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Ecosystem services in practice: Challenges to real world implementation of ecosystem services across multiple landscapes – A critical review

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A B S T R A C T

Ecosystem services (ES) and ecosystem services assessment (ESA) have become common parlance in the environmental field. Scientists, policy-makers and activists have promoted the ES approach as a means of conveying the extent of threats to natural ecosystems with the goal of crafting socially acceptable and effective policy to address ecological threats and biodiversity conservation. Yet there are some significant challenges to wide acceptance of the ES approach which hinder its absorption into the mainstream geography literature. This paper reviews the historical development of the ES approach focusing on its relevance to applied geography at different stages of its development, describes the present state-of-the-art of ES, and synthesizes the results from several seminal papers and reports. I posit that there are two major stumbling blocks: 1) the difficulty of simplifying complexities between services so that statutory planning processes can incorporate the approach, and 2) the lack of cross-landscape assessment methods and examples. If we focus on the most immediately surmountable challenges to the ES approach much progress could be made in a short time. The subsequent and final substantive section of this review summaries these challenges and offers some suggestions for moving forward.

Introduction

Ecosystem services (ES) and ecosystem services assessment (ESA) have become common parlance in the environmental field. ESA which provides a way to anticipate the effects of impending changes, has become one of the most prominent frameworks for spatial planning and land use management. The connection of the approach to geography has to do, first and foremost, with the influences of location on the characteristics of services. Yet despite growing attention to ES in geography literature much of it has failed to achieve broad, general appeal.

Among the reasons why the field of geography should concern itself with ES and ESA is because almost all ESAs are dependent on the mapping of ES and on the use of geographic information system (GIS) tools. Further, the evolution of ESA has engendered the broadening of the definition of ES and tools for ESA to account for as many services as possible. It is hoped that this will facilitate reaching the full potential of the approach for conservation as intended by environmental professionals. As such, the ES approach has come to depend on the many sub-fields of geography, including socio-cultural geography, economic geography and biogeography. Even historical geography has relevance for ES; historical environmental conditions can determine current or future services. For example, identification of historical extents of species' dispersal and species' natural habitats aid in assessing present and future additions or losses to ES (Moilanen et al., 2005).

Scientists, policy-makers and activists have promoted the ES approach as a means of conveying the extent of threats to natural ecosystems with the goal of crafting socially acceptable and effective policy to address ecological threats. Environmental geography should recognize and integrate between the fields of spatial ecology and geography to support the practical application of ES as a "language" for environmental protection. According to some environmentalists, ES is the last great hope for making biodiversity and environmental conservation a priority for planning and resource management.

This article looks critically at some of current challenges to the ES approach, challenges that are that are obstacles to its absorption in the mainstream geography literature. Specifically what are the impediments to ES becoming accessible to the widest possible audience, from academics to professionals to laymen? I posit that there are two major stumbling blocks: 1) the difficulty of simplifying complexities between services so that existing institutions

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(including statutory planning) can incorporate the approach, and 2) the lack of cross-landscape unit assessment methods and examples. This paper reviews the historical development of the ES approach focusing on its relevance to applied geography at different stages of its development. It also describes the present state-of-the-art of ES, and synthesizes the results from several seminal papers and reports related to the current challenges mentioned above.

Overview of historical background and evolution

The ES approach rose to prominence starting in the early the 1980s even though references to valuing benefits of natural ecosystems can be found in many earlier publications from various fields (e.g., King, 1966; Ryther, 1969). Within the field of the ecology, there has been recognition of the value of provision of services, functions and structures of ecosystems practically since the field materialized. As early as 1948, Rachel Carson alluded to these services when writing simply about wildlife conservation: “For all people, the preservation of wildlife and of wildlife habitat means also the preservation of the basic resources of the earth, which men, as well as animals, must have in order to live. Wildlife, water, forests, grasslands – all are parts of man's essential environment”(Carson, 1948).

In recent years the ES approach has become a conceptual and empirical link between ecological health and human wellbeing and a vehicle with which to communicate the importance of nature conservation to policy makers and the general public (Carpenter et al., 2009; Collins et al., 2011; Daily, Kareiva, Polasky, Ricketts, & Tallis, 2011; de Groot, Alkemade, Braat, Hein, & Willemen, 2010). The approach has evolved significantly over the past three decades while exhibiting advances on a number of fronts throughout this period.

In the early days, following the publication of seminal breakthrough articles, particularly *Nature's Services* (Daily, 1997) and a cover story in *Nature* (Costanza et al., 1997), ecosystem services became hot news. Stories broke through to the media on the importance of ecosystem services and were featured shortly thereafter in *Newsweek* and in *The New York Times*, on radio talk shows and even in a segment on television's “Nightline” (Salzman, 1998). Developments from various fields led to this broad interest and similarly emanated from the innovative synthesis of economics and ecology (Pimm, 1997). Such developments fit well with neoliberal economics gaining ground in the last decade of the last century. They included the maturation of the field of conservation planning that incorporated the premise that specific regions, areas and landscape types can be clearly valued more than others (e.g., Olson & Dinerstein, 1998). This has close connections to the geographic-dependent, place-based, spatial prioritization techniques with foundations in biodiversity conservation that have continued to gain ground and are contributing to work on restoration ecology, complementarity and resilience (Moilenan, Wilson, & Possingham, 2009).

Other subsequent developments have been the extensive work on particular services, such as crop pollination (e.g., Kremen, Williams, Bugg, Fay, & Thorp, 2004), water flow and hydropower production (e.g., Guo, Xiao, & Li, 2000), and recreation (Naidoo & Admowicz, 2005). A ground-breaking advance on the institutional and social change front has come from the emergence worldwide of small-scale systems of payments for ecosystem services (Food and Agriculture Organization (FAO), 2004). However such arrangements require motivated sets of resources managers to participate in such policies (van der Horst, 2011) and for scaling up, they require complex diplomacy and broad consensus of how to go about ESA.

Another aspect, one related closely to developments in the field of geography, has been the increasing use of GIS and other geospatially advanced methods of analysis for ESA. This includes various types of digital cartography, remote sensing, photometric image analysis, and technologies such as simulation visualization and augmented reality that can be adapted for application to spatial problems. Over the past three decades, GIS applications have become basic tools accessible to professionals beyond highly trained geographers. There have been great improvements in recent years in computer software and hardware, spatial databases and targeted applications that have the facilitated implementation of ESA. One prominent example of the latter is InVest developed by the Natural Capital Project (see Nelson et al., 2009). Generally, such GIS applications, specifically developed for ESA, support overlay analysis that combines data layers, developed through the use of complex modeling algorithms, into composite maps (Hinojosa & Hennermann, 2012; Ng, Xie, & Yu, 2013; Norman, et al., 2012; Sherrouse, Clement, & Semmens, 2011). Temporal and spatial trajectories can then be applied and adjustments made as new information becomes available.

Parallel to these advances, a number of seminal reports catapulted the ES approach into the mainstream of conservation planning and helped propel the concept to prominence in the academic and policy-making communities. The most influential report has been the Millennium Ecosystem Assessment (MEA) carried out during 2001–2005 under the auspices of the United Nations (UN). Its mandate was to establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being. It provides management options and future scenarios for policy makers to consider.

Other initiatives that have made substantial contributions to the ES approach are the work of the Economics of Ecosystem and Biodiversity (TEEB) group. With offices and support coming from the UN Environmental Program (UNEP), TEEB is a global initiative focused on drawing attention to the economic benefits of biodiversity. TEEB's first influential document was the interim report of their study on the economic significance of the global loss of biological diversity published in 2008 (TEEB, 2008). Their work has led to progress on ES such as a standardized classification scheme for valuation (mentioned below) being discussed in the context of the System of Environmental-Economic Accounts of the UN Statistical Division (Haines-Young & Potschin, 2010).

These efforts were accompanied by the concurrent establishment of the UN Intergovernmental Platform on Biodiversity and Ecosystem Services in 2010, and the increase in national-scale ES assessments like those in Great Britain (UK National Ecosystem Assessment, 2011) and Japan (Japan Satoyama Satoumi Assessment, 2010). Recently the European Community has called on its member states to map and assess the state of ecosystems and their services in their national territories with the assistance of the European Commission (Action 5 of the EU Biodiversity Strategy to 2020).[1]

A new discussion paper that serves as a toolkit for implementation of the ES approach (European Commission, 2013) proposes a typology of ecosystems to be assessed and mapped. It also proposes the use of the Common International Classification of Ecosystem Services (CICES) developed for environmental accounting purposes. This is an important step for progress towards a

[1] Action 5 of the Biodiversity Strategy requires Member States to map and assess the state of ecosystems and their services in their national territories by 2014 and to promote the integration of these values into accounting and reporting systems at national levels and at the EU level by 2020 (European Commission, 2013).

common "language" of ES for practical and professional use. As described above there have been some significant advances in the science, economic valuation, institutional design, and social capacity needed for ecosystem-service conservation. However, despite these advances, the importance of ES in view of threats to services and the flood of literature on the topic (see Orenstein & Groner, 2013), leaders have been slow to incorporate the approach into decision making (Chan, Shaw, Cameron, Underwood, & Daily, 2006; Daily et al., 2009) and within frameworks of existing regulatory institutions (Ruhl, 2010b; van der Horst, 2011) (Fig. 1).

Advantages and disadvantages of the ES approach

The vast literature on ES and methods for its assessment provide ample justification of the approach as the "last, best hope for making conservation mainstream – attractive and commonplace worldwide" (Daily et al., 2009). The ES approach has advantages that have contributed to its ubiquitous treatment in ecological research with spillovers to other professions, including geography and urban and regional planning. Among the lesser discussed advantages are its integrative foundations, its compatibility with other resource management tools and its acceptability to the general public because in capitalistic societies money (value) "talks". That is not to say that the approach is without disadvantages; some of these hinder progress as discussed below.

In an article on use of the ES approach within the urban planning context, Kohsaka (2010) contends that an advantage to the use of the ES approach is that it encourages going beyond the species level of biodiversity (a general goal) such that biodiversity can be better defined on the micro or site level. An important tenet for *environmental planning and management* is integration and one of the dimensions of integration is that between disciplines (Portman, 2011). The ES approach requires the coupling of natural and human systems through the infusion of multidisciplinary perspectives.

The ES approach has wide appeal for application to what are considered "wicked" problems (see Randolph, 2011) first and foremost the loss of biodiversity. To address such complex problems, the coupling of natural and human systems is called for and this is in fact the essence of the ES approach. ES has been described as a "core concept of the rapidly developing interdisciplinary field of *ecological economics*" (Kozak, Lant, Shaikh, & Wang, 2011). Other authors contend that the ES approach goes far beyond inter-disciplinarity (research between disciplines); it requires deep knowledge within—and across—multiple disciplines (Burkhard, Petrosillo, & Costanza, 2010; Daily et al., 2011).

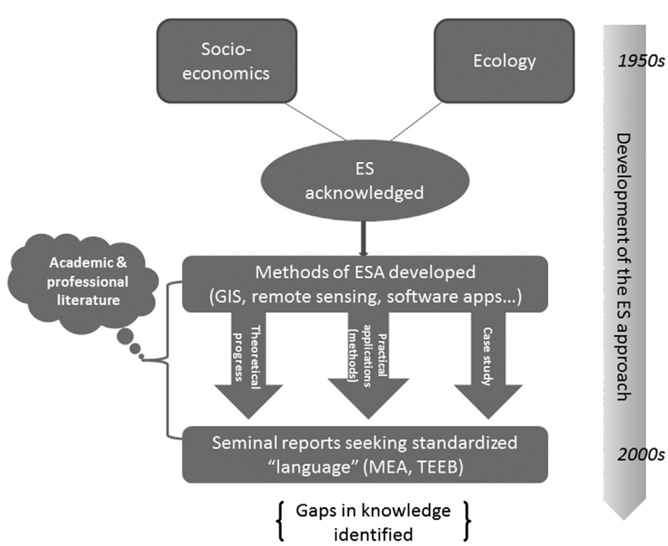

Fig. 1. A schematic illustrating the progress in the ES approach over time as described.

Expertise from three disciplines is usually required for ESA: ecology, economics, and sociology. Ruhl (2010b) suggests that the three are ecology, economics and geography; the latter because one needs to know where the services are located. Chan et al. (2006) contends that use of the approach requires expertise in biology, chemistry, physics, economics, finance, geosciences, geography, and particular analytical tools. The integration of theoretical understanding and empirical expertise from these diverse fields requires a multidisciplinary team of experts working in close communication, spearheaded by trans-disciplinary scholars and practitioners (Chan et al., 2006).

As mentioned, the ES approach can be readily used in conjunction with other environmental planning tools. For example, Kohsaka (2010) writes that the "DPSIR" model became one of the most dominant modifications used in discussions on ecosystem services. Other case studies on ES integrate participatory GIS (PPGIS) methods (Delphi method and Analytic Hierarchy Process) (Nahuelhual, Carmona, Lozada, Jaramillo, & Aguayo, 2013), models of cultural geography (van der Horst, 2011) and economic valuation models (i.e., stated or revealed preferences) (Kozak et al., 2011).

In a recent review Lester et al. (2010) describe ecosystem services as a critical research area for a transition to the ecosystem-based management (EBM) model promoted to tackle the declining state of oceans and coasts. EBM, widely accepted as a best practice for the burgeoning field of marine spatial planning, is defined as an *integrated* approach to management that includes humans within it (Ehler & Douvere, 2009). The goal of EBM is to maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the *services* humans want and need (McLeod, Lubchenco, Palumbi, & Rosenberg, 2005) and thus it fits well with the ES approach.

One of the draws to the use of a system founded on values, whether or not these are monetary, is that they are generally understandable to layman. Once explained, ES are understandable to professionals in myriad fields and to the public at large. TEEB distinguishes three main types of ecosystem-based benefits and related values: (i) ecological benefits and values, (ii) sociocultural benefits and values, and (iii) economic benefits and values (TEEB, 2008). Challenges lie in figuring out what these values (relative or absolute) are, and in linking the spatial and temporal dimensions of ES to them. But the overall principle is clear: nature has value.

Although understood, the ES approach is not without its dissenters. The approach draws considerable criticism on both theoretical and empirical grounds, ranging from ethical concerns regarding the commodification of nature (Luck et al., 2012) to suggestions that economic ES assessment reflects a neoliberal fetishism (Kosoy & Corbera, 2010). Empirically, various global assessments are heavily skewed toward natural science disciplines (especially the earth sciences) and the MEA was no exception. Social science content is limited and heavily dominated by economics, the social science that is most dedicated to 'future scenarios' which are usually unknown and debated. Further knowledge about socio-economic values are both hard to come by and expensive.

Even where it seems like economic data would be easy to obtain, there are many challenges. For example, commercial fisheries landings data, which include information on the volume and ex-vessel value of catch are readily available from the US National Oceanic and Atmospheric Agency. However, these are recorded at relatively coarse and ecologically inappropriate spatial scales for reasons that range from concerns for confidentiality of fishing locations to constraints on agency data quality and management. These data are useful starting points that require additional information and analysis to be translated into economic and social value to people for use in ESA.

Evolving concepts and controversy

The Millennium Ecosystem Assessment (MEA, 2005) was the first to classify "ecosystem services" into four categories: provisioning, regulating, cultural and supporting. The use of these categories has been found confusing due especially to difficulties identifying "support" services. Wallace (2007) argues that the four classifications are inadequate because they mix ends with means, whereas the "end" is the service provided and the means are the supporting services.

This problem has also been noted by Granek et al. (2009): although ecosystem services depend on ecosystem functions, the two are not synonymous. An ecosystem service is something that benefits people. Without human demand for a given ecosystem function, there is no ecosystem service. The term "function" can be used instead of support service. It is likely that authors of the Millennium Ecosystem Assessment anticipated this problem with supporting services. Some parts of the Millennium Ecosystem Assessment report intentionally leave out "supporting services" (for example, Table 1 of the report), explaining that they are a category unto themselves since they are not used directly by people (MEA, 2005).

Some suggest that this problem can be circumvented by incorporating the "service providing unit" (SPU) concept. So far applied only to the terrestrial environment, the concept refers to the quantification of biological components of a given ecosystem that ultimately support human activities (Cognetti & Maltagliati, 2010). But what about abiotic components of the ecosystem that support human activities? The more recent Proposal for a Common International Classification of Ecosystem Services (CICES) proposes "distinguish[ing] the material and energetic outputs from ecosystems as 'goods' and the non-material outputs as 'services'.

So we see that the ES approach is still evolving even though its basic usefulness is clear. The utility of the concept for landscape management depends in part on our ability to understand the links between landscape structure, movement of organisms and materials through this landscape, and the subsequent provision of multiple ecosystem services (Mitchell, Bennett, & Gonzalez, 2013). By many accounts, much of our knowledge about aspects of ecosystem services and their assessment remains rudimentary and therefore the effective management of varied landscapes to provide and protect ES is curtailed (Daily et al., 2009; Kremen, 2005; Nicholson et al., 2009; Tscharntke, Klein, Kruess, Steffan-Dewenter, & Thies, 2005).

This perspective coincides with the claim that there is a lack of empirical data to move forward with ESA on a wider scope. Nicholson et al. (2009) contend that the main problems with ES are linked to the failure of scientists to engage fully in the interdisciplinarity of the approach, to address feedbacks between and within both social systems and ecosystems, and to incorporate uncertainty about thresholds and underlying processes. In contrast, Lester et al. (2010) claim that available science is not the bottleneck to moving forward with the use of ES, at least not in the ocean environment. This is surprising in that ESA for the ocean environment is considered less advanced than ESA for the terrestrial environment (Cognetti & Maltagliati, 2010).

Thus it is clear that ES should be multi-disciplinary; the question is whether it succeeds in being so. This relates to the second point discussed in the next section highlighting insufficient ES studies that cross varied landscape units. Most ES papers are also case study specific or scenario based and therefore fail to have a general, universal application. For example, of the ten articles on ecosystem services published in *Applied Geography* in the last three years, seven were either scenario based or case-specific. Most ES papers are published in ecology journals and therefore fail to reach wide exposure to professionals reading planning, geography and general social-science or economics literature. Legal and policy studies on the approach are also needed. But if we put controversy aside, such as that regarding supporting services, and focus on the most immediately surmountable challenges much progress on ES could be made in a short time. The subsequent and final section of this review summaries these immediate challenges and offers some suggestions for moving forward.

Current challenges

Two fundamental advances are needed to sustain and expand the pioneering efforts that are currently underway to broaden the appeal of ES and to give the approach weight in decision making. First, the scientific community needs to deliver the knowledge and tools necessary to forecast and quantify the return from ES, especially through mapping tools, and to aid professionals in explicitly and systematically integrating this knowledge into institutional frameworks. Without these advances, the value of nature will remain an interesting idea, represented in dispersed, limited, and idiosyncratic efforts.

An aspect neglected despite the growing ES literature and the increase in practical applications reviewed and presented is how to scale-up so as to cover greater areas of land and seascape, and to encompass complex and extensive resource systems. This leads to the second necessary advance: the development of methodologies for ESA that can cross of landscape units and be incorporated in decision-making among multiple jurisdictions and at levels of government. Local-scale efforts benefit from being able to effectively engage local communities, yet they lack the purview to drive larger scale protection efforts or to manage ecosystem services in ways that incorporate multiple cumulative impacts that change as ecosystem types vary.

Much has been written about the need for indicators to monitor key services, a problem confounded across landscape units. This is not unlike the need for surrogates to conduct systematic spatial conservation planning (McArthur et al., 2010; Moilenan et al., 2009). Literature reviews have also addressed the lack of social science information *vis a vis* natural science (i.e., biophysical) information on services (e.g., Lester et al., 2010). Yet much of the literature on ES depends on example case studies in particular regions of the world, within a set decision-making context and within limited, bounded landscape units. Similarly much of the recent literature dealing with the legal aspects of ES tends to be service specific (e.g., public lands, watersheds, migratory species) (Hirokawa, 2012; Reynolds & Clay, 2011; Ruhl, 2010a, 2010b). This reinforces the default to treatment of discreet landscape units and specific jurisdictional and institutional contexts and limits broad appeal of the approach as discussed below.

Difficulties incorporating ES into existing regulatory programs

Making ES relevant for policy-making requires an understanding of the interlinked production of services, a good sense of the decision-making processes of individual stakeholders, more research into institutional design and policy implementation, and the introduction of experimentally based policy interventions designed for performance evaluation and improvement over time. To bring about a change in decision making, the values of natural capital must be embedded in institutions (Daily et al., 2009). Without institutional change, communities may well continue to carry on with behaviors that are known to be harmful to society over the long term (e.g., overfishing, intense use of fossil fuels). The right institutions can create incentives, so that the decisions made by individuals, communities, corporations, and governments promote widely shared values.

Take for example applied methodologies recently developed as case studies using advanced geo-spatial mapping. One presents a method for valuing ES based on an area-based functional connectivity index – the "possibility of connectivity". The index is applied to the Shenzhen River watershed to examine the spatial and temporal dynamic of changes in value of ES (Ng et al., 2013). Another case presents a method for assessing ES based on perceptions of development on socio-economic conflicts. The authors apply the method to an area subject to gas exploitation in southern Bolivia (Hinojosa & Hennermann, 2012). A third case makes innovative use of a socio-economic and demographic index – the Modified Socio-Environmental Vulnerability Index – which the authors use to map disparities in environmental risks throughout a bi-national watershed along the US-Mexico border. The authors use the Soil and Water Assessment Tool (generally used to assess erosion and flood potential under different development scenarios for reference to show the advantage of their proposed method which can highlight important environmental justice concerns related to ES (Norman et al. 2012). These papers have much to contribute especially in their use of GIS techniques for synthesizing qualitative and quantitative information and coupling of socio-economic data with physical geographic data. However, they fall short of outlining how to integrate such methods into existing regulations, in essence what is needed to synthesize methods with policy.

Fisher et al. (2008) conducted a survey of the literature on ES and administered a questionnaire to researchers regarding how ecosystem service research can be integrated into the policy process. As for all aspects of the ES approach, the crossing of disciplines is essential. They concluded that integration of economic concepts during ESA would make ecosystem service research more immediately policy relevant. The introduction of economic concepts could aid in the distinction between services and benefits, in understanding the importance of marginal ecosystem changes and in formalizing the idea of a safe minimum standard for ecosystem service provision (Fisher et al., 2008).

Ruhl (2010a) cites both regulations and litigation that incorporate ecosystem services but also points out that such examples are few. To 2010, the only example at the US federal level of the emergence of an environmental services market was the 2008 Farm Bill (Ruhl, 2010a). The 2008 federal Farm Bill requires the US Department of Agriculture to "establish technical guidelines that outline science based methods to measure the environmental service benefits from conservation and land management activities…" Farm advocacy groups have been unsuccessful at convincing lawmakers to formalize ES markets in subsequent reauthorizations of the Farm Bill. Such markets would enable the trading of "credits" for individual units of an ecosystem service by allowing entities that need to reduce their environmental impacts (such as wastewater utilities) to buy credits from entities (such as farms and ranches) that voluntarily reduce their own impacts (Achterman & Mauger, 2010; Ruhl, 2010a, 2010b).

The existence of science-based methods and technical guidelines such as those required by the Farm Bill is no guarantee that recommendations will be adopted. Granek et al. (2009) point out that incorporating ecological and socio-economic data into management and policy decision making is challenging because scientists fail to translate their results into a form that is useful and easily understood by those with different levels of scientific expertise. Furthermore, the pace of knowledge development in science (typically slow) and the timeline on which policy makers and managers need information (typically fast) do not match.

Yet there are some important examples where markets are starting to take hold. According to Achterman and Mauger (2010), US states and other countries are looking to the US Pacific Northwest for lessons on establishing ecosystem service markets because in those states, particularly Oregon, there have been important successes. Some of these local success stories (in the Oregon cases called "ecological service markets") need to be scaled-up in order to achieve greater multi-disciplinarity and cover more services in landscapes of varied types. Larger-scale governments are likely better equipped to achieve this (Achterman & Mauger, 2010).

Despite the attention the ES approach receives in academic and professional literature, leaders in both the private and public sectors have been slow to support incorporating ES into policy and institutions. This issue traces to complex factors, but at the core is the poor characterization of the flow of services in the necessary biophysical and economic terms at the scales most useful to decision makers (Chan et al., 2006). The availability of specific, scattered, local-condition ES case studies that lack universal relevance are often reasons to forgo the use of the approach altogether.

Crossing landscape units – marine vs. terrestrial

The problem of crossing landscape units is not a challenge unique to ecosystem services (e.g., Beger et al., 2010), however, solving it becomes essential for rendering the approach accessible to a wide audience. ESAs often make use of values on a continuum that are relative to each other. The continuity of undisturbed land or sea-scapes is also important to ESA. Mitchell et al. (2013) conduct a semi-quantitative review of the literature that investigates how landscape connectivity affects the provision of specific ES. They emphasize the importance of connectivity and identify gaps in the connectivity-ecosystem services literature. Gaps include a lack of multiple service studies which precludes identification of tradeoffs between services as connectivity changes (Mitchell et al., 2013).

Early on, Costanza et al. (1997) estimated the annual worth of ecosystem services to be at least 33 trillion US dollars. About 63% of this value was estimated to be contributed by marine ecosystem services, with most coming from coastal systems ($20.9 US trillion per year). Terrestrial systems were estimated to make up about 38%, mainly from forests and wetlands. Despite estimates of ES values for marine (especially near-shore) systems surpassing those of the terrestrial environment (Costanza et al., 1997; Turner et al., 2007), terrestrial ES-based concepts and service provision have been elaborated and developed, while studies on the marine environment have lagged behind (Cognetti & Maltagliati, 2010). This likely relates to the general problem of a lag in attention to marine biodiversity both in practical terms (see Irish & Norse, 1996; Norse & Crowder, 2005) and in the literature (Roach, 2003).

These gaps and lags are often inextricably linked to the disparities between marine ESA and terrestrial ESA, making continual assessments going from land to sea, almost impossible. Some organisms spend part of their life on land and part at sea. For example, anadromous fish live in different landscape units according to their life stages. A single ESA should ideally cover the continuum of these environments, taking into consideration appropriate spatial and temporal dimensions. Sanchirico and Mumby (2009) consider such phenomenon but stop short of defining methods for ESA for the purposes of *marine* development decisions; rather, they focus mostly on *landside* coastal development in species–habitat associations for mapping near-shore ES.

Marine ES were a late addition to the MEA of 2005 and have been subject to a situation of "catch-up" and "fit-in" since then. While there has been considerable support for the inclusion of marine services the overall dominance of "terrestrial" issues and expertise is clear (Cognetti & Maltagliati, 2010; UNEP, 2006). This has hindered the incorporation of both marine and terrestrial ES in single, unified assessments that cross terrestrial-coastal-marine landscape units. In addition to the expense and complexity of applying the ES to the marine environment, coastal ecosystem which lie at the interface between marine and terrestrial ecosystems provide an array of services and benefits to different groups (Granek et al., 2009). The coastal environment is well known as an interface that engenders intensified conflict between stakeholder groups (Sas, Fishhendler, & Portman, 2010).

Comparing characteristics of marine environments to those of terrestrial environments can help raise awareness of unique challenges for ESA in the marine environment. Marine systems have nebulous boundaries, large spatial scales and fine temporal scales. Consideration of the three-dimensional living space of organisms is essential for the marine environment as are relatively unstructured food webs, nonlinear system dynamics and finally the understanding that the marine environment is less well-studied than the terrestrial environment. Relative to marine systems, terrestrial systems have clear boundaries, small spatial scales, temporally coarse spatial scales, relatively two-dimensional living space, structured food webs, linear system dynamics and are better studied (Agardy, 2000).

Furthermore, elements of marine ecosystems are more transient than terrestrial. By and large marine areas are held in public trust or at least comprise areas of public domain without the constraints and opportunities that private property hold for ES protection and management. Marine data collection is more complex, more expensive and often not available to the public or even to research authorities. Remote sensing techniques for the marine environment are different and more sophisticated. For terrestrial ESA, remote sensing, GIS and image analysis easily provide measures of indicators such as Leaf Area Index (LAI) or Normalized Difference Vegetation Index (NDVI) (Nunes, Almeida, & Coelho, 2011). Satellite photos or flyovers can give good indications of various terrestrial services, whereas these types of straight forward imaging of the marine environment appear as monolithic watersheet.

Cognetti and Maltagliati (2010) highlight the differences between marine and terrestrial environments and call for unique ESA models for each based on these differences. For the latter, they envisage a matrix of a human-altered landscape with fragments of original biodiversity; conversely, for the marine model the matrix is represented by the original biodiversity with fragmented areas of human activities. Cognitti and Maltagliati's "original biodiversity" refers to original biotopes, for example areas of "original" habitat left adjacent to cultivated land that provide supporting services for honey bees. Whether or not one accepts this type of differentiation between models, it is clear that more work is needed, especially on the mapping of terrestrial-coastal-marine continuums.

The lack of adequate marine data is aggravated by problems of scale when the ES analysis progresses to mapping once ES indicators have been identified. The National Ocean Economics Program (http://noep.mbari.org) for example, compiled data from the Bureau of Labor Statistics into a database that allows users to search for annual contributions of ocean or coastal-based activities, including tourism, recreation, fisheries and transportation, to the economy at county and state scales. These estimates have important flaws, including underestimating fisheries employment and accruing value to land-based counties rather than showing where in the ocean these services are derived (Lester et al., 2010). Such problems hinder the use of ES approach in places where resources are relatively plentiful for data collection, data management and where research institutions are established and well-equipped for long-term study. Problems are vast on these fronts in developing countries or in parts of the world where regional cooperation for study is essential, but severely impeded due to lack of institutional capacity.

An influential article that bridges the two challenges highlighted here, reviews the environmental regulatory programs in the US such as the Oil Pollution Act, the National Environment Policy Act, the Clean Water Act, and the Endangered Species Act, in view of their ability to address ES. Some of these laws require those responsible for damage to ES to provide compensation, both physical and monetary, for benefits lost (Levrel, Pioch, & Spieler, 2012). Although Levrel et al. analyzes reactionary mechanisms and not proactive decision making institutions with bearing on policy, it is a welcome addition to the literature. It has broad implications, much greater than for the marine policy milieu in which it appears.

Unfortunately, the results of the research presented in Levrel et al.'s (2012) study show that the analysis criteria for the equivalencies between ecosystem services lost due to damage and ecosystem services gained due to compensatory measures are questionable. Also the compensation monitoring is for a relatively brief period of time and the data obtained may be insufficient for assessing the net effect of compensatory measures (Levrel et al., 2012). The weaknesses regarding the equivalencies and the uncertainty about the relevant time-scale can be counter-balanced by increasing the area of compensation, a problematic solution at best due to scaling-up and cross-landscape unit issues. In summary, much more work is needed both on the incorporation of ES in regulatory and planning institutions, particularly those that simultaneously treat terrestrial, coastal and marine landscape units.

Conclusion

Scientists and resource practitioners interested in promoting greater use of the ES approach face many challenges, with some of them yet to be defined. Two major stumbling blocks discussed here are the difficulty of integrating the ES into policy-relevant institutions, either existing or evolving, and the lack of cross landscape assessment methods and examples. The lag in development of marine ESA is particularly acute and in some cases, hinders the characterization of the flow of services in the necessary biophysical and economic terms at the scales most useful to decision makers.

Ecosystem services may in fact be the last great hope for making conservation mainstream. However, for this to really happen, the ecosystem service language needs to expand beyond the academic literature of ecologists and branch out to a variety of fields and professional activities. Despite all its complexity there are many basic advantages to the approach that should convince myriad professionals, especially those in place-based fields that have important spatial applications such as applied geography, that ecosystem services are relevant and deserve their attention.

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