly depends on the type of ES being considered. Indeed, ESs occur at different spatial scales, and their consideration is now recognised as an important issue in the valuation process.

2.2. Ecosystem service spatial scales

Ecosystems themselves vary in spatial scales since they can have the shape of small individual patches, large continuous areas or regional networks. Services delivered by 'service providers' are also generated at a range of ecological scales, and organisational levels for service(s) production can vary from populations of single species to ecological communities (Luck et al. 2009). It has therefore become common practice to distinguish spatially defined *ecological* scales such as the global scale, the landscape scale or the plot plant before the assessment of a particular service. The most relevant ecological scale per ES has been identified, adapted from Hein et al. (2006), and is presented in Table 1.

As for the service delivery, ES demand can occur within a range of socio-economic scales. The assessment of a change requires identification of the scale and of the beneficiaries of the systems' services. For locally demanded services, it is considered in the literature that there is a spatial limit outside of which individuals no longer benefit from the service. When goods are local and ordinary, some studies have proven that they are distance-dependent, presenting distance decay (Bateman et al. 2006;

Table 1 Most relevant ecological scales for ecosystem services (adapted from Hein et al. 2006). Note that some services may be relevant at more than one scale

Ecological scale	Dimension (km^2)	Services
Global	> 1,000,000	Global climate regulation through carbon sequestration and storage; regulation of albedo, temperature and rainfall patterns
Landscape	10,000–1,000,000	Freshwater provisioning
Biome, watershed	1–10,000	Flood regulation; regulation of water flows; regulation of erosion and sedimentation; regulation of species reproduction; nutrient retention; hunting recreation; pollination; aesthetic information and recreational activities
Ecosystem or plot plant	< 1	Food and raw material provisioning; air quality regulation; local climate regulation; waste treatment; biological control; freshwater fishing recreation

Schaafsma et al. 2012, 2013). Distance decay highlights two important aspects: (1) It delineates the ES demand zone, i.e. the distance from which the ES will no longer be demanded, and (2) it indicates the spatial decay rate at which WTP declines when distance increases. Many studies have proven that distance decay may nevertheless depend on the direction in which it is being measured (Cameron 2006; Schaafsma et al. 2012). Distance decay is explained by different factors. The first factor is an effect of knowledge, assuming that individuals living near a site are more likely to have knowledge about it than those further from a site. A high degree of interest about the states and level of influence of ecosystem services on the environmental decision-making process also has an impact on the scale of individuals' demands, as recently shown by García-Nieto et al. (2015). Moreover, this is explained by the fact that the more distant a site is, the greater is the increase in costs to reach it, and the substitute sites' availability, with decreasing net benefits, also needs to be considered (Bateman et al. 2006; Hanley et al. 2003). Hence, this applies in particular to the direct use of goods and services. Indirect uses do not necessarily imply a travel cost. However, the benefits derived from these services can be local depending on the extent of the service delivery and on the proximity of the beneficiaries.

The spatial extent of the demand for non-uses is assumed to be greater than that for uses because it does not require travel for beneficiaries. This is particularly true for exceptional goods and services (existence of symbolic species, national parks), which can be considered as global (Costanza 2008). Indeed, in that case, goods are globally known and do not have any or very few substitutes. They are considered to be distance-independent.

To conceptualise the relationship between the supply side and the demand side, we have to imagine two overlaid maps (Brander et al. 2012). The first one represents the spatial extent of the capacity of the service-providing unit (SPU) to deliver an ES in a given quantity and quality, according to its ecological and spatial conditions and management. The other one represents the spatial extent of potential beneficiaries given their preferences, the distance to the environmental good and the spatial context, indicating the service-benefiting area (SBA) (Bagstad et al. 2013a; Palomo et al. 2013). Service-providing units and service-benefiting areas can overlap or be connected by flows. Flow directions between supply and demand also differ according to the service considered. For instance, water-provisioning services rely on watershed functions and farmers benefit from the service at stream access points, whereas in the case of recreational services, beneficiaries may have to travel to the ecosystem that presents a recreational potential. In some cases, the service is delivered and enjoyed at the same location (e.g. a home's viewshed).

Finally, when land use changes occur, impacts on ES capacity, flows and demand can be observed at different scales. Non-linear and abrupt changes may occur. For instance, Tardieu et al. (2015) addressed indirect impacts on some ESs caused by transport infrastructure construction.

2.3. Mapping ESs for communication and decision-making

Representing ESs in maps makes it possible to present complex information in a pedagogical way. It provides intuitive methods for communicating multiple types of information among different stakeholders: scientists, policymakers, resource managers, project designers and citizens. However, spatial assessments with ES maps are strongly dependent on the question addressed by researchers. In the literature, studies can be divided into two types of analysis (Verburg et al. 2008; Polasky and Segerson 2009).

The first type of analysis is aimed at designing solutions for specific issues by calculating the optimal land use allocation given a set of objectives, for example, the optimisation of ES delivery (Polasky et al. 2008). This is a *normative* type of analysis. ES maps can identify, in a spatially explicit way, potentially forgone benefits (opportunity costs) incurred by a public or business decision that impact land and water resources. This makes it possible to spatially target locations with comparative advantages for investment, i.e. the one that provides the greatest returns at the least cost (Crossman and Bryan 2009; Naidoo and Ricketts 2006). Issues such as comparison of the joint generation of multiple ESs in a region vs. the specialisation in one ES can also be analysed (Ruijs et al. 2015). This may improve the efficient use of limited funds by targeting areas with a particular synergy between environmental and social dimensions ('win-win' areas) from a conservation perspective.

The second type of analysis aims to provide information on existing situations, or on present/future land use alternatives, by producing assessments in a *positive* type of analysis. This is principally done by producing ES maps of ecological, social or economic value that is gained or lost under different scenarios. This first helps in rendering the spatial repartition of ES values visible to society. Further, it informs resource managers and policymakers in several ways:

- Pinpointing ES hotspots or cold spots to take better advantage of ES benefits (Egoh et al. 2008; Rodríguez et al. 2015)
- Monitoring the consequences of different public or business investment strategies or determining new opportunities for public/business investments
- Reducing risks for companies that depend on ESs (Tardieu and Crossman 2017)
- Identifying areas with ES benefits specific to economic sectors (e.g. tourism) or identifying ES use across stakeholder groups (Darvill and Lindo 2015)

More broadly, ES maps can improve existing evaluation tools commonly used in the public and private sectors, such as environmental impact assessments, life cycle assessments, risk assessments, cost-benefit analysis, land use plans and off-site mitigation plans (for a general review and recent advances, refer to Geneletti 2016).

The type of analysis will be dependent on the kind of scenario used in different studies. In the context of ES assessment, a small but increasing number of studies consider scenarios (Seppelt et al. 2011; Landuyt et al. 2016). Constructing scenarios makes it possible to model ES delivery on the basis of different land distributions, used as a predictor, compared to the business-as-usual (BAU) scenario. Scenarios determine the future amount of land that will be assigned to each type of land use given different drivers, including policy and land management drivers (Verburg et al. 2008; Kareiva et al. 2011; Bateman et al. 2014). Multiple land use change models have been used, including ruled-based GIS such as LCM for Geneletti (2013), CLUE for Verburg et al. (2014), IMAGE and GLOBIO for Alkemade et al. (2009) and historical changes for Chakir and Le Gallo (2013). Scenarios can also be combined with climate change models (e.g. IPCC scenarios) or with impacts caused by invasive alien species (Schröter et al. 2005). Normative scenarios can also be constructed showing which desirable future can be achieved (Brunner et al. 2016). Uncertainty can be included in the scenarios. Partial insights are obtained through sensitivity analysis, as in Johnson et al. (2012). Uncertainty can also be accounted for in outcomes to alternative land use configurations (Landuyt et al. 2016).

3. Mapping ecosystem services: current approaches in the environmental and social sciences

Different disciplines can be involved in ES modelling and mapping: the life sciences (ecology and agronomy, hydrology and geography) and the social and human sciences (sociology and economics). The supply side is generally represented using geographical information systems, remote sensing data and different ecological models to generate spatially explicit maps (Polasky et al. 2005; Egoh et al. 2008; Naidoo et al. 2008; Nelson et al. 2009; Kareiva et al. 2011). The demand side is principally represented using social valuation methods and economic valuation techniques in order to elicit and spatially differentiate sociocultural and economic values (Costanza et al. 1997; Eade and Moran 1996; Bateman et al. 1999; Kreuter et al. 2001; Troy and Wilson 2006; García-Nieto et al. 2013, 2015).

For all of the approaches, the precision level and representativeness of maps vary depending on data availability, data quality, methodologies, the models developed, and the scope of the study (precision increases when the study scale becomes more local). Study precision varies according to the underlying biophysical data, land use and land cover typology (Global Land Cover, Corine Land Cover, among others), the accuracy and robustness of economic values, and the consideration of spatial variation (Kandziora et al. 2013).

3.1. Mapping ecosystem service supply and associated opportunity costs

3.1.1. Ecosystem service supply in biophysical terms

Four main techniques can be distinguished for modelling potential ES supply in biophysical terms. They differ in terms of the level of precision and the data requirement.

The first technique is a qualitative assessment based on expert opinion, professional judgement and rankings used when primary data for the study region are not available (Baral et al. 2013). Qualitative assessments mainly use participatory mapping tools and expert views converted into indicators representing professional judgements on ES conditions and temporal trends. Qualitative indicators (such as high, moderate or low provision of ESs and increasing, decreasing or stable trends) are used and transferred into GIS to produce maps (Haines-Young et al. 2012; Scolozzi and Geneletti 2012). However, the results of these analyses are criticised because of their subjectivity since they depend on the knowledge and experience of the experts for a particular landscape.

The second technique is based on ES one-dimensional proxy (e.g. tons of carbon per hectare, tons of timber per hectare). There are many different kinds of ESs, and different metrics are therefore used to monitor them (for a review on ES indicators, see De Groot et al. 2010). Provisioning services are the easiest to map with representative data because they are directly quantifiable (particularly for raw material provisioning), and most of the time, these data are readily available in national statistics (Maes et al. 2012). However, metrics for other services are lacking and are likely to be less reliable. Regulation and cultural services are less directly quantifiable, and ecosystem capacities have to be approached by proxies such as ecosystem components (environmental and spatial data, information on habitats, biodiversity, etc.) (UNEP-WCMC 2011). Proxies are mainly used for large-scale assessments. More recent approaches, namely, downscaling or dasymetric mapping, are used to study the ES supply (Verkerk et al. 2015). This method is based on non-precise data on a service proxy (at the administrative or national scale) that is then related to land use and land cover data, biophysical variables etc. The statistical relationship between the proxy and the other variables is then used to predict the capacity of each cell to deliver the service in high resolution (Carré et al. 2009).

The third approach is used when representative data for time and location are unavailable or when service delivery depends on multiple ecosystem functions (as is the case for ESs supplied at the landscape scale). These techniques rely on ecological models based on GIS tools.

Deterministic models are based on an ecological production function which aim to model the ES output supplied by an ecosystem given its conditions and processes (Naidoo et al. 2008; Nelson et al. 2009; Kareiva et al. 2011). An ecological production function specifies the potential ES outputs that are provided by an ecosystem. It uses georeferenced data (e.g. land use and cover raster, digital elevation models, soil depths, potential evapotranspiration) and environmental information (e.g. plant-available water content or nesting habitats for pollinators) to assess ES supply in biophysical terms. Once production functions are specified, it is possible to quantify the impact of a particular policy on the ES delivery (Boyd 2008; Nelson et al. 2009; Polasky et al. 2011; Daily et al. 2011). Changes are evaluated using future scenarios (potential future states of the natural environment) of land use changes (Swetnam et al. 2011) or by comparison to historical references. Probabilistic models, by contrast, recognise that random behaviour can be part of a system. Rather than using single values as input parameters, probability distribution functions are used (e.g. Barraquand and Martinet 2011).

The use of Bayesian belief networks is increasing because of their particular accuracy when data sets are incomplete or have a high degree of uncertainty (Marcot et al. 2006; Villa et al. 2014; Balbi et al. 2015). Bayesian models rely on causal networks that link parent and child nodes. Nodes have assigned probabilities of occurrence (in the absence of real data) and prior probabilities that are conditional. Probabilities are generally assigned according to expert knowledge or based on the literature. When real data are available, they replace prior and conditional probabilities. In Fig. 2, a very simple Bayesian belief network is presented as an example. In this diagram, an ecological function is explained by two factors, A and B. The likelihood that a particular state of the ecological function is reached is represented by a probability. As Carpenter et al. (2009) and Baveye et al. (2016) argue, the use of probabilities makes it possible to use uncertain input variables and to address uncertain causal relationships. Uncertainties spread into the network and influence model outputs, making it possible to present probability maps that can be particularly useful for decision makers (see, for example, Landuyt et al. 2015).

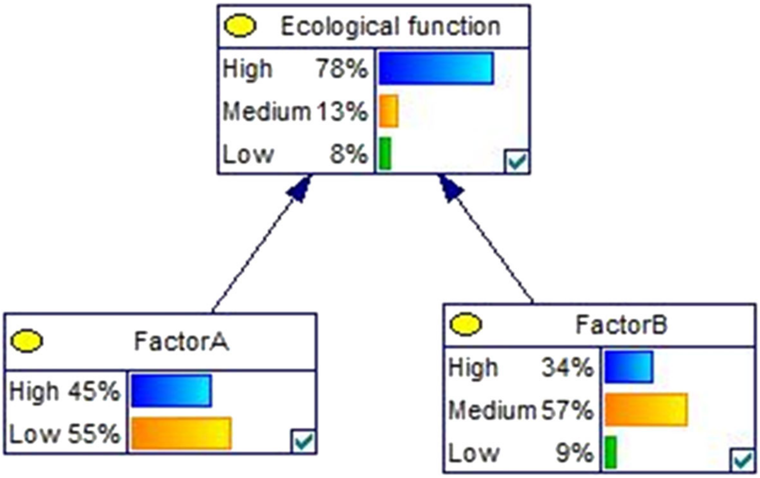

Fig. 2 Example of a simple Bayesian belief network (developed with Genie software)

Different mapping tools have been developed to help practitioners, including ARIES and InVEST.[10] InVEST is based on the use of ecological production function models (Nelson et al. 2009; Tallis and Polasky 2011). ARIES can use deterministic models, but the use of Bayesian belief networks is also possible.

The last mapping technique of ES supply in biophysical terms is based on biological data such as functional traits within species groups or ecosystem structures such as plant heights, leaf dry matter contents or other detailed biological data (Luck et al. 2009; Lavorel et al. 2011). For the most accurate maps, these can be based on primary data, measurements and observations from remote sensing or field studies. Such precise models principally aim to study the relationship between biodiversity and services or trade-offs among services.

Because biophysical maps may be sufficient, economic valuation is not applied when government agencies look for environmental standards. Knowing how ESs will change in biophysical terms is informative because it makes it possible to visualise hotspots or cold spots of potential ES supplies (e.g. overlap analysis as in Häyhä et al. 2015) and to statistically analyse ES trade-offs and/or synergies (e.g. by using Pearson’s correlation as in García-Nieto et al. 2013). However, when these changes are not measured in monetary terms such as costs or benefits, models may not give full weight to decisions because we cannot make comparisons with other costs and benefits. In this case, it can be very useful to combine biophysical, social and economic valuation methods to estimate and report the monetary value of the change in ecosystem service quality or quantity.

[10] ARIES: Artificial Intelligence for Ecosystem Services. For more information on the tool, refer to Villa et al. (2009). InVEST: Integrated Valuation of Ecosystem Services and Trade-offs (Kareiva et al. 2011). For a mapping tool comparison, the reader can refer to Bagstad et al. (2013b, c).

3.1.2. Representing opportunity costs associated with ecosystem service supply

Recent studies now map ES supply values, i.e. the cost associated with service provision for land managers (forgone benefits from land use known as opportunity cost). These models are used to analyse spatial variations of opportunity costs that are not necessarily similar to benefits (Murdoch et al. 2007; Schröter et al. 2014a, b). These approaches are mainly applied to manage ecosystems such as agricultural land or forestry (Polasky et al. 2008; Ruijs et al. 2015; Pennington et al. 2017). One option to derive opportunity costs is to use transformation functions representing the possible combinations of ESs generated in a region. This makes it possible to represent the trade-offs between different land uses, contingent on the production function shape at each point. Transformation functions can be empirically estimated using a two-step, semi-parametric distance function as in Ruijs et al. (2015). They are based on biophysical data providing state-of-the-art data on interrelationships between land uses, agricultural productions and ES synergies or substitutability. Transformation functions are then used to represent the different relationships between production, ES interactions and the effects of spatial differences of biotic and abiotic characteristics. It can be noted here that there is this implicit assumption that the marketed service is the baseline (the value of the reduced provision of timber service vs. increasing carbon storage and sequestration). Within a general framework, it will then be more important to consider trade-offs between services.

3.2. Mapping ecosystem service demand and associated benefits

3.2.1. Mapping ecosystem service demand and sociocultural values

In comparison to ES supply, a small number of studies analyse and map ES demand, and this constitutes one of the main research gaps in the ES modelling and mapping literature. The conceptualisation and the definition of demand remain fuzzy and can differ among studies. A major difference is whether demand is conceptualised as the direct use/consumption of an ES or as the required level by society (Wolff et al. 2015; Baró et al. 2016). Microeconomic foundations would define the demand only as the required/desired level of ESs, depending on individual preferences (utility) and budget constraints. The demand can thus be satisfied or unsatisfied depending on the level of ES supply. The satisfied part of the demand is thus the realised flow of ES between capacity and demand (Villamagna et al. 2013; Martín-López et al. 2014; Schröter et al. 2014a, b; Brunner et al. 2016; Egarter Vigl et al. 2017). A brief summary of mapping demand techniques is given in Table 2.

According to Wolff et al. (2015), most studies on demand mapping focus on regulating services, followed by recreation and eco-tourism services. For provisioning services, the demand is sometimes assumed to be equal to the supply (or capacity). This is the case when the capacity is directly converted into monetary terms (e.g. through the use of market price). Many studies agree that other cultural ESs are still often neglected in ES assessments (Darvill and Lindo 2015; Baró et al. 2016; Pascual et al. 2017). This can be explained by the difficulties of basing this type of ES on biophysical proxies and of formalising cultural benefits, even if some methodologies and subjective well-being indicators are developed (e.g. Bryce et al. 2016), including ‘engagement with nature’,

Table 2 Brief overview of available methods for mapping ES demand (based on Brown and Fagerholm (2015) and Wolff et al. (2015))

Method	Description
Public participatory GIS (PPGIS)/participatory GIS (PGIS)	Spatially explicit method that aims to capture spatial information in participatory planning processes that can involve different types of stakeholders (users, experts, decision makers). It uses random sampling methods/purposive sampling (through interviews, work-shops or surveys) and digital/non-digital mapping technology.
Volunteered geographic information	Geographic data provided voluntarily by individuals by reporting georeferenced locations, points of interest etc. (e.g. geotagged images through social media platforms)
Process-based approaches	Process-based models are based on the theoretical understanding of ecological processes and are mostly used to quantify regulating services (e.g. dependency of agricultural production on pollination).
Empirical methods	Uses real-world observations or those obtained through surveys, based on a demand indicator of different locations (e.g. fishing licences, recreational visit counts)

'place identity' and 'therapeutic value'. An emerging literature indeed recognises the sociocultural values representing the direct or indirect contribution of ESs to the user's cultural identity and heritage, spiritual values or good social relationships (Gómez-Baggethun et al. 2014; Martín-López et al. 2014).

Studies that map ES demand first aim to identify service beneficiaries and then aim to represent the importance of ESs for these groups. This is mainly done with participatory GIS mapping techniques (public participatory GIS, participatory GIS, volunteered geographic information) (see Brown and Fagerholm 2015 for a review). The technologies used are digital mapping and non-digital mapping, where sampled participants (through interviews, workshops or surveys) are asked to mark by points or polygons where they particularly benefit from ESs. The importance is revealed through indicators representing the importance that users attribute to ESs in a value-elicitation context, usually ranging from 0 to 1 or from 0 to 5 (de Groot et al. 2010; Burkhard et al. 2012; García-Nieto et al. 2013, 2015; Martín-López et al. 2014; Bryce et al. 2016). Demand indicators can also be assessed by making assumptions about the proximity to the service-providing units, e.g. proximity of an agricultural area to hedges for local climate regulation or population density and spatial distribution of NO_2 concentrations (Baró et al. 2016). This type of study is generally used to analyse the spatial distribution of ESs and flows between service-providing units and service-benefiting areas (Syrbe and Walz 2012; Bagstad et al. 2013a; García-Nieto et al. 2013; Palomo et al. 2013). It can also be used to analyse which groups of stakeholders particularly benefit from an ES or to reach social equity objectives (Pascual et al. 2014; García-Nieto et al. 2015; Darvill and Lindo 2015).

Second, ESs are quantified and mapped. This can be done through all of the mapping techniques described in Table 2. Process-based models are implemented for demand related to services directly dependent on ecological processes (such as pollination, erosion control, global climate regulation, flood control or water demand). Demand is also assessed through empirical methods (surveys relying on a demand indicator, such as fishing licences). These methods are usually integrated as a step in the monetary valuation techniques that specifically quantify demand (e.g. travel cost method quantifying the number of visits; see Section 3.2.2).

3.2.2. Mapping well-being changes and associated economic values

1. Primary economic valuation techniques Different techniques are used to map economic values associated with human well-being variations.[11] Until recently, assessments were rarely described in a spatial manner, and values were aggregated across local or large areas without the ability to determine where individuals were benefitting from the service. The first applications of economic value mapping were conducted in the 1990s (Eade and Moran 1996; Bateman et al. 1999; Kreuter et al. 2001). Since then, the number of publications on the mapping of ES economic values has grown exponentially, with almost 60% published after 2007 (Schägner et al. 2013). The quality of studies varies depending on the underlying biophysical data, the accuracy and robustness of the economic assessment and the consideration of spatial context (Eigenbrod et al. 2010).

Different primary valuation techniques have been developed to assess beneficiaries' WTP for non-market goods and services (see Table 3). Clearly, some valuation methods may be more suitable for capturing the values of different elements of the total economic value (Gómez-Baggethun et al. 2010). For example, market price and cost approaches are more generally used to assess provisioning services and the majority of regulation services. Revealed preference techniques might be more suitable to capture use values such as recreation (e.g. the travel cost method which uses information about the costs incurred travelling to a biodiversity-rich area to assess the recreation value of that area) (Navrud and Mungatana 1994; Shrestha et al. 2002) or landscape amenity values measured with hedonic pricing techniques. Stated preference techniques would be more suited to capture non-use values. For example, the contingent valuation method or choice modelling can be used to assess how much people are willing to pay for a biodiversity protection programme (Nunes and van den Bergh 2001).

Spatial patterns of WTP in primary valuation techniques are generally studied using spatial econometrics and GIS data to construct variables that describe local environmental characteristics, such as views on amenities, distance to road, and share of land use. Initially, such approaches were principally used in hedonic pricing models, but they are now applied to various economic valuation techniques (Panduro and Veie 2013; Schläpfer et al. 2015). For instance, Czajkowski et al. (2016) incorporate GIS data into a discrete choice experiment, Baerenklau (2010) and Baerenklau et al. (2010) incorporate the data into a travel cost approach, and Jørgensen et al. (2013) use GIS data as an explanatory variable in a contingent valuation study.

Spatial statistical approaches are mostly used to address unobserved heterogeneity in the population and to demonstrate spatial relationships and interactions that explain WTP spatial heterogeneity. Campbell et al. (2008) and Lee and Schuett (2014) applied Moran's *I* statistic and confirmed significant global spatial clustering of the demand and

[11] In economics, the concept of a change is fundamental to defining values. Economists measure the value of a change from a baseline (pre-policy) level of some price or quantity variable to an alternative (post-policy) level (Polasky and Segerson 2009).

Table 3 Available economic techniques for non-market valuation

Economic valuation techniques	Description
Market prices	Changes in ES quality or quantity are valued by using directly observed market prices from actual markets as a proxy for the value.
Market cost approaches	
Avoided damage costs	Uses the costs associated with the mitigation of environmental damage as the proxy for the value
Replacement costs	Uses costs of replacing an environmental service as a proxy for the value
Opportunity costs	Explicitly considers the value that is lost in order to protect, enhance or create a particular environmental asset
Production function	Focuses on the (indirect) input costs of a particular environmental service for the production of a marketed good
Revealed preferences	
Travel cost method	Uses data on people's actual behaviour in real markets that are related to the environmental good. The behaviours studied are the number and distribution of trips that people make to outdoor recreation sites as a function of the cost of a trip. The travel cost is the weak complement (a complementary marketed good) of the outdoor recreation value.
Hedonic pricing	Weak complementarity is assumed between the price of a property and the quality of the surrounding environment. The non-market value is revealed through observations on the demand of residential properties.
Stated preferences	
Contingent valuation	Estimates values by constructing a hypothetical market and asking survey respondents to directly report their willingness to pay to obtain a specified good or their willingness to accept giving up a good
Choice modelling	Based on a hypothetical market where respondents have a series of choice tasks in which they are asked to choose their preferred option (including status quo). Each option is described in terms of a set of attributes describing the good (including a price attribute) presented at various levels according to an experimental design. The analysis of the respondents' choices is based on random utility maximising (RUM) theory.

WTP estimates and that these estimates exhibited positive spatial auto-correlation, even over relatively large spatial areas.

2. Secondary economic valuation techniques: benefit transfer Collecting new data and conducting primary valuations for multiple services are likely to be costly and time-consuming. Therefore, methodological approaches for applying original valuation results in other spatial policy and decision-making contexts, usually referred to as benefit transfers, are increasingly developed and tested. Four transfer techniques are used in the mapping literature, showing increasing complexity and data requirement.

The simplest transfers are made by applying estimates from study sites *per unit area* and *per ecosystem type*, found in the economic literature, to the same ecosystem types in the application site (Troy and Wilson 2006; Mendoza-González et al. 2012). Ecosystem types are located with land use/land cover typologies. In some cases, the values transferred are expressed as a function of the total economic value per ecosystem type rather than estimates of values per individual service, impeding the analysis of how the provision and value of each ES will change under different conditions. Plummer (2009) highlighted that errors generated by this unit value transfer are likely to be high due to generalisation errors. First, errors can be attributed to the extrapolation of economic values between different sites. Differences may concern social, demographic or economic information or differences in markets and substitutes (Loomis and Rosenberger 2006). In addition, errors can be made by considering the spatial constancy of biophysical measurements (Bateman et al. 2011). This approach indeed assumes that the ES delivery is spatially homogeneous across ecosystem types and, consequently, that every hectare of a given ecosystem is of equal value, regardless of flows, ecosystem spatial configuration, size or quality (Nelson and Kennedy 2009).

Values transferred can also be expressed *per physical unit* (e.g. euro/ton of carbon), relying on a unidimensional proxy of ESs stemming from national statistics or ecological models. This makes it possible to account for the reality of ES supply in the application site that specifies the supply conditions underlying the service flow. However, it does not account for beneficiaries nor for the supply effects on demand (presence of substitutes, marginal value or threshold effects). Nevertheless, this method can be suitable for some regulating or provisioning services. Many studies have mapped ESs using this method (Naidoo and Ricketts 2006; Abson et al. 2014).

In the value transfer *function* (third transfer technique), the WTP function assessed with primary valuation techniques of the study site is applied to parameter values of the application site to assess ES values. Bateman et al. (2011) suggest that many value function transfer attempts have failed because they used ad hoc, empirically driven specifications of utility functions that fit the study site data well but that appear to be over-parameterised when they are applied outside of the sample framework to application sites. Due to the multiplicative role of coefficients, this type of parameterisation can result in major transfer errors.

Finally, the fourth transfer technique is the *meta-analytic function* which can be estimated based on multiple studies and transferred to the application site. This approach accounts for differences in results and explanatory variables in relevant studies valuing a particular ES in order to estimate a WTP function for the service. It requires collecting available accurate studies that assess the service and then coding the study characteristics in terms of WTP estimates, the non-market technique used, study site and population characteristics. A regression model is estimated from these data, with WTP per unit (for a particular base year) as the dependent variable and the study site characteristics, methodological attributes and socio-economic variables as the independent variables. The meta-regression function is then used to predict welfare estimates in the application site by inserting levels of the independent variables that describe the policy site. This approach is now mainly used to scale up ES values. Scaling up consists of using the existing values of local ecosystem services for an assessment of these values at a larger geographical scale: regional, national or global (Brouwer et al. 1999; Brander et al. 2012; Woodward and Wui 2001). A difficulty in using this method is the multitude of original studies that may differ in at least three ways: (1) in range (changes from reference to target levels), (2) in spatial and temporal scales, and (3) in the number of explanatory variables that may affect the suitability of including these studies in the meta-analysis. In addition, opportunities to scale up values for land use management decisions are reduced because this technique does not work for applications in multiple ESs and ecosystem types that include all spatial variables. Finally, evidence from the literature (Smith and Pattanayak 2002; Brander et al. 2012) shows that potentially large transfer errors exist and that in some cases, the simple transfer of unit values may be as effective as this less parsimonious model.

3.3. Summary of the different approaches

To sum up this section, the main current approaches are presented in Fig. 3. Assessments linking biophysical data and economic data are increasingly applied (link between the green box and the blue box). As stated in Section 3.2.1, the demand-side assessment is still neglected in the ES mapping literature (orange box). So far, the link between the supply side approaches expressed in monetary terms (pink box) and demand side expressed in monetary terms (blue box) has not been investigated, even if it would make it possible to assess how ES supply changes due to a land use change in monetary terms and how people value these changes (Ruijs et al. 2015).

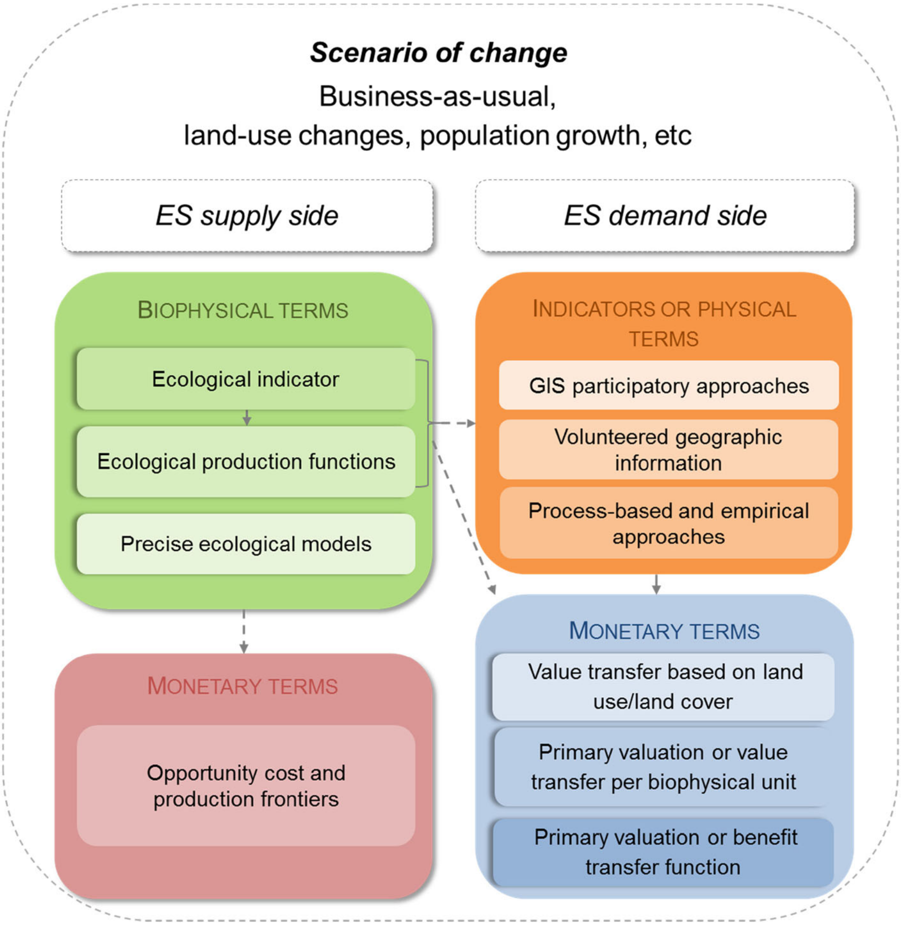

Fig. 3 Main current approaches for mapping ES supply and demand under a scenario of change

4. Selected examples of integrated assessments for ecosystem service mapping

According to Scopus,[12] roughly 120 papers mentioning integrated assessments and ES mapping have been published between 1996 and 2017, with 46 publications in 2016 and 2017. This is thus a new topic of research as shown in Fig. 4, and as for ES mapping, the majority of the papers are classified as life science (81%) and a smaller part is classified as social science (19%). This section presents selected examples of integrated assessments (across value domains and disciplines, scales, services and stakeholders) in order to illustrate what is meant by integration in the different publications and what is the value added in each case.

1. Integrated treatment of values This type of integration is probably the most reported case in the literature of integrated assessments. There is now a formal recognition of multiple values in ecosystem services, and the literature on ES valuation increasingly stresses the importance of integrating ecological, sociocultural, and monetary aspects of ES dimensions for decision-making (Saarikoski et al. 2016, Pascual et al. 2017). According to Gómez-Baggethun et al. (2014), there are multiple values that may be, in principle, equally fundamental but that can be in conflict with each other. Value pluralism recognition is thus required to capture the diversity of needs and wants that ecosystems can contribute to fulfil benefits to society and individuals. Therefore, multiple and often conflicting values may be combined to inform decision makers instead of the result of a reduction of information onto a single metric (Gómez-Baggethun and Barton 2013). Martín-López et al. (2014), for instance, considered a plurality of values. To do this, they first assessed the ecosystem capacity to deliver ESs from a biophysical perspective. Further, they assessed the demand of ESs from a sociocultural viewpoint, analysing the importance that people give to particular ESs. Finally, they assessed the demand of ESs using monetary valuation techniques. This allowed the authors to assess whether these different value domains (i.e. biophysical, sociocultural and monetary) provide similar information regarding ES assessment. The results show that the methods revealed different ES trade-offs. This confirms the conflict that can emerge across different values and thus the need for plural valuations to inform decisions (García-Nieto et al. 2015; Baró et al. 2016).

2. Integration of disciplines This integration requires combining techniques from different disciplines to assess the level of supply and demand of different ESs. Namely, here are examples of studies using biological/ecological models combined with economic valuation techniques.

In their study, Naidoo and Ricketts (2006) spatially evaluated opportunity costs and benefits of conservation for a landscape in the Atlantic forests of Paraguay. Opportunity costs of conservation are defined as the expected agricultural value of each forested parcel of land multiplied by the probability that a given parcel would be converted.

[12] The search entered in Scopus was '(TITLE-ABS-KEY (mapping AND ecosystem AND services) OR TITLE-ABS-KEY (mapping AND ecosystem AND service) AND TITLE-ABS-KEY (integrated) OR TITLE-ABS-KEY (integrated AND valuation) OR TITLE-ABS-KEY (integrated AND assessment))'. The same search results to 92 papers in Web of Science ISI Web of knowledge.

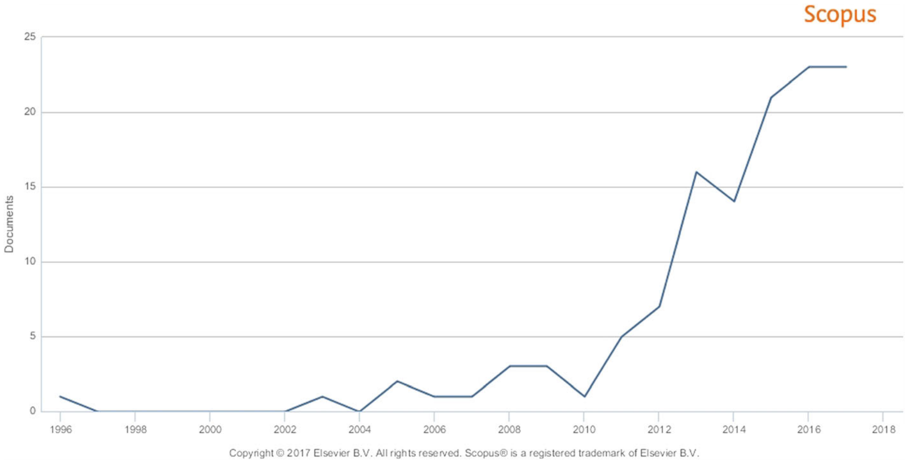

Fig. 4 Number of studies that mapped ESs with integrated assessment approaches from 1996 to 2017

Benefits derived from ESs were calculated through biological models (such as habitat associations of game species for bushmeat harvest) and economic data (essentially, market prices and benefit transfer). Along the same lines, Nelson et al. (2009), Polasky et al. (2011) and Kareiva et al. (2011) modelled ES supply (carbon sequestration and storage, water quality, agricultural production and timber production) with a detailed ecological production function (InVEST tool) and applied benefit transfer or market prices to biophysical quantities across different land use scenarios.

The UK NEA (UK National Ecosystem Assessment, 2011) valued changes in ESs across six different scenarios for the UK (world markets, nature at work, go with the flow, green and pleasant land, local stewardship and national security) (Bateman et al. 2014). They modelled, monetised and mapped three services across the UK: carbon storage (Abson et al. 2014), outdoor recreation (Sen et al. 2014) and urban green-space amenity combining ecological and economic techniques (Perino et al. 2014).

3. Integration across spatial or temporal scales As explained in Section 2.2, mapping of ESs is fundamentally related to the topic of scales (Dick et al. 2014). The spatial scale considered should be chosen in relation to the subject of interest (local, regional, national, global). However, in some cases, the consideration of multiple spatial scales[13] is of importance when the problem or objectives intrinsically require a multi-scale approach to detect some trade-offs and conflicts. The same applies to different temporal scales (short, medium, long terms). Obviously, multi-scaling, meaning doing a study at several scales simultaneously, raises concerns in terms of data availability. As a response to this barrier, Grêt-Regamey et al. (2015a) proposed a tiered approach providing information about relevant variables to be considered in long-term monitoring at different scales. They show that the number of variables needed and their specification depend on the empirical or policy question under investigation and this can be determined by conducting a meta-analysis of the scientific literature on the topic.

[13] Here, the scales described should be distinguished from the map resolution. For an assessment of the effect of resolution on ES assessments, see Grêt-Regamey et al. (2015b).

Brunner et al. (2016) proposed an interdisciplinary method with a backcasting approach to infer land use policy strategies that match the regional supply and demand for ESs over a long time horizon. Backcasting refers to the creation of a future normative scenario and then looking back to identify strategies to achieve this desired future. In their study, they first assessed future demand for ESs with a discrete choice experiment that elicited the preferences of local residents for changes in ESs. They further used formative scenario analysis to define socio-economic and political boundary settings. They followed by simulating land use change with corresponding changes in the supply of the targeted ESs under various land use policy strategies using an economic agent-based land use model. Finally, for each model run, they evaluated how well ES demand was satisfied at the planning horizon.

4. Integration across ESs and ecosystems To understand interconnections between ESs and ecosystems, it may be necessary to combine the models assessing different ESs. Balbi et al. (2015) used the ARIES tool to develop different spatially explicit models for different ESs (crop yields, water supply, climate regulation and air quality) that communicate between each other. This helps obtain a better understanding of ES trade-offs under different management practices and climate change scenarios. This approach provides a more holistic view because it does not function model by model but, instead, functions in a synchronised way. It makes it possible to overcome the service-by-service modelling approach traditionally applied in ES assessment which implicitly considers no relationships between ESs.

5. Integration with stakeholders ES assessment and mapping can be fed by the knowledge of different stakeholders: the research community, practitioners, local actors and indigenous or peasant communities (Tengö et al. 2014). The benefits of involving stakeholders during the process are multiple.

First, the involvement of stakeholders in the assessment process can result in a better understanding of ES and biodiversity dynamics. Second, it provides the opportunity to elicit stakeholders' diverse value perspectives on ESs, which may be critical to reach sustainable outcomes of any decision (Cabral et al. 2016, Pascual et al. 2017). Reyers et al. (2009), for instance, quantified the local-scale consequences of land cover change for ESs by involving different stakeholders in the process (government departments, landowners, non-governmental organisations, municipalities, business representatives and researchers). This allowed them to produce ES maps in a region of data scarcity by harnessing their local or subject-specific knowledge. Furthermore, it helped to collect information on stakeholders' needs and proved to be a useful method for fostering agreement on the future changes in land use activities necessary for a sustainable path of the area.

Third, this allows for developing useful outputs for practitioners to adapt the methodology and the shape of the output results to their particular issues. Tardieu (2016) and Tardieu et al. (2015) expanded the case of ES consideration in the implementation of choices related to transport infrastructure projects. They showed that consideration of ESs can be useful at different stages of transportation projects: (1) during preliminary studies to design the first avoidance measures, (2) in the detailed environmental impact assessment to compare implementation options, and (3) during the cost-benefit analysis to provide indicative evaluations that help to improve the guidance of the decision maker. ES consideration in this case led to new challenges beyond the methodological ones that appeared in mapping exercises. Replicability and usefulness for stakeholders were particularly targeted; issues such as the funnel characteristic of the project processes that progressively define the study area, data availability or adaptation to the legal framework were then considered. To do this, they constructed the methodological framework for and with stakeholders and applied it to a real project.

5. Discussion and future research prospects

ES mapping is becoming an important and powerful tool for the mainstream use of ESs in day-to-day land management decisions. The topic is receiving increasing attention, and many difficulties are progressively being overcome. This paper attempts to provide a review of the latest advances in the literature on ES mapping, from the ecological to the economic perspective, and aims to highlight the progresses made in integrating spatially explicit assessments by giving examples found in the literature. Integration in the ES modelling and mapping literature may refer to different aspects, and each type of integration results in a supplementary comprehension of issues related to the ES concept:

– Integrated treatment of *values* results in a broader perspective of which meaning and importance people and societies ascribe to nature and natural capital. This promotes the different conceptualisations of values and reveals new trade-offs when values appear to be in conflict (e.g. ecological, economic and sociocultural). A pluralistic valuation approach, thus, widens the information upon which decisions are to be made.
– Integration of *disciplines* is the communication between models from different disciplines that give a better understanding of the connection between ES supply and demand. The outputs from ecological models can be directly used to assess the marginal benefit changes in different scenarios.
– Integration of *scales* is the consideration of ESs at various spatial scales that allow for taking into account the off-site effects when local changes (e.g. local environmental policy) affect the flow of distant ESs. The assessment of different temporal scales provides a better perspective of the sequential impacts of scenarios on ES supply and demand.
– Integration across *ESs* and *ecosystems* provides a better understanding of the relationships between various ESs and gives a more holistic view of the side effects of actions impacting various ESs.
– Finally, integration with *stakeholders* not only provides different knowledge for a better understanding of ES dynamics in the study area but also provides the different values and views stakeholders attach to ESs. This also helps to precisely identify issues to be solved, to design methodology and to determine the accurate shape of the output to be produced.

Integrated assessments are therefore more likely to produce realistic results for ESs considered under different scenarios. However, many challenges of producing accurate ES assessments still remain associated with integration and are described hereafter.

A first barrier is the potential non-compatibility between biophysical assessment outputs and inputs required in social assessments and economic valuation techniques. The difficulty of combining across *disciplines* occurs when outputs/inputs of models are non-combinable because they are expressed in different terms or because they have disparate scales. In this case, patching together models may lack relevancy (Carpenter et al. 2009). Combination may also be difficult because of a difference in objectives. For instance, biophysical models assess the service potentially supplied, and economic valuation techniques represent the benefit/loss derived from the change in the service effectively rendered based on individuals' behaviours. Indicators may thus fail to accurately represent the ES flow. In some cases, the difficulty is directly inherent to economic techniques because of the lack of capacity to integrate these variations under some valuation techniques (e.g. unit value transfer, cost and market price approaches). Perfectly functioning markets are thus implicitly considered. This often leads studies to neglect the spatial considerations of beneficiaries.

Interactions and feedback between the economy and the environment are also important issues that are currently under-investigated and that can be seen as another form of integration. When determining economic values, physical changes must be specified, as well as the trade-offs people make by taking into account the different interactions in the economic system. Interactions can be observed between services as well as between complementary market goods, significantly altering the values associated with changes in ESs. One promising new approach is to apply general equilibrium analysis instead of considering that changes are small enough that they can be independent of relative price adjustments (Carbone and Kerry Smith 2013). However, to my knowledge, there is no empirical study applying this type of analysis, probably because of the amount of data it may require.

Furthermore, the shifting points and critical states of ecosystems are remaining challenges that merit deep exploration by the environmental and social sciences to produce accurate ES assessments. Integration here can be described as the consideration of multiple changes or cumulative impacts providing a more inclusive view of ESs or ecosystems under threat (Dunford et al. 2015). In environmental science, the challenge is to clearly identify this shifting point at which a large change in different ES supply may be observed. On the economic side, valuation is only possible when the change in a service is marginal. The determination of the threshold from which ecosystems shift into a critical state can be crucial for policy recommendations (Tardieu et al. 2015). However, economic valuation cannot properly address these cases.

In the end, integration means broadening the vision of ES analysis and mapping in multiple ways that can be very useful to guide decision-making or increasing awareness. In an ideal world, ES assessments should then be as exhaustive as possible. However, performing integrated assessments may require more time and data and is not necessarily accurate for providing relevant information to practitioners and planners affected by a policy process. The type(s) of integration(s) to perform and the required degree of integration depend on the assessment purpose, on the stakeholders' needs and on the policy setting.

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