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**Title:**

# Integration of ecosystem services into a conceptual spatial planning framework based on a landscape ecology perspective

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**Abstract:**

Context: The study of ecosystem services has extended its influence into spatial planning and landscape ecology, the integration of which can offer an opportunity to enhance the saliency, credibility, and legitimacy of landscape ecology in spatial planning issues.

Objectives: This paper presents a conceptual framework suitable for spatial planning in human dominated environments supported by landscape ecological thinking. It seeks to facilitate the integration of ecosystem services into current practice, including landscape metrics as suitable indicators.

Methods: A literature review supported the revision of existing open questions pertaining to ecosystem services as well as their integration into landscape ecology and spatial planning. A posterior reflection of the current state-of-the-art was then used as a basis for developing the spatial planning conceptual framework.

Results and conclusion: The framework is articulated around four phases (characterisation, assessment, design, and monitoring) and three concepts (character, service, and value). It advocates integration of public participation, consideration of “landscape services”, the inclusion of ecosystem disservices, and the use of landscape metrics for qualitative assessment of services. As a result, the framework looks to enhance spatial planning practice by providing: i) a better consideration of landscape configuration in the supply of services ii) the integration of anthropogenic services with ecosystem services; iii) the consideration of costs derived from ecosystems (e.g. disservices); and iv) an aid to the understanding of ecosystem services terminology for spatial planning professionals and decision makers.

**Keywords :**

Ecosystem Services, Landscape Services, Landscape Metrics, Stakeholder Engagement Techniques, Spatial Planning, Nature-based Solutions.

## 1. Introduction

The Millennium Ecosystem Assessment (Alcamo et al., 2003; MEA, 2005) promoted the concept of ecosystem services (ES) due to its capacity to relate nature-human interactions and show the relevance of nature for the maintenance of the main components of human well-being such as health, basic materials, and security. ES is already a central subject in the conservation biology and environmental science disciplines (de Groot, Wilson and Boumans, 2002; Wallace, 2007; Busch *et al.*, 2012), and it is now being integrated in many other disciplines, including spatial planning and landscape ecology (Harris & Tewdwr-Jones, 2010; Wu, 2012; Geneletti, 2015). However, while this integration is accelerating, general aspects of the concept are still being debated, such as:

- The classification of ES and a precise definition of basic concepts (ecosystem functions, services, benefits and goods): Should there be one type of ES classification only, or different classifications depending upon the application (Wallace, 2007; Costanza, 2008; Fisher and Turner, 2008; Frank *et al.*, 2012)? Where are the boundaries between the basic concepts? Should we account for the final ecosystem services only or also intermediate services (Wallace, 2007; Balmford *et al.*, 2008; Costanza, 2008; de Groot *et al.*, 2010a)? What units are appropriate for ES accounting (Limburg et al., 2002; Balmford et al., 2008)?

- Are ES adequate as a stand-alone concept to facilitate communication with different practitioners and policy-makers, particularly when considering other emerging concepts, such as nature-based solutions (Potschin et al., 2015; Shackleton et al., 2016; Schaubroeck, 2017)?

ES can advance current approaches and paradigms in the field of spatial planning and

landscape ecology, but some disciplinary-specific issues need to be addressed. For example, the integration of ES into current spatial planning practice could facilitate the on-going transition from incremental-advocacy approaches towards adaptive-consensual ones (see definitions in Briassoulis, 1989). Whereas the inclusion of ES in landscape ecology could enhance the saliency, credibility and legitimacy of the discipline in societal issues (Nassauer and Opdam, 2008).

Several researchers advocate the expansion of the landscape ecology paradigm, and stress the need for a more applied focus to make the discipline a suitable basis for the resolution of spatial planning issues (Bastian, 2001; Opdam et al., 2001; Wu, 2006; Termorshuizen et al., 2007; Termorshuizen and Opdam, 2009; Opdam, 2010). The role of ES in this domain includes:

- The identification of the relationship between landscape structure (character), functions, and provision of ES; and how to account for this qualitatively and/or quantitatively (de Groot *et al.*, 2010a; Busch *et al.*, 2012; Syrbe and Walz, 2012).
- The definition of ecosystem services and their values in a spatially explicit manner (de Groot *et al.*, 2010a).
- The identification of ES parameter-proxies and tools that can be understood easily by a diverse group of professionals and for different problems, scales, and contexts (de Groot *et al.*, 2010a).
- An increase of design-driven perspectives in landscape ecology, applicable also to ES studies (Nassauer and Opdam, 2008).
- A consistent integration of stakeholders' perception and values in ES studies to facilitate

collaborative approaches in the development of policy, urban/landscape plans, and project alternatives (Frank et al., 2012; Koschke et al., 2012).

In this paper, we present a conceptual framework for spatial planning, in which we address some of the general and discipline-specific ES issues presented above. As part of the framework the link between ES, landscape character (structure) and values (benefits) is made more evident to spatial planning professionals and decision makers, showing that changes in one element may affect the others. This requires a revision of the concept of ES to adapt it to the holistic character of spatial planning and strengthen its interrelation with other key concepts. Concurrently, traditional ES discourse is extended incorporating costs (disservices) and anthropogenic services.

## 2. Reflection on general ES issues

### 2.1. Differences in ES categorisation and basic concepts (function, services, goods and benefits)

Several ES classification systems exist (Haines-Young & Potschin, 2017). Some well-known examples are: the Millennium Ecosystem Assessment classification (MEA), The Economics of Ecosystems and Biodiversity (TEEB), the Common International Classification of Ecosystem Services (CICES), the UK National Ecosystem Assessment (UKNEA), and the US National Ecosystem Services Classifications Systems (NESCS). The diversity in classifications is related to differences in frameworks, disciplinary approaches, and definition of the basic concepts (EPA, 2015; Haines-Young and Potschin, 2017) and are designed to better suit different purposes (Haines-Young & Potschin, 2014; Haines-Young & Potschin, 2017; Heink, Hauck, Jax, & Sukopp, 2016). However, the development of a common and rigorous ES classification and a clear

differentiation of the basic concepts would improve the operationalisation of ES assessments.

Among the proposed classification systems, CICES has been extensively used by scientists and policy makers to define and map ES indicators (Haines-Young and Potschin, 2017; La Notte et al., 2017). Moreover, this classification framework was initially proposed by the European Environment Agency (EEA) and developed for the System of Integrated Environmental and Economic Accounting (SEEA), and is currently employed within the Mapping and Assessment of Ecosystem Services (MAES) reports (Haines-Young & Potschin, 2014; Haines-Young & Potschin, 2017). A straightforward comparison between CICES and the MEA and TEEB classifications is explicitly provided in the last CICES version (5.1), which can help to harmonise results from different studies.

Regarding a clear differentiation of ecosystem function, services, goods and benefits, La Notte et al (2017) proposed a re-interpretation of the "cascade model" of Haines-Young and Potschin (2010), which they applied to CICES to enhance operationalisation of the ES categorisation. An ecosystem function is defined as the set of interactions among components (biotic and abiotic) of ecosystems or biophysical structures, which may affect one or more ES. ES are defined as flows (e.g. carbon sequestration, water purification) generated by ecosystems, as a result of ecological processes and exchanges of information (e.g. genetic information, visual appreciation of natural features). Goods are represented by countable mass units and marketable resources (e.g. amount of biomass extracted from forest ecosystems, or fish resources) and the benefits as the contribution of these goods to a positive change in human well-being (e.g. availability of cleaner air or water). More details about the definition of ES can be found in Potschin and Haines-Young (2011) and La Notte et

al. (2017).

2.2. A lack of a common ES accounting unit

Several studies exist that translate ES into economic values or integrate them into systems of economic accounting (Boyd and Banzhaf, 2007; Balmford *et al.*, 2008; de Groot *et al.*, 2010b; UKNEA, 2011, 2013; SEEA, 2012). These studies address issues, such as double counting, trade-offs, and the establishment of economic values for ES without markets (Balmford et al., 2008; Fisher et al., 2009; SEEA, 2012). Success in solving these problems can allow the use of comprehensive cost-benefit analyses during the decision making process (Busch *et al.*, 2012) and will permit the establishment of clear relationships between economic activities and ecosystem functioning (Haines-Young, Potschin, & Kienast, 2012).

Despite the benefits of transferring ES into monetary units, this valuation is limited, with respect to its assumptions and scope, and only partially expresses the value of services (Limburg *et al.*, 2002; de Groot *et al.*, 2010a; Seppelt *et al.*, 2011). The inclusion of ecological and socio-cultural values is recommended as part of the valuation (de Groot *et al.*, 2010a) integrating the three value-domains considered by the MEA (2005). Nevertheless, it is difficult to find universal units by which to account for and aggregate multiple ES ecological and social values, and the use of several different units is required (Kumar and Kumar, 2008; Chan et al., 2012; Martín-López et al., 2014; Scholte et al., 2015).

For ecological values, biophysical capacity units are usually mentioned (Castro et al., 2014; Martín-López et al., 2014). The CICES classification explicitly enables users to integrate biotic and abiotic categories into the same ES assessment framework, allowing the accounting of

ecological values of intermediate ES flows that operate alongside more basic ecological structures and processes (or “supporting services”) to underpin the output of final services (Haines-Young & Potschin, 2018).

Aggregated public perception is often used (Brown, 2013; Scholte et al., 2015) to support the development of indices to express social values (Sherrouse et al., 2011), such as Quality of Life indices (QoL) (Fleury-Bahi et al., 2012; Hassine et al., 2014). The contribution of ES to final QoL might account for social values as a common social unit. By making use of objective and subjective indicators, QoL surveys can account for a variety of ES in specific socio-cultural contexts and diverse types of well-being: i) related to resources; ii) objective satisfaction of people with those resources; iii) and subjective perception of satisfaction (King et al., 2014).

In spite of recent initiatives promoted to address ES definitions and classification systems, (e.g., TEEB, 2011; Landers and Nahlik, 2013; Munns *et al.*, 2015; Haines-Young and Potschin, 2018) a common unit for ES accounting of ecological and social values is still missing. Firstly, ES vary in typology and beneficiaries (at the level of society and biodiversity components), making it difficult to develop a harmonized assessment framework based on common reference metrics. Secondly, while some ES can easily be quantified (e.g. cultivated terrestrial plants for nutrition purposes) and their values monitored over time, others (e.g. maintenance of nursery populations and habitats) are more difficult to value. Therefore, the use of multiple metrics based on *a priori* identification and definition of the ES beneficiaries is recommended.

### 2.3. ES as a stand-alone concept: revision and complementary concepts

Environmental studies that are based only on the quantification and assessment of ES might

suffer a limited understanding by stakeholders, especially those with a non-technical background. In fact, this issue was identified by Davies *et al.* (2017) as one of the main constraints of applying an ecosystem services approach to the management of urban forests. In addition, only assessing ES might offer a partial view or skew valuations, ignoring costs related to ecosystems (Lyytimäki and Sipilä, 2009). This opens up the question as to whether additional concepts could mitigate these two issues.

The recent concept of Nature-based Solutions (NBS) defined as actions inspired and supported by nature (Bauduceau et al., 2015), which act as an umbrella for other nature-related concepts such as green infrastructure of ecosystem-based approaches (Potschin et al., 2015), might contribute to solving the communication issue. In this sense, NBS could be used as an easy and effective way to enhance the active participation of non-technical stakeholders in assessments of different kinds of socio-ecosystems, since it is an easier concept to grasp by non-technical stakeholders (Eggermont et al., 2015). Additionally, defining the relationships between NBS and ES could permit a better evaluation of alternatives (e.g. ES trade-offs) when implementing different NBS (Nesshover et al., 2017).

With respect to the potential skew in evaluations, some authors propose that acknowledgement of ecosystem disservices, such as an increase of leaf litter, damage to paving caused by tree roots, and allergic reactions to pollen emissions, and not only ES when valuing ecosystems (Shackleton et al., 2016; Schaubroeck, 2017), allows a more comprehensive balance of costs and benefits. The consideration of disservices is especially relevant for valuations in urban contexts (von Döhren and Haase, 2015). In addition, ES categorisations relate only to services provided by biotic (but sometimes abiotic) features.

They do not account for services depending on anthropogenic processes or structures, or acknowledge the contribution of human effort for the delivery of certain services (Maes et al., 2013), which could be critical in human dominated environments.

Moreover, in many ES studies the role of space and spatial interactions between ecosystems and between ES and the people using them are not well considered (e.g. Cortinovis and Geneletti, 2017). However, these interactions could be very relevant in multifunctional fragmented landscapes strongly influenced by humans (Termorshuizen and Opdam, 2009). The provision of ES tends to be only associated with individual ecosystems or land cover patches. This is why several authors advocate the use of the term "landscape services" (Koschke et al., 2012; Termorshuizen & Opdam, 2009; Wu, 2013), explicitly taking into account the role of spatial configuration.

## 3. ES integration and disciplinary-specific spatial planning and landscape ecology issues

### 3.1. Landscape structure, functions, services, and values: relationships, spatial definition, and tools

Landscape, as a physical structure, is defined as a dynamic integrated entity composed of abiotic, biotic and anthropogenic components in continuous evolution (Bertrand and Tricart, 1968; Bolós, 1992; Bastian, 2001). According to the landscape ecology paradigm, landscape patterns are interrelated with landscape functions (Wu and Hobbs, 2002; Schröder and Seppelt, 2006; Termorshuizen and Opdam, 2009; Wu, 2013). Since services are dependent on ecosystem (landscape) functions, they are also inherently dependent on landscape patterns.

Therefore, changes in landscape structure can be also related to changes in ecosystem (landscape) services, and their economic, social and ecological values (Termorshuizen and Opdam, 2009).

A spatially explicit definition of ecological (landscape) functions, services and values, can be achieved by making use of indicators, relatively simple ecology and landscape ecology tools or more complex techniques. Simple tools such as a look-up matrix (Burkhard et al., 2012; Koschke et al., 2012) could help to qualitatively relate and easily visualise the land cover composition of an area as to its capacity to provide ES. Landscape metrics are used to assess how spatial composition and pattern configuration affect changes in ecological processes (Lustig et al., 2015; Borges et al., 2017), to evaluate the supply of ES (Feld et al., 2007; Frank et al., 2012; Syrbe and Walz, 2012), and their social values (Fagerholm and Käyhkö, 2009; Sherrouse et al., 2011). Other techniques such as system dynamics modelling, landscape genetics, or agent-based modelling are also being applied to provide a spatial understanding of ecological functions, services and values (Grimm et al., 2005; Coulon et al., 2015; Etherington, 2016; Turner et al., 2016).

### 3.2. An increase of design-driven perspectives in ES studies

Initially, ES initiatives and programmes have focused on the policy level and only address ES during assessment phases (e.g. Maynard, James and Davidson, 2010; UN and FAO, 2014; Choi *et al.*, 2017). Very few initiatives rigorously consider ES in planning and design stages from design-driven perspectives (e.g. Ahern, Cilliers and Niemelä, 2014). In this sense, Nassauer and Opdam (2008) propose an evolution of the landscape ecology paradigm towards "pattern:process:design", stressing collaboration between scientists and practitioners and

removing the gap between landscape ecology and design.

Ecological design approaches have improved environmental performance of buildings or sites, but these approaches have not yet substantially advanced the enhancement of ecological processes, and their derived services, at landscape level (Nassauer and Opdam, 2008). Researchers know that patterns and processes are also interrelated in urban systems, but they still do not know exactly how changes of patterns and processes affect each other (Alberti, 2016) or the amount of ES delivered or demanded.

### 3.3. Integration of stakeholders' perception and values in ES studies

In spatial planning research, several studies have focused on developing and applying stakeholders' analysis and engagement techniques (Karl et al., 2007; Busquets and Cortina, 2009; Prell et al., 2009; Reed et al., 2009; Ruiz-Frau et al., 2011; Susskind et al., 2012). Stakeholder analysis techniques (e.g. snow ball sampling, interest influence-matrix, social network analysis) are focused on the analysis and diagnosis stages, where stakeholders are identified, categorised, and their interests and influence on others understood. Regarding stakeholder engagement techniques, some are more suitable for assessment stages, improving understanding, and surveying stakeholders' perceptions in a spatial format (e.g. public participatory mapping), whilst others seek to improve communication and collaboration of stakeholder groups during decision making (e.g. mediation techniques, collaborative adaptive management).

The relevance of stakeholders' perception is also being considered in ES studies, building on the advances in practical decision making and applied research. For example, the

UKNEA(2011, 2013) includes some of the previous techniques to integrate stakeholders in decision making, since these techniques could ease a spatially explicit social valuation of ES. As an example, Public Participation GIS (i.e. a type of public participatory mapping), is already being applied in ES studies (Brown and Fagerholm, 2014) and tested on initiatives such as the PPGIS tool developed by ADAS (2017) for Natural England as a way to understand which areas people perceive as more valuable in terms of ES.

**4. Conceptual Framework**

The proposed framework is articulated around the conceptualisation of landscape made by Bertrand and Tricart (1968), the reinterpretation of the cascade model of La Notte et al (2017), the expanded pattern:process:design paradigm of Nassauer and Opdam (2008), and the integration of stakeholders' perceptions. As a consequence, the concept of landscape character, services and values and their interrelationships become essential. A revision of suitable landscape science, public participation and assessment techniques were used for each phase of the framework (characterisation, assessment, design, and monitoring) and their interim stages aligned with general phases of spatial planning practice (Figure 1).

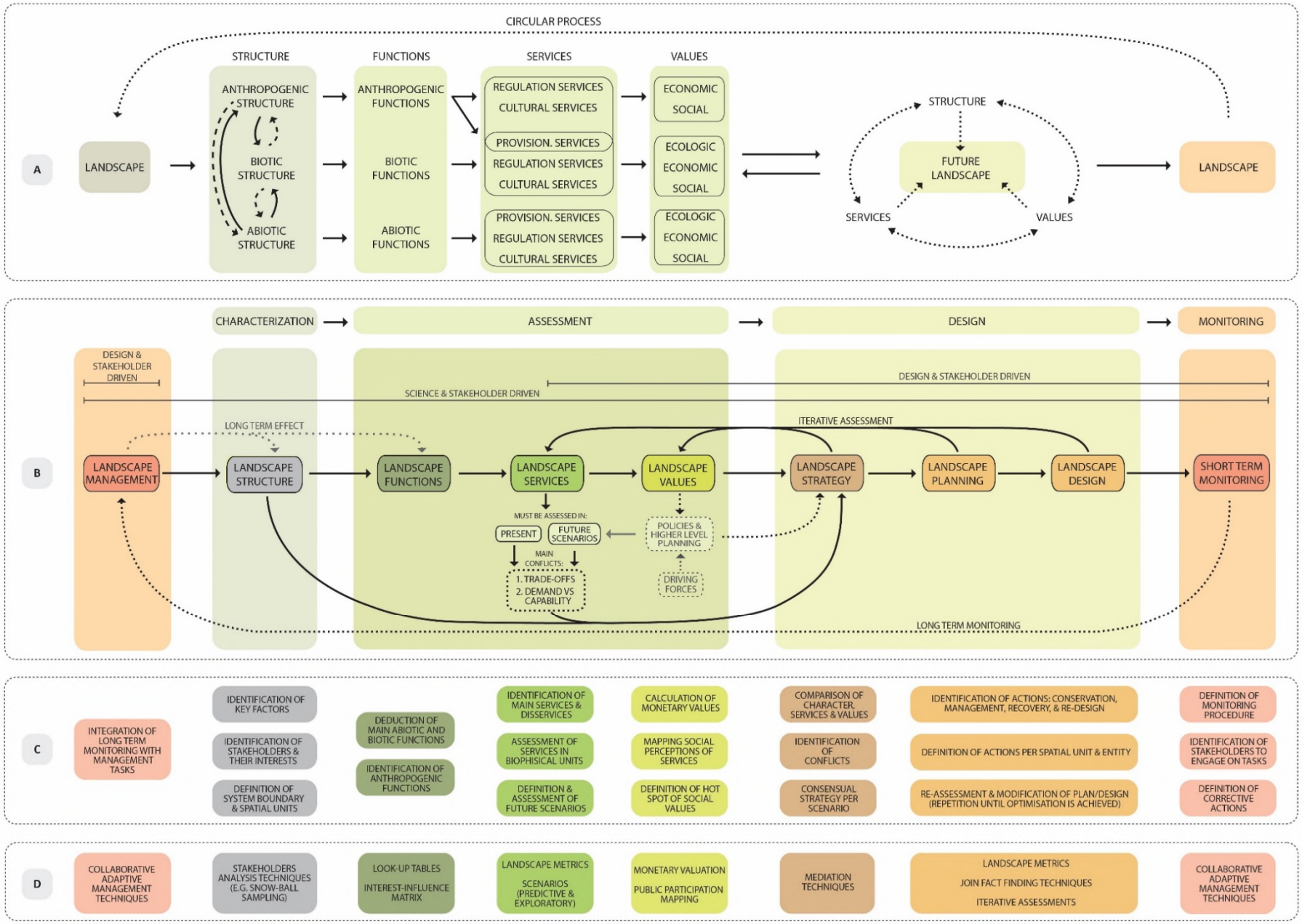



Figure 1. Conceptual Framework. A) Conceptualisation; B) Framework C) Main Tasks of Phases; D) Key Techniques

4.1. Characterisation

The characterisation phase is based on the landscape evaluation framework of Bolós (1992) and European frameworks of landscape character assessment (Brunet-Vinck, 2004; Sala, 2007; Tudor, 2014). Initially, local stakeholders are identified, categorised by their interests and influence on others, and their perceptions and future aspirations are investigated by making use of stakeholder analysis techniques. The spatial system boundary and irregular spatial units are defined together, taking into account the purpose of the plan, dimension (scale), and limits of the area of intervention. The spatial units should be delimitated as coherent areas of landscape structure (landscape character areas in Tudor (2014)) with respect to anthropogenic, biotic, and abiotic factors of interest. During the delimitation, the integration of stakeholders would facilitate identification of social structures or functional dependencies invisible to experts (Sala, 2009). The spatial system boundary should include the landscape character areas inside the area of intervention plus adjacent spatial units sharing strong functional dependencies (social and/or ecological). This characterisation may ensure the relevance of the scale of observation to the processes studied, and minimise mismatches of scale by considering the social and ecological functions together during the system boundary definition (Bergsten et al., 2014).

4.3. Assessment

The assessment phase builds on the work of La Notte et al (2017), their differentiation of the basic concepts (functions, services, values), and the use of CICES for the categorisation of services. However, the importance of landscape configuration in the provision of services is reinforced (Termorshuizen and Opdam, 2009) and ES accounting is extended to abiotic and anthropogenic services under the broader concept of landscape services. Additionally, the

potential disservices as defined by Campagne, Roche and Salles (2018) are included. The concepts of ecosystem functions and values are also substituted by landscape functions and values.

Consistently with the cascade model, this phase is divided into three stages: functions, services, and values. Relevant landscape functions are identified in the first stage. Key abiotic, biotic and anthropogenic functions can be deduced from a look-up matrix, and refined based on the stakeholders' interests.

In the second stage, the functions are focused onto specific ES of interest. Changes in present and future supply and demand for those services, as well as potential disservices, can be spatially assessed qualitatively making use of landscape metrics as simple indicators or quantitatively integrated into modelling tools (see section 4.6). Future alternatives can be defined through predictive and exploratory scenarios (Börjeson et al., 2006) involving stakeholders, acknowledging uncertainty, social aspirations, landscape capability, and conflicting interests.

In the third stage, the services should be converted into monetary units (for quantitative assessments), and social values. The monetary units should be calculated using information from biophysical units and a weighting based on their qualities to provide specific ES (e.g. agricultural soil classes used as a weight for food supply). These would inform market analysis for ES with direct and indirect use value, and contingent valuation or avoided cost methods for existence values (TEEB, 2011). For the social valuation, mapping and aggregating stakeholders' perceptions, making use of techniques such as public participatory GIS, would facilitate the

posterior identification of hotspots (Sherrouse et al. 2011).

### 4.4. Design

The design phase and its relationship with the assessment phase is articulated making use of the pattern:process:design paradigm of Nassauer and Opdam (2008) and the iterative Geodesign framework of Steinitz (2012). The influence of science is extended to the design phase and vice versa, aiming to reinforce design-driven approaches and reduce science:design segregation. Both phases and the role of their professionals become more blurred and a constant iterative assessment is developed as part of the process of design (Cashmore, 2004). This phase is divided into three stages: strategy, planning, and design.

In the strategy stage, the information on character, services and values is integrated and compared to identify potential incompatibilities per scenario (e.g. aspired character vs aspired services). A consensus should be achieved by professionals and stakeholders that simultaneously considers the kind of places where people want to live (aspired character), the services the people wish to have (aspired services), the services the area will be capable of providing (future landscape capability), the kind of values people would like to enhance or preserve (social valuation), and the indications of policies and higher level plans. The use of stakeholder engagement techniques such as mediation and joint fact-finding techniques could be very valuable for achieving consensus between parts. After the strategy is defined, four types of actions might be assigned to different landscape character areas or entities: conservation (keep current status), enhancement through management, physical recovery or restoration, and re-design. The last three actions are equivalent to the NBS types proposed by Cohen-Shacham *et al.* (2016). During the planning stage actions should be defined per

landscape character area and class of entity, specifying the individual entities at the design stage according to the European Landscape Convention guidelines. For each scenario and stage different alternatives can be created and reassessed iteratively until consensus is achieved.

### 4.5. Monitoring

The monitoring phase builds on the framework of van Oudenhoven *et al.* (2012) to better acknowledge the relevance of land management and related policies on spatial plans. Monitoring is extended beyond the short term to better understand consequences derived from the implementation, indirect effects of policies, and/or the combination of both. The long term monitoring would be integrated with management tasks, engaging users and land managers, extending collaboration, and making the monitoring more cost-effective.

### 4.6. Landscape metrics as potential landscape services indicators.

Indicators must be representative, reliable, comparable, cost-effective, accessible and simple to measure, and used by different types of professionals (Cornforth, 1999; Heink and Kowarik, 2010; Bottero, 2011; Heink et al., 2016). Some of the indicators for spatial planning purposes should be spatially explicit and measured in physical units, showing the land affected by landscape changes (Haines-Young & Potschin, 2005). In addition, to enhance collaboration between users, indicators need to be in a "common language" understandable by natural and social scientists, designers, planners, stakeholders, and decision makers.

Landscape metrics have been used for the last two decades by landscape ecologists to understand changes in ecological patterns and how these might affect processes in mainly

natural or semi-natural ecosystems (Jaeger, 2000; Uuemaa et al., 2005; Schindler et al., 2008). They provide composition and spatial configuration information (McGarigal, 2013) through basic geometric information (e.g. shape, area, length of perimeter), have low resource demands, are simple to use, and most are spatially explicit. Additionally, spatial outputs are a "language" understandable by built environment professionals and can easily be represented on maps aiding the comprehension of results by stakeholders and decision makers.

In this sense, landscape metrics could be adequate indicators to assess qualitative changes in landscape services for spatial planning purposes, since their demand and supply is dependent on spatial patterns, and could be easily complemented with other indicators. For example, landscape metrics could identify qualitative changes in the provision of regulation services (e.g. temperature and humidity, maintaining nursery populations and habitats) produced by a masterplan, making use of the land use/cover information provided by the plan itself. In fact, once the strategy is established, landscape metrics could be a simple and cost-effective way of undertaking quick iterative assessments during landscape planning and design to identify whether the proposal would produce the qualitative changes in character, services and values expected.

In contrast, selecting landscape metrics for new study areas requires expert knowledge and the same set of metrics are unlikely to be valid for other landscapes or their processes. Hence, an understanding of the ecological processes or services evaluated, their dependent factors, specific conditions, scale sensitivities of several metrics, and an appropriate classification and aggregation of land cover or habitats to avoid excessive simplification of landscape patterns is required. Also, landscape metrics are not suitable for measuring all of the services, but only

those dependent on structural aspects (Syrbe and Walz 2012). In fact, for complex plans or when high accuracy in the assessment is relevant, the combination of landscape metrics with other indicators as part of modelling tools (e.g. agent-based models or system dynamic models) might be necessary. For example, EnviroAtlas, a web-based tool for evaluating and mapping ES (Pickard et al. 2015), combines the use of 300 indicators (including landscape metrics) with additional toolboxes.

As a consequence, several authors indicate that caution with landscape metrics is needed, since the ecological understanding of them is still missing (Wu and Hobbs, 2002; Corry and Nassauer, 2005). In this sense, Corry and Nassauer (2005) indicate that landscape metrics that are demonstrated to relate to ecological functions for the studied context should be considered useful for comparing the ecological consequences of different plans or designs. Rehm and Baldassarre (2007) show a positive correlation between interspersion of water and vegetation (using edge density and cover to water ratio as parameter proxies) and marsh bird abundance. Viaud *et al.* (2008) also demonstrate that maize pattern (composed of maize area, genetically modified maize area, and maize spatial arrangement) was relevant to explain simulated cross-fertilisation between genetically modified maize and non-modified fields, confirming the results of previous simulations for maize and oilseed rape (Messean et al., 2006; Ceddia et al., 2007). Regarding regulation of temperature and humidity, Chen *et al.* (2014) show that 56% of the urban land surface temperature in Beijing (China) is explained by the percentage of impervious land surface (a composition metric), and configuration metrics such as the landscape shape index could explain an additional 6-12%. Similarly, Park and Cho, (2016) studied the cooling effect of different sizes (from less than 1 ha to 1 $km^2$) of urban green spaces . They demonstrated that cooling distance in Ulsan, Korea, is affected by the

shape of green areas (land shape intensity used as a parameter proxy), where having belt-shaped green areas produced the longest cooling distances compared to compact green areas of the same size. Hence, landscape metrics need to be used with caution to qualitatively assess ecological functions and services. However, research on landscape metrics has advanced in the recent years and previous research could inform their use as service indicators in spatial planning.

**5. Conclusion and outlook**

A conceptual framework supported by landscape ecology thinking is proposed to integrate in a comprehensive way the study of landscape (ecosystem) services into spatial planning. Based on a reflection of general and specific-disciplinary ES issues, the framework differentiates basic concepts and stresses an interrelation between the concepts of service, character and value.

The framework has a flexible structure supported by public participation and landscape ecology techniques indicated for each phase, although more complex tools (e.g. system dynamic models) could also be used. The potential techniques proposed make the framework accessible and applicable to a broad range of public and private organizations, professionals, stakeholders, and decision makers. Additionally, stakeholders should be integrated into each of the stages, empowering their intervention in the spatial planning process, but also enhancing their understanding of potential conflicts between their aspirations (e.g. character vs service aspirations). The structure of the framework would be useful at different scales and contexts to guide several types of urban and landscape plans at different planning stages (e.g. vision, strategy, conceptual plans, detailed plans), and scales. Therefore, the structure and

techniques recommended would facilitate the credibility of the framework in the built environment sector.

The concept of landscape services is proposed for anthropogenic dominated spatial planning contexts, where natural and anthropogenic factors are equally relevant and highly interrelated. This is to ensure a more complete consideration of services by stressing the importance of the spatial configuration of patches on the supply and demand of services.

By integrating the pattern:process:design paradigm as a key element, the framework advocates a tighter relationship between landscape ecology knowledge and its application to the resolution of urban and landscape planning/design issues. This is transposed in the form of a closer relationship between the assessment and design phases, and the implementation of iterative processes where stakeholder perceptions are integrated. Accordingly, the movement of assessment stages toward an environmental design model (Cashmore, 2004) can be allowed, as well as spatial planning toward more consensual and adaptive approaches. This can eventually increase saliency or relevance of the framework in decision making processes, and its legitimacy by integrating stakeholders' values.

Initial applications of the framework proposed will require increased efforts of communication, due to the extension of traditional roles and the iterations along the assessment and design stages. In addition, the adequate identification of relevant actors is a key element of the framework and if those are not identified well during the characterization stages, this might jeopardise the following stages. Moreover, the tools and indicators (e.g. landscape metrics)

proposed should be used with caution and their application should be guided by experts, especially when applied to new contexts and spatial planning issues. On the other hand, the framework offers an opportunity for advancing integrated empirical spatial planning and could act as a roadmap for transdisciplinary empirical research on anthropogenic dominated contexts such as urban ones. A natural follow-up of this paper is the application of the proposed framework to a real spatial planning study, to be possibly conducted in collaboration with professionals in the field of landscape and/or urban management.

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# Integration of ecosystem services into a conceptual spatial planning framework based on a landscape ecology perspective

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