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Multidimensional Approaches in Ecosystem Services Assessment

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CONTENTS

20.1 The Need for a Multidimensional and Interdisciplinary Framework for Ecosystem Services Assessment

Sustainability science, or the science that focuses on human–nature relationships (MA 2005; Perrings 2007; Perrings et al. 2011), is increasing in research forums particularly through the application of the ecosystem service concept in environmental conservation and management (Seppelt et al. 2011; Burkhard et al. 2012a). Over the past two decades, the ecosystem service concept has gained importance among scientists, managers, and policy-makers worldwide as a way to communicate societal dependence on ecological life support systems integrating both the natural and social science perspectives (Bastian et al. 2012). Many international initiatives, such as the Millennium Ecosystem Assessment (MA), The Economics of Ecosystems and Biodiversity (TEEB), and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Carpenter et al. 2009; de Groot et al. 2010; Seppelt et al. 2011; Burkhard et al. 2012a), have developed interdisciplinary frameworks to tackle the different value dimensions in which ecosystems benefit society and, therefore, make the ecosystem service concept operational.

Despite the progress that has been made, many challenges still remain to integrate the ecosystem service concept into an operational framework (de Groot et al. 2010). Seppelt et al. (2011) recently reviewed the ecosystem service field to provide guidance that enhances the applications of the concept and the credibility of results. Despite the increasing number of publications that present innovative ideas and complementary insights from various perspectives, there is growing uncertainty with respect to the appropriate methodologies and techniques used that limits the comparability and the applicability of the studies (Seppelt et al. 2012). Therefore, one of the major challenges to address is developing a comprehensive framework that integrates the multidimensional value of ecosystem services (i.e., biophysical, sociocultural, and economic) (Lamarque et al. 2011; Chan et al. 2012).

Many authors have noted the importance of identifying the ecosystem's capacity to provide services (supply side) and their social demand (demand side), highlighting that the status of an ecosystem service is influenced not only by the ecosystem's properties but also by societal needs (Paetzold et al. 2010; Syrbe and Walz 2012; Burkhard et al. 2012b). On the supply side, ecosystems and biodiversity are experiencing serious degradation with regard to their capacity to supply services. At the same time, the demand for certain ecosystem services is rapidly increasing as populations and standards of living increase (Liu et al. 2010). Burkhard et al. (2012a) defined the supply side as the capacity of a particular area to provide a specific bundle of ecosystem services within a given time period, and the demand side as the sum of all ecosystem services currently consumed, used, or valued in a particular area over a given time period.

In this sense, remote sensing methodologies and techniques have been mostly used in the past two decades for quantifying and mapping the supply side of ecosystem services, in particular quantifying and map provisioning (e.g., timber or food production) and regulating services (e.g., air quality, climate, extreme events, waste treatment, erosion, and soil fertility) (Ayanu et al. 2012). Remotely sensed information has been used most often as a proxy for biophysical variables (e.g., biomass; see Chapter 5) that in turn are used as proxies for a particular ecosystem service (e.g., carbon storage). Ayanu et al. (2012) recently provided a revision of relevant remote sensing systems, sensors, and methods applicable in quantifying the supplies and demands of provisioning and regulating services. Their results showed that the quantification of services through Earth observation techniques used either regression models (by linking remotely sensed information to a limited number of *in situ* observations) or land use/land cover classifications (see Chapter 10) that are subsequently linked to ecosystem services.

Here, we present a conceptual framework for ecosystem service assessment (Figure 20.1) that links the service-providing units (SPUs) and the ecosystem service beneficiaries (ESBs) concepts and how they could be

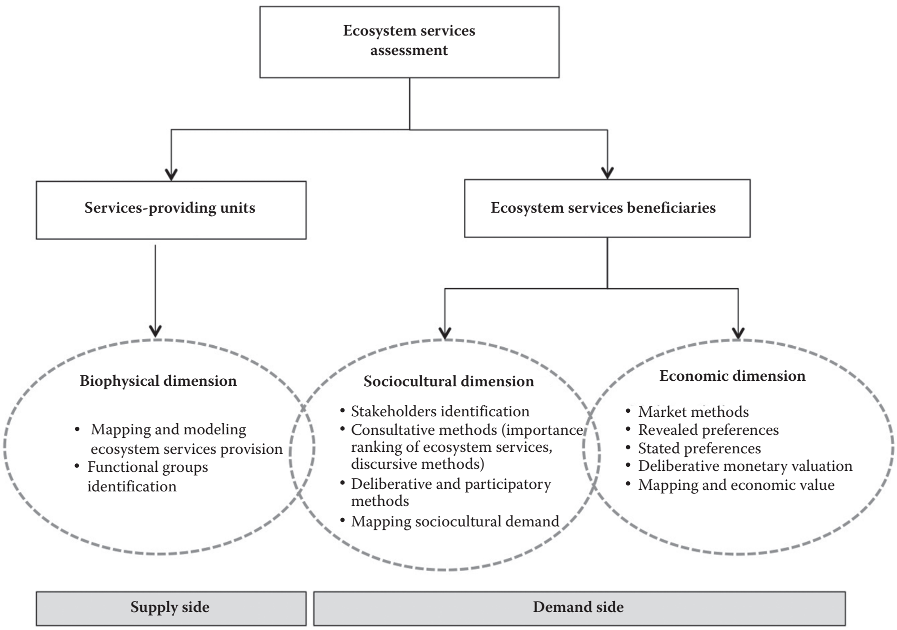

FIGURE 20.1
Conceptual framework showing the links among services-providing units and ecosystem services beneficiaries and how they could be explored through the different dimensions in the ecosystem services assessment.

explored from both the supply side and demand side, considering the multidimensional nature of ecosystem service assessment. Within this framework, we explore which remote sensing methods and techniques are currently available for the assessment of both the supply and the demand side of ecosystem service.

20.2 The Supply Side of Ecosystem Services

20.2.1 Service-Providing Units

Several concepts have been developed to operationalize the specific contributions of ecosystems to the delivery of ecosystem services. One of them is the SPU concept, developed as a method to link species populations with service delivery, primarily at small scales (Luck et al. 2003) but extensible to larger scales. The SPU is defined here as the collection of individuals from a given species and their characteristics necessary to deliver an ecosystem service at the level desired by beneficiaries. This idea was further extended to other levels of organization and scales of analysis. In fact, Luck et al. (2009) suggested the use of social–ecological landscape units as service providers where the capacity of landscapes to provide services would be related to their structural, functional, and social attributes within a socioecological context. There is growing consensus that, among all biodiversity components, it is the functional diversity of the component that mostly ensures the delivery of ecosystem services, particularly regulating services (Díaz et al. 2006; Cardinale et al. 2012; Alcaraz et al. 2013).

Remote sensing may provide useful tools for understanding the SPU concept and its application into regional and global assessments of ecosystem services. Spectral information may help to map land units that are homogeneous in terms of the ecosystem services that they provide. This could be the case with the ecosystem functional types (EFTs) approach. As plant species can be grouped into plant functional types, ecosystems can be grouped into EFTs (Paruelo et al. 2001). EFTs represent groups of ecosystems that share functional characteristics in relation to their exchanges of matter and energy between the biota and the physical environment (Paruelo et al. 2001; Alcaraz-Segura et al. 2006). In other words, EFTs are homogeneous patches of the land that exchange mass and energy with the atmosphere in a common way and, therefore, they can be interpreted as landscape units with the same capacity to provide ecosystem services derived from the ecological process used in their classification (e.g., carbon gains). In this sense, further studies exploring the capacity of EFTs to deliver ecosystem services could be a major breakthrough for Earth observation of ecosystem services.

20.2.2 Biophysical Indicators for Evaluating Ecosystem Services

Several biophysical indicators have been used for assessing ecosystem services. The type of indicator depends on the information available for each case study and the amount of resources willing to be used for evaluating ecosystem services. Some studies have grouped the most common indicators used (Maes et al. 2011; Burkhard et al. 2012b). Table 20.1 presents some of the indicators most often used to assess ecosystem service supply, and it indicates some indicators that might be used for ecosystem service demands. We also provide studies where remote sensing has been or may be used to derive ecosystem service indicators on both the supply and the demand sides.

Due to the urgent need for understanding and mapping the spatial and temporal heterogeneity of ecosystem services, and for maximizing conservation objectives (Polasky et al. 2008; Nelson et al. 2009), remote sensing can be very useful for obtaining biophysical indicators in the supply side of ecosystem service assessment. Approaches from remote sensing can be used for ecosystem service mapping by direct or indirect monitoring and in combination with ecosystem models (Feng et al. 2010). Ecosystem services directly monitored such as habitat for species (see Chapter 3), carbon fixation (see Chapter 6), water provision (see Chapter 12), and climate regulation (see Chapters 17 and 20) require information on aspects of vegetation and water. Indirectly monitored services, such as soil-based services, use surrogate information such as soil status or canopy reflectance. Finally, some services such as flood regulation or soil erosion can be monitored and implemented in ecosystem models through inputs derived from remotely sensed spatial explicit data. Table 20.2 shows some remote sensing products linked to ecosystem services (Feng et al. 2010).

20.2.3 Going Spatial: From Early Approaches to Current Toolboxes

Ecosystem services maps seem to be among the most useful approaches to integrate the ecosystem service concept into ecosystems management (Balvanera et al. 2001; Daily and Matson 2008). They allow the identification of highly valuable areas for conservation (Chan et al. 2006; Naidoo and Rickets 2006) or the identification of ecosystem service supply and demand, which might help in achieving a sustainable use of ecosystem services (Kroll et al. 2012). From the very first ecosystem services maps (Eade and Moran 1996; Costanza et al. 1997), to the current ecosystem service models or toolboxes (e.g., InVEST or ARIES; see Box 20.1), we have witnessed an increase in the different methodologies and purposes for mapping ecosystem services. Nowadays, new tools for mapping services have emerged to support landscape management (see POLYSCAPE in Box 20.1) (Jackson et al. 2013). Mapping works started focusing on the supply side (Burkhard et al. 2012b), but participative methods are being increasingly used to also include the demand side (Bryan et al. 2011; Palomo et al. 2012).

TABLE 20.1

List of Biophysical Indicators and Proxies Used for Evaluating Ecosystem Services Supply and Demand

Ecosystem Services	Biophysical Indicators for the Supply Side	Studies from Remote Sensing	Social Indicators for the Demand Side	Studies from Remote Sensing
Provisioning services				
Food provision	Crop yield production	Doraiswamy et al. (2003)	Crop yield consumption	
Food provision	Number of grazing heads		Livestock consumed	Stephen et al. (2013)
Food provision	Fish population sizes	Chassot et al. (2010)	Number of fish or biomass captured or consumed	Stuart et al. (2006)
Biotic materials	Timber stock	Clementel et al. (2012)	Timber production used	
Water provision	Water availability (i.e., precipitation minus evapotranspiration)	Bahadur (2011)	Water consumed	
Medicinal resources	Number of species from which natural medicines have been derived		Number of drugs using natural compounds	
Genetic pool	Number of crop varieties and livestock breed varieties living in a region/surface	Heller et al. (2012)	Number of crop varieties and livestock breed variety used in a region	
Raw materials	Raw materials existing in nature/surface	Mengzhi (2009)	Quantity extracted, used, or bought/surface/time	
Regulating services				
Air quality regulation	Atmospheric cleaning capacity in ton of pollutants removed		Illnesses avoided related to air contamination	
Water quality regulation	Biomass of nutrients removed by aquatic ecosystems (i.e., N, P)		Illnesses/avoided related to contaminated water or unhealthy water consumption	Lobitz et al. (2000)

Climate regulation	Total amount of carbon (or methane) sequestered-stored	Myeong et al. (2006)	Avoided number of climate refugees, people affected by climate change
Moderation of extreme events	Natural elements or number of species dampening extreme events (flood, storms, avalanches)		Avoided damages caused by flood, storms, avalanches
Erosion protection	Soil erosion rate, erosion related variables (slope, precipitation, vegetation cover, etc.)	Vrieling (2006)	Mass of soil removed from water reservoirs
Pollination	Abundance and species richness of wild pollinators	Schulp and Alkemade (2011)	Benefits for crop production or biodiversity maintenance due to pollination
Cultural services			
Aesthetic enjoyment	View shed extent, iconic elements views		Users of scenic routes
Recreation and tourism	Number of trails or natural areas available for tourism	Nichol and Wong (2005)	Number of visitors
Recreation and tourism (recreational hunting)	Animal population sizes, regeneration rate of species	Ropert-Coudert and Wilson (2005)	Individuals hunted
Intellectual and experiential (scientific knowledge)	Abundance of landscape features or species with scientific value	Mertes (2002)	Number of researchers working with environmental assets

Source: From de Groot, R. S., et al., *Ecological Complexity*, 7, 260–272, 2010; Maes, J., et al. *A spatial assessment of ecosystem services in Europe: Methods, case studies and policy analysis-phase 1*, PEER Report No 3, Partnership for European Environmental Research, Ispra, Italy, 2011; and Burkhard, B., et al., *Ecological Indicators*, 21, 17–29, 2012.

TABLE 20.2

Ecosystem Services Quantified by Remote Sensing Data

Proxy	Ecosystem Services	Spectral Index or Techniques	Sensor Types	References
Carbon storage/ sequestration by different land uses/land cover	Climate regulation; air quality regulation	Advanced Very High Resolution Radiometer (AVHRR) for mapping land covers	LANDSAT	Konarska et al. (2002)
Animal and plant species richness and abundance	Habitat for species; biodiversity conservation	Normalized Difference Vegetation Index (NDVI)—landscape heterogeneity; Light Detection and Ranging (LiDAR)—canopy structure	MODIS	Gould (2000); Balvanera et al. (2006); Carlson et al. (2007)
Carbon uptake through ecosystem carbon gains	Climate regulation	Normalized Difference Vegetation Index (NDVI); net primary productivity (NPP); Enhanced Vegetation Index (EVI); gross primary productivity (GPP); Leaf Area Index (LAI)	MODIS	Gianelle and Vescovo (2007); Olofsson et al. (2008); Gianelle et al. (2009)
Water cycle (green and blue water)	Hydrological regulation; erosion protection; flood control	Normalized Difference Vegetation Index (NDVI)—surface parameters; Leaf Area Index (LAI)—surface parameters	SWIM WEPP LASCAM SEB	Krysanova et al. (2007); Liu and Li (2008); Minacapilli et al. (2009); Williams et al. (2010)
Properties of soil types	Climate regulation; soil fertility	Normalized Difference Vegetation Index (NDVI)— proportion of bare soil; normalized difference wetness index (NDWI)—soil saturation; soil color index (SCI)—soil saturation; LiDAR—soil roughness measurement	LANDSAT MODIS	Lobell et al. (2009); Kheir et al. (2010)

Note: Table offers most common proxies, spectral index or remotely sensed techniques, sensors types used. SWIM = Soil and Water Integrated Model; WEPP = Water Erosion Prediction Project model; LASCAM = Large Scale Catchment Model; SEB = Surface energy balance; LANDSAT = Land-Use Satellite (http://landsat.gsfc.nasa.gov); MODIS = Moderate Resolution Imaging Spectroradiometer (http://modis.gsfc.nasa.gov).

BOX 20.1 MAIN TOOLBOXES FOR ECOSYSTEM SERVICES MAPPING

InVEST: INTEGRATED VALUATION OF ENVIRONMENTAL SERVICES AND TRADEOFFS

InVEST is a family of tools designed by the Natural Capital Project that allow mapping of ecosystem services. It is designed to inform decisions about natural resource management and provides an effective tool for evaluating trade-offs among ecosystem services by estimating the amount and value of ecosystem services that are provided on the current landscape or under future scenarios. InVEST models are spatially explicit, using maps as information sources and producing maps as outputs. InVEST returns results in either biophysical (e.g., tons of carbon stored) or economic terms (e.g., net present value of that sequestered carbon). InVEST allows mapping several ecosystem services These are (1) lands and waters—biodiversity, carbon, hydropower, water purification, reservoir sedimentation, managed timber production, crop pollination; and (2) oceans and coasts—wave energy, coastal vulnerability, marine fish aquaculture, aesthetic quality, overlap analysis (fisheries recreation, habitat risk assessment). The different tools run as an extension of ArcMAP and are available from the Natural Capital Project Web site (http://www.naturalcapitalproject.org/InVEST.html). There is a forum in which users can communicate and exchange experiences.

LAND USE/LAND COVER: REMOTELY SENSING INFORMATION

Land use/land cover has been widely used as a proxy for the quantification and mapping of ecosystem services by assigning ecosystem service values to the different land use/cover types. Remote sensing provides useful data for land use/land cover classification. The classification techniques are based on statistical analysis to obtain discrete classes, and its accuracy depends on the set training areas. Therefore, resulting land use/land cover classification depends on properties of the remote sensing data available. Thus, the accuracy of ecosystem services quantification depends on the accuracy of the land use/land cover classification (Ayanu et al. 2012).

ARIES: ARTIFICIAL INTELLIGENCE FOR ECOSYSTEM SERVICES

ARIES is a web-based technology developed by various institutions (including the University of Vermont) that allows rapid ecosystem service assessment and valuation. It provides an intelligent modeling platform capable of composing complex ecosystem services models from a collection of models specified by the user. It can map the locations and quantity of potential provision of ecosystem services (sources), their human beneficiaries (users), and any biophysical features that can deplete service flows (sinks). Because many models written in the ARIES modeling language are based on a probabilistic, Bayesian approach, they are able to explicitly convey uncertainty about their inputs to their outputs and are capable of operating even in data-scarce conditions where deterministic models cannot run. In the version 1.0 beta release, ecosystem services maps are available for seven areas of the world including the following ecosystem services: water supply, subsistence fisheries, carbon, flood and sediment regulation, coastal protection, aesthetic, and recreation. All information is available at the project Web site (http://ariesonline.org).

POLYSCAPE: EMERGING NEW TOOLS

POLYSCAPE is a geographic information system (GIS) framework designed to explore spatially explicit synergies and trade-offs among ecosystem services to support landscape management. POLYSCAPE currently includes algorithms to explore the impacts of land cover change on flood risk, habitat connectivity, erosion and associated sediment delivery to receptors, carbon sequestration, and agricultural productivity.

Remote sensing provides valuable input data for ecosystem service models or toolboxes commonly used to simulate ecosystem services. Ecosystem service models provide an explicit connection between the ecosystem services to be quantified and the remotely sensed parameters. For instance, the Integrated Valuation of Environmental Services and Trade-offs (InVEST) toolbox quantifies and maps ecosystem services using spatially explicit information on land cover, evapotranspiration, precipitation, and topography that can be derived from remote sensing data. Box 20.1 provides information on the applicability of remote sensing data in characterizing and mapping land use and land covers within InVEST.

20.3 The Demand Side of Ecosystem Services

20.3.1 Ecosystem Services Beneficiaries

Currently, the majority of ecosystem service studies do not explicitly include the preferences and values of different ESBs (Menzel and Teng 2010; Seppelt et al. 2011). However, there is a range of stakeholders with different priorities regarding which ecosystem services are most important for their well-being (McMichael et al. 2003; Díaz et al. 2011) and, consequently, they should be included in the ecosystem service assessment (Egoh et al. 2007). Detaching ecosystem services from their perceived value, as is often currently practiced, implies that these services can be defined without including the values given by those who benefit from them. This approach is not compatible with the ecosystem service definition. Ecosystem services include the direct or indirect contributions of ecosystems to human well-being; thus, including the importance of human preferences in their assessment is a necessary requirement (de Groot et al. 2010; EME 2011).

Recent contributions in Mediterranean protected areas identify three common categories of stakeholders' profiles that should be included in the assessments: tourist population, managers and environmental professionals, and local population or residents with different relationships with the ecosystem services (Martín-López et al. 2007; Castro et al. 2011; García-Llorente et al. 2011a, 2011b). Locals mainly benefit from and enjoy provisioning (related to agropecuarian activities) and cultural services associated with a sense of place and cultural heritage. Professionals are focused on regulating services and those cultural services related to knowledge systems and protected areas (i.e., environmental education, scientific knowledge, or nature tourism). Tourists usually indicate their preferences for cultural services related to recreational activities and aesthetic values linked to urban demands of biophilic stimuli. Understanding this diversity of views improves the analysis of potential social conflicts and the understanding of the ecosystem service trade-offs. On the one hand, the identification and characterization of the different ESBs is a first step to their latter engagement in participatory processes for designing or promoting environmental management policies toward a desirable future (Baker and Landers 2004; Reed 2008; Palomo et al. 2011). On the other hand, the inclusion of different ESBs profiles promotes the combination of different knowledge sources, that is, the experimental (local ecological knowledge) and experiential (technical or scientific knowledge) (García-Llorente et al. 2011b).

20.3.2 Sociocultural Valuation

The number of studies using the sociocultural perspective in service valuation is very limited, and the techniques used have not been as formalized as in the economic assessments (explained here). However, this approach is increasingly gaining attention as a means to value cultural services, address nonmaterial benefits (Chan et al. 2006, 2012), and to reflect a plurality of service values. The sociocultural valuation of ecosystem services includes noneconomic methods to analyze human preferences toward ecosystem service demand, use, enjoyment, and value in which moral, ethical, historical, or social aspects play an important role. Understanding human preferences, attitudes toward nature, and behavioral intentions requires analysis of psychological, historical, and ethical factors in addition to economic approaches (Spash et al. 2009).

In sociocultural valuation, we recognize that ecosystems and their biodiversity provide services related to non-use values, such as the satisfaction of conserving biodiversity, local identity, or local ecological knowledge. The same applies when we want to explore the inherent intrinsic value of species and ecosystems, where the utility function used in the economic dimension does not cover all human motivations (Chan et al. 2006). As Kumar and Kumar (2008) stated, there are issues that transcend the domain of the particular logic of economic choices and, therefore, the different dimensions of human well-being, such as social relationships, health, security, and freedom of choice and action (MA 2003). These cannot be addressed using economic techniques (Wegner and Pascual 2011).

The particular sociocultural values attached to biodiversity and ecosystem services can be explored using qualitative and quantitative techniques that involve direct and indirect consultative methods. Direct consultative methods include techniques that explore individual perceptions and collective preference methods. On the one hand, techniques analyzing individual perceptions of ecosystem service importance or use, apply ranking or rating of preferred ecosystem services through surveys and the use of scales (e.g., Likert scales). In the ranking technique, the respondents usually decide the most important ecosystem services from a panel of existing services in a given ecosystem (e.g., Castro et al. 2011). In the rating technique, respondents rate each service independently (Agbenyega et al. 2009), often using some kind of visual aid (Calvet-Mir et al. 2012). On the other hand, collective preferences, such as discourse-based analysis (Wilson and Howarth 2002), are based on the assumption that the valuation of public goods (e.g., most ecosystem services) should result from a process of free and open public debate incorporating social equity issues and not from the aggregation of individual perceptions. In this set of techniques, a small group of individuals (usually more than 2 but no more than 20) debates and reaches a consensus-based value of ecosystem services.

In indirect consultative methods, respondents are asked to name notions or terms to describe a particular ecosystem; later, the researcher associates these notions with the defined ecosystem service types (e.g., Quetier et al. 2010). In addition, those expressed views could be complemented with the views, ideas, and language included explicitly in communication media (books, articles, laws, conservation programs, webs, etc.). This could be done using content analysis techniques—suitable tools to assess ESB views and values attached to biodiversity and ecosystem service importance (Xenarios and Tziritis 2007; Webb and Raffaelli 2008). Q-methodology is also a promising way to identify ecosystem service values. This technique is focused on how ESBs understand and feel about environmental issues (Sandbrook et al. 2011), with the possibility of exploring ecosystem service priorities and tradeoffs depending on the ESBs profile. Other indirect approaches would consist of the use of social preferences toward landscapes, considering landscape as a representation of the capacity of ecosystems to provide ecosystem services to society, and *a posteriori* identification of ecosystem services using the Delphi method (e.g., García-Llorente et al. 2012).

20.3.3 Economic Valuation

In the past decades, the economic dimension has gained the highest importance in both the academic and the political arenas (Gómez-Baggethun et al. 2010). One of the first steps toward its mainstream occurred in the 1990s with the publications of de Groot (1992), Daily (1997), or the research conducted by Costanza et al. (1997) in which 17 ecosystem services for 16 biomes in the world were valued with an average estimation of US$33 trillion per year. (Meanwhile, the global gross national product total was around US$18 trillion per year.) Consequently, nature "production" was estimated to be 1.8 times more than the human "production." Since then, the predominance of ecosystem service economics could be seen by the rise in the number of scientific articles (e.g., Loomis et al. 2000; Kontogianni et al. 2010), and by different global projects such as TEEB, which quantifies the cost of biodiversity loss and ecosystem service degradation at an international scale (TEEB 2010). This approach has been broadly applied to value the ecosystem services provided by different ecosystems such as forests (e.g., Croitoru 2007; Zandersen and Tol 2009), wetlands (e.g., Woodward and Wui 2001; Brander et al. 2006), marine ecosystems (e.g., Turpie et al. 2003; Ressurreição et al. 2011), or by species (Losey and Vaughan 2006; García-Llorente et al. 2011b).

Some of the principal explanations for the predominance of this dimension are given next. First, considering that biodiversity continues to decline (Burkhard et al. 2012b), there is a call to look for the instrumental arguments to support biodiversity conservation in addition to acknowledging its intrinsic value, a vision traditionally supported by conservationists. It is argued that as far as intrinsic values are not measurable, they tend to be ignored in the decision making (Bateman et al. 2002). Following this line of argument, there is currently a growing enthusiasm for the challenge of giving visibility to the ecosystem services provided by biodiversity that are invisible to the markets (positive externalities) because of their lack of price (but not value). This is the case for a large number of regulating services (e.g., pollination, erosion control, or water regulation) and cultural services (e.g., aesthetic values, local ecological knowledge, or the satisfaction of conserving species) (Rodríguez et al. 2005; Gee and Burkhard 2010; Vejre et al. 2010). Policy-makers are particularly sensitive to the cost of actions, and a majority of land use policies are based on economic studies using a cost–benefit analysis that measures the benefits and costs of a policy measure (Balmford et al. 2011). This argument is based on the idea that the more complete the information in economic terms is regarding the contribution of ecosystems and biodiversity to human well-being, the higher the decision-making success (de Groot 2006). The economics of ecosystem services has proven helpful in demonstrating the importance of those ecosystem services without market price, such as (for example) the contribution of pest control and pollination to agricultural production (Gallai et al. 2009) to set priorities between conservation strategies (Martín-López et al. 2007; García-Llorente et al. 2011b), or to analyze the trade-offs or synergies between different biodiversity and ecosystem management options.

Several case studies around the world have demonstrated the importance of maintaining different natural ecosystems not only for their ecological value but also in economic terms, instead of converting them into intensive uses. This has been the case with studies showing the importance of the tropical forests in Cameroon or the wetland ecosystems in Canada (Balmford et al. 2002), the economic importance of sustainable management in wetlands (Birol et al. 2006), coral reef management strategies (Hicks et al. 2009), or the ecosystem services provided by Mediterranean protected areas (Martín-López et al. 2011). In this sense, considering the context of global change and the mainstream economic thinking, the economic valuation of ecosystem services has created a pragmatic and common language between scientists and policy-makers, and a forum of discussion around the idea of how to make explicit the human dependence on ecosystems and their biodiversity.

Within environmental economics, ecosystem services are considered as positive externalities that could be measured through the total economic value (TEV) framework. Different service categories have different types of values attached to them that could be aggregated and isolated for analysis (Pearce and Turner 1990). The TEV is composed of use and non-use values (Figure 20.2). Use values are related to the direct or indirect contributions we receive from ecosystems; non-use values are related to moral or ethical considerations of maintaining biodiversity and its ecosystem services independent of their use values. At the same time, use values are composed of direct use values, indirect use values, and option values. Direct use values usually have an expression in markets and result from the direct human use of ecosystems and their biodiversity, no matter if this is consumptive or extractive (e.g., timber or freshwater) or nonconsumptive or nonextractive (e.g., nature tourism). Meanwhile, indirect use values are generally not reflected in conventional markets and are derived from ecological processes and regulating services (e.g., water purification by aquatic plants or the role of mangrove ecosystems in erosion control or erosion mitigation). Finally, the option value is related to the importance of maintaining a flow of ecosystem services in the future and by definition is associated with any ecosystem service category. Non-use values could be split into existence values related to the satisfaction of conserving ecosystems and their biodiversity even though we will not enjoy or use them; that is, which ones represent a cultural service or philanthropic values related with the satisfaction of knowing that future generations will have access to ecosystem services (bequest value), and the satisfaction of knowing that other people have access to ecosystem

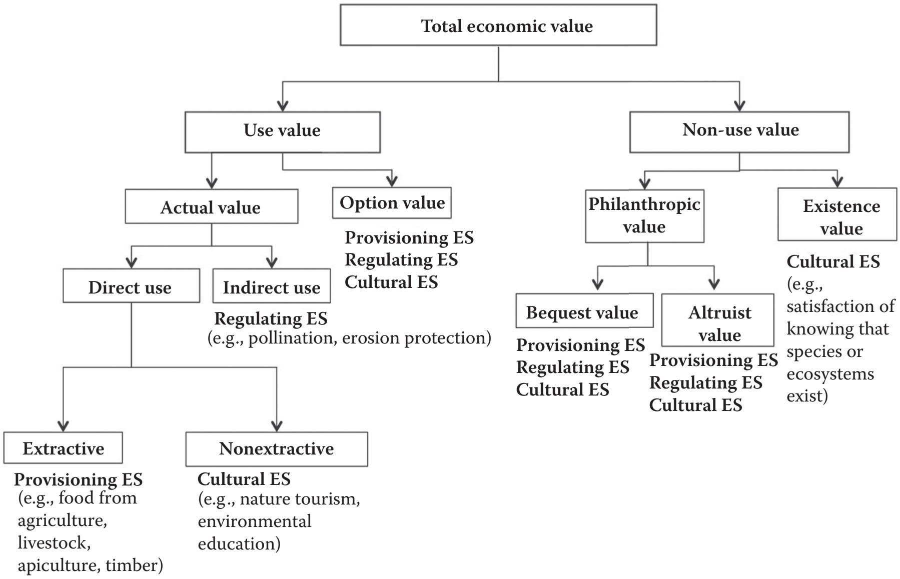

FIGURE 20.2
Representation of the total economic value, its value types, and the ecosystem services categories related to them. ES = ecosystem service.

services (altruist value). Thus, the option value, and bequest and altruist values could be related to all ecosystem services categories (see Bateman et al. 2002; Martín-López et al. 2009; TEEB 2010 for more detail).

In order to assess each value type, a number of different approaches have been designed within environmental economics. These approaches could be classified into three main categories: direct markets, revealed preferences, and stated preferences. The challenge now is to know which valuation method can best fit different biodiversity value types and their ecosystem services with attention to the context and the particular policy purpose (de Groot et al. 2010). In a review of previous works, we provide some guidelines about how to decide which valuation methods are best suited for measuring different valuation types and service categories.

Direct markets use price as a reflection of value and then use data from actual markets to estimate direct use values. These include: (1) market prices used for those provisioning services sold in markets such as the commodities obtained from agriculture or from forest services (such as timber or nontimber forest products); Production function (2) is used to estimate how much a particular ecosystem service that does not have a market price contributes to the delivery of another service that is sold in markets (e.g., the contribution of pollination in apiculture or agricultural production); and (3) the cost approach estimates the expense that would be incurred if ecosystem service contributions needed to recreated through artificial markets using the estimation of avoided cost or replacement cost (TEEB 2010). Both production function and cost approaches are usually used to estimate indirectly the value of regulating services (e.g., how much wetland is saved by providing protection to the coastline against storms and floods) (Figure 20.3).

Revealed preferences estimate the value of a given service without market price through the observation of substitute markets related to the service. The two main techniques are: (1) travel cost (used to estimate recreational services such as nature tourism in a given natural area), which is based on the idea that the cost to arrive to the particular area should be at least equal to the utility obtained (e.g., Shrestha et al. 2002; Martín-López et al. 2009); and (2) hedonic pricing—where a market commodity, usually a property, is described in terms of several attributes including an environmental one (e.g., size, neighborhood, but also the possibility to see an aesthetically pleasant landscape from the window). Then, estimating a demand function for the property, we could infer the value of a change in the environmental attribute (e.g., with landscape views or without them) (e.g., Lansford and Jones 1995; Geoghegan et al. 1997). Travel cost and hedonic pricing are mainly used to estimate the indirect use value of cultural services related to recreational activities (e.g., nature tourism, recreational fishing, recreational hunting, and landscape aesthetic values). However, hedonic pricing

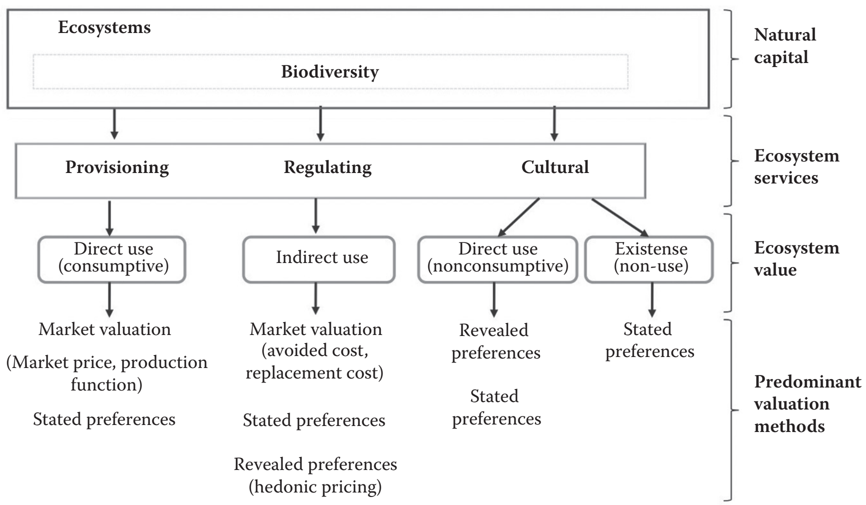

FIGURE 20.3
Graphical representation of the natural capital, the ecosystem services categories delivered, the economic values related to them, and the most commonly used evaluation methods for measuring them.

could be also used to value some regulating services—for example, to estimate the implicit price for air quality service in the price of a property (Figure 20.3).

Finally, stated preference techniques use surveys to create hypothetical markets using to calculate the value of ecosystem services related with both use and non-use values and that could be applied to all ecosystem service categories (Figure 20.3). The three main methods are: (1) the contingent valuation method that elicits public preferences by directly asking people how much they would be willing to pay (or accept) for a change in the quantity or quality of a given ecosystem service in a hypothetical market (Mitchell and Carson 1989). This technique has been one of the most widely used (e.g., Jorgensen et al. 2001; Gürlük 2006; García-Llorente et al. 2011a); (2) choice modeling or conjoint analysis (with the different possibilities of choice experiments, choice ranking, and choice rating) elicits public preferences by asking respondents to choose their preferred option from a series of alternatives of choice sets, each described in terms of different attributes and levels related to the ecosystem services or different environmental plans (e.g., Hanley et al. 2003; Westerberg et al. 2010; Zander and Straton 2010); (3) finally, deliberative monetary methods constitute a hybrid method in which stated preferences techniques are applied in small groups in order to facilitate a participation process (e.g., Zografos and Howarth 2008; Kenter et al. 2011). For an overview of the different valuation techniques, see Bateman et al. (2002), Chee (2004), TEEB (2010), and Turner et al. (2010).

All of these valuation methods present advantages and disadvantages related to information and methodological misspecification, strategic responses, equity problems, unfamiliarity, or sequencing effects, which still need to be improved (Carson et al. 2001; Barkmann et al. 2008; Schläpfer 2008; Turner et al. 2010). However, some of these shortcomings are also related to the inherent limitations of the economic framework itself. Based on neoclassical economics, these techniques assume a utilitarian framework in which individuals in a society are assumed to have rational preferences and try to maximize their profit, advantage, or benefit, and that social interest is an aggregation of individual interests (Dequech 2007; García-Llorente et al. 2011a). Furthermore, the valuation of biodiversity and its ecosystem services differs from the valuation of other goods because of the influence of moral, ethical, or psychological motivations (Hanley and Milne 1996). These values cannot and should not be fully translated into economic terms and have to be complemented or approached using other tools such as sociocultural analysis (TEEB 2010). However, thinking in terms of ecological values, economic valuation dominance implies a risk of being a complexity blinder, hiding the importance of ecosystem processes in the provision of ecosystem services (Norgaard 2010; Sagoff 2011) and having a counterproductive effect on the problem being addressed by favoring the creation of markets that commoditize certain ecosystem services (Gómez-Baggethun and Ruiz-Pérez 2011).

Again, this leads to the necessity of combining or using alternatives to economic valuations with social and biophysical ones.

20.4 Discussion and Future Steps: Toward Hybrid Methodologies and New Concepts

Recent developments in the field include hybrid methodologies that combine nonmonetary and monetary methods or multidimensional methods. One of them is the application of mapping tools in the exercises. On the one hand, this could improve the design and description of multiple economic exercises (Troy and Wilson 2006; Balmford et al. 2011). This is the case of hedonic pricing, travel cost, or choice modeling (e.g., Geoghegan et al. 1997; Brouwer et al. 2010), where GISs permit integration of environmental data and of ecological complexity in the economic exercise. In addition, mapping tools permit us to analyze an ecosystem for the provision of ecosystem services in a sustainable way having also taken into account the social value and demand of those services (Sherrouse et al. 2011; Kroll et al. 2012). The resultant maps provide easily interpretable information that facilitates the analysis of different management options that obtain spatially explicit economic data (Bateman and Jones 2003; Goldstein et al. 2012).

Other examples within the realm of stated preference methods would include the choice modeling without including the monetary or cost attribute (Sayadi et al. 2005) or looking for other options in the payment vehicle that are expressed in terms of willingness to give up time to labors related to biodiversity conservation or to ecosystem service maintenance instead of a conventional willingness to pay (Higuera et al. 2013). This option has been supported by the different ESB profiles; in this way, we avoid the equity problem and also we avoid the assignation of monetary value to things that are considered to be incommensurate with economic values (García-Llorente et al. 2011a). As Cowling et al. (2008) stated "because money is the most commonly used interchangeable commodity, valuation in monetary terms may send the message that a service is more easily replaced by human-manufactured providers than it actually is" (p. 9485). Therefore, the analysis of human values of ecosystem services should be tackled from both the economic (combining monetary and nonmonetary methods) and sociocultural perspectives.

The spatially explicit mapping of ecosystem services available in current toolboxes (e.g., InVEST or ARIES) is based on intensive data requirements, which restricts their application to local scales due to the high costs of surveying the data (Ayanu et al. 2012). In contrast, remote sensing provides data for quantifying and mapping ecosystem services at comparatively low costs, and it offers the possibility of frequent and standardized observations for monitoring (e.g., quantification and mapping of carbon storage at regional or global scales). Currently, most of the lack of knowledge in ecosystem services science relates to quantification and mapping in a spatially explicit way. Hence, remote sensing has the potential to address the basic issues of spatial quantification of ecosystem services. Most of such potential is related to provisioning and regulating ecosystem services such as carbon storage or sequestration (Konarska et al. 2002) or erosion protection (Krysanova et al. 2007). Nowadays, though, the sociocultural and economic dimensions of ecosystem services assessments may also make use of remote sensing products that offer a cost-effective technique for collecting spatial information about services delivered. We only found one study—by Lobitz et al. (2000)—that used remotely sensed indicators of the supply and demand sides to indirectly measure the effects of an infectious disease.

Furthermore, advances in the field have introduced new concepts such as *service benefiting areas* (referring to the demand of ecosystem services from certain areas) and *service connecting areas* (referring to the areas that link ecosystem service supply and demand) (Syrbe and Walz 2012). In this sense, the spatially and explicit quantification of SPUs and service benefiting areas and the importance of linking service providers with ESBs can be considered as opportunities to develop better valuation scenarios for biodiversity conservation (Luck et al. 2003, 2009) and to promote stronger and well-informed valuation applications.

20.5 Conclusions

The assessment of ecosystem services has gained increasing importance among scientists worldwide including the natural, the social, and the economic science perspectives. However, despite the academic progress, many challenges remain to integrate the ecosystem service concept into an operational framework that can be useful for decision making. This chapter presents a review of the current status of assessment of ecosystem services based on the interdisciplinary nature of the methodologies and techniques used today. In particular, we pay attention to the multidimensional nature of ecosystem services (i.e., biophysical, sociocultural, and economic), and we assess them from the supply side as well as the demand side.

Regarding ecosystem service mapping, we provided a list that included some of the common indicators for assessment ecosystem service supply (e.g., precipitation minus evapotranspiration for freshwater provision), and we noted some that might be used to assess ecosystem service demand (e.g., water consumed per surface and time). We also described the main existing toolboxes for mapping ecosystem services (e.g., for example the InVEST toolbox designed by the Natural Capital Project). We also identified the most common indicators derived from remote sensing for services mapping from local to global scale, such as the NDVI index for the quantification of the net primary productivity at multitemporal and spatial scales.

The use of remote sensing data has been mostly utilized for the quantification and mapping of the supply side of provisioning and regulating ecosystem services. Usually, these studies simply characterize the ecological process, such as net primary productivity, without linking it with the potential benefits associated with it, such as food production or climate regulation. Our study reveals the necessity for using a standardized nomenclature of ecosystem services that not only focuses on the ecological process or function but also on the subsequent ecosystem services that the general public perceives as beneficial. Our study also shows that there is still a lack of use of remote sensing data for the assessment of the demand side of ecosystem services. This is probably due to the difficulty in tracking sociocultural aspects, such as social preferences or perceptions, from remotely sensed data. Therefore, further research is needed to provide guidelines that assist in linking remote sensing information with the quantification and mapping of ecosystem services from the supply side with the benefits that society perceives (i.e., the demand side).

Despite the limited number of studies using sociocultural valuation, we described some procedures, including using nonmonetary methods to analyze human preferences. We analyzed some of the reasons why the economic dimension has gained the highest importance in the academic and the political arenas, and we provided some guidelines within the TEV framework about how to decide which valuation method is better for measuring different value types and service categories. In conclusion, we recommend the inclusion of ecosystem service maps evaluated from the biophysical, the sociocultural, and the economic perspectives by combining monetary and nonmonetary methods.

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