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Urban and agricultural soils: conflicts and trade-offs in the optimization of ecosystem services

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Abstract On-going human population growth and changing patterns of resource consumption are increasing global demand for ecosystem services, many of which are provided by soils. Some of these ecosystem services are linearly related to the surface area of pervious soil, whereas others show non-linear relationships, making ecosystem service optimization a complex task. As limited land availability creates conflicting demands among various types of land use, a central challenge is how to weigh these conflicting interests and how to achieve the best solutions possible from a perspective of sustainable societal development. These conflicting interests become most apparent in soils that are the most heavily used by

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humans for specific purposes: urban soils used for green spaces, housing, and other infrastructure and agricultural soils for producing food, fibres and biofuels. We argue that, despite their seemingly divergent uses of land, agricultural and urban soils share common features with regards to interactions between ecosystem services, and that the trade-offs associated with decision-making, while scale- and context-dependent, can be surprisingly similar between the two systems. We propose that the trade-offs within land use types and their soil-related ecosystems services are often disproportional, and quantifying these will enable ecologists and soil scientists to help policy makers optimizing management decisions when confronted with demands for multiple services under limited land availability.

Keywords Agriculture · Ecosystem services · Land use · Management optimization · Soil · Urban · Trade-off

Introduction

Landscapes around the world have experienced substantial changes over the last 80 years (Vitousek et al. 1997). In particular, intensification of agriculture and increased urbanization to support a growing human population have caused profound changes in the structure and functioning of ecosystems (Ellis and Ramankutty 2008). Various traditional land use systems have been lost or diminished, as land uses have polarized either towards extensification or intensification (Plieninger et al. 2006). For example, in the European Union during the decade 1990–2000 the area converted to housing, industrial, transportation, and commercial uses was nearly 1,000,000 ha. In the U.S., urban, suburban and exurban areas now cover an estimated 148 million ha and during the period 1950–2000, creation of these urbanized environments caused an 11 % decline in cropland area (Brown et al. 2005). Projections of future landscape change in the U.S. suggest that an additional 24 to 70 million ha might be urbanized by 2020 to 2025 (Theobald 2005).

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Thus, while urban land conversions are made to satisfy the demands for living space of an increasing human population, the amount of soil sealing (i.e. covering soil with impervious surfaces) has greatly increased thus reducing the ability of soil to provide ecosystem services other than ones related to sealed soils (Frazer 2005). As land is a finite resource, there is an urgent need then to optimize the potential of multiple services from within a given area of land so as to not degrade their long-term sustainability. The question to be addressed in this paper is how the various, dwindling soil-related ecosystem services associated with agricultural and urbanized landscapes can be optimized given trade-offs that occur within and between them. We argue that (i) the maximization of one ecosystem service may be at the expense of other services, which often occurs as a disproportionate trade-off of those services, and (ii) because the increasing demands for land surface and soil-related ecosystem services occurring at local, regional, and global scales creates conflicts, the trade-offs of ecosystem services that occur between and within agricultural and urbanized landscapes need to be analysed more closely to better enable landscape design and management that leads to their optimization. We present a conceptual model in this paper to explore these two issues.

Even the most highly altered urbanized and agricultural ecosystems can deliver important ecosystem services other than the primary functions for which they are created and managed. For instance, urban green spaces and their permeable soils can support regulation of the hydrological cycle, sequester carbon, and protect human health by retaining pollutants when managed appropriately (Pouyat et al. 2010; Pataki et al. 2011). Intensive crop production involves the maximization of a provisioning ecosystem service (MEA 2005) at the expense of other services revealing the trade-offs that exist among many ecosystem services (Foley et al. 2005). Nonetheless, farmed landscapes contain a substantial part of biodiversity and they influence ecosystem services such as the provisioning of drinking water, carbon sequestration, and they may influence emission of greenhouse gases. Moreover, both agricultural and urban uses offer cultural and aesthetic values (Turbé et al. 2010; Mitchell and Popham 2007). Likewise, ecosystem services provided by trees in urban landscapes often have trade-offs such as those related to water use efficiency (Pataki et al. 2011).

Because of conflicting land use policies that promote conflicting goals, and because ecosystem services can covary or be antagonistic (Daily 1997; Bennett et al. 2009), a major challenge is how to manage, evaluate and optimize trade-offs among multiple ecosystem services (Carpenter et al. 2009; Raudsepp-Hearne et al. 2010; Pataki et al. 2011). This will be especially effective and efficient when a slight reduction of one service will disproportionally enhance other services (Bennett et al. 2009). For example, field margins may enhance biological control of pest insects in crops and thus reduce the need for spraying herbicides and fertilizers close to ditches and canals (Tscharntke et al. 2007). Another example is that of reducing flows and purifying urban runoff water, which can also be increased disproportionally by decreasing the area of sealed soil or impervious surfaces (McPherson et al. 1999). The opposite effect can also be found: if urban or agricultural soils are poorly managed, they may generate “disservices” to adjacent or downstream ecosystems.

When considering agricultural and urban ecosystems from a soil perspective, disproportionate trade-offs (see “Trade-offs among ecosystem services along land use intensity gradients in urban and agricultural systems”) among ecosystem services are conceptually the same among these land use types. In both cases a slightly changed level of one service (e.g., agricultural production maintaining a field margin) may disproportionally affect another service (e.g., field margins as a place for survival of natural enemies that control crop pests). Therefore, we propose that management decisions for both agricultural and urban soil systems have many similarities but at the same time, decision-making in each can impact the other system in unexpected, indirect and complex ways (Fig. 1; see Seto et al. 2012 for further discussion). Thus, when considering environmental and economic trade-offs and dependencies within and between both systems, land use planners and managers may benefit by comparing these systems simultaneously and holistically using a broader landscape context instead of viewing them as separate systems that can be managed effectively in isolation. This is particularly true for soil-related ecosystem services which can have strong, but not readily apparent, relationships across larger spatiotemporal scales (e.g., Lewis et al. 2006). To develop a holistic cross-system view of land- and soil-use trade-offs, we take an ecological approach, as this field of research has a long tradition of quantifying conflicts and trade-offs, to determine both antagonistic and synergistic relationships among ecosystem processes that are vital for maintaining soil-related ecosystem services in urban and agricultural landscapes.

Our aim is to consider how ecosystem services in two seemingly different systems can be conceptually integrated to optimize their benefit to society as demands for land surface and soil-related ecosystem services increase with the exponential increase in human population. We propose that conflicts in land use between urban systems and agriculture and the inherent trade-offs among soil-related ecosystem services need to be jointly addressed in decision-making. We will first investigate the crucial role of soils in providing an array of life-supporting ecosystem services. Then, we explore the trade-offs in the provisioning of ecosystem services within and between agricultural and urban systems to shed light on the

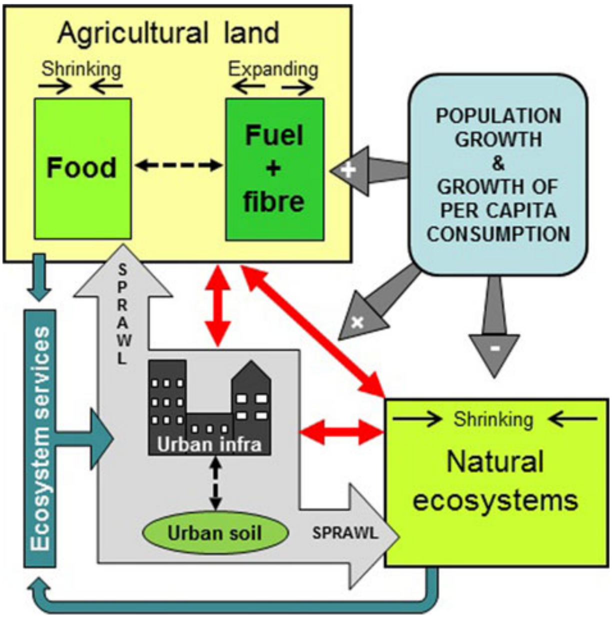

Fig. 1 The growth of the human population and an increase of per capita consumption of resources caused by changing lifestyles increases the demand for urban (*grey shaded area*) and agricultural land (*yellow*) and brings about conflicts in land use between (*solid, red arrows*) and within (*hatched, black arrows*) urban, agricultural and natural ecosystems. The conflicts in land-use often give rise to various trade-offs within a given system–a well-known example being competition for urban land between housing, roads and urban parks. Solving the "local" conflicts often results in transferring the problems to other ecosystems, where the conflicts may become even worse. For example, urban sprawl is known to take place at the expense of food production area. This, in turn, will evidently impair the various ecosystem services upon which urban dwellers depend

various conflicting land use interests in land use systems under strong human control. Next we provide an economic valuation and optimization example of these trade-offs to facilitate the decision-making process. Finally, we address the influence of spatial scale on the optimization of trade-offs among ecosystem services and its effect on policy decision making, as well as the use of incentives to motivate stakeholders to preserve common resources while pursuing their own economic interests.

We acknowledge that our approach is not the first to address the modelling of multiple ecosystem services and their trade-offs. However, besides the study by Eigenbrod et al. (2011) in which changes (densification vs. sprawl) in urban land cover was modelled, to our knowledge there do not appear to be any studies of trade-off comparisons of ecosystem services between and within urban and agricultural landscapes and soils. As a matter of fact, empirical knowledge on trade-offs between any type of ecosystems services is sparse (Cardinale et al. 2012)

Soil ecosystem services

The increasing demand for land for housing, infrastructure and crop and forest production implicitly leads to conflicting uses of soil systems, and land use change driven by this demand often leads to a loss of soil-derived ecosystem services (Vauramo and Setälä 2010). In the Millennium Ecosystem Assessment (MEA 2005) soils were regarded as essential for supporting services (soil formation) that are a prerequisite for provisioning, regulating and cultural services. The MEA, as well as the United Nation's environmental conventions (UNCCD 1994), considered soils as important, but without providing a detailed explanation for how they support human well-being or how their functions can be maintained or even enhanced. Nonetheless, it is well established that the activities of soil organisms contribute to provisioning services such as food and timber production, and regulating services such as carbon sequestration, and pest and disease prevention (Van der Putten et al. 2004).

Both urban and agricultural uses of soils result in drastic changes to their chemical, physical, and biological characteristics, and thus the ecosystem services provided by these soils. Although urban soils are profoundly altered by human activities, they still–when not paved or "sealed"–provide many of the same ecosystem services as do undisturbed and agricultural soils (Effland and Pouyat 1997; Pouyat et al. 2010). For instance, unsealed (pervious) urban soils serve as habitat for soil organisms and plants, and provide key functions such as the degradation of pollutants, storage of carbon and mineral nutrients, and moderation of the hydrologic cycle through absorption, storage, and supply of water (Lehmann and Stahr 2007; Pouyat et al. 2010). While these services are similar to many of those provided by agricultural soils, specific, local land uses that vary in management activities may impact these ecosystem services differently leading to fine-scale heterogeneity of urban soils and associated ecosystem services (Fig. 2). For example, the proportion of biologically active soils within the total land area often differs between the two land use systems. At landscape scales, the majority of urban soils appear impervious with little biological activity, thereby having a restricted ability to provide ecosystems services for urban inhabitants. However, at local scales pervious urban soils often form a biologically active, highly heterogeneous patch structure as a result of a diversity of environmental changes associated with urbanization (Pouyat et al. 2007). Such a mosaic of soils is likely to provide an array of ecosystem services that manifest themselves only when observed at a fine spatial scale. For example, at the scale of individual parcels pervious urban soils (1 m depth) have been found to store up to 14 kg C m^{-2}, while the respective amount at the scale

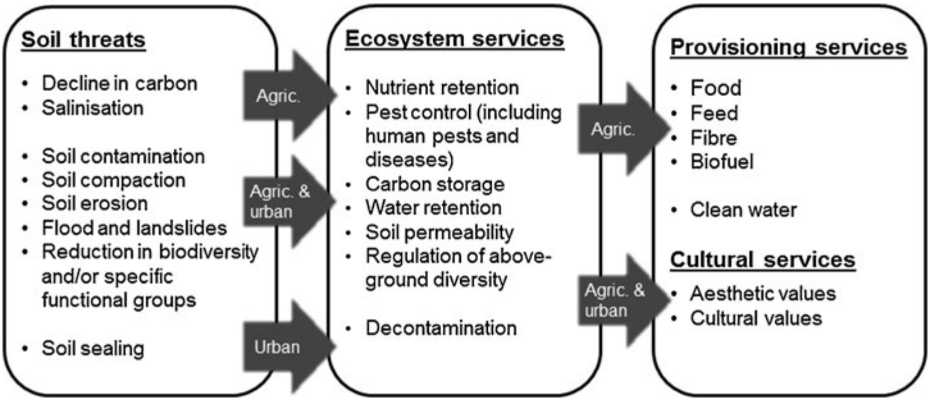

Fig. 2 A scheme presenting potential threats to soil biodiversity that decrease ecosystem services, either specific to agricultural or urban areas or shared by both. These ecosystem services will act together providing provisioning services that can be used for human well fare

of a city (impervious and pervious soils combined) the stored amount is only 6 kg m^{-2} (Pouyat et al. 2006). In agricultural landscapes such scale-dependent variations in ecosystem services are less likely to occur due to the homogeneous management of agricultural soils at larger spatial extents (i.e., field versus urban parcel). However, despite the obvious differences as to how soil-derived ecosystem services are managed and threatened by urbanisation and agriculture, the loss of biologically active soil, its biodiversity and the ability of soils to provide multiple ecosystem services are common to both land use types (Pavao-Zuckerman and Coleman 2007; Birkhofer et al. 2010) (Fig. 2). Thus, examination of how to best conserve and optimize ecosystem services may also be similar between them and benefit from a unifying model.

Trade-offs among ecosystem services along land use intensity gradients in urban and agricultural systems

To address the trade-offs that typically occur with ecosystem services at local or regional scales we present a simple conceptual model. Management to enhance one ecosystem service usually affects the level at which other ecosystem services can be provided (Foley et al. 2005). However, a variety of trade-off relationships are possible (Fig. 3). For the simplest case, we conceptualize two representative ecosystem services: ES1 is assumed to be a provisioning service, the benefits of which (shown on the y-axis; Fig. 3) can be augmented by increasing the intensity of land use/management from 0 to some maximum intensity I_{max} (as seen on the x-axis) and ES2 is an ecosystem service that is also affected by intensification. First, there may be a positive relationship between the two ecosystem services (Fig. 3a). In this case, intensification (including land use intensity and/or management activity) generates higher levels of benefits associated with both services. For example, increasing plant biomass (amount of green space in urbanized areas and crop production in arable systems) can result in enhanced C sequestration in soils (Fornara et al. 2009). Within ecosystems, however, it is more likely that management to promote a provisioning service—represented by ES1—reduces the production of another service—represented by ES2—under intensive land use (Kareiva et al. 2007; Carpenter et al. 2009; Raudsepp-Hearne et al. 2010), which is illustrated in Fig. 3b for the case of a "one-to-one trade-off (i.e. conflicting ES1 and ES2). This is because human demands for

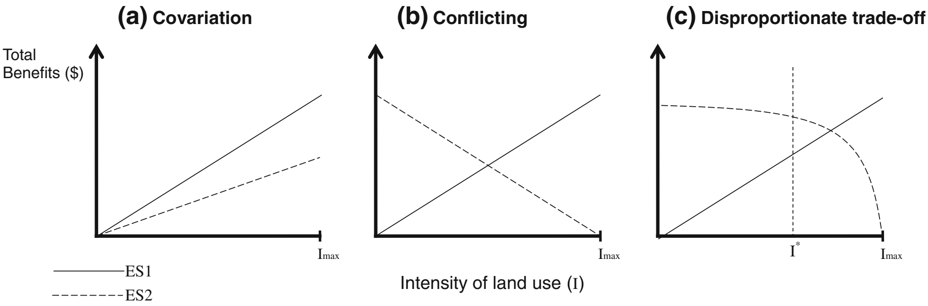

Fig. 3 Conceptual description of different trade-off situations between two ecosystem services (denoted ES1 and ES2). ES1 is, for example, representing a provisioning service that increases with the intensity of land use and ES2 is a supporting service. The x-axis represents the intensity of land-use and/or management activity that can be increased to a maximum I_{max} (intensity might for instance be thought of as the proportion of total urban area allocated to a particular land use). The y-axis measures the benefits generated by ecosystem services in monetary terms. The ES shown in panel **a** covary, **b** are conflicting, and **c** exhibit a disproportionate trade-off

ecosystem services compete for the same space, resources, or because they exclude each other for other reasons. For example, in arable production ecosystems there can be a trade-off between agricultural provisioning services (i.e., yield) and soil ecosystem services such as the conservation of nutrients in the soil (Steinmann and Gerowitt 2008). Long-term organic farming promotes internal nutrient cycling and pest control, but yields can be lower than in farming systems with mineral fertilizer inputs (Birkhofer et al. 2008).

These trade-offs are a complicating factor for decision makers, as the provisioning of one service may exclude other. However, it is worth noting that some apparently incongruous ecosystem services are not necessary conflicting as they can be optimized by appropriate planning and management, such as use of “engineered soils” in green infrastructure (e.g. Grabosky et al. 1996). Furthermore, management intensity can both enhance an ecosystem service (e.g. yield or crop provisioning service), or deplete a service (increase use of fertilizer, soil erosion, etc.). The challenge is how to identify cases where and how multiple ecosystem services may be optimized.

In Fig. 3c we illustrate the concept of the disproportionate trade-off where changing the provisioning service has a disproportionate effect on the supporting service. An example of such a *disproportionate trade-off* is the biological control of above ground insect pests in arable field; removing a small fraction of the land surface along field margins from production (ES1) strongly enhances the provisioning of biological control agents (ES2) (Ostman et al. 2001) and can help prevent surface water contamination, thus reducing economic and environmental costs of pesticide application (Novotny 2003). In both cases decreasing production beyond a certain point (e.g., I*) will cause a more significant increase in ES2 and hence environmental quality for relatively little decrease in the provisioning service, ES1. This can be seen in contrast to initial levels of ES1 that come at relatively small cost in terms of reductions in ES2. Thus the disproportionate trade-off implies a (strong) asymmetric relationship between services, and hence the right hand side in Fig. 3c is far more sensitive from the perspective of ecosystem functioning than the other way around. Therefore, because of the often asymmetrical relationship among ecosystem services it is better to optimize across all services than to attempt to maximize a single service.

Disproportionate trade-offs between ecosystem services also occur in urban ecosystems. For example, leaving ca. 10 % of the soil surface unsealed in urbanized settings has a disproportionately positive effect on prevention of city flooding after rainstorm events, if the runoff is directed to a pervious soil surface that has the capacity to infiltrate added flows (Schueler and Holland 2000). Therefore, we assume that disproportionate trade-offs can have a major impact on soil-related ecosystem services in intensively used environments. Such disproportionate trade-offs will not apply to all soil-borne ecosystem services in urban areas. For example, the ability of soils to store carbon is linearly related to land surface so that this ecosystem service cannot be disproportionally enhanced by leaving a relatively small fraction of the urban soil surface unsealed unless management practices can be developed that enhance carbon storage in the soil (e.g., Prescott 2010). Consideration of urban ecosystem services must be conducted with the caveat that cities by definition are not "production ecosystems" in the same sense as agricultural and forest ecosystems (Folke et al. 2002). However, the provisioning of ecosystem services in cities can nonetheless be significant. Besides water infiltration, microclimate regulation by urban forests (that are intimately dependent on permeable soil) is an ecosystem service that can mitigate urban heat island effects and reduce energy consumption in households. Model estimates for the city of Chicago, IL suggest that even limited amounts of urban green can provide cooling of urban climate. Nevertheless in this case the effects on urban cooling increase linearly with the amount of green area (McPherson et al. 1997).

There may be severe consequences from the competition for space by different land use types (Raudsepp-Hearne et al. 2010). This is well exemplified by urban sprawl–the unplanned urban development with rapid low-density outward expansion–which has greatly impacted ecosystem services provided by agricultural and forested lands and at the same time potentially increasing a disservice such as greenhouse gas emissions. Indeed, as urban areas expand, pressure on arable land increases as cities often transform the most productive soils (Foley et al. 2005), because many cities were established near productive land-water interfaces. If in the future more and more land is transformed by urbanization, the world population will need to be increasingly fed by a progressively shrinking area available for agriculture, unless agriculture expands at the expense of natural areas (e.g., deforestation in Amazonia).

Economic valuation and optimization of multiple ecosystem services

As argued above, asymmetric relationships can lead to overall loss of services if one is maximized at the expense of others. In order to avoid such undesirable developments, the valuation of ecosystem services can be used in directing policies towards conserving soils' biodiversity and ecosystem services in an optimization framework. Before developing an optimization model, the economic value of a particular soil-related ecosystem service needs to be determined. Economic valuation depends on two factors: first there must be some service-related effect of changes in the soil—biological, chemical, aesthetic or otherwise—and secondly there must be a human reaction to that effect. Economists summarize the human reaction to a reduction in an ecosystem service as a loss in welfare (i.e. benefits), and such losses can, theoretically, be expressed in monetary terms. As there are no markets and hence no readily observable prices for most ecosystem services (other than provisioning services), valuation and hence optimization of multiple ecosystem services is challenging (Winkler 2006). However, over recent decades ecologists and economists have made progress in not only defining but also measuring the monetary value of ecosystem services (Heal 2000).

To maximize individual services (or optimize several services) it is necessary to know the approximate marginal values of the different ESs affected by a particular land use. Marginal benefit or value is the change in total benefits generated by a particular service given a small change in land use intensity. Figure 4 shows the marginal benefit curves, MES1 and MES2 respectively, associated with the corresponding panels in Fig. 3 (a horizontal MES line indicates a constant marginal benefit whereas the curve for MES2 in Fig. 4c indicates exponential change in marginal benefits). It is to be noted that this is an example with two factors only; once three or more factors are dealt with, optimization models are needed. For example, in the case of drinking water supplies, managers have many different options to secure water for drinking and therefore must optimize among them (Schlüter et al. 2005).

In Fig. 4a, an increase in intensity (a measure of the extent to which a land parcel, whether urban, agricultural or other type of land, is developed) results in a constant increase in the total benefits generated by both ES (i.e. marginal benefits are constant), which is somewhat higher for ES1 than for ES2. This covarying relationship implies obviously that total benefits are maximized at $I = I_{\max}$ In Fig. 4b, the ESs are conflicting and the optimal trade-off is determined by the difference in absolute values d=|MES1|-|MES2|. If d=0, then no unique solution exists: any increment in ES1 is balanced by an equal reduction in ES2, hence it doesn't matter what intensity is chosen. If d>0 then a marginal increase in benefits generated by ES1 is always greater than the concomitant reduction in benefits generated by ES1, and hence at maximum intensity total benefits are maximized. Alternatively, if d<0 then the loss in benefits generated by ES2 is always greater than the benefit generated by augmenting ES1, hence intensity should be minimized to obtain maximum total benefits, i.e. I=0. In case of the disproportionate trade-off between ESs (Fig. 3c), the marginal benefits generated by ES2 are a nonlinear decreasing function of intensity. Initially, the marginal benefit generated by ES1 through increasing intensity, is significantly larger than the concomitant loss in benefits from ES2. Hence the intensity should be increased so long as d>0 and the optimal trade-off occurs where d=0 at $I=I^*$. Beyond this intensity d<0 and total benefits decline if intensity is increased further.

In many real world situations we can imagine that intensity is greater than the optimum $I = I^*$ required to maximize total benefits (i.e. ES1 + ES2 in Fig. 3). More probable is that intensity is close to $I_{\max}$ and hence a marginal reduction in intensity would result in a disproportionally large increase in total benefits (the resulting reduction in benefits generated by ES1, i.e. *MES*1, is more than compensated by the increase in benefits generated by ES2, i.e. MES2). This is well illustrated in multiple factor optimization models that are used in

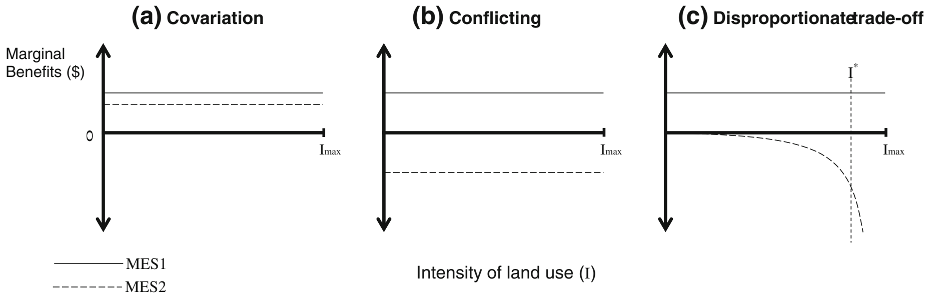

Fig. 4 The optimal trade-off between two ecosystem services (ES1 and ES2) with the corresponding marginal benefit curves associated with the diagram panels in Fig. 3. The marginal benefit curves MES1 and MES2 show the change in total benefit (y-axis) generated by each ES resulting from an incremental change in the intensity of land use/management activity (I). The x-axis represents the intensity of land use that can be increased to a maximum $I_{\max.}$ The ES shown in panel **a** covary, **b** are conflicting, and **c** exhibit a disproportionate trade-off because MES2 is nonlinear

e.g. water usage decision making (Schlüter et al. 2005). Apart from the spatial scale, time can also have important consequences in affecting the shape of the land use intensity–ecosystem service benefit curve. For example, plotting ecosystem services (such as yield) may result in a different curve in the short run compared to the long run.

Having identified the approach for determining the optimal trade-off between ES in different situations, in the next section we discuss various policy options for promoting this solution in the real world. The basic principle that needs to be followed is that the marginal value of all relevant ES should be reflected in the land managers decision processes. We focus on situations characterized by a disproportionate trade-off, since the potential net benefits to society of relatively small changes in land use could be very large.

Policy and planning implications

As previously discussed, to develop policies and economic incentives aimed at optimizing soil-derived ecosystem services across various spatio-temporal scales, scale-dependent trade-offs present a challenge due to potential conflicts between local and regional management goals (Figs. 1 and 3b). For example, fertilizer applications to a lawn can maximize the aesthetic ecosystem services of the parcel or land use unit; however, this management practice may increase disservices if soils in that parcel allow significant losses of N or P to the surrounding environment resulting in the degradation of regional ecosystem services (e.g., Baker et al. 2008). Therefore, appropriate spatial and temporal scales must be clearly identified for any analysis of ecosystem services trade-offs, resulting in the accurate analysis of policies and incentives (Bennett et al. 2009; Raudsepp-Hearne et al. 2010). Nonetheless, different stakeholders will emphasize different spatiotemporal scales as they interpret and value ecosystem services within and among parcels (see Fig. 1; Byrne and Grewal 2008). Inevitably, this will lead to a divergence of opinion by landscape managers about how to best handle trade-offs making it difficult to determine the appropriate trade-offs and ideal optimization scenarios at both local and regional scales (Tallis et al. 2008). For instance, within a land parcel of a certain size (e.g., the yard around a single-family home), an ecosystem process may be perceived to have little or no value as a service because the minimum desired level of service is not provided under optimal conditions (e.g., water infiltration), or it is degraded due to preferential management for other services (e.g., loss of favorable nutrient cycling due to lawn pesticide applications; Fig. 3b). However, the ecosystem process may have significant regional value if it enhances a service that arises from the aggregation of processes across space and/or time (e.g., high levels of regional soil carbon sequestration across many parcels, Pouyat et al. 2006). Similarly, as illustrated in Fig. 1 potential in land use choices at the parcel level can create substantial trade-off situations among parcels at larger spatial scales, perhaps leading to indirect and unpredictable offsite impacts of local-scale management across space (Foley et al. 2005; Seto et al. 2012; Fig. 1). In this regard, the analysis of trade-offs across landscapes composed of parcels managed by many stakeholders requires a more holistic, multi-scale and multi-stakeholder approach than is usually used when individual decisions are made about how to best manage one landscape parcel (Byrne and Grewal 2008).

Economic incentives that encourage land managers to recognize and respond to such cross-scale connections can be applied to balance the demands for different types of ecosystem services (Ostrom 2009). In order to be effective, incentives should take several aspects into account. First, the shape of the relationship between different biophysical as well as economic processes should be reflected adequately, which implies that the curves in

Figs. 3 and 4 should be known at least approximately for a given area. Second, feedback processes between the activities induced by incentive schemes and the costs and benefits for the ecosystem service provider receiving payments from such schemes should be incorporated. For example, the costs incurred by creating a semi-natural habitat in an agricultural area in terms of loss of production could partly be balanced by improved biological pest control, reducing the amount of the incentive necessary to motivate a farmer to establish such habitat. Third, the costs and benefits for different stakeholder groups are likely to vary through space and time (Ineson et al. 2004), warranting a multi-scale perspective when designing incentive schemes. However, existing incentive schemes to promote the provision of certain ecosystem services considered to be suboptimal generally do not take the aspects described above into account (Kleijn et al. 2001; Nelson et al. 2008).

The sociocultural context around a parcel may dictate not only the management of ecological conditions but also affect the valuation of services within a parcel (e.g., community desires for water infiltration lead to the preservation of green space) (see Kaye et al. 2004; Baker et al. 2008). Consequently, the state and valuation of ecosystem services within a parcel and their spatiotemporal variability can be, in part, independent of the characteristics within the parcel. Further, land managers are likely to focus on an analytical scale most relevant to their immediate interests and thus may fail to recognize or understand concerns arising from other scales. In those cases, clear communication about scaling issues to all stakeholders will be essential to enhancing success. This suggests that scenario building and community-wide discussions are needed as components of the ecosystem service optimization toolbox (Zurek and Henrichs 2007). For both urban and agricultural parcel managers, making decisions about how to optimize trade-offs among ecosystem services within a parcel may be challenged by the changing socio-cultural and environmental context around the parcel, much of which are out of that parcel manager's control. From the planner's perspective, it is often larger scale implications of decisions made at the parcel level that is of concern, which makes policy makers and planners seek policies that mandate restrictions at the parcel level.

Further, landscape design and planning tools will also be needed to help create agricultural and urban landscapes in which ecosystem services and their trade-offs are optimized (Lovell and Johnston 2009). For example, if the spatial arrangement of unsealed soil surfaces in a city is designed in relation to the runoff of water instead of being randomly dispersed or arising from unplanned sprawl development, the infiltration of water into soils during storm events can be maximized (Hatt et al. 2004). In an agricultural landscape, planning for a beneficial pattern of hedgerows and organic fields across a larger area could form a network of mini-soil and biodiversity conservation areas and promote the movement of various species into adjacent parcels. The process of implementing successful landscape designs that intentionally consider multi-scale issues related to optimizing ecosystem service trade-offs will require collaboration among diverse parties and compromises among various stakeholders, especially parcel-owners. Ecological-economic approaches on disproportionate trade-offs can play a critical role in the process by helping stakeholders to step outside their spatial mindsets by engaging them in thinking about the scale- and context-dependency of ecological variables and associated ecosystem services (Byrne and Grewal 2008; Tallis et al. 2008).

Conclusions and perspectives

Changes in land use type and land use intensity influence the provision of soil-related ecosystem services. Large-scale conversion of agricultural land to urban land use types, or potentially the other way round, is one important aspect, but the spatial distribution of specific management regimes within each land use type (in the present case urban or agricultural land use) also plays a major role for ecosystem service delivery. In order to optimize between and within trade-offs among ecosystem services, we propose a conceptual model that addresses the non-linearity in relationships of ecosystem services between and within in agricultural and urban environments. This awareness may be useful both in theoretical analyses of social-ecological systems and in the practical implementation of policies to enhance sustainable land use by optimizing ecosystem services.

In our analysis, we have provided examples of so-called disproportionate trade-offs between ecosystem services that exist in contrasting land use types, such as urban and agricultural use. Both of these usage types are quite extreme compared to low-intensity managed semi-natural ecosystems, but also in those systems we expect disproportionate trade-offs to occur. Therefore, we conclude that balancing different ecosystem services is not always a matter of 'either-or', but that it can also be matter of 'and-and' when patterns of optimization are considered and the loss of ecosystem services are minimized (see Swinton et al. 2007). An economic modelling framework can be used to assess the trade-offs between two soil ecosystem services, but in practice multiple ecosystem services will need to be compared and balanced. When the identification of disproportionate trade-offs is of concern, new tools will be needed for optimizing disproportionality in trade-offs among multiple services. This will require collaboration between economists, sociologists, land use planners and ecologists.

Trade-offs that function within one land use type can be influenced by, or influence other land use types; for example ecosystem services in urban areas may be optimized as urban areas spread, but only at the expense of the amount of land available for agriculture, forests or nature conservation. Therefore, spatio-temporal scales need to be explicitly considered in future studies in order to determine how trade-offs among ecosystem services in one system influence those in other systems, i.e., become a disservice. Our conceptual framework can also consider ecological footprints, or indirect effects, of activities in one part of the world for other parts. For example the policy incentives for percentage of biofuel in fossil fuel in industrialized countries will have a strong impact on land and soil use, as well as on the provisioning of food and other services in second and third countries. It is worth noting that, although not explicitly dealt with in our model, by increasing the intensity of management, higher provisional benefits may be realized, but also at the same time increasing ecosystem disservices (Foley et al. 2005; Cameron et al. 2012). This disparity suggests that the optimization between ecosystem services should, in some cases, also consider reducing disservices.

Finally, multi-service analyses of disproportionate trade-off balances need to be made accessible to policy makers and other stake holders and end users to enable them to analyse the consequences of policy options, which becomes even more challenging as the number of land parcels and their diversity in land use increases. There is a multitude of issues that are all connected with each other. For example, increasing urban green and the production of biofuel and food all place a demand on the already limited available amount of land (Smith et al. 2010). Applying disproportionate trade-off optimization analysis can help guide the sustainable management of ecosystem services and ease tensions and unexpected negative outcomes that arise when attempts are made to maximize single services.

We conclude that recent advances in our knowledge of soil processes in urban and agricultural environment provide a promising starting point to extend and apply the conceptual models of trade-offs between ecosystem services presented here in practise. This will facilitate more efficient collaboration between ecologists and socio-economists in the sustainable management of socio-ecological systems (Carpenter et al. 2009).

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