Title
Landscape beauty: A wicked problem in sustainable ecosystem management?

Author
Dronova, Iryna

Publication Date
2019-10-01

DOI
10.1016/j.scitotenv.2019.06.248

Landscape beauty: a wicked problem in sustainable ecosystem management?

Iryna Dronova[1*]

[1]Department of Landscape Architecture & Environmental Planning, 202 Wurster Hall #2000, University of California Berkeley, California 94720-2000, USA

*corresponding author, e-Mail: idronova@berkeley.edu

Abstract

Recent discourses on sustainable ecosystem management have increasingly emphasized the importance of bundling relationships and interactions among multiple ecosystem services supported by similar natural and anthropogenic mechanisms within the total environment. Yet, the aesthetic benefits of ecosystems, playing critical role in management of both wild and anthropogenic landscapes, have been under-represented in these discussions. This disregard contributes to the disconnection between environmental science and practice and limits our understanding of ecological and societal implications of management decisions that either generate aesthetic benefits or impact them while targeting other ecosystem services. This discussion reviews several "wicked problems" that arise due to such limited understanding, focusing on three recognized challenges in present-day ecosystem management: replacement of natural ecosystem functions, spatial decoupling of service beneficiaries from its environmental consequences and increasing inequalities in access to services. Strategies towards solutions to such wicked challenges are also discussed, capitalizing on the potential of innovative landscape design, cross-disciplinary research and collaboration, and emerging economic and policy instruments.

Keywords: ecosystem management, sustainability, aesthetic ecosystem services, wicked problems, decision-making, landscape

1. Aesthetic ecosystem services: the under-discussed component of sustainability

The discourses on sustainable ecosystem management and governance have increasingly acknowledged the importance of bundling relationships among ecosystem services (ES) enabled by the connections among ecological, physical and anthropogenic processes within the total environment (Raudsepp-Hearne *et al.* 2010; Plieninger *et al.* 2013; Saidi and Spray 2018). However, aesthetic ecosystem services (AES; Box 1), a major sub-component of cultural ecosystem services (CES) in the Millennium Ecosystem Assessment (MEA) framework (De Groot *et al.* 2005), have been under-represented in these discussions, despite their significance for multi-functional ecosystem management and conservation (Ehrlich and Wilson 1991; Klein *et al.* 2015; Dronova 2017; Assandri *et al.* 2018), economic value of natural and human-designed landscapes (Nicholls and Crompton 2005, 2018; Kong *et al.* 2007), and broader aspects of human well-being (Velarde *et al.* 2007; Grinde and Patil 2009). As the global human population is shifting towards more regulated and designed landscapes such as cities, the impact of aesthetic values and preferences design on both ecological systems and human well-being will likely keep increasing (Meyer 2008; Lovell and Taylor 2013; Saito 2014; Klein *et al.* 2015; Botzat *et al.* 2016; Hoyle *et al.* 2017a; Opdam *et al.* 2018). Under-representation of aesthetic benefits in the analyses of ES tradeoffs and synergies thus makes it difficult to anticipate the implications of environmental decisions that intentionally or unintentionally produce significant aesthetic impact (Mozingo 1997; Junker and Buchecker 2008; Lim *et al.* 2015).

The objective of this discussion paper is to review several common contexts in which the limited understanding of aesthetic values and of the impact of their provisioning gives rise to "wicked problems" in ecosystem management. Wicked problems are complex challenges that cannot be solved in a predictable, straightforward way and lack generalizable approaches to test for potential solutions, thus requiring more comprehensive, strategic and multi-scale tackling (Rittel and Webber 1973; DeFries and Nagendra 2017). This paper argues that such problems may pose important barriers to sustainable management of ecosystems, and that one of the main common roots among these challenges is the under-studied interconnectedness of AES with other important ecosystem services and functions in the total environment context. This synthesis subsequently reviews strategies towards potential solutions and the relevant research needs.

BOX 1 ABOUT HERE

2. Background: AES & their representation among ES bundling studies

Aesthetic ecosystem services occupy a unique and under-explored niche within the science of the total environment, as they often result from synergistic properties of ecosystems shaped collectively atmospheric, hydrological, biological and geophysical factors – as well as human environmental decisions and policies (Nassauer 1997; Lothian 2017; Dronova 2017). This discussion uses the concept of AES similar to MEA and Common International Classification of Ecosystem Services (CICES v.5.1; Haines-Young and Potschin-Young 2018) as ecosystem characteristics that enable aesthetic experience and appreciation of "the beauty of nature", or negative effects, i.e., aesthetic disservices (Box 1). It is further recognized that "aesthetic experience" is a holistic notion which encompasses not only visual quality, but also broader sensory immersion, sense of place and related concepts, though specific interpretations vary (Gobster *et al.* 2007; Meyer 2008; Carlson 2014; van Zanten *et al.* 2014). Broader dimensions of human aesthetic perception of environment have been studied for multiple decades prior to MEA's ES framework by multiple disciplines including environmental psychology, landscape design and architecture (e.g., Kaplan and Kaplan 1989; Meyer 2008; Lothian 2017) and environmental aesthetics (Carlson 2014; Saito 2014). Not surprisingly, AES have been recognized as pivotal in reconciling stakeholder attitudes towards ecological management and conservation (Ehrlich and Wilson 1991; Cordingley *et al.* 2016; Graves *et al.* 2017; Kiley *et al.* 2017; Vlami *et al.* 2017) and the broader discourse on cultural sustainability (Nassauer 1997; Gobster *et al.* 2007; Meyer 2008; Daniel *et al.* 2012; Opdam *et al.* 2018).

Accordingly, various methods have been proposed to assess both the subjective aspects of the individual aesthetic perceptions, and objective criteria that enable such experiences based on ecosystem structure, landscape configuration, seasonality, and similar properties. For instance, landscape photographs and interviews have been widely used to gauge the human observers' experiences and rank different criteria contributing to varying perceptions (e.g., Kaplan and Kaplan 1989; Nassauer 2004; Kiley *et al.* 2017). Diverse economic approaches have been also employed to assign monetary values to AES as well as other cultural benefits (van Zanten *et al.* 2014, 2016). Recent advances in "big data" analyses, web-based informatics and geographic information systems (GIS) have opened new frontiers for AES assessments (Pardo García and Mérida Rodríguez 2015; van Zanten *et al.* 2016; Vlami *et al.* 2017), such as using social media contributions to study visitation of places and cultural preferences (e.g., Richards and Friess 2015) or modeling landscape-driven criteria for aesthetic quality at different vantage points (Pardo García and Mérida Rodríguez 2015; Martin *et al.* 2016).

Despite such recognition, AES have been prominently under-discussed among the studies focusing on tradeoffs, synergies and other bundling relationships among ES. For instance, a sample of such case studies across different geographic and ecosystem science contexts (Fig. 1a, Tables S1 & S2, Supplementary Material) shows that aesthetic benefits or their close proxies were explicitly selected in only 32% (28 out 87) analyses, although 83% studies included at least one cultural service. Furthermore, the average number of cultural ES per study was significantly higher in the studies including AES than otherwise (p<0.001 for all pairs), and even higher in the CES-only studies (Fig. 1b). This contrast points to a degree of disconnection between cultural ES studies and those focusing on broader dimensions of ES and their bundling (Daniel *et al.* 2012; Milcu *et al.* 2013). These statistics also resonate with the evidence from the global review of ecosystem service values by de Groot et al. (2012) which found only 12 AES estimates, or 1.8% of 665 total estimates across all ES types based on >320 studies and >300 locations. In contrast, both reviews and case studies specific to cultural ES frequently found AES to fall the top 1-2 cultural benefit types (Hernandez-Morcillo *et al.* 2013; Plieninger *et al.* 2013).

FIGURE 1 ABOUT HERE

The observed non-uniformity of AES inclusion based on a limited sample of studies in this example certainly warrants a more in-depth follow-up literature analysis to explicate their inclusion and omission. However, it is notable that only relatively few studies provided any concrete reasons for omitting specific ES types, even when their objectives focused broadly on bundling relationships among different ES. Some of the named reasons for excluding AES were limited availability of data and the dependence of AES on social constructions; in several cases aesthetic qualities were also mentioned in relation to other formal CES types (e.g., recreational), but not as a standalone service (Table S2 and endnotes, Supplementary Material). The latter tendency may reflect the limited detail in definitions of CES types within MEA (Plieninger *et al.* 2013), as well as assessment challenges due to their conceptual overlaps (Daniel *et al.* 2012) and varying individual perceptions (Gunnarson *et al.* 2017; Kiley *et al.* 2017). However, the variety of established methods for AES assessment discussed above suggests that methodological challenges might not be the main reason for their lack of inclusion. Another reason could be the treatment of AES as non-essential and substitutable, in contrast to provisioning and regulating services considered to be essential to human survival and thus integral to ES bundles. Yet, it is precisely as the "luxury" commodity aesthetic value can be a linchpin in the decisions about ecosystem governance and amenity-driven triggers of environmental and socio-economic injustice (Abrams and Bliss 2013; Wolch *et al.* 2014; Anguelovski *et al.* 2018a). Limited attention to AES can thus not only exacerbate the disconnection between environmental science and management practice (Naeem *et al.* 2015), but also produce major barriers to making ecosystem management and conservation sustainable.

3. Limited understanding of AES contributes to wicked problems

A recent review of wicked problems in present-day ecosystem management (DeFries and Nagendra 2017) discussed several reasons for their exacerbating complexity in the 21st century related to 1) the use of management approaches that replace natural functions of ecosystems but fail to re-create their self-regulating properties; 2) spatial separation of production and consumption of ecosystem services which limits the understanding and awareness of the cost and implications of management among service beneficiaries; and 3) inequalities in access to resources, aggravated by differences in stakeholder perspectives and values. These issues gain a special significance in the context of cultural and particularly aesthetic benefits (Fig. 2), as has been acknowledged in the earlier discussions on sustainability in landscape planning and design (Nassauer 1997; Meyer 2008; Opdam *et al.* 2018). Several examples discussed below illustrate that such wicked challenges emerge both when AES represent a central objective in ecosystem management and when their importance is overlooked.

FIGURE 2 ABOUT HERE

3.1. Replacement of natural ecosystem functions

In highly human-regulated ecosystems, provisioning of aesthetic value (and other benefits) via vegetation often relies on the alternatives to natural ecological controls and nutrient cycling, such as pesticides and fertilizers. Practical considerations behind such management choices can be fueled by social pressures for aesthetic norms that may be difficult to implement sustainably at a massive scale and some of the service-oriented landscape practices that follow these norms (Meyer 2008; Saito 2014; Groffman *et al.* 2014; Sisser *et al.* 2016; Aronson *et al.* 2017). An infamous example is the use of chemicals in maintenance of grassy lawns, which may adversely impact not only biological diversity and functioning of adjacent ecosystems, but also the health of humans benefitting from these decisions aesthetically (Robbins and Sharp 2003). This is a wicked problem because markets and financial considerations behind the maintenance of green infrastructure may not immediately favor sustainable solutions in the absence of additional public incentives and top-down regulation (Khachatryan *et al.* 2017). In the long run, prevalence of management pathways perceived to be more economical and practical contributes to regional and national-scale homogenization of urban residential and public spaces with potentially critical ecological implications well beyond their immediate boundaries (Groffman *et al.* 2014).

At the same time, efforts to preserve, mimic or restore ecological functions and processes may sometimes diminish aesthetic quality, leading to disengagement or even repulsion of public attitude (Nassauer 1992, 2004; Kiley *et al.* 2017) and "alienation from nature" (Mozingo 1997). Such outcomes may be difficult to avoid when supporting vulnerable species requires large extents of homogeneous habitat, or when aesthetically unfavorable outcomes result from critical measures such as reduction of irrigation during droughts leading plant senescence (Mozingo 1997; Hilaire *et al.* 2008). Similarly, some renewable energy systems, such as wind farms and photovoltaic structures, have been perceived as unfavorable for scenic value (e.g., Saito 2014; Kienast *et al.* 2017) despite their potential for sustainability. Failure to consider such aesthetic disservices may sometimes directly hinder conservation and management goals; for example, in hydrological restoration, public aesthetic preferences may pose barriers for measures such as introduction of wood to provide the habitat for fish and other aquatic organisms (Piegay *et al.* 2005; Junker and Buchecker 2008; Ruiz-Villanueva *et al.* 2018). Even in a basic sense, appearance of landscapes as messy or untidy may become an important disservice (Nassauer 1995a) and lead to management practices emphasizing orderliness and neatness over potential environmental and ecological implications (Plieninger *et al.* 2013; Chapman *et al.* 2019).

3.2. Separation of ecosystem service beneficiaries and implications of provisioning

The immediate spatial scale of human AES experience is often decoupled from the spatio-temporal footprint of resources, processes and markets enabling this service. As a result, beneficiaries and even providers of AES may not be aware of the full range of resources and costs contributing to provisioning or mitigation of their implications (Thompson 2002; Meyer 2008; Spirn 2014). This disconnection makes it difficult to raise public awareness about such implications or modify social preferences, behaviors and management practices. This concern becomes especially evident in the controversies around the use of non-native plant species in landscaping (Drew *et al.* 2010; Hoyle *et al.* 2017a, b; Epanchin-Niell 2017) with a suite of contributing factors and costs not immediately obvious at the scale of individual projects (Fig. 3). Introduction of non-native species elevates the risk of ecological invasions and associated economic losses, should such species escape to native ecosystems of their new localities and proliferate due to facilitating adaptations and/or lack of natural competitors and predators (McDermott *et al.* 2013; Epanchin-Niell 2017). This risk is magnified by the large pools of available species in horticultural markets, market selection of species with traits that could contribute to invasive potential, propagule pressure and the uncertain time lags between the introduction and the onset of invasion (Drew *et al.* 2010). Ironically, in some cases massive invasions may produce negative aesthetic impact despite the ornamental value of invading species. For instance, flood-induced mortality of invasive giant reed (*Arundo donax*) in coastal areas may lead to massive depositions of dead biomass as floodway dams and beach debris (Loper *et al.* 2005). Another example relates to the uncontrolled spread of the ornamental elephant tree (*Ailanthus altissima*) in some European species, which produces multiple aesthetic disservices and adds to vegetation maintenance and clean-up costs (Casella and Vurro 2013).

FIGURE 3 ABOUT HERE

Yet, cultural benefits provided by non-native species and the complexity of their ecological effects make the decisions about their control, regulation and use a truly wicked challenge. Cultural factors play a prominent role in the human-induced movement of species, which may contribute to provisioning services, sustain heritage and educational values and promote sense of agency and preservation of cultural ties in immigrant communities (Shackleton *et al.* 2019). In some cases non-native species may offer unexpected benefits to native ecosystems, such as riparian bird habitat enabled by invasive woody *Tamarix* in the southwestern U.S. (Sogge *et al.* 2008), or contributions of different alien plants and animals to limiting ecological food sources and critical functions such as pollination or nutrient enrichment (Rodriguez 2006). Aesthetically pleasing non-native plant species tolerant of warm and/or dry conditions can be perceived as potentially sustainable opportunities for adapting human-dominated landscapes to climate change (Hoyle *et al.* 2017b; Alizadeh and Hitchmough 2019). These controversies are augmented by the already occurring climate-induced species range shifts and the fact that some ecosystems, such as warming cities, are already experiencing “future” climates to which the native species of their regions may not be well adapted, as shown in a California, USA study

(McBride and Laćan 2018). Together, these challenges suggest that alien species management and use urgently require a fuller understanding of teleconnections and costs associated with their broad-scale movement for aesthetic and other benefits (Epanchin-Niell 2017).

3.3. Inequalities in access to resources and services

Although still difficult to assess in monetary terms, aesthetic benefits associated with attractive ecosystem features (forests, gardens, waterfronts, scenic vistas) can generate direct economic value for ecosystems and places and stimulate public demand (e.g., Tarrant and Cordell 2002; Kong *et al.* 2007; Jim and Chen 2009; Vejre *et al.* 2010; Nicholls and Crompton 2018). As such, both their targeted provisioning and under-appreciation of their importance may trigger inequalities in human access and exclusion of specific groups, exacerbating social injustice. Examples of such cascading effects are evident in the high amenity value of urban green spaces providing aesthetic services together with recreational, social and health benefits, as well as regulating and supporting ecological functions (e.g., Tarrant and Cordell 2002; Grinde and Patil 2009). In densely populated cities with limited access to "nature", such coupled benefits can increase the market value of properties in associated neighborhoods, which may trigger gentrification and displacement (Wolch *et al.* 2014; Anguelovski *et al.* 2018a) and generate tensions within communities about aesthetic and cultural characteristics of the designed spaces themselves (Gobster 2001; Aptekar 2015).

These issues are not unique to cities; similar concerns about exacerbating inequality and gentrification also arise in rural areas attracting urban citizens or intensified development due to presence of scenic qualities. Examples of such "amenity migration" and subsequent shifts in rural commodities, production and/or cost of living have been reported in various scenic regions of the United States (Ghose 2004; Abrams and Bliss 2013), parts of China (Qian *et al.* 2013) and some other global regions. Obviously, such issues extend well beyond the immediate scope of ecosystem management and lead to profound multi-scale transformations of decision-making landscapes, aggravating other wicked problems (Wolch *et al.* 2014; Anguelovski *et al.* 2018b).

4. Common roots of different wicked problems

Despite the differences in context and scope, the challenges discussed above share several notable commonalities. First, their "wickedness" is often centered on the difficult tradeoff of making the service accessible while also controlling the cost of its provision. When increasing accessibility depends on reducing market value, cheaper options might be easier to provide to many; however, such options risk being less sustainable and rely on "shortcuts" such as the applications of hazardous chemicals in landscape maintenance or using non-native species in landscape design as a lower-cost "material" to achieve a specific aesthetic and experiential impact. Relatedly, once something *is* accessible to many, it might be less valuable as a private good. As a result, when inequitable access becomes a factor in generating economic value at least in the short term, there could be little private incentive to invest in public access, which creates a barrier for resolving such wicked problems at their core. Addressing this tradeoff in a sustainable, lasting way thus requires economic and social incentives that would magnify the benefits of alternative options, which is neither a quick nor an easy change to make.

The second common challenge lies in the earlier mentioned specificity of AES to vantage point, landscape setting or seasonal context of the observer's experience (Gobster *et al.* 2007; Carlson 2014; Saito 2014; Dronova 2017). When AES are conceptualized as a "stable" landscape property, their provisioning may become dependent on maintaining such an experience as an appearance or "form", rather than a suite of dynamic processes (Gobster *et al.* 2007; Meyer 2008). Such an emphasis makes it difficult to control the outcomes and invites more practical and/or fast-acting management options that might not be ecologically sustainable (Spirn 2014). However, changing such practices may be difficult even when they do not accurately represent public values or environmental attitudes (Aronson *et al.* 2017; Khachatryan *et al.* 2017). The latter issue also points to a deeper lack of a dialogue about aesthetic and cultural values between ecosystem managers and the public– and especially of their dynamics and adaptations in response to changing socio-economic context, environmental awareness or transformation of landscapes and ecosystems (Hilaire *et al.* 2008; Nguyen *et al.* 2017; Anguelovski *et al.* 2018a).

The third commonality is that aesthetic benefits are often coupled with multiple other outcomes and functions targeted by ecosystem management, which provides opportunities for win-win strategies of their provisioning (Nassauer 1997; Lovell and Taylor 2013; Howett 2014; Klein *et al.* 2015; Botzat *et al.* 2016; Dronova 2017). To date, however, the evidence of such linkages has not been yet translated into generalizable guiding principles to use AES as a means to enhance the delivery of other services, although the potential of AES to generate cultural value and amplify ecological value has been long recognized (Nassauer 1992, 1995a, 2011; Howett 2014; Spirn 2014). This issue calls for more targeted research efforts to inform holistic management strategies yielding "multi-functional" outcomes within the same spatial, environmental and social contexts (Opdam *et al.* 2013; Meerow and Newell 2017).

5. Moving forward: potential solutions and research needs

Wicked problems do not have straightforward solutions and require innovative approaches to address the complexity of their contributing mechanisms and agents. As such, tackling them requires more than simply bridging together science and practice –a mixed portfolio of strategies addressing different scales of ecosystem management and perception and capitalizing on emerging research, policy and economic instruments. A number of relevant approaches have been highlighted in the literature as discussed below; however, they have not been yet jointly examined through the lens of the ES frameworks and ES bundling relationships. Broadly, such strategies may include: 1) creating aesthetic value to enhance public perception, engagement and support for ecologically focused management; 2) capitalizing on common drivers of different ecosystem service types to promote strategies leading to multi-benefit outcomes and win-win scenarios; 3) employing economic and policy measures to incentivize ecosystem service provisioning options with lower distributed environmental impact; and 4) using environmental education and cross-disciplinary collaboration to enable a longer-term paradigm shift.

5.1. Enabling cultural sustainability by creating and promoting aesthetic value

Creating aesthetic value means expanding the scope of ecosystem management to introduce specific characteristics that enhance visual and sensory quality of the human experience, while maintaining the primary ecological objectives. Measures that promote visual appeal, orderliness and legibility, such as visual “cues to care” (Nassauer 1995a, 2011) can help assign cultural value to ecological processes and functions that may not be otherwise easily comprehended or even “visible” as a landscape experience (Nassauer 1992; Meyer 2008; Spirn 2014). In cases when public access restriction is necessary to protect sensitive ecosystems and habitats, providing such aesthetic experience at the permissible nodes of human interaction can help promote environmental education and awareness and more effectively communicate ecological objectives to the broader audience (Nassauer 2004; Meyer 2008; Mocior and Kruse 2016).

The ultimate challenge, however, lies in making such effects lasting and avoiding the misuse of aesthetic value for shielding and diverting public attention from unsustainable management practices or other wicked issues (Nassauer 1995b; Anguelovski *et al.* 2018a). This means that ecosystem managers, designers and other decision-makers need to understand the public and community values in the first place, as well as the ways by which these values define human interactions with ecosystems (Sterling *et al.* 2017). Individual aesthetic perceptions and preferences may be complex and vary depending on demographic, social and other factors (Gunnarsson *et al.* 2017; Kiley *et al.* 2017); however, the extent to which such preferences are shaped by economic considerations, community-shared aesthetic norms and environmental constraints may provide a useful basis for incorporating such values in decision-making. For example, a study of five residential tree distribution programs in the U.S. (Nguyen *et al.* 2017) reported community preferences for smaller ornamental flowering or fruiting trees which demand less space and maintenance cost than, e.g., large shade trees. Such evidence illustrates that public and community values are dynamic and adapting, and thus should be continuously monitored to enable the "sustained public support for environment" (Meyer 2008).

5.2. Capitalizing on common drivers among different ecosystem services

Aesthetic value can be generated as a direct outcome of ecosystem management due to its dependence on specific ecosystem properties and processes contributing to non-aesthetic benefits, which provides opportunities to manage for such diverse benefits jointly and thus reduce the risk of conflicting priorities in management outcomes (e.g., Felson and Pickett 2005; Spirn 2014; Klein *et al.* 2015; Dronova 2017). Such opportunities can be found, e.g., in diversified agriculture systems, where enhancement of ecosystem services and resilience is achieved via complexity of crop patterning and biological diversity of remnant habitats (Kremen and Miles 2012; Morandin and Kremen 2013), which, in turn, may contribute to scenic quality of rural areas and their attractiveness for agritourism (van Zanten *et al.* 2014). Similarly, in systems with frequent disturbance, spatial and biological heterogeneity may promote diversity of ecological responses and resilience of important functions while also supporting high visual complexity and scenic quality due to diverse and structure of vegetation (Jiang *et al.* 2012; Southon *et al.* 2017; Dronova 2017) along with topographic and hydrological factors (Schirpke *et al.* 2016; Nicholls and Crompton 2018). In residential landscapes, homeowners' opting for wooded front and backyards may provide opportunities for long-term carbon storage under certain design strategies to promote preferred levels of neatness and style (Visscher *et al.* 2016)

Such coupled benefits demonstrate that AES have their right place within the domains of ecosystem service management that traditionally may not have extensively considered cultural and aesthetic impacts. (For example, a review of cultural ES provided by the biodiversity of forest soils across Europe (Motiejūnaitė *et al.* 2019) discussed several benefits and disservices provided by soil-dwelling organisms in terms of aesthetics and sense of place). However, taking fuller advantage of these multi-ES linkages requires more research on specific relationships of AES with ecosystem structure, function and dynamics in different contexts (Fry *et al.* 2009; Dronova 2017). This also means that landscape design practices emphasizing form and visual experience may need to conscientiously shift towards a "trivalent design" paradigm embracing social and aesthetic values together with environmental values and their functional connections (Thompson 2002). This need has been recognized for decades (Nassauer 1997; Gobster *et al.* 2007; Meyer 2008; Howett 2014; Spirn 2014; Opdam *et al.* 2018); however, persisting challenges, such as ornamental use of species with established invasive potential or excessive reliance on harmful chemicals in landscaping, imply that emphasis on form still prevails in certain domains of ecosystem management and does not adequately meet the needs of communities experiencing their outcomes.

5.3. Engaging economic and policy instruments

In cases where wickedness is augmented by the economic appeal of less sustainable options, public awareness alone might be insufficient to make a lasting difference at a large enough scale or to overcome the realities of markets or fashion trends. In such cases, innovative policy and economic instruments could be engaged to create incentives for sustainable alternatives (Suh *et al.* 2016; Khachatryan *et al.* 2017) and reduce the cost of their implementation (McDermott *et al.* 2013). Such instruments may help restrict or de-incentivize decisions with higher distributed social and environmental risks (Hulme *et al.* 2018), or regulate supply and accessibility of amenities while ensuring representation of diverse community voices in complex decision-making. For example, it was shown in the context of lawn maintenance that homeowners may be more willing to pay higher premiums for sustainable and eco-friendly fertilizers in jurisdictions with formal ordinance and regulations supporting such options (Khachatryan *et al.* 2017). Such choices may be further encouraged by educational and informational programs contributing to positive perception and awareness of more sustainable landscaping measures both among individuals and at the neighborhood/ community levels (Suh *et al.* 2016). Similarly, the spread of invasive species may be slowed by economic and policy instruments that combine both subsidies encouraging decentralized control and management, and penalties to ensure collective intervention at the relevant scales (McDermott *et al.* 2013; Epanchin-Niell 2017).

Analytical economic and policy instruments should be also engaged to comprehensively test the outcomes of various environmental planning scenarios to help better identify and anticipate the impacts of urban revitalization strategies on social inequity (Anguelovski *et al.* 2018a, b; Xu *et al.* 2018). Such assessments can be designed as spatially explicit analyses incorporating socio-economic and demographic information highlighting variation priorities and needs among different neighborhoods (Almeter *et al.* 2018) and biophysical characteristics of urban ecosystems and vegetation to identify more ecologically sustainable opportunities for greening interventions (Felson and Pickett 2005; Sass *et al.* 2019). Such efforts may help address other urgent research needs, such as understanding the implications of ecosystem management for public health. While less easily traceable, societal costs of hazardous management practices and broader-scale environmental planning may translate to the actual detectable health impacts (Robbins and Sharp 2003; Ćwik *et al.* 2018), which could be used to guide the regulations and incentives (Sisser *et al.* 2016).

5.4. Longer-term paradigm shift: the potential of collaborative research and education

In the longer-term perspective, even the most cutting-edge analytical instruments and innovative policies still risk being ineffective and unsustainable, unless a more profound paradigm shift occurs to re-establish aesthetic and cultural values as a formal outcome and contributing driver of ecosystem management at various levels of its intensity. Environmental education has a particular power to enable such a paradigm shift (Mocior and Kruse 2016), because deeper understanding of ecological value, conservation potential or risks may affect the individual's definition of scenic and attractive towards greater acceptance of ecologically critical measures (Nassauer 2004; Gunnarsson *et al.* 2017). Economic instruments and research tools can also support such a paradigm shift by daylighting the less obvious benefits of sustainable decisions and their positive externalities emerging at different spatial and temporal scales due to positive synergies between AES and other outcomes (Klein *et al.* 2015; van Zanten *et al.* 2016; Sass *et al.* 2019). For instance, increasing the quality and accessibility of ecosystem services associated with, e.g., urban green infrastructure may not only help address social inequities in environmental quality but also generate important synergistic landscape-scale benefits to public health, pollution regulation, thermal protection, energy saving, in addition to enhancing immediate landscape experiences (Felson and Pickett 2005; Aronson *et al.* 2017). However, to become sustainable, these benefits should visibly outweigh the costs of their provision, which requires a certain critical mass of infrastructure generating these synergistic services as well as explicit recognition of their coupling (Georgescu *et al.* 2015; Meerow and Newell 2017).

These tasks, in turn, require more dedicated research efforts and collaborative integration of environmental sciences with social and landscape design disciplines towards a rigorous multi-way cross-pollination of their academic curricula and practices. Such an exchange is critically important not only to assist ecosystem managers in balancing public social values with ecological necessities (Gobster 2001; Thompson 2002; Nassauer 2004), but also facilitate a more profound revitalization of management and design practices which necessarily requires environmental, social – and aesthetic – literacy (Gobster *et al.* 2007; Opdam *et al.* 2013, 2018; Saito 2014; Spirn 2014). While cultivating such literacy is a challenging task, contingent on multi-lateral cooperation and mutual learning among practitioners, planners, scientists and diverse stakeholder communities, it offers an important promise towards more robust and creative solutions to wicked challenges with sensitivity to dynamic societal values and needs.

Acknowledgments

The author thanks four anonymous reviewers for their helpful feedback and suggestions.

References:

Abrams J and Bliss JC. 2013. Amenity Landownership, Land Use Change, and the Re-Creation of “Working Landscapes.” *Society & Natural Resources* **26**: 845–59.

Alizadeh B and Hitchmough J. 2019. A review of urban landscape adaptation to the challenge of climate change. *International Journal of Climate Change Strategies and Management* **11**: 178–94.

Almeter A, Tashie A, Procter A, *et al.* 2018. A Needs-Driven, Multi-Objective Approach to Allocate Urban Ecosystem Services from 10,000 Trees. *Sustainability* **10**: 4488.

Anguelovski I, Connolly J, and Brand AL. 2018a. From landscapes of utopia to the margins of the green urban life: For whom is the new green city? *City* **22**: 417–36.

Anguelovski I, Connolly JJ, Garcia-Lamarca M, *et al.* 2018b. New scholarly pathways on green gentrification: What does the urban 'green turn' mean and where is it going? *Progress in Human Geography*: 030913251880379.

Aptekar S. 2015. Visions of Public Space: Reproducing and Resisting Social Hierarchies in a Community Garden. *Sociological Forum* **30**: 209–27.

Aronson MFJ, Lepczyk CA, Evans KL, *et al.* 2017. Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment* **15**: 189–96.

Assandri G, Bogliani G, Pedrini P, and Brambilla M. 2018. Beautiful agricultural landscapes promote cultural ecosystem services and biodiversity conservation. *Agriculture, Ecosystems & Environment* **256**: 200–10.

Botzat A, Fischer LK, and Kowarik I. 2016. Unexploited opportunities in understanding liveable and biodiverse cities. A review on urban biodiversity perception and valuation. *Global Environmental Change-Human and Policy Dimensions* **39**: 220–33.

Carlson A. 2014. Ten Steps in the Development of Western Environmental Aesthetics (M Drenthen and J Keulartz, Eds).

Casella F and Vurro M. 2013. Ailanthus altissima (tree of heaven): Spread and harmfulness in a case-study urban area. *Arboricultural Journal* **35**: 172–81.

Chapman M, Satterfield T, and Chan KMA. 2019. When value conflicts are barriers: Can relational values help explain farmer participation in conservation incentive programs? *Land Use Policy* **82**: 464–75.

Cordingley JE, Newton AC, Rose RJ, *et al.* 2016. Can landscape-scale approaches to conservation management resolve biodiversity-ecosystem service trade-offs? *Journal of Applied Ecology* **53**: 96–105.

Ćwik A, Kasprzyk I, Wójcik T, *et al.* 2018. Attractiveness of urban parks for visitors versus their potential allergenic hazard: A case study in Rzeszów, Poland. *Urban Forestry & Urban Greening* **35**: 221–9.

Daniel TC, Muhar A, Arnberger A, *et al.* 2012. Contributions of cultural services to the ecosystem services agenda. *Proceedings of the National Academy of Sciences of the United States of America* **109**: 8812–9.

De Groot R, Ramakrishnan PS, Berg AVD, and et al. 2005. Cultural and amenity services. In: Millennium Ecosystem Assessment Board (ed.), Ecosystems and Human Wellbeing: Current Status and Trends. Washington, DC: Island Press.

DeFries R and Nagendra H. 2017. Ecosystem management as a wicked problem. *Science* **356**: 265–70.

Drew J, Anderson N, and Andow D. 2010. Conundrums of a complex vector for invasive species control: a detailed examination of the horticultural industry. *Biological Invasions* **12**:

2837–51.

Dronova I. 2017. Environmental heterogeneity as a bridge between ecosystem service and visual quality objectives in management, planning and design. *Landscape and Urban Planning* **163**: 90–106.

Ehrlich PR and Wilson E 0. 1991. Biodiversity Studies: Science and Policy. *Science* **253**: 758–62.

Epanchin-Niell RS. 2017. Economics of invasive species policy and management. *Biological Invasions* **19**: 3333–54.

Felson AJ and Pickett ST. 2005. Designed experiments: new approaches to studying urban ecosystems. *Frontiers in Ecology and the Environment* **3**: 549–56.

Fry G, Tveit MS, Ode A, and Velarde MD. 2009. The ecology of visual landscapes: Exploring the conceptual common ground of visual and ecological landscape indicators. *Ecological Indicators* **9**: 933–47.

Georgescu M, Chow WTL, Wang ZH, *et al.* 2015. Prioritizing urban sustainability solutions: coordinated approaches must incorporate scale-dependent built environment induced effects. *Environmental Research Letters* **10**: 061001.

Ghose R. 2004. Big Sky or Big Sprawl? Rural Gentrification and the Changing Cultural Landscape of Missoula, Montana. *Urban Geography* **25**: 528–49.

Gobster PH. 2001. Visions of nature: conflict and compatibility in urban park restoration. *Landscape and Urban Planning* **56**: 35–51.

Gobster PH, Nassauer JI, Daniel TC, and Fry G. 2007. The shared landscape: what does aesthetics have to do with ecology? *Landscape Ecology* **22**: 959–72.

Graves RA, Pearson SM, and Turner MG. 2017. Species richness alone does not predict cultural ecosystem service value. *Proceedings of the National Academy of Sciences of the United States of America* **114**: 3774–1779.

Grinde B and Patil GG. 2009. Biophilia: Does Visual Contact with Nature Impact on Health and Well-Being? *International Journal of Environmental Research and Public Health* **6**: 2332–43.

Groffman PM, Cavender-Bares J, Bettez ND, *et al.* 2014. Ecological homogenization of urban USA. *Frontiers in Ecology and the Environment* **12**: 74–81.

Gunnarsson B, Knez I, Hedblom M, and Sang ÅO. 2017. Effects of biodiversity and environment-related attitude on perception of urban green space. *Urban Ecosystems* **20**: 37–49.

Haines-Young R and Potschin-Young M. 2018. Revision of the Common International Classification for Ecosystem Services (CICES V5. 1): A Policy Brief. *One Ecosystem*: 145.

Hernandez-Morcillo M, Plieninger T, and Bieling C. 2013. An empirical review of cultural ecosystem service indicators. *Ecological Indicators* **29**: 434–44.

Hilaire RS, Arnold MA, Wilkerson DC, *et al.* 2008. Efficient Water Use in Residential Urban Landscapes. *Hortscience* **43**: 2081–92.

Howett C. 2014. Systems, Signs, Sensibilities: Sources for a New Landscape Aesthetic. In: Ndubisi FO (Ed). The Ecological Design and Planning Reader. Washington, DC: Island Press/Center for Resource Economics.

Hoyle H, Hitchmough J, and Jorgensen A. 2017a. All about the "wow factor"? The relationships between aesthetics, restorative effect and perceived biodiversity in designed urban planting. *Landscape and Urban Planning* **164**: 109–23.

Hoyle H, Hitchmough J, and Jorgensen A. 2017b. Attractive, climate-adapted and sustainable? Public perception of non-native planting in the designed urban landscape. *Landscape and Urban Planning* **164**: 49–63.

Hulme PE, Brundu G, Carboni M, *et al.* 2018. Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions (L Flory, Ed). *Journal of Applied Ecology* **55**: 92–8.

Jiang J, Gao D, and DeAngelis DL. 2012. Towards a theory of ecotone resilience: Coastal vegetation on a salinity gradient. *Theoretical Population Biology* **82**: 29–37.

Jim CY and Chen WY. 2009. Value of scenic views: Hedonic assessment of private housing in Hong Kong. *Landscape and Urban Planning* **91**: 226–34.

Junker B and Buchecker M. 2008. Aesthetic preferences versus ecological objectives in river restorations. *Landscape and Urban Planning* **85**: 141–54.

Kaplan R and Kaplan S. 1989. The Experience of Nature. Cambridge: Cambridge University PRess.

Khachatryan H, Suh DH, Zhou G, and Dukes M. 2017. Sustainable Urban Landscaping: Consumer Preferences and Willingness to Pay for Turfgrass Fertilizers. *Canadian Journal of Agricultural Economics/Revue canadienne d'agroeconomie* **65**: 385–407.

Kienast F, Huber N, Hergert R, *et al.* 2017. Conflicts between decentralized renewable electricity production and landscape services - A spatially-explicit quantitative assessment for Switzerland. *Renewable & Sustainable Energy Reviews* **67**: 397–407.

Kiley HM, Ainsworth GB, Dongen WFD van, and Weston MA. 2017. Variation in public perceptions and attitudes towards terrestrial ecosystems. *Science of the Total Environment* **590**: 440–51.

Klein LR, Hendrix WG, Lohr VI, *et al.* 2015. Linking ecology and aesthetics in sustainable agricultural landscapes: Lessons from the Palouse region of Washington, U.S.A. *Landscape and Urban Planning* **134**: 195–209.

Kong F, Yin H, and Nakagoshi N. 2007. Using GIS and landscape metrics in the hedonic price modeling of the amenity value of urban green space: A case study in Jinan City, China. *Landscape and Urban Planning* **79**: 240–52.

Kremen C and Miles A. 2012. Ecosystem Services in Biologically Diversified versus Conventional Farming Systems: Benefits, Externalities, and Trade-Offs. *Ecology and Society* **17**: 40.

Lim SS, Innes JL, and Meitner M. 2015. Public awareness of aesthetic and other forest values associated with sustainable forest management: A cross-cultural comparison among the public in four countries. *Journal of Environmental Management* **150**: 243–9.

Loper C, Cozad D, Katagi W, and Beehler J. 2005. Thinking Native in Southern California: Coastal Benefits of a Watershed Approach to Invasive Exotic Plant Removal. In: California and the World Ocean '02. Santa Barbara, California, United States: American Society of Civil Engineers.

Lothian A. 2017. The science of scenery: how we view scenic beauty, what it is, why we love it, and how to measure and map it.

Lovell ST and Taylor JR. 2013. Supplying urban ecosystem services through multifunctional green infrastructure in the United States. *Landscape Ecology* **28**: 1447–63.

Martin B, Ortega E, Otero I, and Arce RM. 2016. Landscape character assessment with GIS using map-based indicators and photographs in the relationship between landscape and roads. *Journal of Environmental Management* **180**: 324–34.

McBride JR and Laćan I. 2018. The impact of climate-change induced temperature increases on the suitability of street tree species in California (USA) cities. *Urban Forestry & Urban Greening* **34**: 348–56.

McDermott SM, Irwin RE, and Taylor BW. 2013. Using economic instruments to develop effective management of invasive species: insights from a bioeconomic model. *Ecological Applications* **23**: 1086–100.

Meerow S and Newell JP. 2017. Spatial planning for multifunctional green infrastructure: Growing resilience in Detroit. *Landscape and Urban Planning* **159**: 62–75.

Meyer EK. 2008. Sustaining beauty: The performance of appearance (Reprinted from "Journal of

Landscape Architecture”, Spring 2008). *Landsc Archit* **98**: 92-+.

Milcu AI, Hanspach J, Abson D, and Fischer J. 2013. Cultural Ecosystem Services: A Literature Review and Prospects for Future Research. *Ecology and Society* **18**: UNSP 44.

Mocior E and Kruse M. 2016. Educational values and services of ecosystems and landscapes - An overview. *Ecological Indicators* **60**: 137–51.

Morandin LA and Kremen C. 2013. Hedgerow restoration promotes pollinator populations and exports native bees to adjacent fields. *Ecological Applications* **23**: 829–39.

Motiejūnaitė J, Børja I, Ostonen I, *et al.* 2019. Cultural ecosystem services provided by the biodiversity of forest soils: A European review. *Geoderma* **343**: 19–30.

Mozingo LA. 1997. The aesthetics of ecological design: seeing science as culture. *Landscape Journal* **16**: 46–59.

Naeem S, Ingram JC, Varga A, *et al.* 2015. Get the science right when paying for nature’s services. *Science* **347**: 1206–7.

Nassauer JI. 1992. The appearance of ecological systems as a matter of policy. *Landscape Ecology* **6**: 239–50.

Nassauer JI. 1995a. Messy Ecosystems, Orderly Frames. *Landscape Journal* **14**: 161–70.

Nassauer J. 1995b. Culture and changing landscape structure. *Landscape Ecology* **10**: 229–37.

Nassauer JI. 1997. Cultural sustainability: Aligning aesthetics and ecology. In: Placing Nature: Culture and Landscape Ecology. Edited by J.I. Nassauer. Washington, D.C.: Island Press.

Nassauer JI. 2004. Monitoring the success of metropolitan wetland restorations: Cultural sustainability and ecological function. *Wetlands* **24**: 756–65.

Nassauer JI. 2011. Care and stewardship: From home to planet. *Landscape and Urban Planning* **100**: 321–3.

Nguyen VD, Roman LA, Locke DH, *et al.* 2017. Branching out to residential lands: Missions and strategies of five tree distribution programs in the U.S. *Urban Forestry & Urban Greening* **22**: 24–35.

Nicholls S and Crompton JL. 2005. The impact of greenways on property values: Evidence from Austin, Texas. *Journal of Leisure Research* **37**: 321–41.

Nicholls S and Crompton JL. 2018. The contribution of scenic views of, and proximity to, lakes and reservoirs to property values. *Lakes & Reservoirs Research and Management* **23**: 63–78.

Opdam P, Luque S, Nassauer J, *et al.* 2018. How can landscape ecology contribute to sustainability science? *Landscape Ecology* **33**: 1–7.

Opdam P, Nassauer JI, Wang Z, *et al.* 2013. Science for action at the local landscape scale. *Landscape Ecology* **28**: 1439–45.

Pardo García S and Mérida Rodríguez M. 2015. A geospatial indicator for assessing urban panoramic views. *Computers, Environment and Urban Systems* **49**: 42–53.

Piegay H, Mutz M, Gregory KJ, *et al.* 2005. Public perception as a barrier to introducing wood in rivers for restoration purposes. *Environmental Management* **36**: 665–74.

Plieninger T, Dijks S, Oteros-Rozas E, and Bieling C. 2013. Assessing, mapping, and quantifying cultural ecosystem services at community level. *Land Use Policy* **33**: 118–29.

Qian J, He S, and Liu L. 2013. Aestheticisation, rent-seeking, and rural gentrification amidst China's rapid urbanisation: The case of Xiaozhou village, Guangzhou. *Journal of Rural Studies* **32**: 331–45.

Raudsepp-Hearne C, Peterson GD, and Bennett EM. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences* **107**: 5242–7.

Richards DR and Friess DA. 2015. A rapid indicator of cultural ecosystem service usage at a fine spatial scale: Content analysis of social media photographs. *Ecological Indicators* **53**: 187–95.

Rittel HWJ and Webber MM. 1973. Dilemmas in a general theory of planning. *Policy Sciences* **4**: 155–69.

Robbins P and Sharp JT. 2003. Producing and consuming chemicals: The moral economy of the American lawn. *Economic Geography* **79**: 425–51.

Rodriguez LF. 2006. Can Invasive Species Facilitate Native Species? Evidence of How, When, and Why These Impacts Occur. *Biological Invasions* **8**: 927–39.

Ruiz-Villanueva V, Díez-Herrero A, García JA, *et al.* 2018. Does the public's negative perception towards wood in rivers relate to recent impact of flooding experiencing? *Science of The Total Environment* **635**: 294–307.

Saidi N and Spray C. 2018. Ecosystem services bundles: challenges and opportunities for implementation and further research. *Environmental Research Letters* **13**: 113001.

Saito Y. 2014. Future Directions for Environmental Aesthetics (M Drenthen and J Keulartz, Eds).

Sass CK, Lodder RA, and Lee BD. 2019. Combining biophysical and socioeconomic suitability models for urban forest planning. *Urban Forestry & Urban Greening* **38**: 371–82.

Schirpke U, Timmermann F, Tappeiner U, and Tasser E. 2016. Cultural ecosystem services of mountain regions: Modelling the aesthetic value. *Ecological Indicators* **69**: 78–90.

Shackleton RT, Shackleton CM, and Kull CA. 2019. The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management* **229**: 145–57.

Sisser JM, Nelson KC, Larson KL, *et al.* 2016. Lawn enforcement: How municipal policies and neighborhood norms influence homeowner residential landscape management. *Landscape and Urban Planning* **150**: 16–25.

Sogge MK, Sferra SJ, and Paxton EH. 2008. Tamarix as Habitat for Birds: Implications for Riparian Restoration in the Southwestern United States. *Restoration Ecology* **16**: 146–54.

Southon GE, Jorgensen A, Dunnett N, *et al.* 2017. Biodiverse perennial meadows have aesthetic value and increase residents' perceptions of site quality in urban green-space. *Landscape and Urban Planning* **158**: 105–18.

Spirn AW. 2014. Ecological urbanism: A framework for the design of resilient cities (2014). In:

The ecological design and planning reader. Springer.

Sterling EJ, Betley E, Sigouin A, *et al.* 2017. Assessing the evidence for stakeholder engagement in biodiversity conservation. *Biological Conservation* **209**: 159–71.

Suh DH, Khachatryan H, and Guan Z. 2016. Why do we adopt environmentally friendly lawn care? Evidence from do-it-yourself consumers. *Applied Economics* **48**: 2550–61.

Tarrant MA and Cordell HK. 2002. Amenity values of public and private forests: Examining the value-attitude relationship. *Environmental Management* **30**: 692–703.

Thompson IH. 2002. Ecology, community and delight: a trivalent approach to landscape education. *Landscape and Urban Planning* **60**: 81–93.

Vejre H, Jensen FS, and Thorsen BJ. 2010. Demonstrating the importance of intangible ecosystem services from pen-urban landscapes. *Ecological Complexity* **7**: 338–48.

Velarde MaD, Fry G, and Tveit M. 2007. Health effects of viewing landscapes – Landscape types in environmental psychology. *Urban Forestry & Urban Greening* **6**: 199–212.

Visscher RS, Nassauer JI, and Marshall LL. 2016. Homeowner preferences for wooded front yards and backyards: Implications for carbon storage. *Landscape and Urban Planning* **146**: 1–10.

Vlami V, Kokkoris IP, Zogaris S, *et al.* 2017. Cultural landscapes and attributes of “culturalness” in protected areas: An exploratory assessment in Greece. *Science of the Total Environment* **595**: 229–43.

Wolch JR, Byrne J, and Newell JP. 2014. Urban green space, public health, and environmental justice: The challenge of making cities "just green enough." *Landscape and Urban Planning* **125**: 234–44.

Xu C, Haase D, Pribadi DO, and Pauleit S. 2018. Spatial variation of green space equity and its relation with urban dynamics: A case study in the region of Munich. *Ecological Indicators* **93**: 512–23.

Zanten BT van, Verburg PH, Espinosa M, *et al.* 2014. European agricultural landscapes, common agricultural policy and ecosystem services: a review. *Agronomy for Sustainable Development* **34**: 309–25.

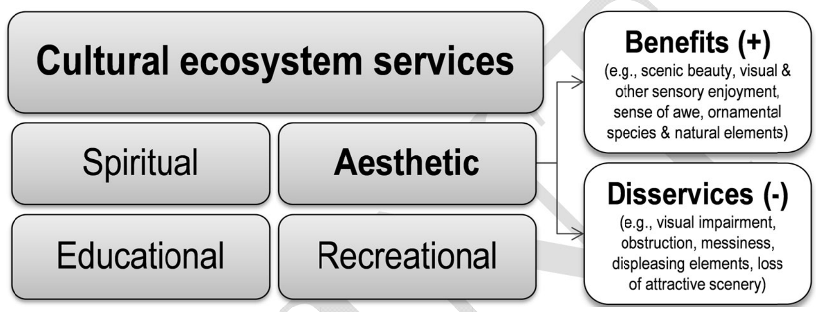
Zanten BT van, Verburg PH, Scholte SSK, and Tieskens KF. 2016. Using choice modeling to map aesthetic values at a landscape scale: Lessons from a Dutch case study. *Ecological Economics* **130**: 221–31.

Boxes and Figures

Box 1

Box 1. The concept of aesthetic ecosystem services.

Millennium Ecosystem Assessment (MEA) designated aesthetic ecosystem services (AES) under "Cultural and Amenity Services" (De Groot *et al.* 2005), which were further formalized in the Common International Classification for Ecosystem Services (CICES). In the most recent version 5.1 of CICES (Haines-Young and Potschin-Young 2018), AES are associated with the code 3.1.2.4 as "characteristics of living systems that enable aesthetic experiences", with a short descriptor "the beauty of nature".



The concept of AES is differentiated from recreational, spiritual, educational and other types of cultural services by explicitly considering aesthetic experiences associated with sensory perception of environment and species (De Groot *et al.* 2005; Daniel *et al.* 2012; Milcu *et al.* 2013). Such experiences may involve both *benefits* and negative effects, or *disservices:*

- *Aesthetic benefits* may be derived from scenic and attractive environmental qualities, ornamental appeal of species and natural elements, and also non-visual sensory pleasure, such as sense of awe and wonder when experiencing nature.
- *Aesthetic disservices* can result both from a particular state of a landscape (presence of displeasing landscape elements, visual messiness and untidiness) and degradation or loss of aesthetic quality due to natural or human-induced processes (e.g., disturbance, pest outbreaks, scenery-obstructing elements).

List of Figures

Figure 1. Representation of aesthetic and cultural ecosystem services (AES and CES, respectively) in a sample of studies focusing on service bundling: a) differential inclusion of AES in studies focusing on any versus cultural-only services ("entries" are individual or combined papers with unique ES sets and study context), and b) number of CES and non-CES per paper depending on emphasis and inclusion of AES (details on the literature sample and search are provided in the Supplementary Material).

Figure 2. Examples of factors contributing to wicked problems associated with provisioning of aesthetic ecosystem services (AES) or their loss in efforts targeting other management objectives.

Figure 3. Examples of effects and implications involved in the general use of non-native plant species in landscaping.

Figure 1

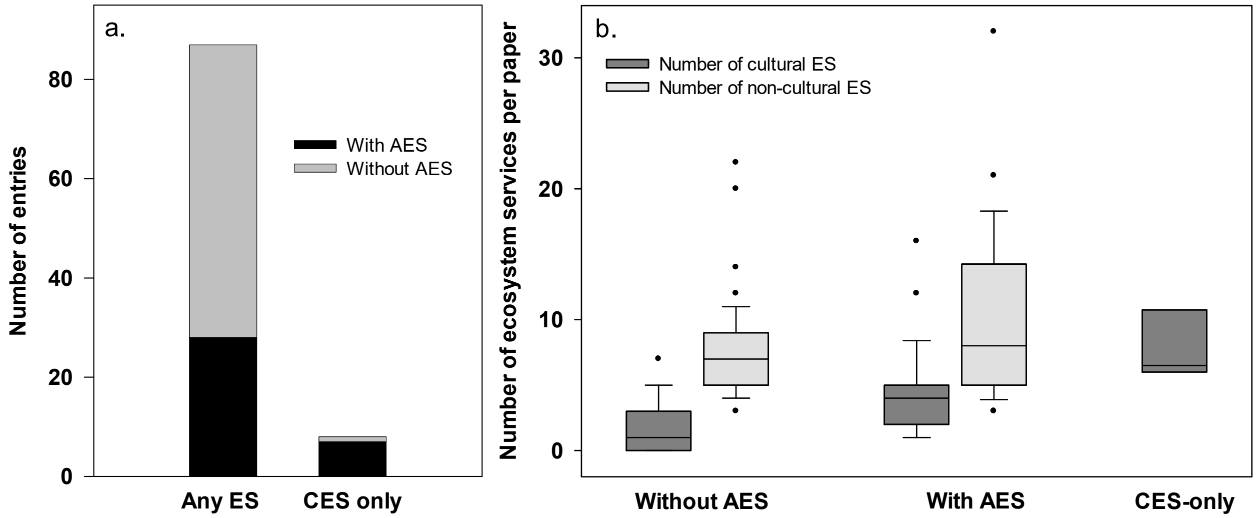

a.
Number of entries
0
20
40
60
80
With AES
Without AES
Any ES
CES only
b.
Number ecosystem services per paper
0
10
20
30
Number of cultural ES
Number of non-cultural ES
Without AES
With AES
CES-only

Figure 2

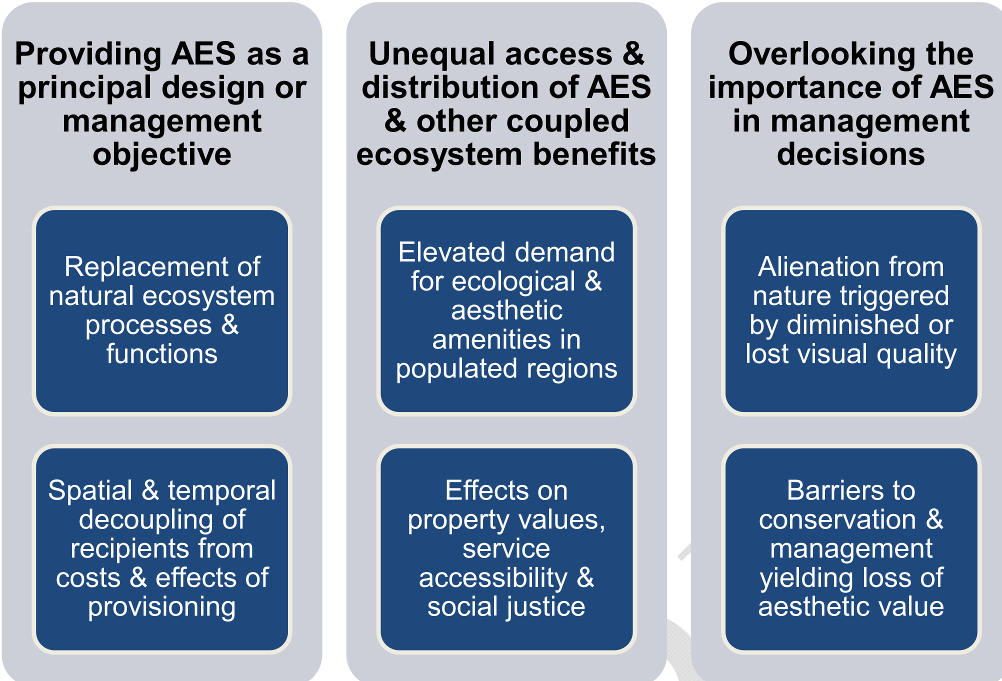
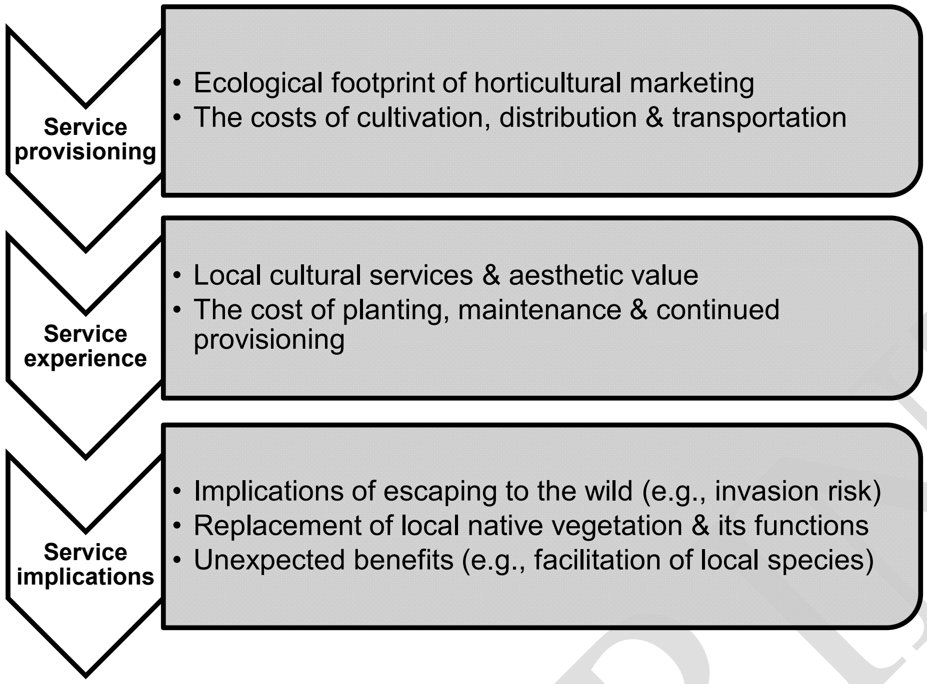

Providing AES as a principal design or management objective
Replacement of natural ecosystem processes & functions
Spatial & temporal decoupling of recipients from costs & effects of provisioning
Unequal access & distribution of AES & other coupled ecosystem benefits
Elevated demand for ecological & aesthetic amenities in populated regions
Effects on property values, service accessibility & social justice
Overlooking the importance of AES in management decisions
Alienation from nature triggered by diminished or lost visual quality
Barriers to conservation & management yielding loss of aesthetic value

Figure 3


Service provisioning
• Ecological footprint of horticultural marketing
• The costs of cultivation, distribution & transportation
Service experience
• Local cultural services & aesthetic value
• The cost of planting, maintenance & continued provisioning
Service implications
• Implications of escaping to the wild (e.g., invasion risk)
• Replacement of local native vegetation & its functions
• Unexpected benefits (e.g., facilitation of local species)