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Review

A review of methods, data, and models to assess changes in the value of ecosystem services from land degradation and restoration

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ABSTRACT

This review assesses existing data, models, and other knowledge-based methods for valuing the effects of sustainable land management including the cost of land degradation on a global scale. The overall development goal of sustainable human well-being should be to obtain social, ecologic, and economic viability, not merely growth of the market economy. Therefore new and more integrated methods to value sustainable development are needed. There is a huge amount of data and methods currently available to model and analyze land management practices. However, it is scattered and requires consolidation and reformatting to be useful. In this review we collected and evaluated databases and computer models that could be useful for analyzing and valuing land management options for sustaining natural capital and maximizing ecosystem services. The current methods and models are not well equipped to handle large scale transdisciplinary analyses and a major conclusion of this synthesis paper is that there is a need for further development of the integrated approaches, which considers all four types of capital (human, built, natural, and social), and their interaction at spatially explicit, multiple scales. This should be facilitated by adapting existing models and make them and their outcomes more accessible to stakeholders. Other shortcomings and caveats of models should be addressed by adding the 'human factor', for instance, in participatory decision-making and scenario testing. For integration of the models themselves, a more participatory approach to model development is also recommended, along with the possibility of adding advanced gaming interfaces to the models to allow them to be "played" by a large number of interested parties and their trade-off decisions to be accumulated and compared.

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Contents

1. Introduction

It is becoming increasingly evident that allowing land to degrade is expensive, both to local owners and to society in general, in the short term, and especially, long-term (Costanza et al., 1997, 2014; Bateman et al., 2013; Trucost, 2013; Von Braun et al., 2013). At RIO + 20 the United Nations Conference on Combatting Desertification (UNCCD) set a target of zero net land degradation (ELD-Initiative, 2013). This need to prevent further land degradation, whether that is natural or human-dominated systems, and to restore degraded lands is especially important now because the demand for accessible productive land is increasing as human population and consumption increase. The geography of these changes are projected to affect mainly tropical regions that are already vulnerable to other stresses, including the increasing unpredictability of rainfall patterns and extreme events as a result of climate change (IPCC, 2007; Foley et al., 2011).

Land degradation is a decline in the processes and productivity of these ecosystems over an extended period of time (Lal, 1997; MEA, 2005; DeFries et al., 2012) and as defined in the Economics of Land Degradation (ELD) Interim Report (ELD-Initiative, 2013) results in “the reduction in the economic value of ecosystem services and goods derived from land as a result of anthropogenic activities or natural biophysical evolution”. In short it is a consequence of poor management of natural capital (soils, water, vegetation, etc.). We need better frameworks to quantify the scale of the problem globally, to calculate the cost of business as usual (ELD-Initiative, 2013), and explicitly and essentially to assess the benefits of ecological restoration. The current methods are often underestimating the cost of change, as they assume restoration will lead to full recovery of ecological functions, which is not necessarily the case. Visionary farmers and business leaders are becoming aware that degradation of ecosystems may become material issues affecting their bottom line and future prosperity (ACCA et al., 2012). However, they lack decision tools to develop robust and effective solutions to the problem (ACCA et al., 2012; ELD-Initiative, 2013). The identification of sustainable management strategies on both farm and landscape levels could be facilitated by the development of integrated decision tools. This could be, for instance, sound cost–benefit frameworks (ELD-Initiative, 2013) accompanied by modeling and simulation techniques that enable the creation and evaluation of scenarios of alternative futures and other decision tools to address this gap (Farley and Costanza, 2002; Costanza et al., 2006, 2013; Jarchow et al., 2012).

The managed land covers more than 60% of the Earth’s land surface and approximately 60% of this is under agriculture (Ellis et al., 2010; Foley et al., 2011). Ecosystems contribute to human well-being in a number of complex ways at multiple scales of space and time (Costanza and Daly, 1992; MEA, 2005; Dasgupta, 2008; Lal, 2012; UNEP, 2012; Costanza et al., 2013). Ecosystem services, including agricultural products, clean air, fresh water, disturbance regulation, climate regulation, recreational opportunities, and fertile soils are jeopardized by the effects of land degradation, and it is a global phenomenon (Walker et al., 2002; Foley et al., 2005; MEA, 2005; UNEP, 2012; Von Braun et al., 2013).

There is a need to integrate agricultural production and other land uses with ecosystem preservation to avoid land degradation in the future and to begin to restore degraded lands (Acevedo, 2011). This involves a standardized framework with methods to quantify and compare the extent of land degradation across political, cultural, biophysical, and managerial boundaries.

The overall development goal of sustainable human well-being cannot be measured in the mere growth of the market economy (Costanza et al., 2013). To obtain sustainable well-being through improved land management depends on the interaction of four basic types of capital assets: built, human, social, and natural. For example, the value of ecosystem services is the relative contribution of natural capital in combination with the other three types of assets to produce sustainable well-being. Although it focuses on natural capital and ecosystem services, it recognizes that the understanding, modeling, and valuing of ecosystem services requires an integrated, transdisciplinary approach which includes all four types of capital and their complex interactions.

The aim of this paper is to identify and discuss the data and methods used to determine global land degradation and to assess the sustainability of alternative management strategies.

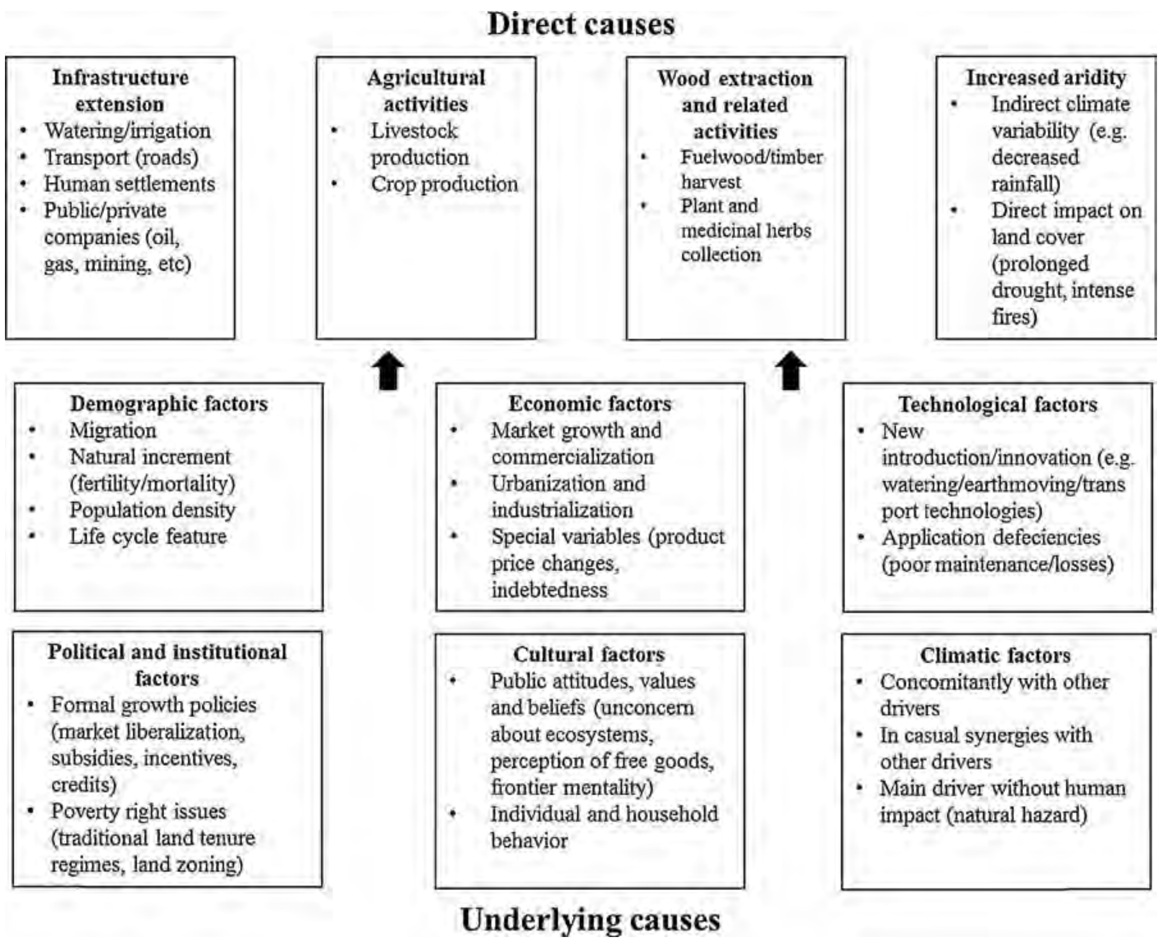

Fig. 1. Drivers of land degradation. Overview of drivers of land degradation in the case of desertification. Six broad clusters of underlying driving forces (fundamental social or biophysical processes) underpin the proximate causes of land degradation, which are immediate human or biophysical actions with a direct impact on in this case dryland cover and thus cause desertification (adapted from Geist and Lambin, 2004).

Assessments on global scale are informative for awareness-raising of the scale of the problem, and aims to inform politicians and policy makers on the extent of this substantial problem. The first section discusses methods for valuing land-use and management options based on the four capitals and reviews the types of databases and information proxies that are available (a comprehensive list is available in the Supplementary information). We argue it is important to take an integrated approach to assess the impact of land degradation on human well-being.

The second section focuses on the valuation methods specifically used to quantify natural capital and ecosystem services. The definition of land degradation is a decline in the production of ecosystem services but these rarely figure in traditional economic assessments and thus misrepresent the state of the natural capital. This section also reviews a selection of available models that range from plot scale to global scale, which can be used to develop a method to analyze the impact of sustainable land management practices. Different models are useful for different decision-makers to assess the effects of management at diverse scales. The final part is a discussion of methodology and knowledge gaps and further actions to take in light of the findings of this review.

2. Methods for valuing land use and management

Valuation is a tool for assessing trade-offs for achieving a common goal (Farber et al., 2002). All decisions that involve trade-offs involve valuation, whether implicitly or explicitly (Costanza et al., 1997, 2011b, 2014). Historically, most land degradation valuations have been focused on marketed physical goods such as food, feed, fibers and fuel production using commodity prices (Barbier, 2000; Cowie et al., 2011; ELD-Initiative, 2013; Nkonya et al., 2013). It is apparent now though, that a more comprehensive assessment of the full range of assets and services of the landscape is necessary to capture its real value for decision making (Costanza et al., 2013).

2.1. Drivers of land degradation

Understanding the drivers of land degradation is one of the major knowledge gaps identified by, among others, the ELD-initiative (Tilman, 1999; Tilman et al., 2002; ELD-Initiative, 2013) and is a crucial component of addressing land degradation as a whole. In addition, assessing the impact of the pressures and patterns of land degradation is also vital (ELD-Initiative, 2013). Drivers of degradation can be many and complex, resulting from a range of different interactions over time and space (Fig. 1) and each case can be distinct (Verburg et al., 2002; Geist and Lambin, 2004). They include proximate drivers, such as topography, land cover and vegetation, soil resilience, climate, and poor management, and underlying drivers, e.g. poverty, decentralization, access to agricultural extension service, land cover changes, and commodity market access (Lambin et al., 2001; Geist and Lambin, 2004; Andersson et al., 2011; Von Braun et al., 2013). Indicators of these biophysical, social, and economic types of drivers are important to identify what the cause of the degradation is and the alternative scenarios that will be part of the Cost Benefit Analysis (CBA). They include, inter alia, measures of vegetation cover, administrative borders, population density, soil properties, biodiversity, climate conditions, land management practices, topography, road density, access to information, land tenure, national policies, institutions, population density, and farmer perceptions.

Agriculture is not surprisingly one of the major proximate drivers of land degradation, but the amount of degradation is amplified by co-occurrence of the other types of drivers, e.g. increased aridity (Geist and Lambin, 2004). Natural processes such as weather variability and extremes increases the need for adaptive management and land restoration processes. If this is not present the combination of highly variable rainfall and lack of adaptive management causes high degrees of land degradation (McIntyre and Tongway, 2005; Stafford Smith et al., 2007). In Australia for instance, agricultural and pastoral processes that increase the proportion of annual plants can cause soil degradation in the form of acidification, salinization and erosion (Bolan et al., 1991; Dalal et al., 1991; Randall et al., 1997; Scott et al., 2000; Sumner and Noble, 2003; Tongway et al., 2003; Brennan et al., 2004; Dunlop et al., 2004; Bouwman et al., 2005; Lavelle and Spain, 2005; Glover et al., 2010; Bell et al., 2013; Pingali, 2013).

It is evident in many places that agriculture is increasingly becoming locked-in to conventional, high input, intensive management systems through changes to genetics of plants and animals (Allison and Hobbs, 2004; Vanloqueren and Baret, 2009). Lack of acknowledgement of (and in extension, payment for) ecosystem services and successful payment schemes restrain investment in such ideas (Van Der Ploeg et al., 2006; Swinton et al., 2007; Wossink and Swinton, 2007; Jack et al., 2008; Vanloqueren and Baret, 2009; Stallman, 2011). See S3 for more information on land degradation databases.

Drivers of management decisions are equally as complex, highly varied and dynamic (Lambin et al., 2001; Douglas, 2006), and indicators need to capture knowledge, opportunity and motivation (Kosmas et al., 2014; Shepherd et al., 2013). Knowledge and motivation for setting and achieving land condition goals are driven by the economy, e.g. private returns from trading private and public goods (Stafford Smith, 1994; Kwansoo et al., 2001; Kroeger and Casey, 2007; Stafford Smith et al., 2007). The ability to import otherwise limited resources such as fertilizers, or export sustainability issues, e.g. timber collection, can disguise natural constraints and thus degradation (Nakicenovic et al., 2000; Geist and Lambin, 2004; Oleson, 2011). The OECD and European Environmental Agency (EEA) has developed a database (http://www2.oecd.org/ecoinst/queries/Default.aspx) that lists all the different public and voluntary instruments and monetary incentives for environmental policies, such as subsidies, taxes, rebates and refund mechanisms for all the member states in EU and OECD. This can help identify the knowledge gaps (ELD-Initiative, 2013, p. 61, box 9, no. 12–14) that surround policies on sustainable land management, and identify what policy measures are appropriate and available for a certain region or approach.

2.2. Four types of capital assets and interrelated services

Methods needed to assess land use degradation and restoration processes will have to encompass the economic, social, and ecological aspects of development and management of the landscape. In short, it focuses on human well-being in the context of the health and well-being of the overall, linked human–natural system. This system consists of four basic types of capital assets (adapted from Vemuri and Costanza, 2006; Costanza et al., 2007).

- *Human capital* is defined as the individual peoples, including the knowledge and information stored in human brains, their physical health, and their labor force.

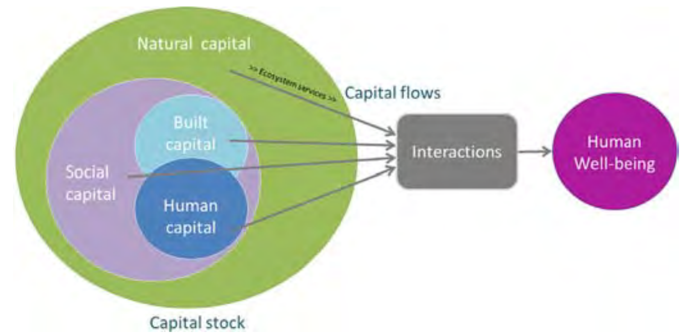

Fig. 2. Human–environment interactions. Interactions between built, social, human and natural capital required to produce human well-being. Built (incl. the economy) and human capital are embedded in society which is embedded in the rest of nature. Ecosystem services are the relative contribution of natural capital to human well-being, they do not flow directly to create well-being and need the other capitals to be able to contribute. It is therefore essential to adopt a broad, transdisciplinary perspective in order to address ecosystem services (adapted from Costanza et al., 2014).

- *Built capital* is manufactured goods such as tools, equipment, roads and buildings.
- *Natural capital* is the natural world, the ecosystems—everything that does not require human agency to be produced or maintained (Costanza and Daly, 1992).
- *Social capital* are those societal networks and norms that facilitate cooperative action. This includes cultures and institutions and the financial system (Putnam, 1995).

These four types of capital are all necessary elements in supporting sustainable human well-being (Fig. 1) and a suitable framework to use for an extensive global approach. All types of capital are influenced by policies and management decisions on land, and thus must be addressed to measure the impacts and costs on each of these four capitals. Thus, understanding the trade-offs in land management requires a holistic assessment of the effects on all four types of capital and their interactions.

This review focus on capturing the broader scope of the ecosystem services production, which are the benefits people derive from functioning ecosystems (Costanza et al., 1997; MEA, 2005). As ecosystem processes may contribute to ecosystem services, it is important to stress that they are not synonymous. Naturally occurring ecosystem processes describe biophysical relationships and exist regardless of whether humans benefit from them or not (Boyd and Banzhaf, 2007; Granek et al., 2010; Costanza et al., 2011a). Ecosystem services, on the other hand, exist if these goods or processes contribute to human well-being and cannot be defined autonomously. Ecosystems cannot deliver any benefits to people without primarily the existence of people (human capital), but also their societies (social capital), and their constructed environment (built capital) are most often an important component. These interactions are shown in Fig. 2. The challenge in ecosystem service valuation is to assess the relative impact of the natural capital stock and flows on the interactions in Fig. 2. This will give us the opportunity to address some of the limitations of understanding the value of ecosystem services in sustaining livelihoods (ELD-Initiative, 2013).

This conceptual valuation framework has been used for the recent UK National Ecosystem Assessment (UK NEA—http://uknea.unep-wcmc.org) and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES—http://www.ipbes.net).

There are several classification schemes that have been used for ecosystem services (Costanza et al., 1997; de Groot et al., 2002, 2012; MEA, 2005; Costanza, 2008; Sukhdev and Kumar, 2010; Haines-Young and Potschin, 2012). The Millennium Ecosystem Assessment (MEA) categorization is used by, or as foundation for, most assessments and categorization frameworks (MEA, 2005; Maes et al., 2013). Here the services are defined as provisioning, regulating, cultural and supporting (see detailed description in S1).

From this initial and somewhat broad definition of the MEA (2005), springs The Economics of Ecosystems and Biodiversity (TEEB), and the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2012), which are two more recent classification schemes, each with their own strengths and weaknesses. The main difference between the earlier MEA and the new additions is the re-categorization of supporting services into regulating services (see S1 for discussion of supporting services). TEEB also operates on two levels (category and service) as the MEA but, additionally, the TEEB is built to incorporate the framework for total economic value (TEV) of ecosystem services. This enables a consistent framework for national and regional ecosystem service assessment, valuation, and incorporation into policy (TEEB, 2010).

CICES (http://cices.eu) further builds on the concepts of TEEB. It is a newer classification tool developed in 2011–2013 by the European Environmental Agency (EEA), as an attempt to revise the System of Economic and Environmental Accounts (SEEA) (Maes et al., 2013). CICES is a hierarchical framework that operates on five different levels (MEA and TEEB only have two levels: section and class), to encompass the needs of cross-referencing between disciplines, such as spatial sciences, environmental accounting and economics (Maes et al., 2013).

The CICES categories parallel the TEEB classification, but are tailored for economic valuation, and the TEV-methods are as applicable to this framework as to TEEB. The CICES framework offers a more detailed definition of function and processes of services with 48 as compared to 22 in TEEB (Maes et al., 2013). It can be argued that there is no need for any more complexity and categories of ecosystem services, but there are some inherent benefits to this framework. For large-scale comparisons like the ELD Initiative it could be valuable because it allows for increasing transparency of methodology in e.g. mapping of services. Increasing the complexity may be valuable for both determining the types of ecosystem services and for communication with and between stakeholders. Furthermore, the hierarchically specified definition of the services may increase comparability between scales and sites.

2.3. *Mapping the capital assets and interrelated services*

To ensure a reliable baseline for this type of analysis, it would be necessary to take the geospatial context into account (Goodchild, 2009). It is far from irrelevant where the assets are located in relation to both the stocks, flows and demand for the capitals, as the spatial context and proximity of the capitals influence many of these interactions (Costanza et al., 2008). For use in analyses it is necessary to have reliable, spatially explicit, and valid data covering these four broad capitals. There has been a lot of work done to collect knowledge on the state of the four capitals and the indicators of land degradation and much of it is readily accessible.

2.3.1. *Human capital*

Human capital is traditionally defined as human qualities (Sen, 1997). Although the initial emphasis was on education and experience, in recent times human capital entails a broader definition: the productive investments embodied in a person in the form of skills, abilities, knowledge and health, often resulting from expenditure on education, on-the-job training and medical care (Todaro and Smith, 2012).

The United Nations Development Programme (UNDP) emphasized the significance of human capital as part of defining human development, defined as a process of enlarging people's choice (UNDP, 1990). The Human Development Index (HDI) (Anand and Sen, 1994), since then, has been considered as a better measure of human capital than Gross Domestic Production (GDP) per capita. HDI includes more components of human capital and is commonly used as its indicator (Anand and Sen, 1994).

Under HDI, the common indicators for knowledge, defined as an education index, are: (1) Adult literacy rate defined as the percentage of population aged 15 years and over who can both read and write with understanding a short simple statement on his/her everyday life, and (2) Gross enrolment ratio defined as number of pupils or students enrolled in a given level of education, regardless of age, expressed as a percentage of the official school-age population corresponding to the same level of education. Since 2010, the education index includes mean years of schooling and expected years of schooling (see S2). All four indicators are available from UNDP dataset at national level. In addition, the UNDP dataset also holds data on percentage of population who have successfully completed the final year of a level or sublevel of education in science and engineering, which can used an indicator for educational attainment. Health is used as a longevity index under HDI, comprised of life expectancy, under 5 mortality rate and maternal mortality rate as measures of health at a national level.

Human capital essentially depends on total population and other characteristics of the population. More importantly, it depends on the number of people who could potentially be economically active. The economically active population is measured as percentage of population between the ages 15 to 64. In a recent report, the World Economic Forum (Forum, 2013) outlined a broader definition of human capital, based on which human capital index is developed at the national level. In addition to current measures of human capital, education, health and employment, the human capital index adds another indicator termed as the enabling environment. This additional indicator captures the factors like legal framework and infrastructure, which enhance the components of human capital.

An interesting source for this is Landscan, a global gridded representation of population density that is produced on an annual basis by researchers at the Oak Ridge National Laboratory (http://web.ornl.gov/sci/landscan/) (Dobson et al., 2000). This data is derived from census data from all nations for which it is available and allocated to grid cells based on transportation infrastructure, slope, elevation, and land cover. This is the fundamental data layer that represents 'ambient population density', a temporally averaged representation of human presence. This dataset can be augmented with national and sub-national measurements such as mentioned above. However it is population density only and will have a significantly different appearance if weighted by national statistics associated with human capital (e.g. education, life expectancy, infant mortality, etc.). The data is available at global scale in extent with annual temporal resolution and a sub-national spatial resolution of 1 km^2 grid cells.

2.3.2. *Built capital*

The 'City Lights' data products derived from the Defense Meteorological Satellite Program's Operational Linescan System (DMSP OLS—http://ngdc.noaa.gov/eog/) and the Visible Infrared Imaging Suite (VIIRS) are available as global 1 km^2 grids at annual temporal resolution dating back to 1992. The Nighttime Lights (NTL) data products have been demonstrated to be good proxy measures of economic activity (Sutton and Costanza, 2002; Costanza et al., 2011c), building volume (Frolking et al., 2013), GDP (Henderson et al., 2012), impervious surface (Elvidge et al., 2007), and the human ecological footprint (Sutton et al., 2012). Just as human capital can be represented primarily via a global population density grid; built capital can be represented using global nighttime lights. This data can be augmented with global radar imagery that informs building volume and other global datasets such as DEMs and national boundaries and their affiliated national statistics. We can represent built capital in several conceptual frameworks at sub-nationally gridded spatial resolution. Built capital and human capital can be highly correlated in space, and have been aggregated to reduce complexity in some past analyses (Vemuri and Costanza, 2006).

2.3.3. Natural capital

Ecosystems are often measured as the amount of each type of land-cover, e.g. forest, tundra, grassland. For mapping on a national level, several global land cover datasets are available and provide basic substrates for representing natural capital, e.g. GlobCover (http://due.esrin.esa.int/globcover/) or the Food and Agriculture Organization (FAO) land cover dataset Global Land Cover-SHARE (http://www.glcn.org/databases/lc_glcshare_en.jsp). This can be augmented with bathymetry data to distinguish open ocean from continental marine shelf as well as incorporate coral reef data. In short, a global representation of natural capital is readily available at 1 km^2 spatial resolution, if it can be satisfactorily represented by relatively coarse land cover types and their associated ecosystem services. These types of land cover data sets have had significant valuation studies conducted on them (Costanza et al., 1997, 2014). It has been prepared from GlobCov, ReefBase (http://www.reefbase.org/main.aspx), and Bathymetry data and published as an updated representation of the total global value of nature's services (Costanza et al., 2014). This data is global in extent with 1 km^2 spatial resolution. The temporal resolution of this sort of data is on the 5 to 10 year update cycle. Of course, for local and regional change these changes can be measured and documented at higher temporal and spatial resolution.

2.3.4. Social capital

Most of the empirical literature on social capital focuses on community level or meso-level measures of trust, networks and collectivism (Putnam, 1995). There are many ways to conceptualize social capital including levels of corruption, effectiveness of government, socio-economic resilience of e.g. a population or community, or levels of collectivism and trust.

The World Values Survey (http://www.worldvaluessurvey.org/wvs.jsp) provides country level data for a range of indicators for collectivism and trust. Collectivism can be measured as active membership in religious, political, labor union, professional, environmental, sports or recreational, art/music/educational organizations. Using e.g. a Likert-scale questionnaire, trust is measured for family, neighborhood, friends, and people from different religions and nationalities.

The UNDP dataset includes information on trust in people and people's perception of safety, both of which can be used as a measure for trust. Data on fixed line and mobile telephone subscription, measured as the sum of telephone lines and mobile subscribers, expressed per 100 people, can be used as a proxy measure for social networks.

Representing social capital as a spatially explicit data set is quite a challenging task because social capital is embedded in relationships between and amongst individuals and institutions. One potential way of doing this is via Facebook connections (Ellison et al., 2011; Pender et al., 2014), or through Twitter feeds as the happiness indicator Hedonometer.org from University of Vermont (Dodds et al., 2011), other suggestions could be the internet connectivity as measured by the Global Submarine Cable map (Malecki and Wei, 2009).

2.3.5. Other spatial data bases for land use and degradation impact assessments

The large databases from the major organizations as Organization for Economic Co-operation and Development (OECD), World Bank, and FAO, hold vast amounts of information (see S12), and are repeatedly used for large scale analyses like the Better Life Index or Human Development Report and would suit this analysis too, if the scale of data is appropriate. This data is available at national scales for the most part but some datasets are available at a finer scale, such as 1 km^2. This is the case for the extensive Global Agro-Ecological Zones (GAEZ) database from FAO (http://www.fao.org/nr/gaez/en/). They have prepared thousands of spatially explicit datasets on climate, hydrology, fertilizer inputs, soils, crop production, levels of irrigation, and growing season, just to mention a few. This has very valuable data for assessing the sustainability of current land uses and agricultural production based on 30 years of data collection. This extensive and well-documented suite of global datasets provides a state of the art spatially explicit representation of the current global agricultural situation. An interesting source of land cover dynamics is the University of Maryland's Global Forest Change map (Hansen et al., 2013). This is a time analysis of the increase and decrease of the global forest cover and associated services.

Quandl.com is a major numerical database of current and historical global datasets on economics, finance, demography, society, markets, energy, health, and education among others. It is an open source collaboration that collects data from all over the world, and has time series data from the large international organizations mentioned above, central banks, e.g. the federal reserve; us bureaus and agencies; non-us statistical agencies; financial data such as exchange rates; think tanks and academia, e.g. Gapminder, Yale Department of Economics; private sector sources, e.g. BP and global energy industry; among others (see https://www.quandl.com/about). Not all the data is spatially explicit though and thus there is a limit to the applicability to some of the datasets.

Another interesting agricultural database is the World Overview of Conservation Approaches and Technologies (WOCAT) database (www.wocat.org). It is an open access database of sustainable land management practices and a platform to register various sustainability projects, developed in conjunction with the Land Degradation Assessment in Drylands (LADA) Project from FAO. The collection is based on extensive questionnaires that detail the perceived benefits of the project by participants and experts. The databases are divided into a technology database and an approach database. Furthermore there is a mapping database, as well as elective modules if the project has a climate change angle to it (watershed management is under development at present time). Collectively, this database has a lot of information on sustainable management technologies and practices. It stems from more than 30 regional and national initiatives, covers 470 technologies and 235 approaches of sustainability of land management. The mapping database has a collection of conservation approaches and land use maps developed by the DESIRE project (on fragile arid and semi-arid ecosystems) for the participating pilot countries. The quality of the individual projects, or at least the reporting, varies however. Specifically, there is a lack of quantifiable information of the effect of the sustainable land management on ecosystem services, and some of the projects lack specific spatial information. Nevertheless, the WOCAT database could potentially be very valuable for identifying cases and technique studies and an alternative livelihood information-bank, and provides an opportunity to incorporate the WOCAT studies into the ELD and thus increase information in both initiatives. Furthermore there is not a lot of information about the external impacts of the WOCAT projects, something the ELD could provide.

The Global Restoration Network (GRN) also hosts a database of hundreds of case studies of ecological restoration projects and initiatives from all over the world (http://www.globalrestorationnetwork.org/database/). This is a collaboration between the International Union for Conservation of Nature (IUCN) and Society for Ecological Restoration to collect information about restoration projects and methods globally. Besides having the case study database, it also includes an extensive library of literature on restoration, with more than 600 references, 150 experts and the main organizations in the field. These latter entries are not up to date but still hold valuable information about land restoration and alternative livelihoods. This database is also based on self-reporting and thus has the same restrictions as the WOCAT database.

An extensive list of identified databases is displayed in S8.

2.3.6. The land use classification systems

Land uses are defined as the anthropogenic practices on an ecosystem (Jansen and Gregorio, 2002). Land management of agricultural areas is in line with this the method of cultivation, and livelihood options are an expression of the different income potential the land holds. Land use change is one of the leading drivers of global change (Sala et al., 2000) including conversions of primary nature and changes in management forms or livelihoods (Cihlar and Jansen, 2001; Foley et al., 2005; Acevedo, 2011; Lautenbach et al., 2011; Monteiro et al., 2011; Polasky et al., 2011; Bahadur, 2012). Land cover maps are readily available but land use is harder to establish because of the multiple land uses that could occur for a given land cover. Land cover datasets are a primary input to assess human, built and natural capital but do not reflect the multifunctionality of landscapes nor do they capture faunal states or interactions (Dirzo et al., 2014). This will need to be augmented with other information such as valuations of ecosystem services, global representations of net primary productivity derived from satellite imagery, and characterizations of ecosystem health and function, including faunal representation (Galetti and Dirzo, 2013).

There are a number of different classification schemes in use throughout the world; some of the most common are the original 1976 Anderson (Anderson, 1976), the NLCD 92 Land Cover Classification Scheme (modified Anderson Level II), and the FAO and UNEP Land Cover Classification System (LCCS 3) Classification.

Efforts to assess land degradation are not limited to deal only with the livelihoods that are based on the terrestrial surface but should also include freshwater and marine sources of livelihood diversification. The state of aquatic ecosystem services can also be a source of increased or decreased pressure on the condition of land. If land is heavily degraded people might seek out aquatic sources to increase their income and well-being. If the aquatic state is depleted, it might increase the pressure on the land based management (Granek et al., 2010; Luisetti et al., 2011) and ultimately on the level of degradation. Examples of ecosystem services that stem from these systems include storm protection, hatcheries, fisheries, coral reef biodiversity and ecotourism, among many others (Costanza et al., 2008; Barbier et al., 2011; Gedan et al., 2011).

3. Valuation methods for natural capital and ecosystem services

To be able to assess the impacts and costs of land degradation on a larger scale, there is a need to take a broad and integrative approach, which includes both the capital stocks and flows that affect human well-being, as well as the linkages to external effects and livelihoods that are not based on the terrestrial surface. The first part of this paper has dealt with spatially specific mapping of the full range of capital stocks. In this section, we will look specifically at the value of the flows of natural capital to human well-being, ecosystem services. This is because the immediate effects of land degradation are directly evident in the supply of ecosystem services, and thus the impact on well-being.

Box 1: Different methods for ecosystem service valuations, includes conventional economic valuation and non-monetizing valuation or assessment (adapted from Farber et al., 2006)

Conventional economic valuation
Revealed-preference approaches
- Travel cost: valuations of site-based amenities are implied by the costs people incur to enjoy them (e.g., cleaner recreational lakes)
- Market methods: valuations are directly obtained from what people must be willing to pay for the service or good (e.g., timber harvest)
- Hedonic methods: the value of a service is implied by what people will be willing to pay for the service through purchases in related markets, such as housing markets (e.g., open-space amenities)
- Production approaches: service values are assigned from the impacts of those services on economic outputs (e.g., increased shrimp yields from increased area of wetlands)

Stated-preference approaches
- Contingent valuation: people are directly asked their willingness to pay or accept compensation for some change in ecological service (e.g., willingness to pay for cleaner air)
- Conjoint analysis: people are asked to choose or rank different service scenarios or ecological conditions that differ in the mix of those conditions (e.g., choosing between wetlands scenarios with differing levels of flood protection and fishery yields)

Cost-based approaches
- Replacement cost: the loss of a natural system service is evaluated in terms of what it would cost to replace that service (e.g., tertiary treatment values of wetlands if the cost of replacement is less than the value society places on tertiary treatment)
- Avoidance cost: a service is valued on the basis of costs avoided, or of the extent to which it allows the avoidance of costly averting behaviors, including mitigation (e.g., clean water reduces costly incidents of diarrhea)

Nonmonetizing valuation or assessment
Individual index-based methods, including rating or ranking choice models, expert opinion
Group-based methods, including voting mechanisms, focus groups, citizen juries, stakeholder analysis

The ecosystem services are the benefits people derive from ecosystems (Costanza et al., 1997; MEA, 2005). The value of ecosystem services is the *relative* contribution of ecosystems to that goal, as pictured in Fig. 1. The value of ecosystem services reveals the importance of that contribution.

Once the interrelated ecosystem services have been identified, quantified, and mapped, several techniques are available to conduct a total economic valuation (TEV). There are multiple ways to assess this contribution (Box 1), some of which are based on individual's perceptions of the benefits they derive. These methods can be divided into revealed preference or stated preference. Revealed preference methods use market prices as a proxy for benefits. Of course, this approach only works for goods and services that are traded in markets. Only a small subset of ecosystem services (mostly provisioning services) are traded in markets. Stated preference methods attempt to construct pseudo markets via surveys that ask people to state their willingness-to-pay for ecosystem services that are not traded in markets. These include various versions of contingent valuation and choice modeling (ELD-Initiative, 2013). However, stated preference approaches have severe limitations when applied to ecosystem services (Liu and Stern, 2008). This is mainly an issue of imperfect information individuals hold about ecosystems and their connections to human well-being, as well as their discomfort with stating trade-offs for ecosystems in monetary units. Furthermore, the problem also includes the potentially big difference between the stated willingness to pay and the real payment when it comes to that point. These inaccuracies become especially prominent when it comes to valuing the critical natural capital—ecosystem services that are vital and essential (Farley et al., 2014). Since the individual's perceptions are limited and often biased (Kahneman, 2011), valuation methodology needs to take

Table 1
Categories of ecosystem services and appropriate methods of valuation (from Farber et al., 2006).

Ecosystem services	Amenability to economic valuation	Most appropriate method for valuation	Transferability across sites
Provisioning services			
Water supply	+++	AC, RC, M, TC	++
Food	+++	M, P	+++
Raw materials	+++	M, P	+++
Genetic resources	+	M, AC	+
Medicinal resources	+++	AC, RC, P	+++
Ornamental resources	+++	AC, RC, H	++
Regulating services			
Gas regulation	++	CV, AC, RC	+++
Climate regulation	+	CV	+++
Disturbance regulation	+++	AC	++
Biological regulation	++	AC, P	+++
Water regulation	+++	M, AC, RC, H, P, CV	++
Soil retention	++	AC, RC, H	++
Waste regulation	+++	RC, AC, CV	++/+++
Nutrient regulation	++	AC, CV	++
Cultural services			
Recreation	+++	TC, CV, ranking	+
Aesthetics	+++	H, CV, TC, ranking	+
Science and education	+	Ranking	+++
Spiritual and historic	+	CV, ranking	+

AC = avoided cost, CV = contingent valuation, H = hedonic pricing, M = market pricing, P = production approach. RC = replacement cost, TC = travel cost; High: +++; Medium: ++; Low:+.

Table 2
Four levels of ecosystem service value aggregation (adapted from Croen et al., 1991).

Aggregation method	Assumptions/approach	Examples
1. Basic value transfer	assumes values constant over ecosystem types	Creasy (1993), Costanza et al. (2002)
2. Expert modified value transfer	adjusts values for local ecosystem conditions using expert opinion surveys	Batker et al. (2010)
3. Statistical value transfer	builds statistical model of spatial and other dependencies	de Groot et al. (2012)
4. Spatially explicit functional modeling	Builds spatially explicit statistical or dynamic systems models incorporating valuation	Boumans et al. (2002), Costanza et al. (2008), Nelson et al. (2009)

that into account. A mitigation option for this problem it should be incorporated into the method of valuation, which is addressed in Section 4.

This is an on-going challenge in ecosystem services valuation, but some of the existing valuation methods like avoided and replacement cost estimates are not dependent on individual perceptions of value. For instance, choice experiments, where individuals are asked to rank alternative scenario outcomes, seem to be an easier way for people to think about trade-offs (Farber et al., 2002). The degree of replication needed to produce an estimate of value is achieved by sending different versions of the scenario-ranking questionnaire to a number of participants.

Table 1 shows the relationship between these valuation methods and the ecosystem services they are most appropriately applied to. Note that there is generally not one correct approach, but a range of approaches that should be used and compared.

3.1. Aggregation and scaling

Ecosystem services are often assessed and valued at specific sites for specific services. However, some uses require aggregate values over larger spatial and temporal scales (Table 2). Producing such aggregates suffers from many of the same problems as producing any aggregate estimate, including macroeconomic aggregates such as GDP.

Most aggregates are "accounting measures" of the quantity of ecosystem services (Howarth and Farber, 2002). In this accounting dimension the measure is based on virtual non-market prices and incomes, not real prices and incomes. This degree of approximation is appropriate for some uses, (Table 2) but ultimately a more spatially explicit and dynamic approach would be preferable and more accurate to scale up. Regional (at e.g. watershed or provinces such as the EU NUTS 3 levels) aggregates are useful for decision-makers to assess land use change scenarios. National aggregates are useful for revising national income accounts. Global aggregates are useful for raising awareness and to trigger action by emphasizing the importance of ecosystem services relative to other contributors to human well-being.

Modeling at a scale appropriate for land management, for instance farm scales, provides important information about farmer choice and decision-making. Different frameworks can be adapted by different schools, groups, and disciplines based on data availability (cf. Boyd and Banzhaf, 2007; Carpenter et al., 2009; Fisher et al., 2009; Sukhdev and Kumar, 2010; Seppelt et al., 2011, 2012). These models can then be aggregated and scaled up to cover whole landscapes, regions, national and global levels. While there has been a call for a unifying and standardized ecosystem service accounting framework that allows for transparent aggregation between scales and disciplines (Costanza, 2008; Lamarque et al., 2011), it is clear, however, that no one valuation framework will work for all purposes either (Costanza, 2008).

There are, however, issues that are considered cross-scale (Hein et al., 2006; Fisher et al., 2009; Scholes et al., 2013), e.g. top-down policies on global issues that affect local management strategies, like the water framework directive in the EU, or international fishery or biodiversity policies. Bottom up effects can also result in cross-scale issues where regional problems drives management practices on a global scale, for instance with droughts or flooding and following global food shortages (Scholes et al., 2013). These aspects can have a considerable effect on the management practices, but can be difficult to capture. Thus, as scale is fundamental to describe context, any assessments or models should take these cross-scale issues into consideration, whether that is by multi-scale synthesis assessments or decidedly cross-scale assessments (Scholes et al., 2013).

3.2. Methods to assess land degradation

There are some significant efforts to assess the extent of land degradation across different case study sites and for different ecosystem types (ELD-Initiative, 2013). One way of approaching this is by conducting a cost–benefit analysis of the current land management type and alternative options. The ELD Initiative has developed such a methodology, based on the 6 + 1 steps action plan established by the United Nations Convention to Combat Desertification UNCCD Global Mechanism (ELD-Initiative, 2013). It is intended to allow the estimation of the overall benefits of addressing land degradation and implementing ecosystem restoration. Such estimates will enable businesses and policy makers to test the economic implication of land management decisions, based on a scenario-driven, net economic benefit decision making framework (ELD-Initiative, 2013). The 6 + 1 steps are designed to ensure a thorough and applicable knowledge base is established for the valuation and subsequent cost–benefit analyses that are the base of the decision making process (Table 3).

Table 3
The 6 + 1 methodology of the ELD Initiative (adapted from ELD-Initiative, 2013).

1. Inception	Identify the context and framework of the study
2. Geographical characteristics	Establish the geographic and ecological limits of the study area
3. Types of ecosystem services	Identify, classify, and map stocks and flows in each ecosystem
4. Roles of ecosystem services and economic valuation	Link the role of ecosystem services to livelihoods and economic development in the area. Estimate TEV of ecosystem services
5. Patterns and pressures	Identify patterns of degradation and pressures and drivers of SLM to inform scenarios. Potential revision of previous steps
6. CBA and decision making	A cost–benefit analysis of each scenario to assess whether the proposed land management changes have net benefits
7. Take action	Implement the most economically desirable option(s)

This framework is intended to provide decision makers with transparent information to adopt economically sound sustainable land management (ELD-Initiative, 2013). By comparing the economic costs versus the benefits of action, impacts on human well-being and the long term effects of decisions, one is equipped to choose sustainable land management solutions. Nevertheless, there are extensive knowledge gaps in both the conceptual and procedural approach of this 6 + 1 steps methodology (ELD-Initiative, 2013). These deficits center on different technological, environmental evaluation, policy and institutional problems. Some of the previous attempts to value land degradation, see e.g. (Nkonya et al., 2011, 2013), relies mostly on use values in their case study assessments, whereas the ELD aims for a more comprehensive valuation process that engages spatially explicit models for valuation of ecosystem services and other capital stocks and flows.

This approach will give a more accurate and reliable output for e.g. spatial aggregation and accurate accounting. Nkonya et al. furthermore use the comparison of cost of action of adopting sustainable land management practices versus the inaction of business-as-usual, to analyze the most beneficial approach to land management. The cost of inaction has the possible risk of overestimating the benefits of action (Stern, 2007). Not all approaches to ecological restoration will result in 100% functional land, which might very well be impossible to regain, and if that is the case, the cost of inaction will overestimate the benefit from action (see e.g. ELD-Initiative, 2013, Fig. 5).

The ELD, on the other hand, goes beyond what has earlier been done. The main goal is to determine the economic costs and benefits of action, and compare these costs and benefits for multiple livelihoods options, land values, as well as the impact on overall human well-being (ELD-Initiative, 2013; Nkonya et al., 2013). The 6 + 1 methodology and the net economic benefit decision making framework (Table 3) makes comparisons between different scenarios and land use options transparent and intuitive for decision making, and thus also improves the applicability of the methodology in both theory and practice. It does, however, mean we have to overcome the caveats and the knowledge gaps identified by the Interim Report (ELD-Initiative, 2013, box 7).

3.3. Selected examples of models for sustainable land management assessments

When the level of land degradation and ecosystem service provision is assessed, the next step is to identify a different approach, a sustainable land management path. In order to make sure this approach is in fact sustainable before implementation, it can be helpful to assess the effects using a model. There are of course many things to consider when running such models. For instance, scale and purpose of the analyses must align, as mentioned in the aggregation and scaling section. The heterogeneity of spatial and temporal scale highlights the complexity of the human–nature interactions (Costanza et al., 2002; Costanza and Voinov, 2003; Aertsen et al., 2012; Dale et al., 2013), and matters like the lack of harmonized methodology to conduct an environmental evaluation, as discussed above, complicate the analyses and this is important to acknowledge and prioritize. There is probably no 'one-size-fits-all' modeling technique that can serve all the needs of ecosystem services analysis. In each case the choice of a particular modeling tool will be driven by the particular goals of the study, the available data and scale of the problem (Jakeman et al., 2013).

Table 4 is a collection of representative and widely used mathematical models available for different systems and scales. Even though the main focus of this paper is to identify global assessment techniques, we have chosen to include models that deal with management of smaller scales. This is mainly because there is no such global model today, and the development of such a model would most likely require aggregation of existing models. The models in Table 4 operate on temporal and spatial scales that could be used for studies of local, national, or global systems (or parts thereof). Fig. 3 depicts Table 4, the spatial extent with the degree of comprehensiveness. The models have been selected to represent tested and widely used models that could serve as a toolbox for researchers that look into sustainable land management studies, and we stress this is not a comprehensive sample, as there are simply too many models to include in this paper. Table 4 give an overview of the models selected, a detailed description and discussion of the models can be found in S4–S7.

The models have been divided into sections by the level of spatial aggregation and integration of a wider amount of the capitals. As mentioned above, different scales of analyses are useful for different objectives, whether that is for local or global processes and stakeholder representations. Some examples at the larger scale, include Dynamic Global Vegetation Models (DGVM) while at smaller scales (regional, farm and site scale) include process based and simulation models.

Integrated global models (IGMs) attempt to build quantitative understanding of the complex, dynamic history and future of human–environment interactions at the global scale, and are thus very comprehensive and complex. There is now a 30-year history of this approach. Over this period, computer simulation modeling has become a well-accepted technique in scientific analysis, but truly integrated simulation models – those that deal with the dynamics of both the natural and human components of the system and their interactions – are still relatively rare, and those that do this at the global scale are even rarer.

One of the major problems with using models is the trade-off between specificity and applicability, which the user must assess when determining the transferability between sites (Wenger and Olden, 2012). As Nordhaus (1973) also states, a model is only as good as the assumptions it relies on, and therefore choosing the right type of model for the problem and the scale of the analysis is a crucial step. Another problem to take into account is the fact that some of the models might be able to operate on many different levels but the amount of data they need can be a restriction for the applicability, simply because they are too data hungry for many sites around the world.

3.4. Models specifically aimed at ecosystem services

In addition to the GUMBO global unified metamodel of the biosphere (Boumans et al., 2002), several other models and

Table 4
Representative selection of models to evaluate the effects of sustainable land management. Divided by aggregation scale from plot to global. For additional and more comprehensive details, including URL, see the Supporting information.

Model	Scale	Type	Brief description
CropSyst	Field	Process based model	The model was developed as an analytic tool to analyze the effect of management on both the productivity and the environment
DNDC	Plot to field	Biogeochemistry computer simulation model in agro-ecosystems	This model is process-based and concurrently models (trace-) gas emissions, soil carbon storage, as well as crop yield, in agricultural
APSIM	Field to farm	Agro-ecosystem process based model	Consists of three different modules, plant, soil and management. Each of them include a diverse range of crops, pastures and tree production, soil processes, as well as water balance, N and P transformations, soil properties, erosion and a large range of management settings
CENTURY	Field to farm	Agro-ecosystem process based model	Simulates macro nutrient dynamics, on both farm and field scale. It embodies the best understanding of the biogeochemistry of C, N, P, and S, available today
EPIC	Field to farm	Agro-ecosystem model	Cropping systems model that estimates the effects of management decisions on soil, water, nutrient and pesticide movements
APEX	Watershed	Landscape model	Watershed analysis to evaluate a range of land management strategies and take into consideration sustainability, erosion, economics, water supply and quality, soil state, plant community competition, weather, and pests
DSSAT	Farm to regional	Cropping system model (CSM) Software application program	On-farm model developed to handle precision management, with a component to analyze regional assessments of climate variability and climate change impacts
STICS	Plot to regional	Process based model	Calculates the properties of the agricultural output, and assess environmental impacts, such as nitrate leaching and greenhouse gas emissions
LPJmL	Global	Dynamic global vegetation models process based	Simulates the terrestrial carbon cycle and the effect on vegetation patterns under climate change for both natural and agricultural ecosystems
ORCHIDEE	Local to global	Dynamic global vegetation models process based	Models both natural ecosystems and human managed carbon, water, and energy dynamics from site to globe scale on sub-daily to centennial scales
CARAIB	Regional	Dynamic global vegetation models process based	This model estimates the net primary productivity (NPP) of the continental vegetation
World3	Global	Integrated global model	Systems dynamics model with five sectors: population, capital, agriculture, nonrenewable resources, and persistent pollution. Limited growth model
IMAGE	Global	Integrated global model	Incorporates different components of the earth system, including oceans, biosphere, atmosphere and anthroposphere (water use and land degradation)
IF	Regional	Integrated global model	Consists of seven submodels: a population, economy, agriculture, energy, social, international policy, environment, and a technical submodel
TARGETS	Global	Integrated global model	Five submodels: population and health, energy, land and food, and water. Each of those submodels is a DPSIR model but they are linked through a socioeconomic scenario generator, in which policy responses are explicitly incorporated
GUMBO	Global	Integrated global model	Five modules: Atmosphere, Lithosphere, Hydrosphere, Biosphere, and Anthroposphere. The Earth's surface is further divided into eleven biomes or ecosystem types. The first global model to include the dynamic feedbacks among human technology, economic production and welfare, and ecosystem goods and services

analytical tools have been developed that are specifically aimed at assessing ecosystem services. These models, or adaptations thereof, can address knowledge gaps such as lack of mapping and specific types of non-marked values of ecosystem services and offer a robust low cost method of quantifying ecosystem services (ELD-Initiative, 2013). Most of the models are spatially explicit, some are dynamic. The best summary of these approaches is in Bagstad et al. (2013). Table 5 is an extract from that paper, listing these models and tools along with brief descriptions.

Each of the tools have its strengths but to highlight one, the Multiscale Integrated Models of Ecosystem Services (MIMES) framework works on many scales to assess the dynamics, magnitude and spatial pattern of ecosystem services. It is an extension of the GUMBO model described in the general modeling section above but is more scalable and spatially explicit. It explicitly addresses the linked dynamics of natural, human, built and social capital, and allows one to integrate site-specific information with regional and global surveys, GIS, and remote sensing data. MIMES is process-based, spatially explicit, dynamic, non-linear simulation model (including carbon, water, nitrogen, phosphorous, plants, consumers (including humans) and a range of ecosystem services) under various climate, economic, and policy scenarios. MIMES is spatially scalable in that it can be applied at multiple spatial and temporal scales from farms to watersheds to countries to globally.

Each "location" in MIMES includes the percent of the land surface in eleven biomes or ecosystem types: Open Ocean, Coastal Ocean, Forests, Grasslands, Wetlands, Lakes/Rivers, Deserts, Tundra, Ice/rock, Croplands, and Urban. The relative areas of each biome at each location change in response to urban and rural population growth, economic production, changes in temperature and precipitation and other variables. Among the biomes, there are exchanges of energy, carbon, nutrients, water and mineral matter. The model calculates the marginal product of ecosystem services in both an economic production and welfare function as estimates of the shadow prices of each service. The number of "locations" or cells in MIMES is variable and can include either grid or polygon representations of multiple locations. For each of these applications, the basic MIMES structure remains the same but the parameters must be recalibrated.

Conventional economic valuation presumes that people have well-formed preferences and enough information about trade-offs that they can adequately judge their "willingness-to-pay". These assumptions do not hold for many ecosystem services. Therefore, we must firstly inform people's preferences (for example by showing them the underlying dynamics of the ecosystems in question using models like MIMES; secondly, allow groups to discuss the issues and "construct" their preferences (again using the MIMES framework to inform the discussions); or thirdly, use other techniques that do not rely on preferences to estimate the

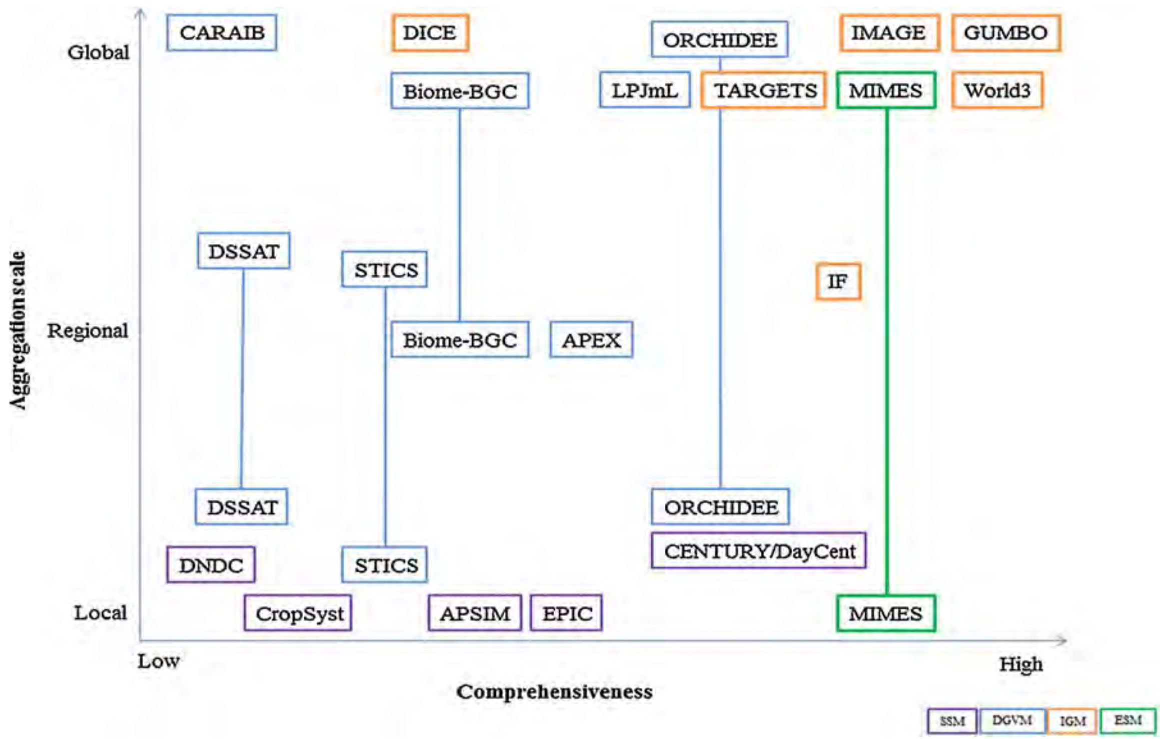

Fig. 3. The distribution of models in regards to comprehensiveness and aggregation scale. Color corresponds to model types Small Scale Models (SSM), Dynamic Global Vegetation Models (DGVM), Integrated Global Model (IGM), Ecosystem Service Model (ESM). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 5
A survey of ecosystem services tools (adapted from Bagstad et al., 2013).

Name	Tool, URL, and references	Brief description
Ecosystem Services Review (ESR)	http://www.wri.org/, (World Resources Institute (WRI), 2012)	Publicly available, spreadsheet-based process to qualitatively assess ecosystem services impacts
Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST)	http://www.naturalcapitalproject.org, (Kareiva et al., 2011; Tallis et al., 2013)	Open source ecosystem service mapping and valuation models accessed through ArcGIS
Artificial Intelligence for Ecosystem Services (ARIES)	http://www.ariesonline.org (Bagstad et al., 2013; Villa et al., 2011)	Open source modeling framework to map ecosystem service flows; online interface and stand-alone web tools under development
LUCI (formerly Polyscape)	http://www.polyscape.org (Jackson et al., 2013)	Open source GIS toolbox to map areas providing services and potential gain or loss of services under management scenarios
Multiscale Integrated Models of Ecosystem Services (MIMES)	http://www.afordablefutures.org	Open source dynamic modeling system for mapping and valuing ecosystem services
EcoServ	Feng et al. (2011)	Web-accessible tool to model ecosystem services
Co$ting Nature	http://www1.policysupport.org/cgi-bin/ecoengine/start.cgi?project=costingnature	Web-accessible tool to map ecosystem services and conservation priority areas
Social Values for Ecosystem Services (SolVES)	http://solves.cr.usgs.gov (Sherrouse et al., 2011)	ArcGIS toolbar for mapping social values for ecosystem services based on survey data or value transfer
Envision	http://envision.bioe.orst.edu, (Guzy et al., 2008)	Integrated urban growth-ecosystem services modeling system; has used external models, including InVEST, or created new ecosystem service models as appropriate
Ecosystem Portfolio Model (EPM),	http://geography.wr.usgs.gov, (Labiosa et al., 2013)	Web-accessible tool to model the impacts of alternative land uses on economic, environmental, and quality of life
InFOREST	http://inforest.frec.vt.edu/	Web-accessible tool to quantify ecosystem services in Virginia
EcoAIM	Waage et al. (2011)	Proprietary tool for mapping ecosystem services and stakeholder preferences
ESValue	Waage et al. (2011)	Proprietary tool for mapping stakeholder preferences for ecosystem services
EcoMetrix	http://www.parametrix.com (Parametrix, 2010)	Proprietary tool for measuring ecosystem services at site scales using field surveys
Natural Assets Information System (NAIS)	http://www.sig-gis.com, (Troy and Wilson, 2006)	Proprietary valuation database paired with GIS mapping of land-cover types for point transfer
Ecosystem Valuation Toolkit	http://www.esvaluation.org (Ecosystem Valuation Toolkit, 2012)	Subscription-based valuation database paired with GIS mapping of land-cover types for point transfer
Benefit Transfer and Use Estimating Model Toolkit	http://www.defenders.org (Loomis and Rosenberger, 2006)	Publicly available spreadsheets, use function transfer to value changes in ecosystem services in the U.S.

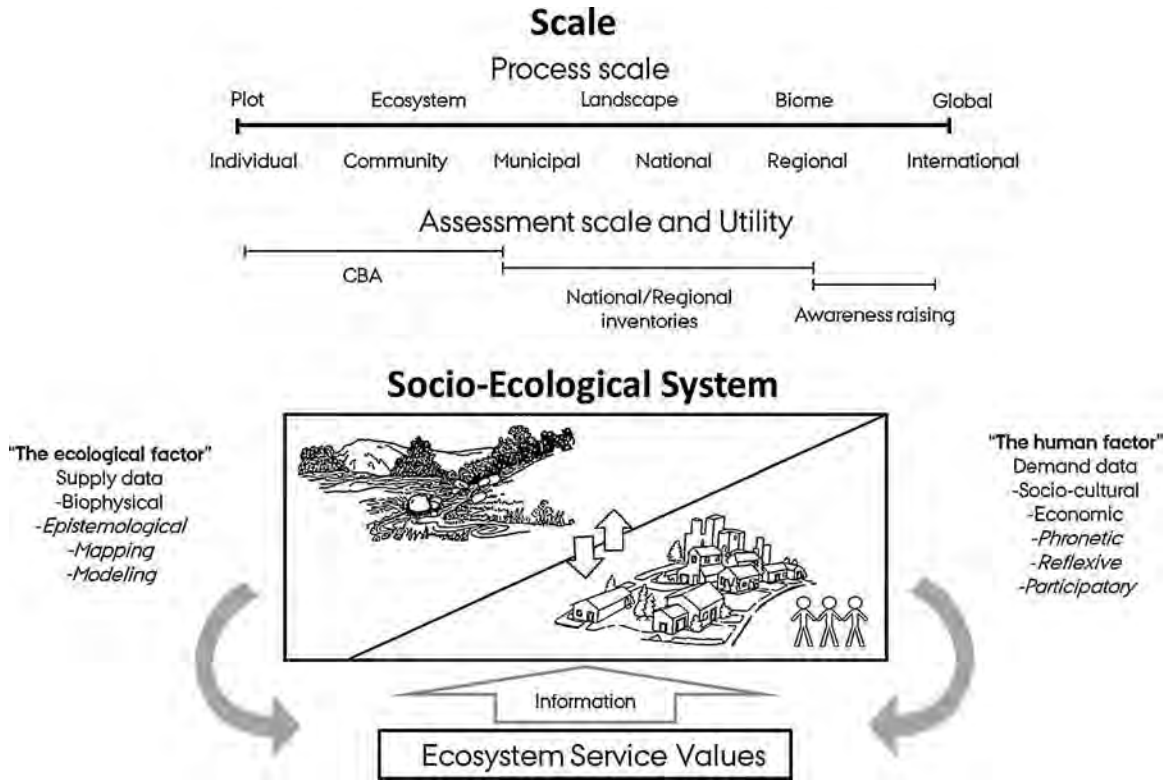

Fig. 4. Conceptual framework developed for mapping and analyzing dynamics and interactions in socio-ecological systems including the scale considerations that are inherent to such assessments (CBA = Cost Benefit Analysis) (Costanza, 2000; Flyvbjerg, 2005; Cumming et al., 2006; de Groot et al., 2010; Willemen et al., 2012; Martín-López et al., 2014).

contribution to human welfare of ecosystem services (i.e. using MIMES to directly infer marginal contributions to welfare). These three methods can be combined to develop new and more integrated methods to value ecosystem services.

In summary, these types of integrated, spatially explicit models are key to describe complex human–nature coupled systems at the global scale. As mentioned in Section 2.2 the model must address the three tiers of sustainability; economy, ecology and society, as well as being spatially explicit.

This is especially because of the strong geographical component in a global model, as well as their utility for dissemination and communication (Sui and Goodchild, 2001). Depending on the context of the chosen site and the integral part it plays in the over-all biosphere (Foley et al., 2011), the choice of model or component will vary with the specific pros and cons, which are listed in the Supplementary information S4–S7. Causal relationships between system variables are not simple and are affected by context, exogenous factors and positive and negative feedback loops (Sterman, 2000). This implies that adequately modeling land degradation, sustainable management practices and restoration require an approach capable of handling these features, such as systems dynamics modeling. The holistic approach that is possible by creating a "whole-systems" perspective model could avoid some of the caveats mentioned in for instance the valuation techniques and the aggregation caveats, and would avoid treating the system as an additive series of disaggregated components (Dale et al., 2013).

4. Discussion

Although there is an impressive suite of models and frameworks to date that can help represent and analyze the context of land degradation, more information is needed to improve on the limitations mentioned. By adopting holistic spatially explicit models to improve on the shortcomings of current valuations of ecosystem services, and a broader estimation of human well-being, it will thus give a better reference point of the real global problem of land degradation than previous collective analyses.

Overall, we found that the largest knowledge gap in this field is assessing the human demand for ecosystem services and the social context. There is a strong tendency to favor measurements of biophysical supply type data over socio-cultural demand data of ecosystem services (Hernández-Morcillo et al., 2013; Iniesta-Arandia et al., 2014; Martín-López et al., 2014). Natural science is oriented toward epistemology and to determine ecological factors—supply, health, and resilience of the ecological system, map and model the biophysical values, or supply value, to provide generalized knowledge that define these matters (Flyvbjerg, 2005; Martín-López et al., 2014). This means that research favors the left side of the socio-ecological box in Fig. 4, and thus there is a lack of information about how the "Human Factors" change with the context they appear in. Methods and models that use epistemological methodologies to describe social context often fall short (Flyvbjerg, 2005; Norton, 2008; Fisher et al., 2009). Instead, Flyvbjerg (2005) argue that a phronetic approach, which describe the value of services as a social context through socio-cultural values and participatory development and assessment, is a better approach to assessing socio-ecological systems (Flyvbjerg, 2005). In practice, this means that the social factors should be considered to a larger extent in the valuation process and further model building.

Such a model complex would need to address connectedness from the field or farm scale to national and global scale economies, as visualized in Fig. 4. Suitable spatial and temporal modules need to be established to reduce the variability and increase the comparability between different management options. This is not an easy task, but one suggestion to combat these shortcomings is to include the 'Human Factor', for instance increase the participatory stakeholder involvement and generate an extended "peer community" (Funtowicz and Ravetz, 1994). Participatory methods seeks to assess benefits to individuals, including the ones that are not so well perceived, benefits to whole communities, and benefits to sustainability (Costanza, 2000). Participatory approaches are becoming more and more common, whether it is to include it in the model (as parameterized interactions between agents), or through participatory model development and scenario building, as described in the following paragraphs.

4.1. The need for participatory model development

Working together with multiple stakeholders in open dialogue at workshops can increase knowledge of a complex system significantly (Kenter et al., 2011). There are various levels of stakeholder involvement in the modeling process (Pretty, 1995). Stakeholders can inform participants on the bigger picture, and individuals' incorrect or insufficient knowledge can be corrected in dialogue and training sessions. Through participatory mapping and valuation exercises it is possible to engage groups of stakeholders in generating a better picture that takes all the components of these systems into account (Costanza and Ruth, 1998; Walker et al., 2002; Mendoza and Prabhu, 2005). At the most elaborate level of stakeholder participation we find participatory modeling, which engages stakeholders in all stages of model development, testing and use (Van den Belt, 2004). Land managers' knowledge is a valuable asset in this approach (Pretty, 2008). Furthermore, stakeholders are critical in developing and testing the scenarios for exploring the future of sustainable land management and finding viable solutions to the problems it is facing. This method enables all parties to participate in understanding and valuing the landscape, from decision makers, academics and farmers, to business leaders and recreationalists, in a holistic approach to land management. Through these participatory engagements it is possible to explore the design for ecosystem services to integrate with the production of conventional commodities, and thus solving issues of ecological degradation. Training local researchers and business personnel to use modeling for ecological assets is expected to accelerate solution adoption and improve innovation and collaboration capacity. Participatory modeling in this context is a powerful tool of stakeholder capacity building and an instrument for 'leveling the playing field' that creates potential for consensus and trust (Voinov and Bousquet, 2010).

4.2. The need for better land use classification systems and definition of scale

Classification systems can be a valuable measure to understand, communicate and compare spatial data, such as land cover data from satellites. The classification systems we have today need to be expanded or redesigned to express the multiple uses that often are associated with the actual land uses. The well-known problem that spatially aggregated measures of geographic attributes tend to conceal local patterns of heterogeneity applies for ES and thus also land use mapping (Troy and Wilson, 2006). The availability of these datasets in time series for change detection is crucial. At the global level, the world 'next year' looks a lot like it did 'last year'; however, with spatially explicit datasets important significant local changes can often be easily identified. Time series of these kinds of datasets used in conjunction with time series of 'event' data such as floods, famines, hurricanes, and civil wars will be useful for identifying areas in which we might want to focus effort and resources.

Land use maps are often based on land covers identified through remote sensing techniques. Major events such as deforestation, forest fragmentation, recent slash-and-burn agriculture, major canopy fires, monoculture, hydroelectric dams and large-scale mining are quite assessable using remote sensing data. However, more subtle land uses and management practices or indicators of changes are not that suitable for this type of determination. This could be e.g. selective logging, subtle edge-effects, soil carbon loss, small scale mining, harvesting of most non-timber plants, implementation of eco-tourism, expansion of narrow unpaved roads (especially in forest), and invasion of non-endemic species (Herold et al., 2011). In these cases data will rely on on-the-ground observation, natural historical collections of material and accounts.

Clearly, there is an abundant amount of data already available. Our collective challenge is to distinguish that which is significant from that which is not. We have the data to see the trends now. The real question is: Can we envision various realistic and probable scenarios and diligently implement, enforce, and maintain the policies and practices needed to achieve those scenarios we collectively decide are most desirable? The scheme needed for this project is one that can deal with land cover and land use and yet also classify based on ecosystem function (or ecosystem dysfunction as in the case of land degradation). Understanding the ecosystem function of a given land cover can often not be determined from aerial photographs or satellite imagery alone. The classification scheme optimized for this type of project will likely need to have inputs from other sources including representations of human, built, and social capital.

Scale is at the center of the land use classification problem. A classification scheme needs to be able to be aggregated up from subcategories to more general categories, as in the CICES protocol mentioned earlier (Haines-Young and Potschin, 2012; Maes et al., 2013). For example, in an urban environment at finer spatial resolution 'commercial', 'transportation', and 'residential' often get aggregated to 'urban' at coarser spatial resolution just as 'ponds', windbreaks of trees', and 'row crops' can be aggregated to 'agriculture'. The enormous quantity of satellite data at many spatial resolutions suggests there is data available at 'theoretically' all scales all the time. These scales (or resolutions) vary from sub-meter to kilometers and meet fundamentally different needs. However, there are times when you need to compare and utilize different resolutions to solve problems, the classification scheme needs to account for this reality. Further complications arise when you have pasture which appear as a grassland (land use versus land cover) and these may have very different economic, and ecosystem service values.

4.3. The need for alternative futures assessments

As shown there are much data to access and multiple models at multiple scales to be employed for sustainable land management assessments. One way to use the models and data described above without being captured by the modeling caveat of complexity vs. applicability would be to generate and evaluate scenarios that embody different possibilities. Bateman et al. (2013) is a good example of this approach. They evaluated six scenarios for UK agriculture and ecosystem services using a range of models. Results showed that scenarios (and their underlying policies) that included the value of ecosystem services were an order of magnitude more valuable than those that focused only on conventional, marketed agricultural products.

For projects based on evaluating land use and management scenarios based on an inclusive assessment of the total value of the land, the methodology addresses solutions to many of the knowledge deficits raised by for instance the ELD Initiative (ELD-Initiative, 2013). However, new questions are raised and the following challenges for this type of assessment should be considered:

1. The TEV is very complicated and costly to calculate, and often the results of a valuation are not included in decision making.

2. There is not yet one standard methodology to TEV, different land and services are measured differently across studies.
3. There are no simple ways of conducting a robust TEV of land.
4. Only partially complete studies have been conducted on land, not considering the full range of services, and therefore comparisons are highly problematic.

The challenge for sustainable land management assessment projects is to overcome these problems and produce a set of tools and techniques that can better assess the true value of land in a consistent, credible, and relatively inexpensive way. Only then will we be able to manage our land sustainably and well. Based on this review of methods, data and models we infer what a system that meets these needs could look like. It would be based on an integrated, dynamic, spatially explicit, scalable computer simulation model (something like MIMES) that has been developed in a participatory way and calibrated at a number of sites around the world.

This general model could be parameterized with global GIS data sets over the internet. The user interface would allow individual users to specify any area of the earth and be able to run a version of the model that would provide dynamics and values of natural capital and ecosystem services along with built, human, and social capital. It would also allow land use policy scenarios to be quickly run and compared. An advanced gaming interface would allow the model to be "played" by a large number of people and their tradeoff decisions (and the valuations they imply) to be accumulated and compared. It would allow the true value of land to be assessed in a consistent and relatively inexpensive way. This will allow us to manage our land sustainably and well.

Substantial literature on simulation modeling for ecosystem services is available at these scales that can form the basis for these games (Boumans et al., 2002; Costanza and Voinov, 2003; Craft et al., 2008; Arkema and Samhouri, 2009; Nelson et al., 2009) including the MIMES framework mentioned above, but these models have yet to be used for integrating gaming. By combining our scientific understanding of how ecosystems function (embedded in dynamic landscape models) with the ability to quickly and cheaply solicit input from a broad range of participants, this approach could have a huge impact on how ecosystem service valuation and management is done. This would dramatically improve our ability to understand and manage ecosystems and benefit society at large.

4.4. The need for multi-scale user interfaces and games

The development and use of models also opens the option of using advanced user interfaces and games in research designed to test for preferences and choices in natural resource management. Land and urban management games have been around for many years (e.g. SimCity in 1989 or SimFarm in 1994), and are considered classics in computer gaming. Using a computer-based system allows for participants to make decisions based on information provided by a user interface, and allows for information and the resulting behaviors to be presented rapidly, allowing more information provision, decision making points, and greater number of repeated trials. Behind the user interface, there is an underlying dynamic simulation model that updates the games database as decisions are made. Janssen et al. (2010) conducted a series of computer-based experiments to test the impact of communication and punishment in common-pool resource management, finding that respondents are willing to engage in punishing defectors even at personal cost, however punishment without communication does not increase overall payoffs.

Heckbert et al. (2011) present an integrated GIS-based agent-based model and an experimental economics platform where participants take on the role as an agent, or avatar, in a dynamic simulation model of agricultural land management. This example shows the progression of experiments toward multi-player gaming, however, experimental economics usually recommends omitting context and complexity from the decision making situation in order to isolate the influence of a given tradeoff decision. For example in the Heckbert et al. (2011) application, experiment participants applied 'inputs' to 'production' rather than 'fertilizer' to 'agriculture' to avoid social biases regarding the system they were managing. In the Janssen et al. (2010) example, participants use an abstract gridded board with tokens to represent a common-pool resource—not unlike common board games (e.g. backgammon, checkers, go).

Dynamic modeling's strength is simulating links between human decisions and ecosystem functioning, but, used alone, it still lacks the human component. To gain this element it could be useful to convert the ecosystem model onto a gaming platform. That would allow for stakeholders to create their own version of a desired "best outcome" through playing the game. Since the model can embed the trade-offs between, for example, a better environment and food production, the choices players make during the game will reflect how they value tradeoffs like these. Information that reveals the nature of interactions among players may also prove useful. This is somewhat like a conventional choice experiment (Wilson and Carpenter, 1999; Colombo et al., 2013) except that players are able to create their own scenarios rather than the researcher presenting them with a fixed set. Preferences can also emerge as a result of learning about the system or from group interactions. The advantages of using a computer game to capture data about preferences include:

1. The ability to quickly replicate experiments;
2. A large potential data pool, with potential sample sizes in the thousands;
3. The ability to track and record player decisions through in-built survey mechanisms;
4. Removing a degree of interviewer bias by enabling users to interact directly with an experiment rather than via an interviewer (albeit, the experiment is one step removed from the game developers or surrogate interviewers);
5. The ability to place people's choices within contexts that are (potentially) more realistic than the typical context-neutral experimental form; and
6. Assessing whether increasing degrees of game 'realism' (complexity), or introducing different elements within a game, affects people's valuations and preferences?

There is of course the unresolved question of whether players will value things in the game context in the same way they do in real life, but this problem is also applicable to experimental economics studies, so is not unique to this type of valuation technique.

5. Conclusions

We have collected and reviewed the valuation methods, the drivers and the models that can and are used in analyzing the effects of land management practices. Furthermore we include a collection of some of the most extensive global databases that harbor spatially explicit data on the biophysical, economic, institutional and practical drivers of sustainable land management. There are a lot of data, methods and models available, and the sheer amount warrants for extensive consolidation and assessment. Modeling at an appropriate scale for land management, policy making or global awareness making are equally important and necessary to gain a full perspective of the problem. Different frameworks can be adapted by different disciplines based on data availability.

Some of our current tools for consolidating data are not equipped to handle the diversity and transdisciplinarity these types of analyses include. This is noted, for example, in our current land use classification schemes. The most important lesson learned is the fact that models in their imperfection should have a measure of the "human factor" included in some way to increase applicability. In practice this means they should be models – preferably whole systems – that are validated on some level with the participation of key stakeholders. A further way proposed to overcome their caveats is to use them as scenario building tools to test alternative futures, and/or taking the research a step further, creating a platform that utilizes choice experiments of gaming communities. These measures are tools to increase the sustainability of our current management practices and curb the development of short term decision making.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolmodel.2015.07.017

References

ACCA, Fauna & Flora International, KPMG, 2012. Is Natural Capital a Material Issue? An Evaluation of the Relevance of Biodiversity and Ecosystem Services to Accountancy Professionals and the Private Sector. ACCA, Fauna & Flora International and KPMG.

Acevedo, M.F., 2011. Interdisciplinary progress in food production, food security and environment research. Environ. Conserv. 38, 151–171.

Aertsen, W., Kint, V., Muys, B., Van Orshoven, J., 2012. Effects of scale and scaling in predictive modelling of forest site productivity. Environ. Modell. Softw. 31, 19–27.

Allison, H.E., Hobbs, R.J., 2004. Resilience, adaptive capacity, and the lock-in trap of the Western Australian agricultural region. Ecol. Soc. 9 (1).

Anand, S., Sen, A., 1994. Human development index: methodology and measurement. In: United Nations Development Programme (UNDP). Human Development Report Office (HDRO).

Anderson, J.R., 1976. A Land Use and Land Cover Classification System for Use with Remote Sensor Data. US Government Printing Office.

Andersson, E., Brogaard, S., Olsson, L., 2011. The political ecology of land degradation. Annu. Rev. Environ. Resour. 36, 295–319.

Arkema, K.K., Samhouri, J.F., 2009. Linking ecosystem health and services to inform marine ecosystem-based management. In: American Fisheries Society Symposium, pp. 9–25.

Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. Ecosyst. Serv. 5, 27–39.

Bahadur, K.C.K., 2012. Spatio-temporal patterns of agricultural expansion and its effect on watershed degradation: a case from the mountains of Nepal. Environ. Earth Sci. 65, 2063–2077.

Barbier, E.B., 2000. The economic linkages between rural poverty and land degradation: some evidence from Africa. Agric. Ecosyst. Environ. 82, 355–370.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monograph. 81, 169–193.

Bateman, I.J., Harwood, A.R., Mace, G.M., Watson, R.T., Abson, D.J., Andrews, B., Binner, A., Crowe, A., Day, B.H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A.A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., Termansen, M., 2013. Bringing ecosystem services into economic decision-making: land use in the United Kingdom. Science 341, 45–50.

Batker, D., de la Torre, I., Costanza, R., Swedeen, P., Day, J., Boumans, R., Bagstad, K., 2010. Gaining ground: wetlands, Hurricanes and the Economy: The Value of Restoring the Mississippi River Delta. Earth Economics, Takoma, WA.

Bell, M., Cox, H., Harch, G., Kirby, D., O'Mara, B., Pilcher, S., Smith, L., Want, P., 2013. In: E.D.a.I.A.-S.Q. Department of Employment (Ed.), Improving Phosphorous Fertiliser Management on southern Queensland Grain Farms. Department of Employment.

Bolan, N.S., Hedley, M.J., White, R.E., 1991. Processes of soil acidification during nitrogen cycling with emphasis on legume based pastures. Plant Soil 134, 53–63.

Boumans, R., Costanza, R., Farley, J., Wilson, M.A., Portela, R., Rotmans, J., Villa, F., Grasso, M., 2002. Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model. Ecol. Econ. 41, 529–560.

Bouwman, A.F., Van Drecht, G., Van der Hoek, K.W., 2005. Global and regional surface nitrogen balances in intensive agricultural production systems for the period 1970–2030. Pedosphere 15, 137–155.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ. 63, 616–626.

Brennan, R.F., Bolland, M.D.A., Bowden, J.W., 2004. Potassium deficiency, and molybdenum deficiency and aluminium toxicity due to soil acidification, have become problems for cropping sandy soils in south-western Australia. Aust. J. Exp. Agric. 44, 1031–1039.

Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., Defries, R.S., Diaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., Sarukhan, J., Scholes, R.J., Whyte, A., 2009. Science for managing ecosystem services: beyond the millennium ecosystem assessment. Proc. Natl. Acad. Sci. U.S.A. 106, 1305–1312.

Cihlar, J., Jansen, L.J.M., 2001. From land cover to land use: a methodology for efficient land use mapping over large areas. Prof. Geographer 53, 275–289.

Colombo, S., Hanley, N., Christie, M., 2013. What are the Consequences of Ignoring Attributes in Choice Experiments? An Application to Ecosystem Service Values. Ecol. Econ. 96, 25–35.

Costanza, R., 2000. Social goals and the valuation of ecosystem services. Ecosystems 3, 4–10.

Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. Biol. Conserv. 141, 350–352.

Costanza, R., Alperovitz, G., Daly, H., Farley, J., Franco, C., Jackson, T., Kubiszewski, I., Schor, J., Victor, P., 2013. Building a Sustainable and Desirable Economy-in-Society-in-Nature. ANU E Press, Canberra, Australia.

Costanza, R., Daly, H.E., 1992. Natural capital and sustainable development. Conserv. Biol. 6, 37–46.

Costanza, R., dArge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., Oneill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Costanza, R., de Groot, R., Sutton, P.C., van der Ploeg, S., Anderson, S., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. Global Environ. Change 26, 152–158.

Costanza, R., Fisher, B., Ali, S., Beer, C., Bond, L., Boumans, R., Danigelis, N.L., Dickinson, J., Elliott, C., Farley, J., 2007. Quality of life: an approach integrating opportunities, human needs, and subjective well-being. Ecol. Econ. 61, 267–276.

Costanza, R., Kubiszewski, I., Ervin, D., Bluffstone, R., Boyd, J., Brown, D., Chang, H., Dujon, V., Granek, E., Polasky, S., 2011a. Valuing ecological systems and services. In: F1000 Biology Reports., pp. 3.

Costanza, R., Kubiszewski, I., Ervin, D., Bluffstone, R., Boyd, J., Brown, D., Chang, H., Dujon, V., Granek, E., Polasky, S., Shandas, V., Yeakley, A., 2011b. Valuing ecological systems and services. In: F1000 Biology Reports., pp. 3.

Costanza, R., Kubiszewski, I., Roman, J., Sutton, P., 2011c. Changes in Ecosystem Services and Migration in Low-lying Coastal Areas over the Next Fifty Years UK Government's Chief Scientific Advisor Sir John Beddington, Crown copyright, http://www.foresight.gov.uk, 11/1175, London.

Costanza, R., Mitsch, W.J., Day, J.W., 2006. A new vision for New Orleans and the Mississippi delta: applying ecological economics and ecological engineering. Front. Ecol. Environ. 4, 465–472.

Costanza, R., Pérez-Maqueo, O., Martinez, M.L., Sutton, P., Anderson, S.J., Mulder, K., 2008. The value of coastal wetlands for hurricane protection. AMBIO: J. Hum. Environ. 37, 241–248.

Costanza, R., Ruth, M., 1998. Using dynamic modeling to scope environmental problems and build consensus. Environ. Manage. 22, 183–195.

Costanza, R., Voinov, A., 2003. Landscape simulation modeling: a spatially explicit. In: Dynamic Approach. Springer, New York, NY.

Costanza, R., Voinov, A., Boumans, R., Maxwell, T., Villa, F., Wainger, L., Voinov, H., 2002. Integrated ecological economic modeling of the Patuxent River watershed, Maryland. Ecol. Monograph. 72, 203–231.

Cowie, A.L., Penman, T.D., Gorissen, L., Winslow, M.D., Lehmann, J., Tyrrell, T.D., Twomlow, S., Wilkes, A., Lal, R., Jones, J.W., Paulsch, A., Kellner, K., Akhtar-Schuster, M., 2011. Towards Sustainable Land Management in the Drylands: Scientific Connections in Monitoring and Assessing Dryland Degradation, Climate Change and Biodiversity. Land Degrad. Dev. 22, 248–260.

Craft, C., Clough, J., Ehman, J., Joye, S., Park, R., Pennings, S., Guo, H., Machmuller, M., 2008. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. Front. Ecol. Environ. 7, 73–78.

Creasy, R., 1993. Preterm labor and delivery. In: Creasy, R., Resnik, R. (Eds.), Maternal-fetal Medicine: Principles and Practice. WB Saunders, Philadelphia.

Croen, L., Shaw, G., Jensvold, N., Harris, J., 1991. Birth defects monitoring in California: a resource for epidemiological research. Paediatr. Perinat. Epidemiol. 5, 423–427.

Cumming, G.S., Cumming, D.H., Redman, C.L., 2006. Scale mismatches in social–ecological systems: causes, consequences, and solutions. Ecol. Soc. 11, 14.

Dalal, R.C., Strong, W.M., Weston, E.J., Gaffney, J., 1991. Sustaining multiple production systems. 2. Soil fertility decline and restoration of cropping lands in sub-tropical Queensland. Trop. Grassl. 25, 173–180.

Dale, V.H., Kline, K.L., Kaffka, S.R., Langeveld, J.H., 2013. A landscape perspective on sustainability of agricultural systems. Landscape Ecol. 28, 1111–1123.

Dasgupta, P., 2008. Nature in economics. Environ. Resour. Econ. 39, 1–7.

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L.C., ten Brink, P., van Beukering, P., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosyst. Serv. 1, 50–61.

de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393–408.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complexity 7, 260–272.

DeFries, R.S., Ellis, E.C., Chapin III, F.S., Matson, P.A., Turner II, B.L., Agrawal, A., Crutzen, P.J., Field, C., Gleick, P., Kareiva, P.M., Lambin, E., Liverman, D., Ostrom, E., Sanchez, P.A., Syvitski, J., 2012. Planetary opportunities: a social contract for global change science to contribute to a sustainable future. Bioscience 62, 603–606.

Dirzo, R., Young, H.S., Galetti, M., Ceballos, G., Isaac, N.J., Collen, B., 2014. Defaunation in the anthropocene. Science 345, 401–406.

Dobson, J.E., Bright, E.A., Coleman, P.R., Durfee, R.C., Worley, B.A., 2000. LandScan: a global population database for estimating populations at risk. Photogramm. Eng. Remote Sens. 66, 849–857.

Dodds, P.S., Harris, K.D., Kloumann, I.M., Bliss, C.A., Danforth, C.M., 2011. Temporal patterns of happiness and information in a global social network: hedonometrics and twitter. PLoS ONE 6, e26752.

Douglas, I.A.N., 2006. The local drivers of land degradation in South-East Asia. Geogr. Res. 44, 123–134.

Dunlop, M., Turner, G.M., Howden, S.M., 2004. Future Sustainability of the australian grains industry. In: CSIRO Sustainable Ecosystems. Canberra, Australia.

Ecosystem Valuation Toolkit, 2012. http://esvaluation.org (accessed 04.08.15).

ELD-Initiative, 2013. The rewards of investing in sustainable land management. In: Interim Report for the Economics of Land Degradation Initiative: A Global Strategy for Sustainable Land Management.

Ellis, E.C., Klein Goldewijk, K., Siebert, S., Lightman, D., Ramankutty, N., 2010. Anthropogenic transformation of the biomes, 1700 to 2000. Global Ecol. Biogeogr 19, 589–606.

Ellison, N.B., Steinfield, C., Lampe, C., 2011. Connection Strategies: Social Capital Implications of Facebook-enabled Communication Practices. New Media & Society.

Elvidge, C.D., Tuttle, B.T., Sutton, P.C., Baugh, K.E., Howard, A.T., Milesi, C., Bhaduri, B., Nemani, R., 2007. Global distribution and density of constructed impervious surfaces. Sensors 7, 1962–1979.

Farber, S., Costanza, R., Childers, D.L., Erickson, J., Gross, K., Grove, M., Hopkinson, C.S., Kahn, J., Pincetl, S., Troy, A., Warren, P., Wilson, M., 2006. Linking ecology and economics for ecosystem management. Bioscience 56, 121–133.

Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecol. Econ. 41, 375–392.

Farley, J., Costanza, R., 2002. Envisioning shared goals for humanity: a detailed, shared vision of a sustainable and desirable USA in 2100. Ecol. Econ. 43, 245–259.

Farley, J., Schmitt, A., Burke, M., Farr, M., 2014. Extending market allocation to ecosystem services: moral and practical implications on a full and unequal planet. Ecol. Econ.

Feng, M., Liu, S., Euliss Jr., N.H., Young, C., Mushet, D.M., 2011. Prototyping an online wetland ecosystem services model using open model sharing standards. Environ. Model. Softw. 26, 458–468.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643–653.

Flyvbjerg, B., 2005. Social science that matters. Foresight Eur. 2, 38–42.

Foley, J.A., Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–574.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockstrom, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342.

Forum, W.E., 2013. The Human Capital Report.

Frolking, S.M.T., Seto, K.C., Friedl, M.A., 2013. A global fingerprint of macro-scale changes in urban structure from 1999 to 2009. Environ. Res. Lett. 8 (2), 024004, 1–10.

Funtowicz, S.O., Ravetz, J.R., 1994. Uncertainty, complexity and post-normal science. Environ. Toxicol. Chem. 13, 1881–1885.

Galetti, M., Dirzo, R., 2013. Ecological and evolutionary consequences of living in a defaunated world. Biol. Conserv. 163, 1–6.

Gedan, K.B., Kirwan, M.L., Wolanski, E., Barbier, E.B., Silliman, B.R., 2011. The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. Clim. Change 106, 7–29.

Geist, H.J., Lambin, E.F., 2004. Dynamic causal patterns of desertification. Bioscience 54, 817–829.

Glover, J.D., Culman, S.W., DuPont, T.S., Broussard, W., Young, L., 2010. Harvested perennial grasslands provide ecological benchmarks for agricultural sustainability. Agric. Ecosyst. Environ. 137, 3–12.

Goodchild, M.F., 2009. The quality of geospatial context. In: Quality of Context. Springer, pp. 15–24.

Granek, E.F., Polasky, S., Kappel, C.V., Reed, D.J., Stoms, D.M., Koch, E.W., Kennedy, C.J., Cramer, L.A., Hacker, S.D., Barbier, E.B., Aswani, S., Ruckelshaus, M., Perillo, G.M., Silliman, B.R., Muthiga, N., Bael, D., Wolanski, E., 2010. Ecosystem services as a common language for coastal ecosystem-based management. Conserv. Biol. 24 (1), 207–216.

Guzy, M.R., Smith, C.L., Bolte, J.P., Hulse, D.W., Gregory, S.V., 2008. Policy research using agent based modeling to assess future impacts of urban expansion into farmlands and forests. Ecol. Soc. 13 (1), 37.

Haines-Young, R.H., Potschin, M., 2012. Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August–December 2012.

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O., Townshend, J.R.G., 2013. High-resolution global maps of 21st-century forest cover change. Science 342, 850–853.

Heckbert, S., Bishop, I., Marceau, D., Beneson, I., 2011. Empirical calibration of spatially explicit agent-based models. In: Marceau, D., Benenson, I. (Eds.), Advanced Geo-Simulation Models. Bentham, p. 92.

Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol. Econ. 57, 209–228.

Henderson, J.V., Storeygard, A., Weil, D.N., 2012. Measuring economic growth from outer space. Am. Econ. Rev. 102, 994–1028.

Hernández-Morcillo, M., Plieninger, T., Bieling, C., 2013. An empirical review of cultural ecosystem service indicators. Ecol. Indic. 29, 434–444.

Herold, M., Roman-Cuesta, R., Mollicone, D., Hirata, Y., Van Laake, P., Asner, G., Souza, C., Skutsch, M., Avitabile, V., MacDicken, K., 2011. Options for monitoring and estimating historical carbon emissions from forest degradation in the context of REDD+. Carbon Balance Manage. 6, 13.

Howarth, R.B., Farber, S., 2002. Accounting for the value of ecosystem services. Ecol. Econ. 41, 421–429.

Iniesta-Arandia, I., García-Llorente, M., Aguilera, P.A., Montes, C., Martín-López, B., 2014. Socio-cultural valuation of ecosystem services: uncovering the links between values, drivers of change, and human well-being. Ecol. Econ. 108, 36–48.

IPCC, 2007. IPCC Fourth Assessment Report (AR4). Intergovernmental Panel on Climate Change, Cambridge.

Jack, B.K., Kousky, C., Sims, K.R., 2008. Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. Proc. Natl. Acad. Sci. U.S.A. 105, 9465–9470.

Jackson, B., Pagella, T., Sinclair, F., Orellana, B., Henshaw, A., Reynolds, B., McIntyre, N., Wheater, H., Eycott, A., 2013. Polyscape: a GIS mapping toolbox providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. Landscape Urban Plan. 112, 74–88.

Jakeman, A.J., Barreteau, O., Borsuk, M.E., Elsawah, S., Hamilton, S.H., Henriksen, H.J., Kuikka, S., Maier, H.R., Rizzoli, A.E., Van Delden, H., 2013. Selecting among five common modelling approaches for integrated environmental assessment and management. Environ. Modell. Softw. 47, 159–181.

Jansen, L.J., Gregorio, A.D., 2002. Parametric land cover and land-use classifications as tools for environmental change detection. Agric. Ecosyst. Environ. 91, 89–100.

Janssen, M.A., Holahan, R., Lee, A., Ostrom, E., 2010. Lab experiments for the study of social-ecological systems. Science 328, 613–617.

Jarchow, M.E., Kubiszewski, I., Larsen, G.L.D., Zdorkowski, G., Costanza, R., Gailans, S.R., Ohde, N., Dietzel, R., Kaplan, S., Neal, J., Petrehn, M.R., Gunther, T., D'Adamo, S.N., McCann, N., Larson, A., Damery, P., Gross, L., Merriman, M., Post, J., Sheradin, M., Liebman, M., 2012. The future of agriculture and society in Iowa: four scenarios. Int. J. Agric. Sustainability 10, 76–92.

Kahneman, D., 2011. Thinking Fast and Slow. Farrar Straus and Giroux, New York, NY.

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S., 2011. Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford University Press, Oxford.

Kenter, J.O., Hyde, T., Christie, M., Fazey, I., 2011. The importance of deliberation in valuing ecosystem services in developing countries-Evidence from the Solomon Islands. Global Environ. Change 21, 505–521.

Kosmas, C., Kairis, O., Karavitis, C., Ritsema, C., Salvati, L., Acikalin, S., Alcalá, M., Alfama, P., Atlhopheng, J., Barrera, J, et al., 2014. Evaluation and selection of indicators for land degradation and desertification monitoring: methodological approach. Environ. Manage. 54, 951–970.

Kroeger, T., Casey, F., 2007. An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecol. Econ. 64, 321–332.

Kwansoo, K., Barham, B.L., Coxhead, I., 2001. Measuring soil quality dynamics: a role for economists, and implications for economic analysis. Agric. Econ. 25, 13–26.

Labiosa, W.B., Forney, W.M., Esnard, A.-M., Mitsova-Boneva, D., Bernknopf, R., Hearn, P., Hogan, D., Pearlstine, L., Strong, D., Gladwin, H., Swain, E., 2013. An integrated multi-criteria scenario evaluation web tool for participatory land-use planning in urbanized areas: the ecosystem portfolio model. Environ. Model. Softw. 41, 210–222.

Lal, R., 1997. Degradation and resilience of soils. Philos. Trans. R. Soc. London, Ser. B: Biol. Sci. 352, 997–1010.

Lal, R., 2012. Climate change and soil degradation mitigation by sustainable management of soils and other natural resources. Natl. Acad. Agric. Scie. 3, 199–212.

Lamarque, P., Quetier, F., Lavorel, S., 2011. The diversity of the ecosystem services concept and its implications for their assessment and management. C.R. Biol. 334, 441–449.

Lambin, E.F., Turner, B.L., Geist, H.J., Agbola, S.B., Angelsen, A., Bruce, J.W., Coomes, O.T., Dirzo, R., Fischer, G., Folke, C., George, P.S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E.F., Mortimore, M., Ramakrishnan, P.S., Richards, J.F., Skånes, H., Steffen, W., Stone, G.D., Svedin, U., Veldkamp, T.A., Vogel, C., Xu, J., 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environ. Change 11, 261–269.

Lautenbach, S., Kugel, C., Lausch, A., Seppelt, R., 2011. Analysis of historic changes in regional ecosystem service provisioning using land use data. Ecol. Indic. 11, 676–687.

Lavelle, P., Spain, A.V., 2005. Soil Ecology. Springer, Dordrecht.

Liu, S., Stern, D.I., 2008. A meta-Analysis of Contingent Valuation Studies in Coastal and Near-Shore Marine Ecosystems MPRA Paper. University Library of Munich, Germany, http://mpra.ub.uni-muenchen.de/11720/3/MPRA_paper_11720.pdf (accessed 04.08.15).

Loomis, J.B., Rosenberger, R.S., 2006. Reducing barriers in future benefit transfers: needed improvements in primary study design and reporting. Ecol. Econ. 60, 343–350.

Luisetti, T., Turner, R.K., Bateman, I.J., Morse-Jones, S., Adams, C., Fonseca, L., 2011. Coastal and marine ecosystem services valuation for policy and management: managed realignment case studies in England. Ocean Coast. Manage. 54, 212–224.

Maes, J., Teller, A., Erhard, M., Liquete, C., Braat, L., Berry, P., Egoh, B., Puydarrieux, P., Fiorina, C., Santos, F., 2013. Mapping and Assessment of Ecosystems and their Services—An Analytical Framework for Ecosystem Assessments under Action 5 of the EU Biodiversity Strategy to 2020 Discussion paper, http://dx.doi.org/10.2779/12398, ISBN: 978-92-79-29369-6.

Malecki, E.J., Wei, H., 2009. A wired world: the evolving geography of submarine cables and the shift to Asia. Ann. Assoc. Am. Geographers 99, 360–382.

Martín-López, B., Gómez-Baggethun, E., García-Llorente, M., Montes, C., 2014. Trade-offs across value-domains in ecosystem services assessment. Ecol. Indic. 37 (Part A), 220–228.

McIntyre, S., Tongway, D., 2005. Grassland structure in native pastures: links to soil surface condition. Ecol. Manage. Restor. 6, 43–50.

MEA, 2005. Ecosystems and Human Well-Being: Synthesis. Island Press, Washington, DC.

Mendoza, G.A., Prabhu, R., 2005. Combining participatory modeling and multi-criteria analysis for community-based forest management. For. Ecol. Manage. 207, 145–156.

Monteiro, A.T., Fava, F., Hiltbrunner, E., Della Marianna, G., Bocchi, S., 2011. Assessment of land cover changes and spatial drivers behind loss of permanent meadows in the lowlands of Italian Alps. Landscape Urban Plann. 100, 287–294.

Nakicenovic, N., Alcamo, J., Davis, G., De Vries, B., Fenhann, J., Gaffin, S., Gregory, K., Grübler, A., Jung, T.Y., Kram, T., 2000. Special report on emissions scenarios, working group III. In: Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge.

Nelson, E., Mendoza, G., Regetz, J., Ploasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Dailey, G.C., Goldstein, J., Dareiva, P.M., Lansdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M.R., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4–11.

Nkonya, E., Gerber, N., Baumgartner, P., Von Braun, J., De Pinto, A., Graw, V., Kato, E., Kloos, J., Walter, T., 2011. The economics of desertification, land degradation, and drought: toward an integrated global assessment. In: ZEF Discussion Papers on Development Policy.

Nkonya, E., von Braun, J., Mirzabaev, A., Le, Q., Kwon, H., Kato, A., Kirui, O., Gerber, N., 2013. Economics of land degradation initiative: methods and approach for global and national assessments. In: ZEF-IFPRI Discussion Paper. Center for Development Research (ZEF), pp. 34.

Nordhaus, W.D., 1973. World dynamics: measurement without data. Econ. J. 83, 1156–1183.

Norton, B.G., 2008. Beyond positivist ecology: toward an integrated ecological ethics. Sci. Eng. Ethics 14, 581–592.

Oleson, K.L., 2011. Shaky foundations and sustainable exploiters problems with national weak sustainability measures in a global economy. J. Environ. Dev. 20, 329–349.

Parametrix, 2010. An Introduction to EcoMetrix: Measuring Change in Ecosystem Performance at the Site Scale. Parametrix, Portland, OR.

Pender, J.L., Weber, B.A., Johnson, T.G., Fannin, J.M., 2014. Rural Wealth Creation. Taylor & Francis, New York.

Pingali, P.L., 2013. Green revolution: impacts, limits, and the path ahead. Proc. Natl. Acad. Sci. U.S.A., 109.

Polasky, S., Nelson, E., Pennington, D., Johnson, K.A., 2011. The impact of land-use change on ecosystem services, biodiversity and returns to landowners: a case study in the state of Minnesota. Environ Resour. Econ. 48, 219–242.

Pretty, J., 2008. Agricultural sustainability: concepts, principles and evidence. Philos. Trans. R. Soc. London, Ser. B: Biol. Sci. 363, 447–465.

Pretty, J.N., 1995. Participatory learning for sustainable agriculture. World Dev. 23, 1247–1263.

Putnam, R.D., 1995. Tuning in, tuning out: the strange disappearance of social capital in America. Polit. Sci. Polit. 28, 664–683.

Randall, G.W., Huggins, D.R., Russelle, M.P., Fuchs, D.J., et al., 1997. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. J. Environ. Q. 26, 1240.

Sala, O.E., Stuart Chapin III, F., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M.n., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the Year 2100. Science 287, 1770–1774.

Scholes, R.J., Reyers, B., Biggs, R., Spierenburg, M.J., Duriappah, A., 2013. Multi-scale and cross-scale assessments of social–ecological systems and their ecosystem services. Curr. Opin. Environ. Sustainability 5 (1), 16–25, ⟨http://dx.doi.org/10.1016/j.cosust.2013.01.004⟩ (ISSN 1877-3435).

Scott, B.J., Ridley, A.M., Conyers, M.K., 2000. Management of soil acidity in long-term pastures of south-eastern Australia: a review. Aust. J. Exp. Agric. 40, 1173–1198.

Sen, A., 1997. Editorial: human capital and human capability. World Dev. 25, 1959–1961.

Seppelt, R., Dormann, C.F., Eppink, F.V., Lautenbach, S., Schmidt, S., 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. J. App. Ecol. 48, 630–636.

Seppelt, R., Fath, B., Burkhard, B., Fisher, J.L., Gret-Regamey, A., Lautenbach, S., Pert, P., Hotes, S., Spangenberg, J., Verburg, P.H., Van Oudenhoven, A.P.E., 2012. Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. Ecol. Indic. 21, 145–154.

Shepherd, K.D., Farrow, A., Ringler, C., Gassner, A., Jarvis, D., 2013. Review of the evidence on indicators, metrics and monitoring systems. In: Research Program on Water, Land and Ecosystems. World Agroforestry Centre.

Sherrouse, B.C., Clement, J.M., Semmens, D.J., 2011. A GIS application for assessing, mapping, and quantifying the social values of ecosystem services. Appl. Geogr. 31, 748–760.

Stafford Smith, D.M., McKeon, G.M., Watson, I.W., Henry, B.K., Stone, G.S., Hall, W.B., Howden, S.M., 2007. Learning from episodes of degradation and recovery in variable Australian rangelands. Proc. Natl. Acad. Sci. U.S.A. 104, 20690–20695.

Stafford Smith, M., 1994. Sustainable production systems and natural resource managementin the rangelands. In: ABARE Outlook 1994. Alice Springs.

Stallman, H.R., 2011. Ecosystem services in agriculture: determining suitability for provision by collective management. Ecol. Econ. 71, 131–139.

Sterman, J.D., 2000. Business Dynamics: Systems Thinking and Modeling for a Complex World. McGraw-Hill Higher Education, USA.

Stern, N., 2007. The Economics of Climate Change: The Stern Review. Cambridge Press, Cambridge, UK.

Sui, D.Z., Goodchild, M.F., 2001. GIS as media? Int. J. Geogr. Inf. Sci. 15, 387–390.

Sukhdev, P., Kumar, P., 2010. The Economics of Ecosystems and Biodiversity, (TEEB). European Communities Brussels.

Sumner, M., Noble, A., 2003. Handbook of Soil Acidity. Marcel Dekker AG, New York, NY.

Sutton, P.C., Anderson, S.J., Tuttle, B.T., Morse, L., 2012. The real wealth of nations: mapping and monetizing the human ecological footprint. Ecol. Indic. 16, 11–22.

Sutton, P.C., Costanza, R., 2002. Global estimates of market and non-market values derived from nighttime satellite imagery, land cover, and ecosystem service valuation. Ecol. Econ. 41, 509–527.

Swinton, S.M., Lupi, F., Robertson, G.P., Hamilton, S.K., 2007. Ecosystem services and agriculture: cultivating agricultural ecosystems for diverse benefits. Ecol. Econ. 64, 245–252.

Tallis, H.T., Ricketts, T., Guerry, A., Wood, S.A., Sharp, R., Nelson, E., Ennaanay, D., Wolny, S., Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G., Aukema, J., Foster, J., Forrest, J., Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C., Verutes, G., Kim, C., Guannel, G., Papenfus, M., Toft, J., Marsik, M., Bernhardt, J., Griffin, R., 2013. InVEST 2.5.3 User's Guide. The Natural Capital Project, Stanford, CA.

TEEB, 2010. The economics of ecosystems and biodiversity: mainstreaming the economics of nature: a synthesis of the approach, conclusions and recommendations of TEEB. In: The TEEB Synthesis Report. Nagoya, Japan.

Tilman, D., 1999. Global environmental impacts of agricultural expansion: the need for sustainable and efficient practices. Proc. Natl. Acad. Sci. U.S.A. 96, 5995–6000.

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671–677.

Todaro, M.P., Smith, S.C., 2012. Economic Development.

Tongway, D.J., Freudenberger, D.O., Noble, J.C., Hodgkinson, K.C., Ludwig, J.A., Griffin, G.F., MacLeod, N.D., Brown, J.R., 2003. Landscape Ecology. CSIRO Sustainable Ecosystems, Canberra.

Troy, A., Wilson, M.A., 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecol. Econ. 60, 435–449.

Trucost, 2013. Natural Capital at Risk: The Top 100 Externalities of Business. TEEB for Business Coalition.

Undp, 1990. Human Development Report 1990. UNDP, New York, NY.

Unep, U.-I.a., 2012. Inclusive Wealth Report 2012. Measuring Progress Toward Sustainability. Cambridge University Press, Cambridge.

Van den Belt, M., 2004. Mediated Modeling: A System Dynamics Approach to Environmental Consensus Building. Island press, Washington, DC.

Van Der Ploeg, J.D., Verschuren, P., Verhoeven, F., Pepels, J., 2006. Dealing with novelties: a grassland experiment reconsidered. J. Environ. Policy Plann. 8, 199–218.

Vanloqueren, G., Baret, P.V., 2009. How agricultural research systems shape a technological regime that develops genetic engineering but locks out agroecological innovations. Res. Policy 38, 971–983.

Vemuri, A.W., Costanza, R., 2006. The role of human, social, built, and natural capital in explaining life satisfaction at the country level: toward a national well-being index (NWI). Ecol. Econ. 58, 119–133.
Verburg, P.H., Soepboer, W., Veldkamp, A., Limpiada, R., Espaldon, V., Mastura, S.S., 2002. Modeling the spatial dynamics of regional land use: the CLUE-S model. Environ. Manage. 30, 391–405.
Villa, F., Bagstad, K., Johnson, G., Voigt, B., 2011. Scientific instruments for climate change adaptation: estimating and optimizing the efficiency of ecosystem services provision. Economia Agraria y Recursos Naturales 11 (1), 54–71.
Von Braun, J., Gerber, N., Mirzabaev, A., Nkonya, E., 2013. The Economics of Land Degradation ZEF Working Paper Series. In: Working Paper 109.
Voinov, A., Bousquet, F., 2010. Modelling with stakeholders. Environ. Model. Softw. 25, 1268–1281.
Waage, S., Armstrong, K., Hwang, L., http://www.bsr.org/reports/BSR_ESTM_WG_Comp_ES_Tools_Synthesis3.pdf (accessed 04.08.15) 2011. New Business Decision-Making Aids in an Era of Complexity, Scrutiny, and Uncertainty: Tools for Identifying, Assessing, and Valuing Ecosystem Services, BSR, San Francisco, California.
Walker, B., Carpenter, S., Anderies, J., Abel, N., Cumming, G., Janssen, M., Lebel, L., Norberg, J., Peterson, G.D., Pritchard, R., 2002. Resilience management in social–ecological systems: a working hypothesis for a participatory approach. Conserv. Ecol. 6, 14.
Wenger, S.J., Olden, J.D., 2012. Assessing transferability of ecological models: an underappreciated aspect of statistical validation. Methods Ecol. Evol. 3, 260–267.
Willemen, L., Veldkamp, A., Verburg, P.H., Hein, L., Leemans, R., 2012. A multi-scale modelling approach for analysing landscape service dynamics. J. Environ. Manage. 100, 86–95.
Wilson, M.A., Carpenter, S.R., 1999. Economic valuation of freshwater ecosystem services in the United States: 1971–1997. Ecol. Appl. 9, 772–783.
World Resources Institute (WRI), 2012. The Corporate Ecosystem Services Review: Guidelines for Identifying Business Risks and Opportunities Arising from Ecosystem Change, Version 2.0. World Resources Institute, Washington, DC.
Wossink, A., Swinton, S.M., 2007. Jointness in production and farmers' willingness to supply non-marketed ecosystem services. Ecol. Econ. 64, 297–304.