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Chapter 9

THE ASSESSMENT OF ECOSYSTEM SERVICES PROVIDED BY BIODIVERSITY: RE-THINKING CONCEPTS AND RESEARCH NEEDS

Berta Martín-López[1], Erik Gómez-Baggethun, José A. González, Pedro L. Lomas, and Carlos Montes

Social-Ecological Systems Laboratory, Department of Ecology.
c. Darwin, 2. Edificio Biología. Universidad Autónoma de Madrid.
28049 Madrid, Spain

ABSTRACT

Recent research illustrates the essential role that biodiversity plays in both ecosystem functioning and the provisioning of ecosystem services for human well-being.

Despite the acknowledged necessity to include the social and economic dimensions into biodiversity conservation research, integrative approaches based on ecosystem services assessment have scarcely been used. This might be in part because ecosystem services have usually been approached from traditionally separated disciplines in the absence of a shared theoretical framework. This chapter is intended to develop such a comprehensive conceptual framework for incorporating ecosystem services assessment into biological conservation research. In doing so, we first reviewed the different existing approaches to ecosystem services assessment, looking for unifying concepts in order to provide an integrative framework. Our proposal focuses on the service-providing functions as the key element to tackle the relationships among society, ecosystems and biodiversity. In addition, an interdisciplinary approach is proposed for the valuation of ecosystem services provided by biodiversity, which integrates the ecological, socio-cultural and economic values of biodiversity. Finally, we reflect on the research needs for evaluating the ecosystem services provided by biodiversity and their relationship with biological conservation.

1 Correspondence author: B. Martín-López Tel: +34 91 497 80 08 Fax: +34 91 497 80 01 e-mail: berta.martin@uam.es .

INTRODUCTION

The links between biodiversity and ecosystem services have been attracting increasing attention in scientific literature over the past few years (Chapin et al., 2000; Díaz et al., 2005; Hooper et al., 2005). Recent publications from the Millennium Ecosystem Assessment (MA) (Díaz et al., 2005; MA, 2005) provide an updated picture of the fundamental messages and key challenges regarding biodiversity (Díaz et al., 2006a). Although the MA (2005) established a conceptual framework to be used for understanding ecosystem services and for assessing their current state and trend, we still lack a robust theoretical basis for linking biodiversity to the ecosystem services underlying human well-being (Carpenter et al., 2006).

The ecosystem services approach involves scientists acknowledging the need for interdisciplinary collaboration between ecologists and social researchers. In order to investigate ecosystem services, ecologists must recognize the human dimension of ecosystem dynamics (Carpenter and Folke, 2006). Ecologists need to know the essence of ecosystem services trade-offs, competing uses in ecosystem services and conflicting choices over temporal and spatial scales. On the other hand, social researchers need to understand ecosystem functioning in order to better recognize the ecosystem condition responsible for the flow of ecosystem services (Kumar and Kumar, 2008). This emerging role of expanding transdisciplinary collaboration among ecologists and social scientists is likely to transform conservation research.

In this context, there is a current scientific recognition of the urgent need to improve approaches for assessing ecosystem services (Carpenter and Folke, 2006), and to develop conceptual frameworks for incorporating ecosystem services into conservation decision-making (e.g. Chan et al., 2006; Egoh et al., 2007). This chapter constitutes an attempt to develop a comprehensive framework to incorporate ecosystem services into conservation research. In doing so, we first review the different existing approaches to ecosystem services assessment and the fundamental concepts that underlie the relationships among ecosystems, biodiversity and human well-being.

Our main objectives are: (1) to review from an ecological perspective the key concepts related to ecosystem services assessment, and (2) to create a conceptual framework capable of reflecting the social value of the ecosystem services provided by biodiversity.

RETHINKING CONCEPTS

Natural Capital, Ecosystem Functions and Ecosystem Services

Although natural capital, ecosystem functions, and ecosystem services have been defined on numerous occasions, there is not a standardized meaning for these concepts (Boyd and Banzhaf, 2007; Wallace, 2007; Fisher et al., 2009). In order to develop an integrative conceptual framework for incorporating ecosystem services assessment into conservation research, we considered it necessary to state what we will understand hereafter by each of these terms (Box 1).

Box 1. Key concepts

Natural capital refers to those ecosystems that have the capacity to exert *ecosystem functions* and provide *ecosystem services* to society.

Ecosystem functions refer to the capacity of ecological processes and structure to provide services that satisfy human well-being (de Groot, 1992).

Ecosystem services are the benefits provided by ecosystems that contribute to making human life both possible and worth living (Díaz et al., 2006a).

Capital is a controversial concept which has given rise to several economic reviews dealing with its meaning during the last century (Naredo, 2003). Although this concept is still subject of debate among economists, *capital* is usually understood as it was defined in neo-classical economics, as the 'stock of real goods, with the capacity to produce further goods or utilities in the future' (El Serafy, 1996), recognizing three production factors: land, labour and man-made infrastructure (Hinterberger et al., 1997). Costanza and Daly (1992) set these economic production factors within the debate on sustainability, using the terms natural capital, human capital and manufactured capital.

We identify *natural capital* and *socially-created capital*, which includes: human, manufactured, financial and socio-cultural capitals (Figure 1). The evolution of human economy has passed from an era in which socially-created capital was the limiting factor in socio-economic development, to the current era in which the remaining natural capital has become the limiting factor (Costanza, 2000).

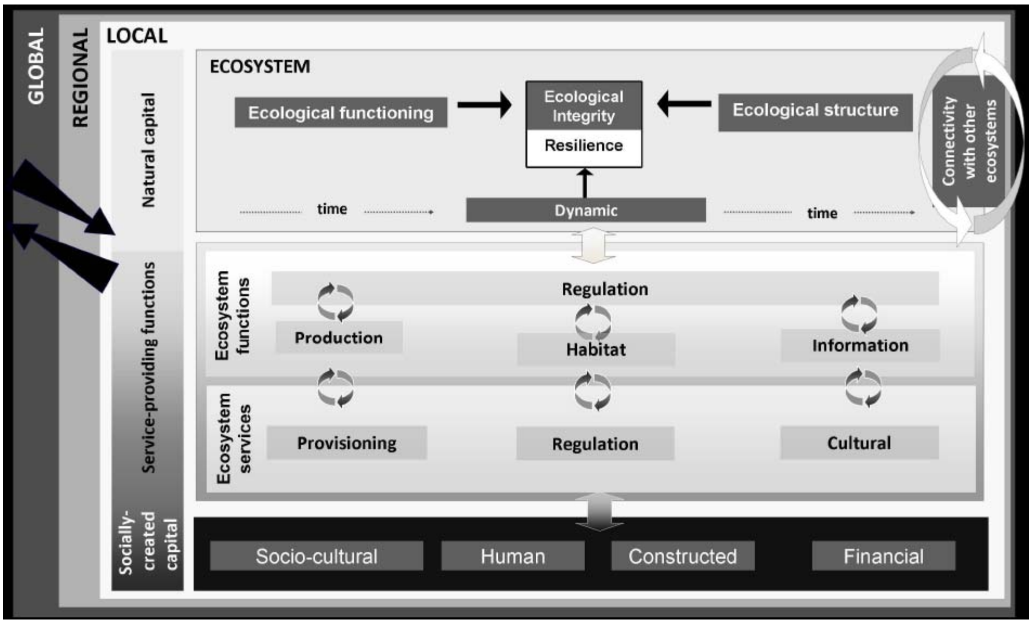

Figure 1. Linking ecosystems to human well-being by the service-providing functions. Ecosystems constitute a natural capital that delivers a stream of services nurturing socially-created capital. Here, the term ecosystem covers both natural and semi-natural ecosystems.

At present, *human capital* is interpreted within a broader scope than the classical one related to the labour factor; including also aspects like knowledge, education, or health. *Manufactured capital* encompasses all material goods generated through economic activity or technological change (de Groot et al., 2003). *Financial capital* relates to the exchange value of other types of capital. *Socio-cultural capital* includes elements such as socio-political institutions, social values, environmental ethics, and social resilience (Ekins, 1992; Berkes and Folke, 1994), but also cultural diversity, common rules and norms, connectedness in networks and groups, and relations of trust among the members of the community and between these and the policy makers (Pretty and Smith, 2004).

The pioneering work of Pearce and Turner (1990) defined *natural capital* as 'any stock of natural resources or environmental assets capable of providing a flow of useful goods and services, now and in the future'. This definition has persisted over the years and has been used in several studies (e.g. Costanza and Daly, 1992; Costanza et al., 1997). We argue that biodiversity conservation research requires an interpretation of the concept with a broader ecological basis. Beside the stock (mainly reflecting structure), ecosystem functioning should also be considered as an essential part of natural capital, because it determines the ecosystems capacity to perform ecosystem functions and provide services. Therefore, we will refer to *natural capital* as those ecosystems that have the capacity to exert *ecosystem functions* and provide *ecosystem services* to society.

This definition of natural capital gives rise to the concept of *ecological integrity* (Figure 1), which is a controversial term that has been defined in different manners (e.g. De Leo and Levin, 1997; Kay and Regier, 2000; Pimentel et al., 2000), becoming an 'umbrella concept' that incorporates aspects such as biodiversity, stability or sustainability. In this chapter, we will refer to *ecological integrity* as the minimum configuration of the *ecological structure* (i.e. geotic and biotic components) and *functioning* (i.e. ecosystem processes, such as primary production, water cycle, and biogeochemical cycles), that characterize a stability domain of an ecosystem. *Ecological integrity* concerns the current organizational state of the ecosystem. However, there is not necessarily one optimal stability domain. Multiple stability domains are possible in a given situation, where each domain represents a different regime for the ecosystem (Gunderson et al., 2002). Each of these stability domains exerts a set of *ecosystem functions,* whose performance depends in the long-term on the ecosystem resilience.

The term *ecosystem function* has been subject to several interpretations, sometimes referring to the internal ecosystem functioning and sometimes relating to the benefits derived by humans from the ecological integrity (de Groot et al., 2002). In this chapter, we understand *ecosystem functions* as the capacity of ecological processes and structure to provide services that satisfy human well-being (de Groot, 1992). Although there are several classifications of ecosystem functions (e.g. Pearce and Turner, 1990; de Groot et al., 2002), here we stick to the one provided by de Groot (1992) in which functions are grouped into four categories: regulation, habitat, production and information. As regulation functions are related to the capacity of ecosystems to regulate essential ecological processes, they are considered as the core functions that maintain the performance of habitat, production, and information functions (Figure 1). Habitat functions refer to the provision of spatial conditions for the maintenance of biodiversity. Production functions are the capacity of ecosystems to provide provisioning services for human use. Information functions offer opportunities for reflection, spiritual enrichment and cognitive development (de Groot et al., 2003).

The terms *ecological functioning* and *ecosystem functions* have occasionally been used in the same manner (e.g. Costanza et al., 1997; Boyd and Banzhaf, 2007), causing some confusion. Herein we establish an important difference between both concepts. Whereas *ecological functioning* is inherent to the intrinsic properties of the ecosystem, as a key component of ecological integrity, *ecosystem functions* are the bridge between natural capital and socially-created capital, through its capacity to provide ecosystem services (Figure 1).

Finally, the concept of *ecosystem services* has been defined several times in Ecology and Economics, but both have failed to standardize its definition (Boyd and Banzhaf, 2007). The role of ecosystems in terms of delivering services to society was first described in 1970 in the 'Study of Critical Environmental Problems' (Mooney and Ehrlich, 1997). The concept was subsequently identified in the ecological forum as 'public services of the global ecosystem' (Ehrlich et al., 1977), as 'nature's services' (Westman, 1977), as 'ecosystem services' (Ehrlich and Ehrlich, 1981), or more recently as eco-services (Bulte et al., 2005). Indeed, the terms ecosystem functions, ecological services, ecological functions, environmental services, and environmental functions are sometimes used in the same sense as the term *ecosystem services* (Egoh et al., 2007).

Daily (1999) conceptualized ecosystem services in the ecological academic forum as the production of goods, regeneration and stabilizing processes, life-fulfilling functions, and conservation of options. Recently, some authors (e.g. Boyd and Banzhaf, 2007) have advocated a stricter definition of ecosystem services as end-products of nature that fits with the neo-classical economic view, in which the benefits obtained from ecosystems are referred to as *ecosystem goods* and *services* (e.g. Daily, 1997; Wilson and Carpenter, 1999). While ecosystem goods are related to the tangible end-products, such as seafood, forage, timber, natural fibre, biomass fuels, and many pharmaceutical products, ecosystem services offer many intangible benefits to society, such as a favourable climate, water quality, soil fertility, as well as aesthetic and cultural enrichment (Daily, 1997).

Although such definitions can be useful for ecosystem services accounting, they convey an important risk in the decision-making process because ecosystem services assessment could be biased toward services that are easily quantifiable, but not necessarily the most critical ones (Díaz et al., 2006a; DeFries et al., 2005). If we want to incorporate ecosystem goods and services into conservation research, we need a broader interpretation of the concept. Some attempts have been made to incorporate both terms, goods and services, under the ecosystem service term, but mostly for simplification reasons (e.g. Costanza et al., 1997). It was not until 2003, when the MA incorporated ecosystem goods into the ecosystem services set, defining ecosystem services as the benefits people obtain from ecosystems (MA, 2003). Based on this definition, Díaz et al. (2006a) explicitly acknowledged the importance of ecosystem services for human well-being, not only for making life possible, but also worth living -i.e. health, security, basic material for life, good social relations, and freedom of choice and action (MA, 2003)-. In our conceptual framework, we rather use this broader definition of ecosystem services because it is much more integrative and consistent with the proposed concept of natural capital.

We also use the widely accepted classification adopted by the MA (2003): supporting, provisioning, regulating and cultural services. Nevertheless, as Hein et al. (2006), we do not distinguish the category of supporting services because it represents the ecological processes underlying the functioning of ecosystems. The inclusion of these supporting services in any valuation process may lead to double counting, as their value is reflected in the three other types of services. Provisioning services are the products obtained from ecosystems, which were usually known as ecosystem goods. Regulating services are the benefits obtained from the regulation of ecosystem processes. Cultural services are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, recreation, and aesthetic experiences.

Research into ecosystem services requires the identification of beneficiaries, the way they use the service, and their location in order to translate a function into a service (Egoh et al., 2007). Thus, the main difference between *ecosystem functions* and *ecosystem services* is related to their use by humans, either consciously or unconsciously. While ecosystem functions exist independently of human enjoyment, use, or valuation; services are necessarily used, enjoyed or valued by society.

Service-Providing Functions in the Context of Social-Ecological Systems

Ecosystem functions and services operate at different spatial and temporal scales. While functions are directly linked to larger and longer-term scales of ecological processes, services are more related to current short-term socio-cultural and economic processes. Therefore, usually there is an important spatial and temporal scale mismatch between the capacity of supplying a service (ecosystem function) and the use and enjoyment of this service by society (Martín-López, 2007). For that reason, it is necessary to jointly investigate the functions and respective services, i.e. to work with *service-providing functions* (Figure 1). This concept refers to those ecosystem processes, functional groups (functional diversity), and species, which have the capacity to supply ecosystem services in a given area. Conservation management focused on service-providing functions ensures a better matching of the different spatial and temporal scales on which ecosystems and social-economic systems operate.

Ecosystems and socio-economic systems share many characteristics. Both are complex networks of components linked by dynamic processes and are open to exchanges across their boundaries by connectivity (Limburg et al., 2002). Despite these similarities, we have tended to treat humans and nature separately, usually under a development-versus-conservation perspective. However, social and ecological systems are interlinked and their separation is arbitrary when analyzing sustainable use and enjoy of ecosystem services (Berkes and Folke, 1998). This linked human-nature systems have been referred in recent literature as social-ecological systems (Figure 2, Anderies et al., 2004). Herein, the concept of service-providing functions emerges as the key element for connecting both ecosystems and socio-economic systems.

Ecosystems and social systems are characterized by containing (1) components (which as a whole constitute the structure of a system), (2) the interactions between them, which generate the processes of the system (functioning), and, (3) in open systems such as ecological or social ones, fluxes crossing the system boundaries. Among these characteristics, interactions (of components, processes or systems with other systems) are responsible for the emergence of complex behaviours. Therefore, social-ecological systems are complex systems, which consist of heterogeneous individual components that interact locally and evolve (physically, behaviourally or spatially) as a result of those interactions (Janssen, 2000).

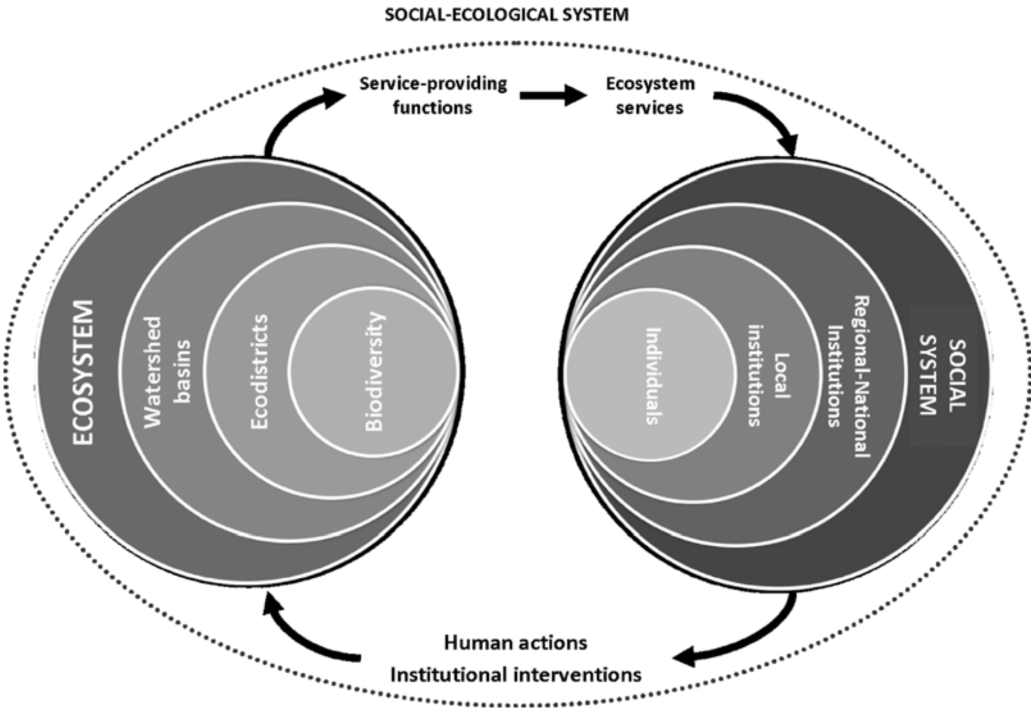

Figure 2. Conceptual diagram of a social-ecological system. Social systems comprise individuals, local groups, and institutions at broad-scale, as well as their relationships. Social systems interact with ecosystems at different scales through individual actions (e.g. fishing, hunting, etc.) or institutional interventions (e.g. conservation, restoration, etc.), in order to obtain ecosystem services. (Modified from Resilience Alliance, 2007).

As ecosystem dynamics are complex and multi-causal, and since causes can be remote in space and time from the event, uncertainty is an intrinsic characteristic of ecosystem services assessments.

The capacity of ecosystems to maintain their structure and functioning in the face of disturbances is determined by their resilience. Resilience is currently defined as the capacity of a system to absorb disturbance and reorganize while undergoing change so as to essentially retain the same functioning, structure and feedbacks (Walker et al., 2004). Resilience reflects the degree to which a complex adaptive system is capable of self-organize and the degree to which the system can build and increase the capacity for learning and adaptation in a changing world (Carpenter et al., 2001). Thus, the reliability of service-providing functions appears to depend on resilience, which itself depends, in complex ways, upon features of the ecosystems (Carpenter and Folke, 2006; Folke, 2006).

The Role of Functional Diversity to Provide Ecosystem Services

Biodiversity is essential for the self-organizing capacity of complex-systems (Levin, 1999), both in terms of absorbing disturbance and of subsequently regenerating and reorganizing the system (Folke et al., 2004).

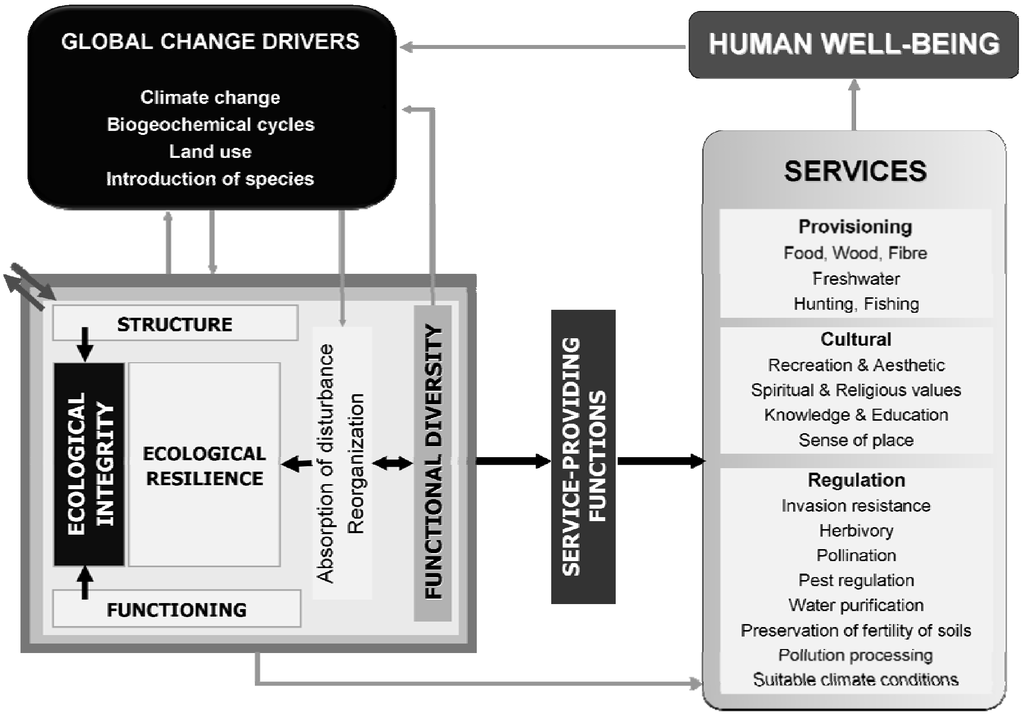

Figure 3. Functional diversity involves both a variable response to disturbances promoted by change drivers and a factor that influences human well-being through the provision of ecosystem services. Black arrows show the links that this article deals with. (Based on Chapin et al., 2000).

Specifically, *functional diversity*, i.e. the kind, range, and relative abundance of functional traits present in a given community (Díaz et al., 2006b), is one of the major factors involved in maintaining ecological integrity (Chapin et al., 2000; Hooper et al., 2005), and, in turn, in the provision of ecosystem services (Figure 3). Recent studies show the important role of functional diversity as the main ecosystem-services providers (Luck et al., 2003; Kremen et al., 2004; Andersson et al., 2007; Vandewalle et al., 2008).

Species are often grouped together according to their *functional traits* to understand general mechanisms of ecosystems. Traits that determine how a species responds to a disturbance or change in environmental factors -*functional response traits*- may differ from those that determine how that species affects ecosystem functioning -*functional effect traits*- (Lavorel et al., 1997; Walker et al., 1999; Lavorel and Garnier, 2002). A *functional group* (or functional type) is a set of species that have similar effects on a specific ecosystem process or similar responses to environmental conditions (Díaz and Cabido, 2001; Hooper et al., 2005). The functional groups of species and their interactions are sources of reorganization for ecosystem resilience in the face of change (Peterson et al., 1998). This role is related to the diversity of functional groups and the species diversity within these functional groups –i.e. functional redundancy (Walker et al., 1992)-.

If multiple species contribute to the same ecosystem function, it is possible that the loss of one or more species might not result in ecosystem function disruption if the rest of species are able to compensate for the loss (Luck et al., 2003). Hence, the presence of dominant and minor species within a functional group provides resilience against perturbations or environmental changes. On one hand, species that dominate under a given set of environmental conditions serve to maintain the ecosystem function under those conditions. On the other hand, minor species which are functionally similar to dominant species, but with different environmental requirements and tolerances, maintain resilience in ecosystems by carrying on the maintenance of function under changing conditions (Walker et al., 1999). Consequently, the variability in responses of species to environmental changes within a functional group is critical to maintain ecosystem resilience, and therefore the performance of service-providing functions in the long-term. Elmqvist et al. (2003) called this property *response diversity*, which is defined as the diversity of responses to environmental change among species that contribute to the same ecosystem function.

As functional diversity declines, the ecosystem becomes vulnerable, and progressively smaller external events can cause shifts. This might result in simplified ecosystems, which are vulnerable to disruptions in their capacity to generate service-providing functions. After a system shift, the new emergent properties of the current state can be, from a societal perspective, either positive or negative. From a management perspective, one must be clear about which system is desired, i.e. what ecosystem services are preferred and more valued by society.

Additionally, it is important to note that all ecosystem functions are the consequence of supporting processes operating at various spatial and temporal scales. Furthermore, species that perform the same ecosystem function at different spatial-temporal scales provide resilience to ecosystem services supply (Peterson et al., 1998). For example, different species of pollinators can begin flight at different times of the day as determined by their body size or warm-up rates, affecting daily and seasonal activity periods and the delivery of pollination functions under different microclimate conditions (Kremen, 2005). Consequently, studies on service-providing functions should be driven within multi-scale approaches considering the interaction among multiple nested scales (Figure 2).

ECOSYSTEM SERVICES VALUATION: A TOOL FOR INCORPORATING ECOSYSTEM SERVICES INTO CONSERVATION POLICY

All components of biodiversity, from the genetic to the community level, may play a role in the long-term supply of at least some ecosystem services (Díaz et al., 2006a). Therefore, the social objective should focus on the maintenance of quality and quantity of biodiversity attributes in order to attain a rich and diversified flow of ecosystem services.

While the valuation of ecosystem services is certainly difficult, it has become essential for the decision-makers who face, on one hand, the trade-offs among different ecosystem services, and on the other hand, the competing use for these ecosystem services (Kumar and Kumar, 2008). Valuation provides a way for decision-makers to assess the impacts and trade-offs of ecosystem and biodiversity change, and elucidates which stakeholders gain or loss benefits at distinct spatial and temporal scales. Ecosystem management certainly involve trade-offs among services, and the weighting of those trade-offs requires some form of valuation (Farber et al., 2006). Quantifying the value of services has become an important tool for assuring the social recognition of the importance of biodiversity conservation.

Incorporating Value Systems into the Decision-Making Process

Ecosystems provide a plurality of benefits to societies, involving the ecological, socio-cultural and monetary dimensions of value. Therefore, the decision-making process regarding biodiversity conservation should jointly consider these three types of value (Figure 4).

The *ecological value* concerns only those 'purely' ecological components that do not depend on human preferences (Straton, 2006), i.e. the intrinsic value of ecosystems or biodiversity. As it was explained above, the ecological value is essentially determined by the integrity and resilience of ecosystems and, therefore, by their functional diversity. As this dimension of value is not associated with human preferences, it is independent of ecosystem services, being directly related to ecosystem functions. Additionally, ecological value influences socio-cultural and monetary values, as the state of the ecosystems –i.e. ecosystem integrity- can shape social preferences.

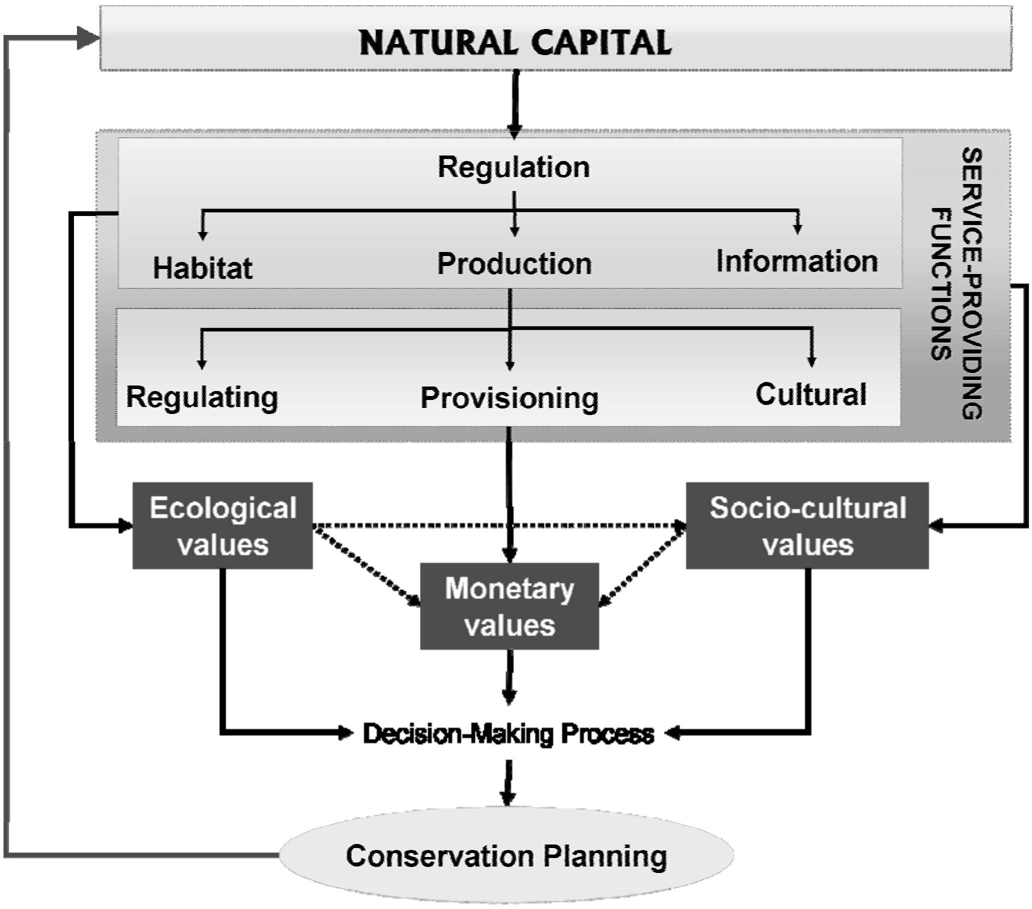

Figure 4. A multi-dimensional framework for incorporating the values of natural capital into conservation planning within an adaptive management process. Discontinuous rows indicate that ecological values partially influence socio-cultural and monetary values; as well as socio-cultural values influence monetary values.

The *socio-cultural value* is related with people's underlying motivations to value nature (Spash, 1997; Johansson-Stenman, 1998). People are attached to moral, ethical and cultural principles that differ from utilitarian criteria (Chiesura and de Groot, 2003), and therefore, we do not express them in monetary terms. We can identify at least three different types of social motivations which are related to social-cultural values:

1) Altruistic motives to future generations and within the current generation (Sagoff, 1988).
2) The recognition of the intrinsic value of non-human species and ecosystems (Spash and Hanley, 1995), which is characterized by beliefs such as 'all species simply have the right to exist'.
3) Finally, both previous motivations are related to social responsibility, from which people derive moral satisfaction by contributing to a worthy cause, such as participation in environmental organizations.

Socio-cultural values partially determine monetary values, because the monetary value that people award to biodiversity usually depends on non-economic motives related to social awareness for biodiversity conservation (Martín-López et al., 2007).

Finally, the *monetary value* also plays an important role in determining whether the importance of the ecosystem service is perceived or not by society (de Groot et al., 2002). In order to capture the monetary value that society allocates to ecosystem services, environmental economists have developed the concept of Total Economic Value (Pearce and Turner, 1990), which comprises the following values:

1) Direct use values, which result from the direct human use of biodiversity, which may be either a consumptive or extractive use, such as food, timber, fibre, etc.; or a non-extractive use, such as certain recreational or educational activities.
2) Indirect use value, which derives from the regulation services provided by biodiversity.
3) Option value, which relates to the importance that people give to a safe future -the future availability of ecosystem services-, either within their own lifetime (option value in a strict sense) or for future generations (sometimes called bequest value).
4) Existence value, which derives from the satisfaction of knowing that an ecosystem or species continues to exist.

Estimating the monetary value of marketed products (i.e. the majority of provisioning services) is usually easier than estimating the value of non-consumptive direct uses or indirect uses, and these are easier to estimate than the non-use value. The absence of financial and legal institutions for many ecosystem services calls for different valuation techniques in order to incorporate their value in the decision-making process.

Consequently, for incorporating the value of natural capital into conservation decision-making, we should contemplate both intrinsic values (ecological values) that do not depend on social preferences, and instrumental values (socio-cultural and economic values), which are related to the indirect or direct benefits that society obtain from biodiversity.

Different Methodological Approaches for Ecosystem Services Valuation

The different dimensions of value can conflict with each other in any decision-making process because the social-ecological system is a site of controversy among competing values and interests of different social communities (Martinez-Alier et al., 1998). These environmental and social conflicts strongly emphasize if we do not properly understand the 'different languages of valuation' (Martinez-Alier, 2002).

The value conflict is rooted in the debate of the commensurability (Figure 5). From a philosophical perspective, it is possible to distinguish between *strong commensurability* and *weak commensurability* (O'Neill, 1993). While strong commensurability implies a common measure of the different consequences of an action based on a cardinal scale of measurement, weak commensurability implies a measure based on an ordinal scale of measurement. Strong commensurability involves *strong comparability* –i.e. there exists a single comparative measuring unit by which all different values can be ranked-, and weak commensurability entails *weak comparability* –i.e. irreducible value conflict is unavoidable but compatible with a rational choice employing practical judgement- (Martinez-Alier, 1998).

Within the strong commensurability context we can distinguish two approaches for valuing ecosystem services: monetary valuation and physical valuation. The utilitarian approach to the monetary dimension of value is enclosed within the Environmental Economics, whereas the Ecological Economics place importance on alternative values such as those based on matter and energy flow analysis (Pritchard et al., 2000).

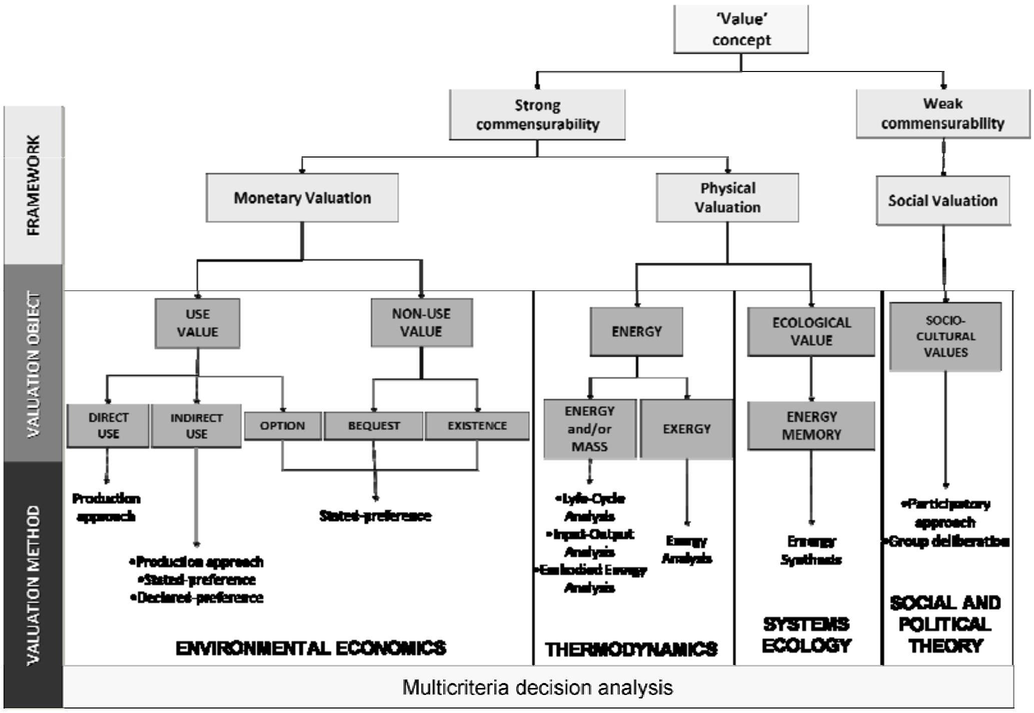

Figure 5. Conceptual framework reflecting the different languages of valuation and the most frequently used valuation methods.

Ecological economists have sometimes used energy and matter flow analysis complementary or alternative to monetary valuation, with the view that energy and matter flow analysis reflects the biogeophysical preconditions for social and economic development (Pritchard et al., 2000). When ecosystems and economic systems are conceptualized in the same language of matter and energy flows, it can be stated that the economy is embedded in the ecosphere of the Earth (Ropke, 2005). The main valuation methodologies developed under this framework are:

1) Material and energetic approaches, which include Material Flow Accounting, Energy Flow Accounting, Life-Cycle Analysis, Input-Output Analysis, and Embodied Energy Analysis. For a revision of these methods, see Daniels (2002), Daniels and Moore (2002), and Herendeen (2004).
2) Exergy approach, which includes Exergy Analysis, Extended Exergy Accounting (Sciubba, 2001), and Ecological Cumulative Exergetic Consumption Analysis (Bakshi, 2002). These techniques are based on *exergy* or *available energy* –i.e. the potential work that can be extracted from a system by reversible processes as the system equilibrates with its surroundings- (Ayres, 1998).
3) Emergy Synthesis, which supposed one of the most important contributions from ecology to physical approaches. In 1983, the term emergy, suggestive of ‘energy memory’, was proposed to eliminate confusion with other energetic valuation concepts, such as embodied energy or exergy. Emergy synthesis characterizes all types of energy in equivalents of solar energy, that is, how much energy would be needed to do a particular task if solar radiation were the only input (Odum and Odum, 2000). For a revision of this method and comparison with other energetic approaches, see Hau and Bakshi (2004), Sciubba and Ulgiati (2005), and Herendeen (2004).

These methodologies have been criticized for the risk of energy reductionism (Georgescu-Roegen, 1982), and because they do not consider social aspects in the valuation (Hau and Bakshi, 2004).

On another hand, Environmental Economics estimates the anthropocentric-based ‘instrumental value’ because it reflects the trade-off between allocation decisions in relation with environment and the resulting change in the economic welfare of the individuals (Venkatachalam, 2007). For environmental economics, valuation is made by individuals according to their own preferences, through something that resembles a marketplace (although it may be hypothetical –i.e. contingent valuation). Different economic valuation tools have been developed to reflect the monetary value of ecosystem services. The main techniques employed by environmental economists are based on three different approaches:

1) The production approach includes techniques based upon current markets, such as direct market analysis, and production function analysis.
2) The revealed-preference approach infers values from data on behavioural changes in real markets related in some way to the missing markets for ecosystem services -i.e. they are based on surrogate markets-. The main techniques developed within this approach are avoided cost, replacement and restoration costs, travel cost, and hedonic pricing.

3) The stated-preference approach avoids conventional markets and explores simulated markets (Adamowicz et al., 1998). Contingent valuation and contingent choice method are used within this approach.

Some valuation methods are more appropriate for valuing particular ecosystem services than others. Regulation services have been mainly valued through avoided cost, replacement and restoration costs, or contingent valuation; cultural services through travel cost (recreation, tourism or science), hedonic pricing (aesthetic information), or contingent valuation (spiritual benefits –i.e. existence value-); and provisioning services through methods based on the production approach. For more details on economic valuation techniques, see Pearce and Turner (1990), Pearce and Moran (1994), or NCR (2004).

The conventional vision of economic value is deeply rooted within the utilitarian perspective, based on individual preferences for ecosystem services. This neo-classical vision assumes that preferences are static; however preferences change over time and under the influence of factors such as education, advertising, consumption trends, etc. If individual preferences change, then value cannot be wholly estimated by personal preferences. Value essentially originates within a community, with its set of interrelated individual and social goals. Thus, economic valuation may focus on any group that can affect or is affected by the supply of ecosystem services -i.e. on stakeholders- (Hein et al., 2006). Furthermore, these valuation techniques assume that societal preferences can be adequately represented by aggregating valuations obtained from isolated individuals. This might be a reasonable assumption when ecosystem services are enjoyed in the purely individual sense and one person’s use creates non external impacts. However, it has been pointed out that this is not appropriate in most real situations, where values depend on communal interaction, where preference formation is partially a social process and where valued services have social implications (Farber et al., 2002). For these reasons, another approach to ecosystem services valuation that has gained increasing attention involves group deliberation (Sagoff, 1998; Wilson and Howarth, 2002). Derived from social and political theory, this valuation approach is based on principles of deliberative democracy and the assumption that decision-making should result, not from the aggregation of separately measured individual preferences, but from open public debate (de Groot et al., 2002). Participatory and discourse-based approaches in conservation decision-making are aimed to accomplish wider community understanding, social equity and greater legitimacy for policies (Wilson and Howarth, 2002). Societal interests may be better attained by encouraging citizen science and participatory approaches, which allow for broad-based debates involving social learning through formation, discussion, negotiation and reconciliation of interests (Chee, 2004). It is important to note that group-deliberation techniques generate complementary information to monetary or energy-based valuations.

Each of this type of valuations may play an important role as an input to the policy-making process, but not as exclusive and ultimate decision-making tools. In this sense, the methodology based on multi-criteria decision analysis constitutes an emerging alternative (Figure 5). This method of policy analysis takes into account a wide variety of relevant information under the weak comparability approach (Martinez-Alier et al., 1998); however it also poses problems such as weak theoretical foundation or biases in ranking criteria (Venkatachalam, 2007).

As the extensive literature on ecosystem services valuation has shown, each of these methodological approaches has its strengths and weaknesses (see Farber et al., 2002; Wilson and Howarth, 2002; Turner et al., 2003; Venkatachalam, 2007).

CONCLUSION

Although the use of ecosystem services in biodiversity conservation has been broadly documented (e.g. Chee, 2004; Mertz et al., 2007; Wallace, 2007), an important ambiguity in the definitions of the key terms –natural capital, ecosystem functions, and ecosystem services– exists, causing difficulties in developing an effective conceptual framework to evaluate ecosystem services. This chapter has tried to clarify those definitions and to develop a conceptual framework capable of assessing the ecosystem services provided by biodiversity.

The ecosystem services framework explicitly states the complex relationships and feedbacks existing among ecosystems and human systems. Consequently, investigating ecosystem services necessarily requires working with social-ecological systems. However, in the current ecosystem services research and management, there are important spatial and temporal scale mismatches between the capacity of supplying a service and the use of this service. For that reason, it seems reasonable to investigate together ecosystem functions and services. In doing so, we have attempted to provide a comprehensive overview, using service-providing functions as the key element for the management of social-ecological systems. By exploring the ecosystem functions and services from different perspectives within this chapter, we have found that the idea of service-providing functions allows us to refocus the way of investigating the relationships between ecosystems and social systems. Understanding and managing service-providing functions requires information about the biophysical nature of services as well as information about their social, economic, and cultural dimensions.

A number of issues for future conservation research emerge from this review:

1) There is a need to extend theoretical, experimental, and observational work on biodiversity effects for an array of ecosystem functions that can be linked to ecosystem services (Balvanera et al. 2006). Information about the magnitude of ecosystem function exertion is pressing. Although one can often determine which services people enjoy, the rate or level of service production is usually difficult to quantify. Policy makers need information about the capacity of biodiversity to produce ecosystem services, as well as how service provision could vary in the face of human actions (Brauman et al., 2007).
2) Spatial and temporal scales in which ecosystem services are supplied and demanded also represent an important knowledge gap, particularly in relation to regulation services. With information about the places in which the services are provided by ecosystems and enjoyed by humans, conservation policies can effectively protect biodiversity and ecosystem services. In this sense, research on services mapping is need to the ecosystem services assessment.
3) Further research is need to understand if the use and enjoyment of ecosystem services produce synergistic or competitive interactions among stakeholders and at what extent those trade-offs occur. In this sense, consideration of stakeholders at

different spatial and temporal scales facilitates the identification of potential social cross-scale conflicts and provides us with clues for multi-scale based decision-making.

4) Further research is required in order to integrate different values of ecosystem services into decision-making. In this context, multi-criteria analysis offers a tool for integrating monetary value with other forms of value. Multi-criteria analysis needs to be used in conjunction with discursive participatory methods (Chee, 2004), where stakeholders hold discussion on the conflicting ecosystem services.
5) Further experimental studies should be developed under the framework of social-ecological systems, which apply this type of approaches to incorporate ecosystem services assessment into biological conservation research.

The conceptual framework proposed in this chapter connects people and biodiversity, moving toward greater interdisciplinarity –an issue that has been highlighted by several authors (e.g. Mascia et al., 2003; Carpenter and Folke, 2006)-, and also breaking significant barriers between natural and social researches in terms of having different concepts, values, and divergent views about what constitutes a useful answer to a problem (Campbell, 2005; Lélé and Norgaard, 2005; Balmford and Cowling, 2006). This framework illustrates how data generated from socio-economic evaluations of ecosystem services might provide useful insights for biodiversity conservation.

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