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Ecosystem service evaluation to support land-use policy

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ABSTRACT

Regular economic activity takes into account ecosystem goods and services that are exchanged for money in the market (e.g. food, fibre, water) but normally ignores more intangible ones left away from market transactions (e.g. soil protection, climate regulation, disturbance control, habitat provision), even in cases when they become irreversibly impaired. However, because of the increasing pressure brought by the public opinion, the attempts to assign an economic, yet volatile, valuation to ecosystems assets has multiplied in recent years, and policy communities are increasingly compelled to incorporate them into land use planning initiatives. Based on contributions to this special issue, we discuss how the perspective of ecosystem services can contribute to develop sound land-use policies and planning actions. Beyond valuation, several practical implications emerge from the contributions. A myriad of potential tradeoffs must be analyzed because since the provision of some services can be accompanied by the emergence of unexpected dis-services. For example, carbon accumulation based on increasing net primary production rates may simultaneously cut water yields and, hence, water provision. Various existing mechanisms ranging from state-controlled to market-controlled for rewarding the provision of ecosystem services are analyzed and discussed in terms of their capacity to connect nature to land-use planning.
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1. Introduction

Modern views on land-use policy aim at getting a long-lasting harmonization of economic, social and environmental interests in the society at the regional to local levels. In the practice, however, such attempt may be jeopardized by conflicting interests among sectors regarding land-use. Societies often obtain economic benefits at the expense of ecological functions and processes such energy flows, nutrient cycling and water processes (Ring, 2008) associated with essential ecosystem service (ES) provisioning (Haygarth and Ritz, 2009). Because concepts are used many times in this article, in line with MA (2005) a brief definition of ecosystem services, functions and processes is necessary: (i) ecosystem services are the benefits that humans get from nature (e.g. food, climate regulation), (ii) ecosystem functions (e.g., energy flow, nutrient cycling) involve essential processes that contribute to ecosystem service provision, and (iii) ecosystem processes comprise the transference of energy (e.g., chemical energy), material (e.g., nutrients, water) and information (e.g., genes, cultural information) among functional compartments. In this context, despite the economic growth and social development are normally considered a priority in most societies the increasing sensitivity of people regarding the provision of goods and services by nature poses new challenges in policy making (Eickhout et al., 2007). Since the mid-1980s, the greater public awareness about nature (Stoate et al., 2001; Robinson and Sutherland, 2002) has led to recognize the need of regulations, financial instruments and technologies to address the conservation of ecosystem services (Burgess and Morris, 2009).

The idea of ecosystem services (ES), generally defined as the benefit that people obtain from ecosystems (MA, 2005), has had an increasing presence in the discussion of environmental issues since its introduction thirty years ago (Mooney and Ehrlich, 1997). The relative simplicity of the concept, its relationship with natural capital valuation (Daily et al., 2009), its direct association with ecosystem functions (Costanza et al., 1997), the possibility of a better understanding with economists and social scientists, and the

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expectation to make it operative to solve environmental conflicts and to assess the multiple consequences of land-use changes, have likely contributed to spread the ES concept. The report of the Millennium Ecosystem Assessment (MA, 2005) clearly links the level of provision of ES to human welfare, stressing the anthropocentric nature of the ES idea (Goulden and Kennedy, 1997). An appealing characteristic of ES from a biophysical viewpoint is the direct link with structural and functional aspects of ecosystems (Costanza et al., 1997). Fisher et al. (2009) makes this point explicit by defining ES as the aspects of ecosystems utilized (actively or passively) to generate human well-being (see also Boyd and Banzhaf, 2007; Boyd et al., 2001).

A growing concern in land-use policy is how to equally incorporate economic, social indicators and ecosystem-service valuations in a well-balanced decision-making matrix (Kremen et al., 2005). Both economic and social indicators usually offer simple measurements that are understandable and manageable by the general public (Daily, 1997). In contrast, the valuation of ecosystem services still shows several restrictions in approach and methods that hinder adoption and explain the large disparity between initiatives to incorporate the value of ecosystem service into land-use policies (Wunder et al., 2008). Such differences may reflect, among other things, socio-economic and cultural gaps among communities, underestimation of nature as provider of ecosystem services, and disagreement about valuation methods (Kosoy et al., 2007; Engel et al., 2008).

This article aims at providing: (i) an overview about the conflict between economic and biophysical evaluation of ES, (ii) a background discussion on the evaluation of ES and the way it can underpin land-use policy and (iii) a brief discussion on how the results of the articles within this special issue contribute to assess and integrate ES into land use policy. Most of the articles of this special issue highlight the importance of the biophysical evaluation of ES provision and their response to land use changes. In the following sections we discussed the shortcomings of monetary approaches to value ES and the potentials of the biophysical evaluation to incorporate the concept into decision making and land use planning process.

2. Ecosystem service valuation: beyond economics

Humanity has a long history of managing provisioning services (food, fibre, energy, water, raw materials) as commodities that are traded in markets. So they are considered the strongest data-based indicators that can be integrated into the formal market economy and can be easily incorporated into the statistical accounts (Layke, 2009). Therefore, price became one of the most frequent indicators used in ES studies.

Regulating and cultural ES, on the other hand, are services which cannot be easily quantified as occurs with the provisioning ones (MA, 2005). Only few regulating services have been integrated into economic markets. In some cases, the cost of engineering projects for water-flow regulation, water cleaning and waste treatment have been used to indirectly estimate the market value of regulating ES such as those of floods regulation, water purification, water supply and waste elimination (EPA, 2000). Likewise, the economic cost around recreation and tourism (travel, accommodation, and other recreational goods and services) is used as a tool to estimate the economic value of cultural services that contribute to human well being (Carpenter et al., 2009). The article by Costanza et al. (1997) is probably the most comprehensive and well known analysis of this type that includes methods such as “willingness to pay”, “replacement value”, “avoided cost”, “travel costs “or” hedonic price”, have spread extensively. Given that all of them are based on the concept of utility to humans, increasing criticism arose among ecologists because the intrinsic “non-value” use of some natural assets, like species diversity, is ignored. For example, setting aside the intrinsic value of the existence of species or their difficult-to-estimate value, conservation of redundant species is hardly justified taking into account their current contribution to essential services that are of human interest. The economic valuation of the ES has been criticized as a mechanism for making decisions when information and data are the result of subjective estimations collected, for example, among different consumer categories (Carpenter et al., 2009). In spite of this criticism, economic valuation remains as one of the most studied aspects of ES and the lack of a proper monetary valuation is identified as an important driver of decreasing ES provision (MA, 2005).

Despite the effort applied to develop practical and effective methods of ES valuation for policy making, successful examples are relatively rare (Ruffo and Kareiva, 2009). Most of them looked at services associated with water provisioning, as it happens with the well-known case of Catskills Basin in the state of New York (Chichilnisky and Heal, 1998).

Besides the problem of pricing ES when no markets exist, a critical limitation is the difficulty to establish quantitatively to what extent changes in land use, and the associated shifts they impose on ecosystem functions, will modify the level of provision of a given ES (i.e., production functions of ES). Production functions have been characterized for goods and services with market value in cultivated and forest ecosystems (Daily et al., 2009), including the links between primary productivity and meat production or those that assess the level of soil protection and quality and forest yields. However, for most ES they are not clearly identified and quantified. Another problem associated with economic valuation of ES is that it assumes marginal changes (small changes in the amount of an ES affects its monetary value), which are difficult to define for most ecosystem processes (Heal, 2000). As Carreño et al. (2012) emphasized, the economic methods for pricing single services in monetary terms are markedly subjective and may say nothing if money cannot be related to ecosystem functions that are associated with service provision. They conclude, as some authors did before (Odum, 1988, 1996; Costanza et al., 1998; Odum and Odum, 2000), that the use of biophysical approaches and methods is necessary to get objective and comparable valuations.

An important issue of assigning monetary values to ecosystem services has to do with the ideological choice, not always explicitly assumed, that is made when the ES analysis is enrolled into a market rationale. The power asymmetries of the process of production, exchange and consumption of ecosystem services may contribute to reproducing rather than reduce inequalities in the access to environmental benefits (Liverman and Vilas, 2006; Kosoy and Corbera, 2010).

3. Biophysical evaluation of ecosystem services

Several articles in this special issue showed that stocks of biomass and water, and their associated fluxes in terrestrial ecosystems, can provide useful biophysical metrics to assess service provision. They focused assessments in different areas of Argentina, like the Southern Pampas (Laterra et al., 2012; Barral and Maceira, 2012), the highly productive Rolling Pampas (Caride et al., 2012; Gavier-Pizarro et al., 2012), the transitional region between the Western Pampas and the Espinal region (Nosetto et al., 2012) and the transition between the subtropical humid forests (Yungas) and the subtropical dry forests (Chaco) in NW Argentina (Volante et al., 2012). They assumed that some ecological functions of vegetation and water in the landscape can be associated with a number of aggregated services (e.g., carbon sequestration + soil protection + habitat and shelter) that can alternatively be expressed in absolute or relative terms. One article in this special issue (see Volante et al., 2012) provides convincing arguments to demonstrate changes in intermediate services (sensu Fisher et al., 2009) with land clearing in Chaco. Not only cleared land lost biomass C in these systems, but also non-cleared fragments displayed lower inter-annual variability than large forest areas in response to climate oscillation. This is a consistent proof of the capacity of natural vegetation to buffer external disturbances.

An attempt to support their biophysical estimations with global biodiversity data (MA, 2005) was done by Carreño et al. (2012), assuming that the number of higher families (reptiles, mammals, birds and amphibians) is comparatively larger in biomes that supply large amounts of services. Although the use of global data to validate local estimations is better than nothing, this operation is not fully consistent because local estimations need local data to yield a reliable validation. In such a way they provided (at different levels) preliminary production functions of final services from ecological processes (intermediate services) that can be measured in field.

What are the advantages and disadvantages that the economic and biophysical evaluation of ES offers to policy making? It is frequently argued that most ecosystem service valuations are neither comprehensive nor suitable to capture the whole complexity and benefits that ES provide (Layke, 2009). Certainly, approaches and methods to analyze and provide information on ES are relatively young and still evolving.

Beyond the facility to measure, one advantage of economic valuation of provisioning services through market prices is that they are considered the strongest data-based way to integrate the value of nature into the formal economy and the national accounts. However, this economic approach does not properly capture the intangible value of regulating and cultural ES that cannot be easily quantified in terms of price (Ludwig, 2000; Chee, 2004). Furthermore, the economic valuation considers only indicators of the flow of an ES (food, fibre, energy, raw material) rather than the stock of natural goods that measure the capacity of the ecosystem to deliver a service (Layke, 2009). Regarding this, the biophysical estimation of stocks of biomass and water in ecosystems (expressed as absolute or relative values) is an advantageous option to indirectly assess fluxes associated with regulating and cultural services delivered by biomass and water stocks. When determining a market price is not possible, the biophysical estimation of stocks conveys a range and quantity of benefits that people derive from ecosystems at multiple spatial and temporal scales (EPA, 2000). According to MA (2005), given the support of modern technology (satellite images, database management, GIS), the biophysical evaluations accomplish two effectiveness criteria suitable for policy making: (i) ability to convey and communicate information to non-technical policy makers, and (ii) data availability to monitor ecosystems at different spatial and temporal scales. Beyond these advantages, one possible disadvantage attributable to biophysical evaluations is that frequently they offer aggregate values of various ES, preventing a desirable discrimination among them.

Despite some advantages of ES evaluation through simple metrics based on local biomass and water bodies (e.g., wetlands), additional information is required for the assessment of functions that depend not only on local but also on broad-scale properties (King and Wainger, 1999). Aiming to map the capacity of a rural landscape to provide clean water, in this issue Laterra et al. illustrate how to integrate the spatial context for the evaluation of the water holding and water depuration capacity by wetlands, as well as the runoff filtration capacity by riparian vegetation. Therefore, landscape models of aggregated ES were greatly improved when not only the composition but also the spatial configuration of the landscape was taken into account.

4. Tradeoffs and synergism between ecosystem services

Depending on management practices, agriculture can also be the source of numerous disservices, including loss of wildlife habitat, nutrient runoff, sedimentation of waterways, greenhouse gas emissions, and pesticide poisoning of humans and non-target species. The tradeoffs that may occur between provisioning services and other ecosystem services and disservices should be evaluated in terms of spatial scale, temporal scale and reversibility (Power, 2010).

An environmental challenge for land-use policy is how to manage multiple ecosystem services. Enhancing the production of provisioning ecosystem services, such as food and timber, often leads to tradeoffs between regulating (nutrient cycling, soil protection, flood control, etc.) and cultural ES (aesthetics, recreation, eco-tourism, etc.). Therefore, the estimation of aggregated services may bring about another trouble: not all services are synergistic and many may actually display trade-offs. As shown before by other authors (Jackson et al., 2005; Nosetto et al., 2005; Jobbágy et al., 2008) Nosetto et al. (2012), show how water yields decline in afforested systems, highlighting the conflicting relations that can emerge between C sequestration, successfully achieved under tree plantations, and water provision, best supported by herbaceous covers. C gains or losses associated with changes in vegetation often modify evaporative water losses. Synthesizing more than 600 observations, Jackson et al. (2005) documented that C gain through tree plantations caused substantial losses in stream flows, and increased salinization and acidification. This means that some services like C sequestration may generate dis-services (Zhang et al., 2007) that affect the water economy and cause collateral negative effects. Another example of conflicting service relation in wetlands was reported by Verhoeven et al. (2006). Heavily fertilized agricultural fields and grasslands show substantial leaching of NO_3 to the groundwater, which is transported laterally to streams through subsurface runoff and deeper groundwater fluxes. Nitrates entering anaerobic soil zones are easily subject to denitrification, which removes nitrogen permanently from the through-flowing water. The same process that removes N excess and purifies water causes a dis-service by emitting N_2O (a powerful greenhouse gas) that contributes to global warming. So, the simple aggregation of various services does not necessarily imply that they are additive or keep a synergistic relation. On the contrary, trade-offs among ecological functions and services may be more common than expected in nature, and this may discourage “one size fits all” approaches seeking single biophysical indicators of aggregated services. Recently, Raudsepp-Hearme et al. (2010) developed a framework named “ecosystem service-bundle analysis” for analyzing the provision of multiple ecosystem services across landscapes. Studying at the landscape scale the spatial patterns of 12 ecosystem services in 137 municipalities in Quebec, Canada, they showed the occurrence of tradeoffs between provisioning and almost all regulating and cultural ecosystem services. They demonstrated as well that a greater diversity of ecosystem services was positively correlated with the provision of regulating services.

Based on estimations for Argentina, Carreño et al. provided in this issue a sole example of tradeoffs analysis between the estimated annual gross margin per hectare and the value of ecosystem services in three historical periods that differed largely in the area allocated to annual cultivation. They show that, in percentage, benefits from economic services increase exponentially as ecosystem services decline linearly in time and space. In terms of human perception, the provision of tangible economic benefits obtained through clearing and cultivation of land overwhelmed the loss of intangible ecological services. They note that this imbalanced response occur at the expense of essential ecosystem services that being irreversibly lost still remains invisible to the public eyes.

The analysis of tradeoffs among ES across scales is a challenging issue both for land-use policy and decision making. Evidence (Rodrĭguez et al., 2006; Mander et al., 2007) shows that tradeoffs among ES increase at small and decrease at broad scales. While the smaller scales are associated with a homogeneous landscape, larger scales are associated with a complex and heterogeneous one. Furthermore, the up-scaling analysis shows that ES tend to be complementary as the complexity of the landscape increases giving room to multi-functional views in land-use policy and management. In this issue, Laterra et al. analyzed tradeoffs cases across scales in one agricultural landscape of Argentina.

5. Ecosystem service evaluation, socio-economic issues and policy making

As a general criterion, in systems that are relatively intact, functioning well and resilient, policies should aim at maintaining a stable provision of ES; but in systems that have been altered and are less resilient, policies should aim to keep these systems away from their critical thresholds in terms of services that can be lost (Chee, 2004).

Roughly speaking, land-use policies can be classified as centralized (strong intervention of the state) or decentralized (incorporation of market mechanisms). It should be noted that the methodological shortcomings for ecosystem service valuation did not halt the incorporation of incentives for nature conservation in land-use policies. But methods improvement to give reliable values to ES will contribute to improve the scientific basis of land-use policies, independently of the fact that policies are centralized or decentralized.

There are several reported cases on centralized policies based on mechanisms of fiscal transfer to the local level. Since 1990s, for example, several states in Brazil have introduced the notion of fiscal transfer to explicitly compensate municipalities for the provision of ecosystem services that benefit jurisdictions outside the municipal boundaries (Bernardes, 1999; Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002). More recently, Joly et al. (2010) described an ongoing program in the state of São Paulo, Brazil, that involves investment in biodiversity research, personnel training, and policy-impact assessment. Scientists jointly to policy-makers set up an agro-ecological zoning ordinance that prohibits sugarcane expansion to areas where biodiversity conservation is prioritized. Policies try to link this initiative to demands from the international ethanol market, which requires compliance with environmentally sound production practices. In Portugal, policy makers implemented a fiscal transfer system to reward municipalities that, within their boundaries, have adhered to a preservation-site program named Natura 2000 (de Melo and Prates, 2007). In Switzerland, Köllner et al. (2002) reported another case in regions where biodiversity was integrated into an intergovernmental system of fiscal transfers. In Germany, policy attempts have been made for the integration of scientifically sound ecological indicators into inter-governmental fiscal transfers to the local level (Perner and Thöne, 2005).

In terms of decentralized policies, the payment for ecosystem services (ES) is an innovative approach, increasingly applied both in developed and developing countries, for conservation and management of critical resources. Frequently, the state may not be absent and may act as auditor of the process. ES payments are conceived as voluntary transactions where a well-defined ES or a particular land use that warrants its sustained provision is 'bought' by a service buyer from a service provider as long as provision is secured (Wunder et al., 2008). Often, funds go directly from users to providers, but government-financed programs in which the service buyers are a third party are common as well. The access to funding programs is fully voluntary for both ES providers and users, who can enter and exit contracts voluntarily. Many programs are hybrids that mix public and private funds. Costa Rica's payment program for ecosystem services (PSA), for example, is financed primarily from government funds, but also includes payments from service users, international agencies and NGOs (Pagiola, 2008). An example of private arrangements is that of the Vittel (Nestlé Waters) watershed protection program in France (Perrot-Maitre, 2006). Other reported cases with varying approaches take place in Bolivia (Asquith et al., 2008), Ecuador (Wunder and Albán, 2008), México (Muñoz-Piña et al., 2008), Zimbabwe (Frost and Bond, 2008), China (Bennet, 2008), USA (Claassen et al., 2008) and other Central American countries (Kosoy et al., 2007). There are large differences in prices paid for the same service. For example, for the service of watershed protection ranges between (values in US\$ ha^{-1} $year^{-1}$) 1.5 and 3.0 in Bolivia, 6.0 and 12.0 in Ecuador, 27.0 and 36.0 in Mexico, and 16.0 and 20.0 in UK.

In socio-economic terms, the same notion of service that is useful and meaningful to human sciences like sociology and economics may sound elusive to biology and ecology. Various social and economic indicators like employment, salary, price, income, tax, and so on can be rigorously measured and represented by figures that people capture and manage daily; so they became accessible and useful tools to be handled by policy makers. But, on the other hand, ecosystem services are neither familiar to common people nor easily assessable. This asymmetry probably explains the over-dimensioning of socio-economic in relation to the environmental ones in policy making.

A critical point is the introduction of the socio-economic dimension in the analysis. Scheffer et al. (2000) define the categories of "affectors" and "enjoyers" of ecosystem services. The affector is the stakeholder that, intentionally or not, adversely alters the level of provision of a particular ES, affecting "enjoyers" as members of society. Depending on the ES of concern, benefits will be perceived locally (in the case of water supply to a particular area), regionally (basin water regulation) or globally (regulation of atmospheric gases composition). For many services the same stakeholder can be affector and enjoyer. This is the case of a farmer who produces annual crops and honey and obtains drinking water from the farm. The application of insecticides by this farmer may impair pollination, honey yields and water quality. In a case like this there is an effective mechanism of feedback that may maximize the provision of both, goods with market value and ecosystem services. However, a delay in the perception of ES declines could bring the feedback too late (e.g. farmer perceives water pollution after a decade). A more problematic and common situations arises when affectors are not direct enjoyers of ES, something that can be even more problematic when they do not share the same territory. Redman (1999) identifies the physical disconnection between the decision-making processes and the territory as one of the most common causes of environmental problems throughout the history of mankind.

Economic valuation of the ES and decision making associated with cost-benefit logic assumes that affectors and enjoyers contribute equally to finding an optimal social welfare. This rarely occurs. Generally, affectors have more economic and political influence than enjoyers (Scheffer et al., 2000). The capacity of the industry or other affectors to define what costs and what benefits are considered is significantly higher than enjoyers, which commonly is an atomized society. The political component, as it reflects power relations, tends to further distort the prospect for the benefit of affectors. Thus, perception is a critical point to define the relative importance of different ES.

The relative success of local to national payment schemes for water services may not only stem from their relatively simple pricing based on avoiding cost approaches but also due to its identification of water as a vital resource for life and the perception that its provision may be highly impaired. Interestingly, the regulation of atmospheric carbon is the first intangible ecosystem service that is recognized by free formal global markets. In fact carbon regulation is the first service based on the issuance of bonds traded in the global free market that has expanded as a key tool to boost the Clean Development Mechanism (IPCC, 2007).

How to generate a viable mechanism that includes ES in land use policy and decision making without affecting essential socio-economic interests? Cowling et al. (2008) presents a general scheme where they identify three stages: assessment, planning and implementation of management. In this article in particular and in the special issue in general we focus on the assessment stage, emphasizing the: (1) identification of ecosystem processes and associated ES affected under different ongoing, expected or plausible ecosystem interventions, (2) quantification of ES provision change in response to human intervention, (3) aggregation of ES services and recognition of possible tradeoffs among them, and (4) quantification of benefits appropriation and losses by each stakeholder.

6. Is the ecosystem-service valuation useful and necessary to land-use policy?

Is the biophysical valuation of aggregated services, as most authors suggest in this special issue, useful to design suitable land-use policies? Even taking into account constraints mentioned in the above section, this provides a rough metrics to be compared with economic and social indicators. Tradeoffs analysis is the tool that can help assess the relative individual response of indicators per unit of land-use change. Both the economic valuation of provisioning services and the biophysical assessment of regulating and cultural services can provide useful input to tradeoffs analysis. In the meantime, approaches and procedures to evaluate ecosystem services can improve.

Case studies on land use and ES provision presented in this special issue cover a variety of land-use configurations, diverse environments and different spatial and temporal scales in Argentina (South America). Despite the geographical specificity of examples, it seems that many outcomes in this paper can be globally relevant because they are based on sound scientific principles. For example, Nosetto et al. (2012) focus on the transition zone between the Pampas and the Espinal biome in temperate central Argentina where native grasslands and forest coexist with growing areas of grain crops and tree plantations. They show how water yields decline in afforested systems, triggering conflicting relations that emerge between two regulating ES: C sequestration, achieved under tree plantations, and water provision, best supported by herbaceous covers. Likewise, Volante et al. unequivocally demonstrated that cleared patches are more sensitive to climate disturbances than large non-cleared forest areas in NW Argentina, showing the capacity of natural vegetation to provide a buffering ES that ameliorate external disturbances. On the other hand, applying a regional scale approach, Carreño et al. assessed the provision of ecosystem services across regions in Argentina that differ largely in their environmental features. They showed the emergence of significant tradeoffs between ES provision and the economic income as croplands expanded at the expense of natural lands: the economic income increased allometrically as ecosystem services declined. These examples comprise general principles that can be valid and meaningful in Argentina as well as in many other regions of the world.

Independently of the land-use policy and the financial incentive to reward the conservation of valuable ecosystems, the development of unified methods for valuing ecosystem services remains as an unsolved problem. Besides, two additional problems contribute to estimate values and distort price payments: land property rights (Sandberg, 2007) and the speculative value of land in different countries (Barbier et al., 2010). Land property rights affect conservation programs and the relative value of land may vary largely in response to opportunistic financial operations. The last two are particularly concerning in some developing countries and regions.

Beyond disparities among societies, the increased sensitivity of people to local and global environmental problems has helped to spread the notion of environmental governance across the world. Environmental governance is a relatively novel concept applicable to policy that connotes that all human activities – political, social and economic should be understood and managed as subsets of the environment and ecosystems. It involves the establishment, reaffirmation or change of institutions to resolve environment-related conflicts (Pavoola, 2007). However, governance is a concept easier to define than to implement.

To what extent can the valuation of ecosystem services be useful to design land-use policies and preserve environmental governance? This challenge requires from researchers more than good ideas and ingenious techniques in order to influence public opinion and policy. To deal with this, research has to become problem driven rather than tool driven (Hahn, 2000). But to what extent it is wise to ask the researchers to become problem solvers rather than technical supporters? Answers may vary, but they will be increasingly compelled to understand why and how the multiple interests of different stakeholders and sectors in society and policy-makers can affect the application of their methods and techniques to land-use policy and governance. The mere use of concepts like "priceless", "invaluable" or "intangible" has proven to be insufficient for stopping ecosystem degradation.

To preserve governance on the other hand, policy makers and multiple land-users and stakeholders should understand that ecosystem services are becoming increasingly scarce and that it is not possible to manage what was not valued. At the same time, they should recognize that the demand for ecosystem services will rapidly increase as populations and standards of life increase worldwide. So they have no choice; ecosystem service valuation is the tool to do this.

Reality indicates that society needs a mutual effort of understanding among policy makers, stakeholders and scientists to establish an effective dialogue. While problem solving and governance in society is not possible without the intervention of multiple actors and policy makers, successful decisions regarding land use may not be viable without the support of the best scientific knowledge available. The argument that current methods for valuing ecosystem services are still imperfect is indisputable, but they rely on the best body of knowledge that is available. To reduce uncertainty around valuation in coming years, the scientific community must make a considerable effort to reconcile the best-known methods of economic valuation to the emerging bio-physical ones such as those described in this special issue. Combination and complementation between economic and bio-physical valuation will probably be the best way to minimize uncertainty.

7. Concluding remarks

Given that the notion of environmental governance is spreading across the world policy makers, land-users, stakeholders and scientists should recognize that ecosystem services will probably be at the centre of future land-use policies. Current methodological shortcomings for ecosystem service valuation and monitoring are not today a valid argument to differ the implementation of land-use policies. This issue is beyond centralized or decentralized policy views, and even beyond developed or developing countries, that demand the harmonization of sensitive ecological and environmental assets with socio-economic aims.

Despite still being uncertain and non familiar to general people, both economic and bio-physical values provide a rough metrics for ecosystem services that can be useful to compare with social and economic indicators and make decisions. In terms of supporting tools for policy making, tradeoffs analysis can help assessing the relative individual response of social and economic indicators on the one hand, and ecosystem-service estimations on the other hand, per unit of land-use change.

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