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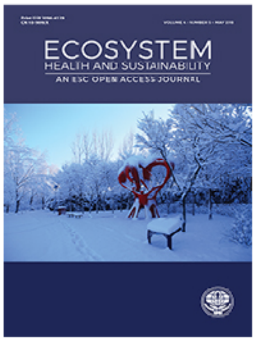

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REVIEW

Democratization of ecosystem services—a radical approach for assessing nature's benefits in the face of urbanization

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ABSTRACT

Objectives: (1) To evaluate how ecosystem services may be utilized to either reinforce or fracture the planning and development practices that emerged from segregation and economic exclusion; (2) To survey the current state of ecosystem service assessments and synthesize a growing number of recommendations from the literature for renovating ecosystem service analyses.
Methods: Utilizing current maps of ecosystem service distribution in Bushbuckridge Local Municipality, South Africa, we considered how a democratized process of assessing ecosystem services will produce a more nuanced representation of diverse values in society and capture heterogeneity in ecosystem structure and function.
Results: We propose interventions for assessing ecosystem services that are inclusive of a broad range of stakeholders' values and result in actual quantification of social and ecological processes. We demonstrate how to operationalize a pluralistic framework for ecosystem service assessments.
Conclusion: A democratized approach to ecosystem service assessments is a reimagined path to rescuing a poorly implemented concept and designing and managing future social-ecological systems that benefit people and support ecosystem integrity. It is the responsibility of scientists who do ecosystem services research to embrace more complex, pluralistic frameworks so that sound and inclusive scientific information is utilized in decision-making.

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Introduction

From its recent revitalization in the 1990s, the ecosystem services concept has been promoted as a powerful agent for expressing a wide range of direct and indirect benefits that humans derive from nature (Costanza et al. 1997; Daily 1997; Millennium Ecosystem Assessment 2005; Fisher, Turner, and Morling 2009; Braat and De Groot 2012; Costanza et al. 2014). Despite the theoretical value of this concept, its current applications have been challenged on a number of fronts. There is a plethora of literature providing in-depth critiques of ecosystem service assessments (Cooper et al. 2016). Key studies have shown that such assessments are plagued by the use of erroneous indicators of ecosystem function, a focus on ecosystem services that are easily quantified, simplistic assumptions employed in economic evaluations, and results biased toward the values of experts (Cowling et al. 2008; Raymond et al. 2009; Peterson et al. 2010; Braat and De Groot 2012; Lele et al. 2013; Hernández-Morcillo, Plieninger, and Bieling 2013; Verkerk et al. 2014; Kenter et al. 2016). In reviewing these critiques, one might conclude that the concept of ecosystem services has failed to become a useful instrument to link human and natural systems for planning, management, and policy.

In fact, in Costanza et al.'s (2017) latest review of ecosystem services, they conclude that practical applications of ecosystem services are still limited. They further elaborate on how scientists need to employ methods that overcome four main barriers to the effective implementation of ecosystem service assessments in management and policy. We need to: (1) establish consistent approaches to evaluating ecosystem services and (2) apply methods that adequately answer questions. Furthermore, bridging the science and policy gap will entail (3) accounting for the appropriate institutional frameworks; and (4)

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building trust among scientists and a broader community of stakeholders.

Transdisciplinary science is obviously needed to transcend these barriers. Transdisciplinary scientists can craft assessments of ecosystem services that provide a sound foundation for the development of conservation policies, planning, and decision-making (Cowling et al. 2008; Daily et al. 2009; De Groot et al. 2010). There is little doubt that a pluralistic model is urgently needed to improve the inclusiveness of ecosystem service assessments (Nahlik et al. 2012; Reyers et al. 2013; Costanza et al. 2017). Yet, this more inclusive approach is only a starting point and a paradigm shift is needed so ecosystem service assessments will be relevant in a highly urbanizing world. Take Africa, for instance, where urbanization is a predominant force, and where a 12-fold increase in urban land area is expected in the next 50 years (Angel et al. 2011). Of all the studies on ecosystem services globally, only a small fraction are conducted in Africa; meanwhile, across the continent there are not enough resources to support current livelihoods, and the distribution of valuable resources is constrained, and the availability of benefits from the available resources is highly heterogeneous (Wangai, Burkhard, and Müller 2016).

In a literature review of ecosystem service assessments in Africa, Wangai, Burkhard, and Müller (2016) found that most studies occurred at the regional scale, and did not address more local tradeoffs and synergies in ecosystem service provisioning. This may be consistent with Costanza et al.'s (2017) conclusions that we need better ecosystem service assessments that address questions relevant to management and policy; however, with urbanization pressure across the continent, the important question is, "how do scientists produce better ecosystem service assessments in the face of this massive change?" For example, analyses should not only focus on multiple scales, and particularly a local scale, if they aim to guide management and policy, but they must measure, monitor, map, and value ecosystem services at the relevant spatial resolution. Moreover, while it may be true that ecosystem analyses need to address the right institutional spaces, but they must also confront the historical injustices that are still strongly a part of institutional infrastructure (e.g., colonial influences, the legacy of apartheid in South Africa, and so on).

Our goal is to consider a way forward, and we posit that the field of urban ecology, in particular, may serve as a guide for a paradigm shift in scientific analysis of ecosystem services, as it is through the study of cities as complex social-ecological systems that a nuanced scientific understanding of heterogeneity relevant to ecosystem services has evolved (Cadenasso, Pickett, and Schwarz 2007; McHale et al. 2013; Pickett et al. 2017). We contend that acknowledging social and ecological heterogeneity, in both science and practice, is necessary to produce ecosystem service analyses that are both accurate and useful. Further, strategic adaptive management (SAM) can provide a model for long-term evaluation of ecosystem service assessments and the effects of decision-making on people and the environment.

In this paper, we first focus on Bushbuckridge, South Africa, a region where urbanization threatens to perpetuate historical social and environmental injustices. We evaluate how ecosystem services may be utilized to either reinforce or fracture the planning and development practices that emerged from segregation and economic exclusion. In the context of our case study, we evaluate the current state of ecosystem service assessments. Rather than contributing to the growing collection of critiques, we synthesize a number of adroit recommendations from the literature for renovating ecosystem service analyses (i.e., Cowling et al. 2008; De Groot et al. 2010; Nahlik et al. 2012; Reyers et al. 2013; Andersson et al. 2015; Reyers et al. 2015). From these, we propose interventions for creating a more pluralistic framework for assessing ecosystem services. Although there are a plethora of ecosystem service frameworks available in the literature (e.g., Tallis et al. 2008; Fisher et al. 2014), they routinely lack an implementation plan and therefore remain purely informative or theoretical (Nahlik et al. 2012). In contrast, we describe the steps needed to operationalize this democratized approach to ecosystem service assessments, to enhance its utility in policy, planning, and decision-making, based on experience in an actual dynamic urbanizing landscape.

Learning from current practices: ecosystem service assessments in Bushbuckridge Local Municipality (BLM), Mpumalanga Province, South Africa

BLM—a hot spot for biodiversity, population growth, tourism, and contentious land ownership debates

Bushbuckridge Local Municipality (BLM) is in the Mpumalanga Province of South Africa. This 10,250 km^2 area is nestled against Kruger National Park, one of the world's signature conservation areas (Figure 1). The BLM is a matrix of state forestry and conservation areas, communal lands, rural villages, and urbanizing centers surrounded by private game reserves and tourism facilities. More than 500,000 people live in the municipality, but only 11% of households have piped water (http://www.localgovernment.co.za/locals/view/142/bushbuckridge-local-

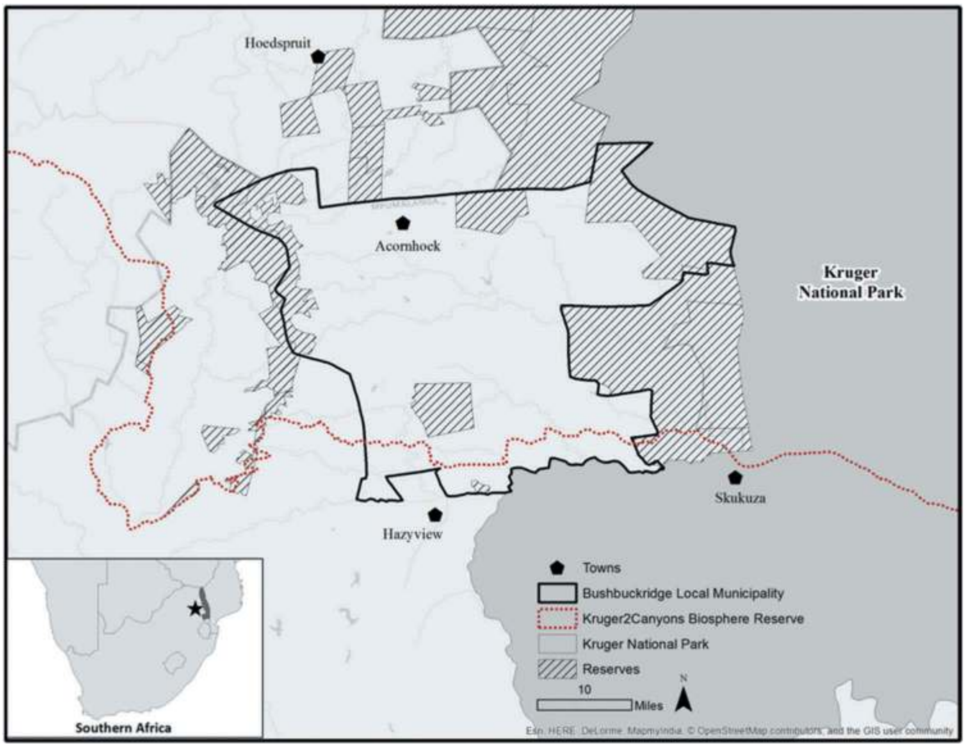

Figure 1. Bushbuckridge Local Municipality (BLM) is a highly urbanizing region of the Mpumalanga Province, South Africa. BLM is nestled against the border of Kruger National Park and surrounded by a host of other private game reserves in the region.

municipality). In fact, a majority of the households are still dependent on local natural resources to support their livelihoods (Twine et al. 2003; Kirkland, Hunter, and Twine 2007).

As a hot spot for biodiversity, human population growth, and tourism, BLM's municipal government is feeling pressure to manage natural resources and land use change. BLM is in the heart of the world's third largest biosphere reserve (designated by the United Nations Education, Scientific and Cultural Organization's Man and Biosphere Program) and an area with animals and plants of unique interest. The goal of designating a biosphere reserve is to promote sustainable use of natural resources; however, this ideal is not easily achieved in BLM. For instance, communal land in the area (i.e., state land under communal tenure) is managed by traditional leaders who are responsible for allocating areas for cultivation, grazing, and housing, and for regulating the use of communal natural resources. However, political, socio-economic, and cultural transitions, exacerbated by burgeoning human populations, are eroding this traditional power structure and its effectiveness (Twine 2005; Kirkland, Hunter, and Twine 2007). At the same time, the municipal government is primarily concerned with the delivery of basic essential services to people and the provincial conservation authorities who are responsible for environmental protection in the region are underresourced.

Further, household dynamics can play a significant role in determining the nature and flow of ecosystem services (Shackleton, Paumgarten, and Cocks 2008). There is a tendency in BLM toward decreasing number of people in each household. As socio-economic status rises with increasing urbanization, there is evidence that houses are getting larger, and the number of people living in them all year is decreasing, leading to disorganized and uncontainable growth of villages in the region. The changes in household size and configuration affect land cover and land use, natural and commodity resource use, waste generation and disposal, the type and management of domestic animals, and a host of other environmentally relevant processes and structures.

Making matters of land management more complex are ongoing land claims. Essentially, local communities are fighting legal battles for property that they claim was theirs historically, before apartheid and its land management policies forced people off of their land. For the most part, the land that is being disputed is dedicated to large farming operations, private game reserves, or conservation areas owned by the government (Kepe 2008). Contentious debates over land ownership and land use underlie almost every development and conservation effort in BLM. Its location along the western border of Kruger National Park only complicates the situation.

The wildlife economy and conservation planning in BLM

The BLM and the other municipalities along the western edge of the Kruger National Park also struggle with an exceptionally high unemployment rate (over 50% in BLM) (Bushbuckridge Local Municipality 2016). Some conservationists and environment-focused organizations contend that building a strong ecotourism-based "wildlife economy" will create jobs and preserve biodiversity (Hackel 1999; Hulme and Murphree 2001). Fundamentally, if the region were to fully invest in this wildlife economy model, much of the land that is currently held by local communities (i.e., communal lands), and managed in traditional ways, would likely be incorporated into private game reserves and other conservation easements, with nebulous forms of benefit-sharing between the reserves and the local communities. These land-for-conservation schemes, however, are quite controversial, since they are largely justified on the erroneous assumption that communal lands are degraded and provide few if any ecosystem services (Bushbuckridge Local Municipality 2016).

Converting these degraded lands into restored conservation areas that provide wildlife habitat and effectively increase the size of Kruger National Park seems enticing to some. In fact, though, residents of these communal lands use their land in many ways, and derive numerous ecosystem services from their land that support their livelihoods (Shackleton and Shackleton 2004). Further, a wildlife-economy-driven land management strategy may return to the colonial and Apartheid model where land is largely under the ownership or management of wealthy whites, leaving many in the black settlement communities vying for low paying jobs with unreasonable hours and subpar living conditions. Initiatives aimed at sharing the tourism profits from the incorporated communal lands run a high risk of being captured by local elites (Child and Barnes 2010). Ironically, those that have the most to lose in such deals, such as poor households heavily dependent on their communal landscapes for fuelwood, medicinal plants, wild foods, and animal husbandry, have the least power in negotiations.

Although there is little evidence that the wildlife economy is capable of being a win-win solution that balances human needs with conservation priorities, promoting conservation in BLM to support the health and well-being of both communities and the environment remains a utopian goal. Recently, the government (i.e., the Department of Environmental Affairs, DEA) hired a consulting company to create a Master Plan that was supposed to be an "integrated, multi-stakeholder sustainable development strategy for the Bushbuckridge area." Notably, the plan is titled, "*Growing a wildlife economy in Bushbuckridge*" and is focused on identifying areas worthy of future conservation efforts. The development strategies outlined in this BLM master plan are then legally implemented through the Integrated Development Plan for the municipality (i.e., Bushbuckridge Local Municipality 2016).

How maps guide future development in the BLM master plan

A series of maps are incorporated into the BLM master plan to help guide future development. All maps were created by a consulting firm external to the community. The first set of maps are mostly descriptive in nature, and primarily show the location of traditional authorities, conservation areas, basic vegetation types, and topography in the region (Figure 2). However, the final maps that present recommendations for creating corridors that support local conservation efforts (Figure 3) are informed by a map that shows the distribution of ecosystem services across the BLM (Figure 2).

A closer look at this ecosystem services map shows that areas providing "essential" or "very important" ecosystem services are conservation areas and parks (e.g., Kruger National Park, the Blyde River Canyon, and Bushbuckridge Nature Reserve) and privately owned game reserves (e.g., Sabi Sands). The areas designated as "other" on the map, are the communal lands where local people are living. We were unable to locate any documentation of how the maps in the master plan representing ecosystem services were created, what ecosystem services were considered "essential," or how the various grades of ecosystem services were valuated, quantified, or compared. However, this distribution of services suggests that the image was created from a land use map, and perhaps primary vegetation land cover types, as proxies for ecosystem service provisioning (Figure 2). In short, it rates areas that are currently wildlife habitat, or might be in the future, as high in ecosystem service provision while it rates areas where the landscape supports the local human population through provision of a diversity of benefits as low in ecosystem service provision.

The results are unsurprising. Simply stated, the main recommendations that emanate from this plan and its maps are to focus human density and urban development in the four main regions, or "urban nodes," that currently have higher density development, and to keep certain higher quality lands from development, with the main goal being to connect conservation areas with wildlife habitat corridors. One such corridor is called the "Sabi Sand Game Reserve Corridor" which as suggested by the name, would link the Sabi Sands consortium of private of game reserves with several other privately-owned wildlife operations and the Bushbuckridge Nature Reserve (Figure 3).

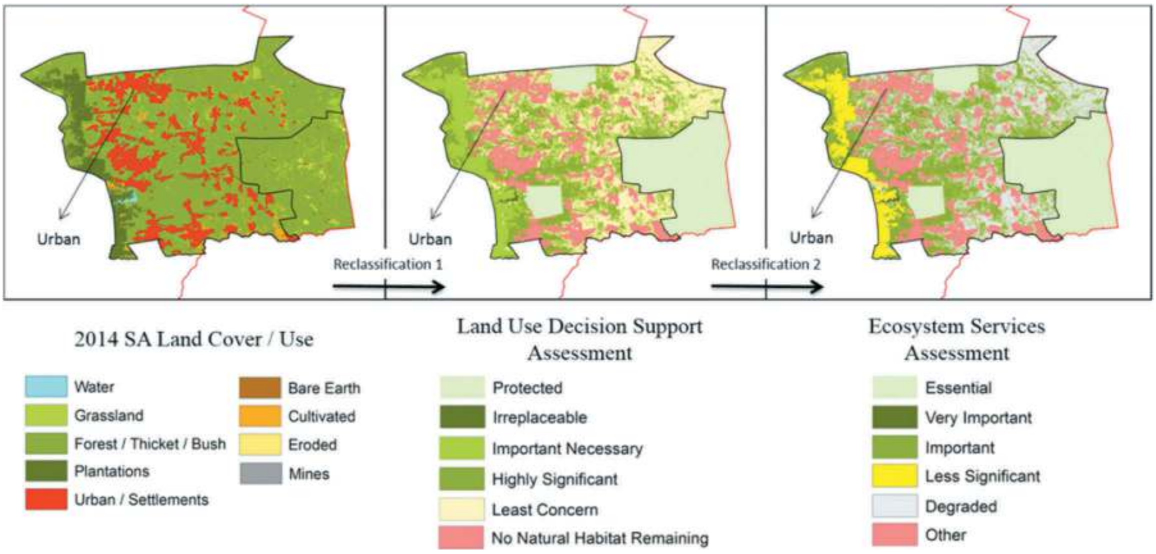

Figure 2. Ecosystem Services in the BLM. This figure provides a visual example of how ecosystem service maps in the Master Plan may have evolved. Land use maps are reclassified to produce a biodiversity assessment, and then reclassified to produce an ecosystem services map. Townships and villages, and their surrounding communal lands, are classified as "No Natural Habitat," "Least Concern," "Degraded," and "Other."

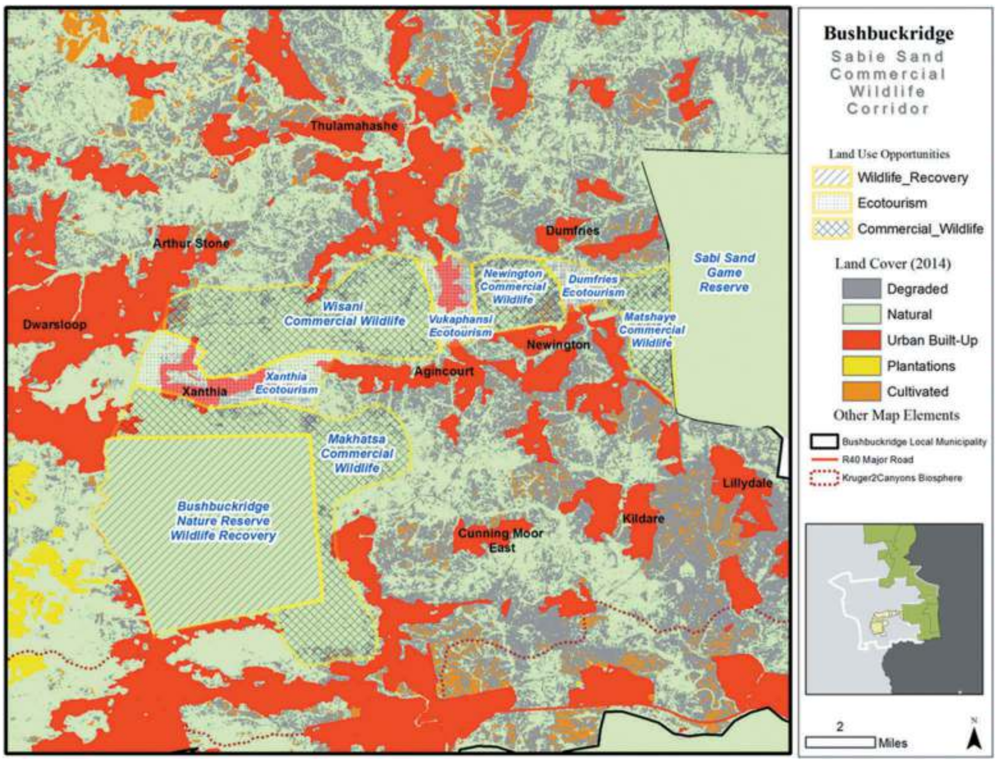

Figure 3. The Sabi Sands Game Reserve Corridor as recommended in the BLM master plan. (This map was recreated to represent a similar map utilized in the Bushbuckridge Master Plan.)

Logical conclusions/irresponsible planning/ serious consequences

Creating corridors among already existing conservation areas and other wildlife-focused land uses is a fundamental philosophy in conservation planning (Bennett 1999). Numerous studies have supported corridors, and demonstrated how more connectivity and less fragmentation is "good" for the environment (Wikramanayake et al. 2004; Damschen et al. 2006; Bailey 2007). However, justifying these corridors with maps that suggest the only high-quality ecosystem services in this region are provided by the conservation areas is irresponsible. Furthermore, we argue that management strategies and land use decisions that are based on these kinds of maps are misguided.

First and foremost, land use does not equate to ecosystem service provisioning. Just because land is currently designated a conservation area, does not mean that it is providing essential ecosystem services—except perhaps to a small number of ecotourists. Some privately owned conservation areas are not managed well—the land is overgrazed by wildlife, fire is underutilized as a management tool, and ecosystem functions have been compromised. However, even if a landscape is maintained in a semipristine state, there are a number of presumed ecosystem services that it does not provide (e.g., we still cannot confirm with any confidence that savanna ecosystems serve as long-term carbon sinks (Pellegrini et al. 2017)). At the same time, the heavily utilized savannas in the communal lands continue to harbor a wide range of biodiversity (Shackleton 2000; Smart, Whiting, and Twine 2005), and locally valued indigenous tree species such as marula (*Sclerocarya birrea*) are protected in homestead yards and cultivated fields (Paumgarten, Shackleton, and Cocks 2005; Anthony and Bellinger 2007).

Frequently, for security reasons or for the sake of an enhanced eco-tourism experience, protected areas are restricted from use by locals. This constrains their ability to support their livelihoods. The wildlife economy is premised on the assumption that conservation areas are providing jobs to local people (Linkd Environmental 2013), presuming that local communities benefit despite restricted access. Yet it is easy to imagine that many of the low-paying jobs associated with the wildlife economy are not benefiting communities as much as land owners or their well-off clientele. Even accounting for the increased economic activity stimulated by attracting tourists to the region, these changes are not necessarily a big advantage to local businesses. In fact, this part of the world still tends toward segregation, with white people driving north out of BLM to shop at white-owned businesses in Hoedspruit, and black communities inhabiting the business district of near-by Acornhoek.

Finally, the idea that communal lands do not provide ecosystem services is entirely erroneous. The ecosystem services concept at its very foundation is human-centric (Costanza et al. 1997). If services are not directly received by people, they simply cannot be valued. The communal lands, or developed land, is where most people in BLM live, and scientists have documented numerous ecosystem services that these lands are providing to them (Cousins 1999; Shackleton and Shackleton 2004; Paumgarten, Shackleton, and Cocks 2005; Shackleton et al. 2007). In terms of ecosystem services, communal lands serve far more people than do the conservation areas, and people living in these communal areas are more likely to perceive these locally produced and realized benefits than tourists or those in the tourism industry.

It could be argued that the conservation goals in the BLM master plan are not primarily driven by accurate evaluations of ecosystem services. Rather, the plan seems to have the overarching goal of using new corridors to preserve biodiversity in a UNESCO biosphere reserve (Linkd Environmental 2013). It is too often the case that biodiversity and ecosystem services are confused and deemed interconnected, despite the lack of evidence for this outside of a few, small scale and controlled experiments (Brose and Hillebrand 2016). Further, the assumption that the developed areas have less biodiversity is often not true. In many instances, places with people have increased biodiversity, especially in low-density developments such as these communal lands and villages (Shackleton 2000; Maestas, Knight, and Gilgert 2003).

Bridging the divide between theory and practice

The case study described earlier is a beacon, indicating a growing, and potentially dangerous, divide between theory underpinning ecosystem services as concept and practical implementation of ecosystem service assessments. This is not the first example of a highly regarded, ecologically oriented concept being used in practice to reinforce injustices that alienate already marginalized communities. A dramatic early instance of this is the history of Clemensian successional theory as it was applied to racial segregation policies and their implementation leading to apartheid through the agency of South African Prime Minister Jan Smuts. (See Anker 2009). Nonetheless, scientists and managers must directly address the challenges of implementing ecosystem service assessments head on if the concept is ever to play a positive role in conservation policy. In the next section of this article, we conceptualize how specific key features of current ecosystem service assessments, like the analyses utilized in planning for BLM, are an obstacle to bridging the divide between theory and practice. Then we identify interventions and operationalize a pluralistic framework for ecosystem service assessments.

The current state of ecosystem service assessments

The ecosystem services literature is replete with studies that have attempted to quantify the benefits provided by ecosystems or ecosystem service supply (Cimon-Morin, Darveau, and Poulin 2013; Burkhard et al. 2015), people's perceptions and values of those benefits or ecosystem service demand (Raymond et al. 2009; Gould et al. 2014), the flows of services or who receives the benefits (Gaston, Avila-Jimenez, and Edmondson 2013; Bagstad et al. 2014), and the tradeoffs among multiple ecosystem services (Tallis and Polasky 2009; Wegner and Pascual 2011). The science behind these assessments usually begins with a focus on quantifying potential benefits provided by an ecosystem (supply) or the benefits people value (demand), but rarely fully captures both standpoints (Figure 1). The studies that do try to analyze both supply and demand, for instance, usually capture demand as the "amount of a resource used," which is easily quantified ecologically or economically, but this is not necessarily how people actually value or perceive the benefit of a particular service. On the other hand, studies that begin with an emphasis on human values typically focus on the notoriously unquantifiable services, such as esthetic, cultural, and spiritual values (e.g., Plieninger et al. 2013; Pascua et al. 2017).

Often, methodological differences among disciplines are the reasons for one-sided approaches to assessing ecosystem services (Bunse, Rendon, and Luque 2015; Pascua et al. 2017). For example, many such assessments of carbon sequestration—a classic focus in the ecosystem services literature—are based on disciplinary approaches that quantify benefits provided by plants removing carbon from the atmosphere, while ignoring the conundrum that most people benefitting from this ecosystem process do not perceive it as a benefit, making the service effectively invisible (Figure 4). Meanwhile, the research that aspires to fully capture a wide range of people's values, often steers away from quantification techniques, and especially avoids monetary quantification (Milcu et al. 2013).

Similarly, the extent to which services are visible or invisible can simply be a function of the people that are a part of the assessment process (Turner et al. 2008). Ecologists, consultants, planners, policy makers, or people in positions of power often conduct these analyses. Although their decisions on what services to measure may simply be a function of their ability to quantify certain services in a repeatable way (Pascua et al. 2017), this distinction of what then becomes a visible versus invisible service often biases the perception, measurement, and management of the environmental structure and function (Figure 4). In this way, many ecosystem service assessments are guided by a narrow, overly technical, and systematically biased agenda, and can thus easily be misused to promote the interests of a select group of people.

Interventions for pluralistic ecosystem service assessments

Since the idea of ecosystem services is by definition a human-centric concept, many have argued that the process of assessing them in any given location should begin with the people actually receiving the benefits, and with an in-depth understanding of their perceptions and values (Cowling et al. 2008; Turner et al. 2008; Carpenter et al. 2009; Maynard, James, and Davidson 2010; Nahlik et al. 2012; Pascua et al. 2017). Chan, Satterfield, and Goldstein (2012) build a framework for engagement, and provide an extensive list of methods that can be used to involve the public in ecosystem service evaluation. There are scientists who propose that ecosystem service assessments begin with a focus on cultural services in particular, since these services are at the epicenter of human and environment relationships (Asah, Blahna, and Ryan 2012; Gómez-Baggethun et al. 2013; Milcu et al. 2013; Plieninger et al. 2013; Asah et al. 2014; Pascua et al. 2017), however, cultural-based assessments usually do not make their way into decision-making (Daniel

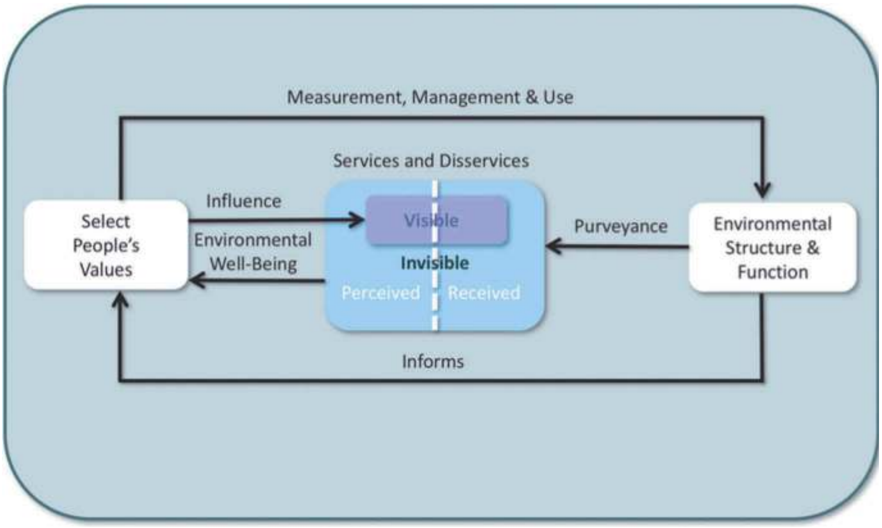

Figure 4. Current State of Ecosystem Service Assessments. The values of a select group of people (left) are the basis for most ecosystem service assessments, and these results lead to the biased measurement, management and use of the environment (top). For this reason, the focus then continues to be on the purveyance of only a few ecosystem services that are visible to a minority of people (purple box), while many actual services and benefits remain invisible (blue box).

et al. 2012; Milcu et al. 2013; Pascua et al. 2017). For this reason, Scholte, Van Teeffelen, and Verburg (2015) suggest that when engaging with stakeholders, the discussion on services should not necessarily be limited to cultural ecosystem services alone; instead, it should address a broad suite of socio-cultural values that can be linked to a variety of ecosystem services.

When an assessment process begins with surveying the local perceptions and values of the people who will be affected by the planning and policies that result from the assessments, the measurement and management of the local environment will be better focused (i.e., the upper arrow in Figure 4) (Turner et al. 2008; Chan, Satterfield, and Goldstein 2012; Pascua et al. 2017; Pascual et al. 2017). Therefore, with this new starting point, we propose, more services will be visibly perceived and received by a variety of different stakeholders (i.e., the box representing perceived and received values in Figure 4).

The importance of directly addressing perceptions of landscapes, the services they provide, and how those services are differentially valued is specifically addressed by Scholte, Van Teeffelen, and Verburg (2015), who provide an overview of the available literature on perceptions research (e.g., Zube, Sell, and Taylor 1982; Ulrich 1986; Nassauer 1995; Daniel 2001; Tveit, Ode, and Fry 2006; Bell 2012). An important part of this process is an open discussion of conflicts and synergies in perceptions and values between stakeholders, including scientists and policy makers. In fact, participatory governance, collaborative planning, and decision-making based on coproduction of knowledge has been shown to result in novel interventions and long-term engagement of a variety of participants in achieving sustainable solutions to natural resources challenges (Turner et al. 2008; Bunse, Rendon, and Luque 2015; Reyers et al. 2015).

Overall, because the connections and tradeoffs among benefits, values, and ecosystem services are complex, pluralistic approaches are well supported by the literature (Norton and Noonan 2007; Kumar and Kumar 2008; Spangenberg and Settele 2010; Chan, Satterfield, and Goldstein 2012); nonetheless, even when these approaches are embraced, there is still difficulty in linking ecosystem service to specific landscape characteristics (Scholte, Van Teeffelen, and Verburg 2015). This is where the field of urban ecology, and its focus on accounting for heterogeneity, can provide some valuable insights into a revised process for evaluating ecosystem services. We contend that a refined concept of heterogeneity, one that takes into account the nuances of scale, time, and complex interactions among different scales and times will be necessary (e.g., Andersson et al. 2015; Pickett et al. 2017). For example, urban ecologists have long known that capturing environmental variability at a 30-meter resolution, often available from satellite imagery, is inadequate. It is not possible to measure biophysical heterogeneity, let alone social-ecological heterogeneity, in human-dominated systems at these low resolutions (Cadenasso, Pickett, and Schwarz 2007). Yet ecosystem service assessments have often begun with analyses of the biophysical landscape using such coarse scales and low resolutions (Burkhard et al. 2012; Zhao and Sander 2015), including the analysis presented in the most often cited ecosystem services paper (i.e., Costanza et al. 1997).

Coarse-scale and coarse-resolution ecosystem service assessments are useful for a "big picture" view of the concept, and for raising awareness (Costanza et al. 2014), but this is not a useful approach for assessing, and then managing, ecosystem services that are being provided by particular social-ecological landscapes (Costanza et al. 2017). Furthermore, these coarse-scale and resolution assessments of services in urban and other human-dominated landscapes often lead to the conclusion that no services are being provided at all, even though these are the very places where people regularly and routinely interact with the environment and thus receive a multitude of benefits (i.e., Cimon-Morin, Darveau, and Poulin 2013). Finally, as we have demonstrated in the BLM case study, the use of these kinds of partial assessments of ecosystem services for planning and policy in the Global South—where a majority of urbanization is occurring today—further entrench historic systems of oppression exacerbating current social and environmental injustices (Simone 2004).

Finally, we propose that ecosystem service assessments not only begin by accounting for socio-cultural perceptions and values of stakeholders, but that these assessments also frame the ecological analysis, and in an iterative way. An iterative procedure, where results are presented back to the stakeholders, will facilitate a process where perceptions and values of actual benefits are continually reassessed (following the lower loop in Figure 4). This process will capture a wider array of community values in ecosystem service assessments and, therefore, should lead to a place where stakeholders' knowledge is incorporated into management and planning. We call this new model, the "democratization" of ecosystem services, because the "benefits before functions" approach focuses on giving a voice to stakeholders about their perceptions and values regarding the ecosystem services they derive from their immediate environment.

In this model the definition of "stakeholders" is important. We place particular emphasis on those who live in and are part of the social-ecological system being assessed. Identifying and capturing place-based perspectives from the broader community is essential to effective long-term natural resource management (Turner et al. 2008; Chan, Satterfield, and Goldstein 2012; Pascua et al. 2017).

We are not suggesting that everyone will perceive or value the same services, as that will likely never be the case. Instead, we are suggesting that all of the participants will have an opportunity to provide knowledge about benefits. In this way, various groups of stakeholders can at least be made aware of the values of others, decreasing invisibility and increasing transparency in decision making (Turner et al. 2008; Davies et al. 2015). This should lead to a broader acknowledgement that some benefits are important to some people—even if those people do not perceive themselves receiving those benefits.

In this process of colearning (i.e., Berkes 2009; Davies et al. 2015), recognizing multiple, and at times conflicting values and perceptions, provides a platform for the later discussion of tradeoffs in ecosystem services planning. Any planning and management of the environment that is based on such a democratized assessment should lead to the purveyance of more net benefits to local communities (Turner et al. 2008; Plieninger et al. 2013; Reyers et al. 2015). Finally, this inclusive process will feed back and influence the measurement, management, and use of ecosystem services, and thus contribute to healthier ecosystems and the enhanced well-being of the communities that depend on them (Figure 4). In the following text, we detail the specific steps involved in a democratized approach to ecosystem service assessments.

Steps for operationalizing a democratized approach to ecosystem service assessments

The case for democratizing ecosystem services is essentially a synthesis of many perspectives and literatures (e.g., Turner et al. 2008; Berkes 2009; Carpenter et al. 2009; Chan, Satterfield, and Goldstein 2012; Nahlik et al. 2012; Reyers et al. 2015; Bunse, Rendon, and Luque 2015; Scholte, Van Teeffelen, and Verburg 2015; Pascua et al. 2017). We suspect that scientists working on evaluation of ecosystem services may be interested in an interdisciplinary and inclusive approach and there is every reason that a pluralistic worldview should advance, in theory, science and practice. In reality, however, operationalizing the framework may be more challenging than it first seems. In order to ease these challenges, we provide a roadmap of the stages necessary to the development of a more democratized approach to ecosystem service assessments.

Stage 1—a wholistic assessment of values

As stated earlier, a democratized approach for assessing ecosystem services starts by evaluating the perceptions and values of people living in the landscape, the human system, capturing a broad picture of knowledge and interests. In this phase, it should be acknowledged that neither biophysical nor social scientists can be presumed to be neutral participants. Working on the question of bias is central to this process of engagement, because previous studies on "stakeholder processes" have strongly recommended that a neutral party lead these kinds of activities (Cooper et al. 2016). Although achieving neutrality may be challenging in some circumstances, inclusiveness in the initial engagement process should also help move toward reducing biases (Reed 2008). There are also frameworks for achieving neutrality that depend on incorporating the humanities and the arts (Cooper et al. 2016; Edwards, Collins, and Goto 2016). For instance, Kester (2004) describes how art can be used to create an open and accepting space for questions that are usually not tolerated in the realm of science-based decision-making.

Regardless of the method used, any framework for achieving inclusivity and neutrality should ensure that all participants have a chance to describe their understanding of the system and everyone can then work jointly to reveal values underlying each understanding. Some participants may feel intimidated or alienated by the involvement of scientific expertise (Reed 2008), whereas other participants may be suspicious of indigenous and local knowledge. However, diverse stakeholders often share core values that can help ease conflicts in perceptions and preferred knowledge bases (Shirk et al. 2012; Haywood and Besley 2014) and these challenges can potentially be overcome by bridging, and other deliberative methodologies that have been developed to engage citizens in decision making processes (Cowling et al. 2008; Turner et al. 2008; Bunse, Rendon, and Luque 2015; Pascual et al. 2017).

Research on deliberative methodologies is gaining momentum, and provides some insights on how to overcome the challenges of inclusive governance (Abelson et al. 2003; Spash 2007; Kenter et al. 2011; Christie et al. 2012; Shirk et al. 2012; Haywood and Besley 2014; Pascual et al. 2017). Although none of these methodologies are problem free, one critical aspect of this democratized process for ecosystem assessments can be the implementation and study of multiple deliberative methodologies. This kind of honest and repetitive "self-evaluation" will advance the theoretical science as well as increase the value of the assessment outcomes.

In these initial phases of engagement, it is important to determine the scales at which the ecosystem service assessments should be approached. Not everyone "sees" a landscape the same way (Johnson et al. 2004; Buijs, Elands, and Langers 2009). Some will think about broad-scale social or ecological processes while others will be very focused on the parts of the landscape with which they directly interact and that influence them or their household. Similarly, different individuals and groups will have differing boundaries of concern. For example, if a main interest is the long-term acquisition of clean water, the boundaries that influence values and perceptions would likely best align with watersheds, while local government officials maybe be more interested in influencing the health and wellbeing of their constituency, so their boundaries of concern would likely follow existing political boundaries.

Furthermore, although land use and land cover maps can play a major role in the engagement process, like they have in participatory mapping exercises (e.g., Raymond et al. 2009; Sherrouse, Clement, and Semmens 2011; Plieninger et al. 2013), overdependence on these could alienate some participants. Such formal maps embody many conventions and assumptions. Hence, they are not necessarily representations of reality for every person. The scale, resolution, or boundaries represented in any particular map could quickly become a sensitive issue among people with diverging interests (Kitchen and Blades 2002; Lewis and Sheppard 2006). This is especially a concern if participants are not experienced with reading these kinds of maps. Something as simple as esthetic decisions by the map maker regarding color choice or complexity level may keep even the most experienced spatial analyst from interpreting a map effectively. An engagement process that includes map-interpreting exercises, along with other participatory methods that enable stakeholders to discover and express the relevant spatial extent for their own perceptions and values, would serve as a foundational step in ecological and social value assessments.

Stage 2—integrate human and natural systems

The values identified in Stage 1 will guide the search for data at the proper resolution needed to comprehensively represent the social-ecological system (Figure 5). In many cases, useful datasets will already exist, but on other occasions some creative investigation will reveal new sources of data and knowledge. For instance, it is often assumed that national-scale datasets such as census offer the highest resolution information available on demographics, but local government agencies may have even more detailed data at the household or parcel scale.

Information from stakeholders on how they value environmental and social uses may also help biophysical scientists improve the ways in which they identify and quantify landscape heterogeneity and critical ecosystem functions (Ritzema et al. 2010; Fagerholm et al. 2012). Land use and land cover data will play an important role in the quantification of structure and function (De Groot et al. 2010), but not in the same ways that these data have been applied in ecosystem service assessments to date. Frequently land uses are

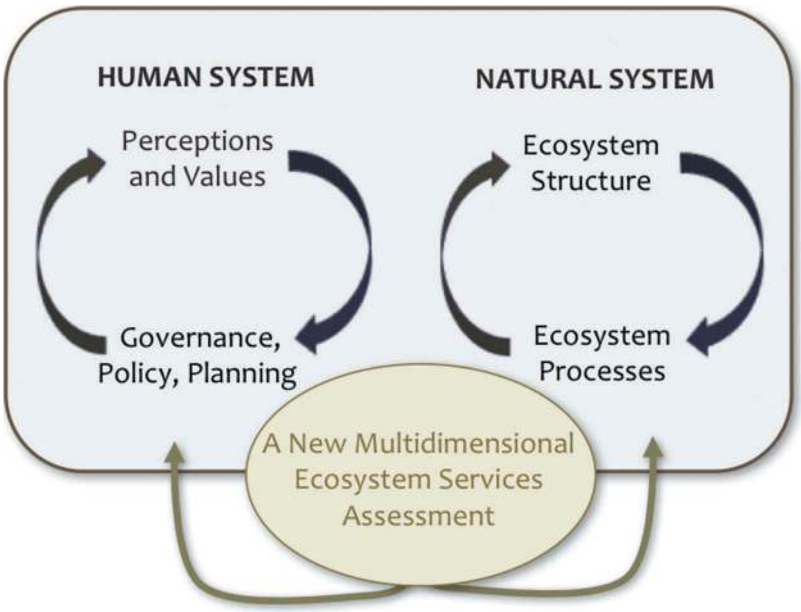

Figure 5. Operationalizing a Pluralistic Ecosystem Services Framework—a democratized approach for assessing ecosystem services starts by evaluating the perceptions and values of people living in the landscape, the human system, capturing a broad scope of knowledge and interests. Input from a broader community will produce some different and unexpected outcomes. Similar to strategic adaptive management, subsequent analyses on the outcomes and feedbacks are necessary.

assumed to uniformly provide predetermined benefits. However, our knowledge of people's perceptions and values may reveal that particular land cover types have a variety of previously unknown and unexpected benefits depending on where they are located in a heterogeneous landscape (Stephenson 2008). For example, trees in people's yards may be valued differently than trees in parks or other common areas (Dwyer, Schroeder, and Gobster 1991; Boone et al. 2010). Using this alternative approach, familiar land use categories become a modifier of the values attached to particular environmental attributes, rather than a true indicator of received benefits. In other words, not all trees or other features of a system provide the same services; rather, the services are context—detailed land use—dependent. Similarly, not all instances of a land use class will provide the same, uniform service. This complexity suggests shifting to a view of actual land covers, in place of land use with the assumed benefits, in a democratized ecosystem service assessment.

This shift to using land cover data rather than land use data in democratized ecosystem service assessments has important advantages. High-resolution land cover data better aligns with values and perceptions of environmental benefits, including at the scale of individual parcels or even individual trees (Cadenasso, Pickett, and Schwarz 2007). This enables us to capture values and benefits at spatial scales that are relevant to the people living in and using the landscape. Notably, these high resolution data can always be aggregated to coarser scales if necessary—for example, to capture larger-scale processes and their potential benefits. Finally, iterative engagement with stakeholder beneficiaries will better inform the process of identifying and mapping the fine scale sources of ecosystem services (Sherrouse, Clement, and Semmens 2011; Fagerholm et al. 2012; Palacios-Agundez et al. 2014). Thus genuine investment by stakeholder participants is enabled when they see that their ideas, opinions, concerns, and places are being addressed and incorporated directly into the process of ecosystem service assessment.

Stage 3—create multidimensional ecosystem service assessments

While this democratized approach may produce some standard spatial outcomes that are familiar to ecologists, we posit that input from a broader community will also produce some different and unexpected outcomes (Figure 5). For instance, one product might be a series of maps that draw attention to the location of certain ecosystem services, enabling a more representative quantification of benefits and values. Such maps are likely to look significantly different from ecosystem service maps developed by only one type of input and expertise. This may especially be the case in circumstances where the range of beneficiaries are subject to cultural norms that do not embrace spatial representations of information common to landscape ecologists and natural resource managers. A potentially innovative outcome would be one that takes into account different perspectives and leads to new forms of visualization not yet utilized in ecosystem service analyses or in the planning and policy sphere. Case-studies utilizing participatory methodologies have shown how the co-production of knowledge leads to the development of novel scenarios and evaluation of their environmental effects that would not have been addressed otherwise, and establishes new long-term collaborations to address social-ecological challenges (Reyers et al. 2015). In fact, the idea that the process may produce many different and unanticipated outcomes is a hypothesis worth testing by carefully documenting the steps of the democratized process and all of its outcomes.

Stage 4— assess outcomes and feedbacks using strategic adaptive management

Decisions are not discrete events and are a function of procedures, habits, and norms (Cowling et al. 2008; Simon 1997). A discourse that addresses historical foundations and institutional processes that influence decision-making is currently missing from the literature on ecosystem services (Cooper et al. 2016). Yet, our ultimate goal is more informed decision-making that leads to a progressively sophisticated management of social-ecological systems. The goal is that a democratized approach for ecosystem service assessments will lead to increased integrity of ecosystems and a larger number of people receiving more benefits from the environment; however, these anticipated outcomes should be evaluated. Community assessments of successes and failures will allow all stakeholders involved to document any unexpected feedbacks (Figure 5). We need to fully understand whether the process has led to an increased awareness of ecosystem services, has identified those services that remain stubbornly unquantifiable or invisible, and has led to design and management decisions that have resulted in improved ecosystem function and integrity.

Engagement in a colearning process often leads to adaptive management, where new learning is valued and incorporated into future management strategies (Berkes 2009; Reyers et al. 2015). In fact, utilizing ongoing assessments of social-ecological systems to reform management strategies over time has also been fundamental to the SAM approach (Rogers and Biggs 1999; Biggs and Rogers 2003; Roux and Foxcroft 2011). This "learning by-doing philosophy," originally developed by Holling (1978), was adapted by scientists and managers working in South African National Parks and is receiving increased international attention (Freitag, Biggs, and Breen 2014). The SAM process is predicated on the fact that the management of natural resources is imperfect and ecosystems are complex; it acknowledges an immediate need for action and is based on a strategy where all the players are committed to a process of experiential learning (Freitag, Biggs, and Breen 2014). We argue that democratizing the process of ecosystem service assessments is effectively a SAM-like evolution, a strategy where colearning and adaptive management merge to become adaptive comanagement (Berkes 2009).

Discussion: implementing the democratization of ecosystem services framework in BLM

What if a democratized ecosystem services framework was implemented in BLM before such important land use and management decisions were made? We have ample evidence that there are many more areas in the BLM that provide numerous and important ecosystem services to local people (Twine et al. 2003; Shackleton and Shackleton 2004; Shackleton et al. 2007; Twine 2013). For instance, the communal lands denoted as "other" (Figure 2) are used to graze livestock and provide a wide range of wild foods, fuelwood, medicinal plants as well as culturally important benefits such as burial, recreation, and male initiation sites. Large trees, such as the marula, are protected and nurtured in yards and fields because they provide fruit that is eaten or used to make beer, and shade under which family members gather in the heat of the day.

Furthermore, despite being substantially modified by human use and management, communal lands continue to yield classic ecosystem services such as carbon sequestration and soil protection (Kakembo and Rowntree 2003; Egoh et al. 2008), and habitat and food for a range of animal species, including pollinators (Hulme and Murphree 1999; Shackleton 2000). Yet, the land use and corridor maps, as part of the BLM master plan classify these developed areas as "degraded" (Figure 2). This assumption continues to be reinforced by claims that people may be overharvesting biomass in communal areas (Banks et al. 1996; Wessels et al. 2013). In several studies, in fact, humans are often compared to local elephants and are considered to be contributing to widespread degradation of ecosystems (Mograbi et al. 2017). Research does not necessarily support the idea that humans are contributing to an overall decrease in available biomass (Mograbi et al. 2015). Twine and Holdo (2016) demonstrate that prolific coppicing (resprouting) by savanna trees that are cut for fuelwood may compensate for the wood removed, providing an explanation for why the anticipated "fuelwood crisis" and total denudation predicted to occur in such areas has seldom materialized.

In reality, while local communities might harvest wood on the communal lands, there is evidence that within their villages they can contribute to planting and maintaining significant forest cover (Paumgarten, Shackleton, and Cocks 2005; Shackleton et al. 2007). We hypothesize that with increasing urbanization local communities might become more dependent on their local residential homesteads to support their livelihoods (Anthony and Bellinger 2007). Focusing on residential management practices could therefore lead to increased biodiversity in the region. Kruger National Park scientists and managers are trying in some ways to fill this void, distributing highly valued and threatened tree species such as pepper bark (*Warburgia salutaris*) to communities so local people can contribute to the maintenance of diversity in the area (Swemmer et al. 2014). Otherwise, ignoring the value added by humans in some landscapes is a missed opportunity for supporting management practices that really are a win-win for people and nature.

Future development plans should integrate human-valued land uses and ecosystem services into plans for conservation and growth management. The potential feedbacks of the democratized approach to ecosystem services are only speculative at this point; however, one can imagine how this process might lead to changes in perceptions and behaviors that are good for people and the environment. Increasing investment in the local health and well-being of these communities and the places where they live could empower people, which would be a welcomed change for a region currently considered a poverty node. Perhaps this approach would relieve pressure on corridor lands dedicated to wildlife management and ecotourism, as people's quality of life increases over time. We acknowledge, too, that there could be some unintended negative consequences to implementing this approach (Agarwal 2001; Mikalsen, Hernes, and Jentoft 2007; Berkes 2009). For example, there are many arguments that suggest inclusivity is too time consuming and does not provide enough benefits (Olsson et al. 2006; Roux et al. 2006). These are all hypotheses worth testing while implementing the democratization of ecosystem services framework.

Conclusion

A democratized approach to ecosystem service assessments is a reimagined path to rescuing a poorly implemented concept and designing and managing future social-ecological systems that benefit people and support ecosystem integrity. It gives a voice to local people to express the multiple, often invisible, values and benefits that they receive from their environment in a way that challenges the very power asymmetries that so often characterize ecosystem assessments conducted by experts alone. An iterative process of ecosystem service assessments, where the human values inform measurement and evaluation of ecosystem structure and function, and vice versa, is needed to guide planning and management of natural resources, and address tradeoffs and synergies among a multitude of benefits and costs during the decision-making process. Assessments must take into account social-environmental heterogeneity and not ignore the places where people live, as that may be the exact location of where they perceive and receive the most benefits. We outline one path of operationalizing this process, but do not deny there are likely many other ways to work toward democratization of ecosystem services. Implementing SAM, where the effects of decision-making on people and environment continue to be evaluated and revised over time, will ensure a democratized process for evaluating ecosystem services provides the most benefits to people over the long-term. Although many scientists have called for including ecosystem services in the planning and policy realm, we believe that it is the responsibility of scientists who do ecosystem services research to embrace complex, pluralistic frameworks, like this one so sound and inclusive scientific information is utilized to make management decisions.

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