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Varela E., Robles-Cruz A.B.

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Ecosystem services and socio-economic benefits of Mediterranean grasslands

E. Varela* and A.B. Robles-Cruz

Center for Agro-food Economy and Development (CREDA-UPC-IRTA)
Edifici ESAB, Parc Mediterrani de la Tecnologia.
C/Esteve Terrades 8 08860 Castelldefels, Barcelona (Spain)
*email:elsa.varela@upc.edu

Abstract. The Ecosystem Services (ES) framework is increasingly being adopted by researchers and practitioners to underline the many services that Mediterranean grasslands provide to society at large, beyond tangible provisioning services. Incorporating the social dimension in the supply and demand of these services implies acknowledging the different preferences and values of stakeholders and the likely spatial-temporal mismatches between providers and beneficiaries. The consideration of grasslands as socio-ecological systems seems crucial to acknowledge the human component needed for their maintenance and hence for the generation of ES. The description of ES provided by Mediterranean grasslands has already been undertaken by several studies; however, their quantification from an economic point of view seems a central issue to raise awareness about the protection and maintenance of these systems. This article presents some key concerns related to the classification of these ES that should be born in mind when attempting to estimate the economic value of these systems, so as not to incur in inconsistencies such as double-counting. Despite the undoubtable benefits of quantification, this article also shows the importance of counting on qualitative assessments of social preferences, especially for services, such as cultural ones, that are difficult to encompass or appraise by standard economic valuation methods.

Keywords. Economic valuation – Double-counting – Social-ecological systems – Qualitative methods.

Les services écosystémiques et les bénéfices socio-économiques des pâturages méditerranéens

Résumé. *Le cadre des Services Écosystémiques (SE) est de plus en plus adopté par les chercheurs et praticiens pour souligner les nombreux services que les pâturages méditerranéens fournissent à la société en général, au-delà des services tangibles d'approvisionnement. Incorporer la dimension sociale dans l'offre et la demande de ces services implique de reconnaître les différentes préférences et valeurs des parties prenantes et les éventuelles discordances spatio-temporelles entre fournisseurs et bénéficiaires. La considération des pâturages en tant que systèmes socio-écologiques semble cruciale pour la reconnaissance de la composante humaine nécessaire à leur maintien et donc pour la création de SE. La description des SE fournis par les pâturages méditerranéens a déjà été entreprise par plusieurs études; cependant leur quantification sous l'angle économique s'avère une question centrale pour sensibiliser quant à la protection et au maintien de ces systèmes. Cet article présente quelques éléments-clés liés à la classification de ces SE dont il convient de tenir compte lorsqu'il s'agit d'estimer la valeur économique de ces systèmes, pour ne pas tomber dans des inconsistances comme le double-comptage. Malgré les bénéfices indubitables de la quantification, cet article montre également l'importance de disposer d'évaluations qualitatives des préférences sociales, en particulier pour les services, tels que ceux d'ordre culturel, qui sont difficiles à aborder ou apprécier par les méthodes standard d'évaluation économique.*

Mots-clés. *Évaluation économique – Double-comptage – Systèmes socio-écologiques – Méthodes qualitatives.*

I – Introduction

Mediterranean grasslands include rangelands, meadows, pastures and fodder crops (Porqueddu *et al.,* 2014). Their existence is tightly linked to extensive grazing, a land use that takes place on areas generally unsuitable for intensive cultivation due to several reasons. They are mainly seen through their provisioning role of food and fibers (Nieto-Romero *et al.*, 2014), although they can provide a multiple array of other services such as erosion control, carbon sequestration, recreational opportunities or cultural identity. In this sense the ecosystem service (ES) approach may help unveiling the contributions of these ecosystems to societal wellbeing.

The qualitative description of the ecosystem services provided by grasslands has to some extent been accomplished, and a significant challenge ahead consists of quantifying them and if possible doing so in monetary units. This quantification is seen as essential to add value to these ecosystems and it is considered a research priority (FAO, 2013).

A crucial aspect when assessing ES of Mediterranean grasslands is their consideration as socio-ecosystems. It highlights the fact that the ecosystem services (ES) they provide are far from being "natural", but on the contrary are tightly linked to human activities (Hutsinger and Oviedo, 2014).

The future societal demands due to increased population and the constraints imposed by climate change in the Mediterranean may increase the pressure on regulating and supporting services to guarantee the flow of provisioning services. Therefore, navigating the trade-offs between provisioning, regulating, cultural, and supporting ecosystem services, as well as maintaining natural capital that is critical to generate future services, is essential for achieving sustainability (Cavender-Bares *et al.*, 2015).

Management of social-ecological systems requires understanding both the biophysical constraints that create trade-offs among ecosystem services and human values to understand the preferences of the stakeholders and the services that contribute to their well-being (Cavender-Bares *et al.*, 2015). In addition, issues such as property rights, thresholds, hysteresis, nonlinear dynamics and resilience of these socio-ecosystems should be incorporated into the ES agenda.

This article revises some basic concepts on the ES terminology and classification. It also provides a review of some studies addressing the social dimension of ES assessment both from a quantitative and qualitative perspective. It also signals some of the caveats in the ES approach and proposes the concepts that should be incorporated in such debate.

II – Ecosystem services as a working framework

1. Introduction: concept definition

The Ecosystem Services (ES) concept has become increasingly popular in the last decades and it is usually employed to emphasize the contributions of ecosystems to human welfare. Although the recognition of the capacity of natural systems to provide benefits to society was already present, the concept of ES provides a framework where the contribution of ecosystems to societal wellbeing is highlighted.

Furthermore, this approach calls for a more fundamental multidisciplinary focus, promoting a dialogue between biology and economics (Lele *et al.*, 2013) by considering both the ecological production and the economic value (Bauer and Johnston, 2013). It allows to distinguishing the contribution of benefits to society supplied by ecosystems from those provided by human capital or labour (Bateman *et al.,* 2011; Brown *et al.*, 2007), offering a framework to link changes in ecosystem processes and outputs to its effects on social welfare.

The most popular and widespread definition of ES was that provided by the Millennium Ecosystem Assessment (MEA, 2005) where ES are defined as the outputs of natural systems that benefit society. However, in the last years, a number of authors have tried to produce a more refined set of working definitions that allow a quantification and mapping of ES in a consistent way, for example avoiding double counting or highlighting where the beneficiaries of a given ES may be.

The review and blueprint proposal for ES assessment in Crossman *et al.* (2013) provides with a comprehensive set of definitions compiled from a number of authors. In this study ES are defined as the contributions of ecosystem structure and function –in combination with other inputs- to human wellbeing (Burkhard *et al*., 2012). A key difference is established between intermediate and final ES. Intermediate ES comprise the ecosystem processes (all the changes and reactions occurring in the ecosystems and includes physical, chemical or biological processes) (MEA 2005) and also the ecosystem functions that give the capacity to the ecosystems to provide services that satisfy human needs, such as pollination, water purification or carbon sequestration. Final ES are the direct contributions to human well-being, such as clean water provision, storm protection or harvest production (Fisher *et al.,* 2009). To transform these final ES into benefits for society typically other forms of capital are required (such as labor or produced assets (e.g. to consume fruits and vegetables or to make water available at domestic level).

Under this view services and benefits are not the same and hence some authors consider the valuation of ecosystem services alone (Fisher *et al.,* 2009; Boyd and Banzhaf, 2007), while others are in favor of defending the valuation of both ES and benefits as separate elements (Wallace**,** 2007).

In an economic sense, ES are distinguished from other ecosystem functions in that there are beneficiaries willing to pay for the use or preservation of those scarce services (Chan *et al*., 2006; Caparrós, 2012). Final outputs are traded in markets or consumed by society as they are, and are usually the focus of economic analysis, while intermediate outputs are used to create final outputs (Caparrós *et al*., 2012).

This distinction is crucial so as to avoid double-counting. In the case of grasslands, if the farmer sells a grass-fed sheep, the final output (i.e. the benefit), is the sheep sold, while the grass intake by the animal is the intermediate input or service. Grass clearly has a value that can be quantified. Hence its value should not be computed twice, once as the ecosystem service ‘‘grazing’’ and a second time as a part of the final benefit, the sheep. Defining an economic value and establishing the methods for valuing ecosystem services requires a precise definition of those services as final or intermediate outputs.

To sum up, the establishment of a more accurate definition allows to highlight that not everything can be considered an ES and that the services are often benefit-dependent, so that the benefits we’re interested in assessing will dictate what we understand as ES.

2. Ecosystem services classifications

The MEA (2005) has achieved perhaps the greatest scientific consensus in the recent years with respect to providing a classification of ES into four main classes:

- Provisioning services: are the products people obtain from ecosystems, such as food, fuel, fiber, fresh water, and genetic resources.
- Regulating services are the benefits people obtain from the regulation of ecosystem processes, such as air quality maintenance, erosion control or water purification.
- Cultural services are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, recreation, and aesthetic experiences.
- Supporting services are those that are necessary for the production of all other ecosystem services, such as primary production, and soil formation.

The perspective we adopt and the classification is crucial, so as to differentiate intermediate ecosystem functions from final ecosystem services and avoid double counting (e.g., Boyd and Banzhaf, 2007; Brown *et al.,* 2007; Fisher *et al.,* 2009; Wallace, 2007)

Because accounting for supporting services may lead us to incur in double counting, some classifications do not consider supporting services as finals services or outputs, but as intermediate processes and functions. Such is the case of the Common International Standard for Ecosystem Services (CICES) (2015). The focus of the CICES on environmental accounting and the risk of incurring in double counting, led to this classification to recognize these outputs from ecosystems to be provisioning, regulating and cultural services. Supporting services are treated as part of the underlying structures, process and functions that characterize ecosystems.

3. Challenges faced by this approach

Describing, quantifying and mapping ES is nowadays challenging due to the scarcity of information or the need to reframe existing data into the ES framework. We still have relatively little understanding of the ecology behind the provision of ecosystem services (Kremen and Ostfeld 2005). The number of studies assessing the services provided by different ecosystems is increasing; however, most services are assessed individually and only in a few instances studies deal with interactions among more than two services (MEA, 2005). This raises a concern on the fact that the emphasis is put on optimizing a small number of ES, which may jeopardize environmental sustainability (Plieninger *et al.* 2015).

Taking into account several services simultaneously brings the difficulty of dealing with trade-offs (Isselstein 2014). The trade-off concept in the context of agrarian systems has been effective to account for the negative effects of production (i.e. when a provisioning service is maximized) and typically affects regulating services or create what is known as disservices (Power, 2010; Sanderson and Wätzold, 2010). As Bennett *et al.* (2005) highlight, it is very frequent to focus on a very narrow set of ES, and try to maximize them, incurring in unexpected trade-offs or sudden declines in other ES.

Provisioning services have been more profusely studied, which are widely recognized as essential for meeting human needs for nutrition, shelter, and safety. Regulating services are more complex but have been brought to public attention by discussions of climate change and recent natural disasters. Supporting services are fundamental to all other services, but their relationship to human needs can be indirect and complex. In contrast, most cultural services are directly experienced and intuitively appreciated, often helping to raise public support for protecting ecosystems (Daniel *et al.*, 2012).

However, quantification of the actual flow and use of these services is a key challenge; without quantification, the value of most services is not easily understood. Quantification has to deal with the spatial mismatch of ecosystems that provide value and the people that enjoy the services, as sometimes these two sides of the chain may be distant from each other. Bagstad *et al.* (2013) among other authors have developed models to link ES with their beneficiaries, highlighting the spatial connectivity between both ends.

Economic quantification of ES is shaped by two economic assumptions (Winthrop, 2014): The first of them is a stock-flow model that assumes a stock of natural capital from which a flow of ES, similar to interest or dividends (de Groot *et al*,.2010), links the ecological and economic systems (Norgaard, 2010). On second place, if we intend to estimate environmental values, these are understood as an aggregation of citizens' willingness-to-pay. These two assumptions may be too simplistic when dealing with complex interactions in the ecosystems, thresholds, hysteresis and non-linearities, which are not taken into account by these previous assumptions rooted on the neoclassical economics.

Furthermore, the adoption of a quantitative approach may not be well suited when it comes to cultural ecosystem services. Hence a qualitative approach may be need for a good characterization of these. Often considered secondary to financial concerns, cultural services can have critical influence on landowner decisions and subsequently on efforts to manage privately owned land. Cultural ecosystem services also motivate rural and urban residents to engage with public or community land. Cultural ecosystem services can also inform landscape planning (Albert *et al.*, 2014).

Last but not least, some attention has to be placed on the ES term. It implies that ESs are a function of natural processes but in many cases human interaction may be key to the production of many of them (Hutsinger and Oviedo, 2014).This risk should especially be born in mind when addressing ecosystems such as grasslands. To varying degrees these ecosystems have been modified or even created by human activity. Hence, problems arise when the human activity is disregarded as a factor in the cogeneration of these ecosystems, either because it is perceived as harmful or because it is not taken into account as a driver. As Hutsinger and Oviedo (2014) propose, thinking of such services as “social-ecological” services can reinforce the importance of human culture, perspectives, and economies to the production of ES and change the conception that rather than thinking of something coming from an ecosystem has cultural value, we indicate that cultural activities cogenerated the service.

III – Mediterranean grasslands and ecosystem services

1. Key features and dynamics of Mediterranean grasslands

According to Peeters *et al.* (2014) grasslands correspond to land devoted to the production of forage for harvest by grazing/browsing, cutting, or both, or used for other agricultural purposes such as renewable energy production. The vegetation can include grasses, grass-like plants, legumes and other forbs; woody species may also be present. This definition encompasses the temporary or permanent character of grasslands and also makes a distinction between meadows and pastures, depending on whether they have been harvested by mowing or by grazing, respectively.

In the Mediterranean, livestock use is in general dominated by sheep and goats (Porqueddu *et al.,* 2014) able to profit from these type of pastures. A key feature of Mediterranean grasslands is the complementarity of herbaceous species with a woody component and with agricultural systems (e.g. stubble grazing) that provides stability to the grazing activity.

Most of them to a greater or lesser extent can be defined as “semi-natural” grasslands, since they have been created and maintained by human activities although their plant communities are natural. Hence the continuation of cutting, and more commonly grazing activities are crucial for the protection of the species they harbor.

Livestock grazing is the key element in these ecosystems (Cosentino *et al.,* 2014) for its dual role in maintaining these ecosystems and strongly driving its dynamics (Perevolotsky, 2005). This calls for their consideration as socio-ecological systems acknowledging that pastoralism is a culture that cogenerates the services. This consideration also helps to understand that the distinctly different socio-economic and political factors in both parts of the basin, have also played a significant role in forming the structure of agricultural practices (Aw *et al.,* 2010). Therefore, conservation efforts should also recognize the need to maintain the human activity to sustain the services (Hutsinger and Oviedo, 2014).

Two main socio economic drivers have played a key role in shaping Mediterranean grasslands and agroecosystems in general: rural abandonment of mountainous and less productive areas, particularly on the northern fringe on the Basin, and land-use intensification of fertile areas where grasslands have been converted into arable land (Bernués *et al.,* 2011).

The reduction in the number of farms in northern Mediterranean countries is associated with two processes depending on the CAP: large increments of herd size and dependency on premiums (Bernués *et al.,* 2011), although at the same time some agro-environmental policies have tried to counteract this process. The abandonment process has brought encroachment of woody vegetation that may in the future produce some environmental benefits (Rey Benayas *et al.,* 2007; Quero *et al.*, 2013). However, at the landscape scale, the loss of mosaic structure and increased homogenization makes these transitional landscapes quite fire prone (Moreira *et al.,* 2011). In absence of appropriate landscape management, it will be the fire suppression capacities that will configure the future landscapes (Regos *et al.*, 2014).

The situation in southern Mediterranean regions shows an increasing demand for animal products derived from the rapid increase in human populations. The contribution of grassland to livestock feed has gradually decreased from 80% in the 1980s to <30%, increasing the feed intake of grains (Ryan *et al*., 2008), while production systems based on extensive grazing are concentrated in arid areas and suffer an increasing pressure (Le Houerou, 2000; Porqueddu *et al*., 2014).

Climate change is expected to have a high impact in the Mediterranean basin due to increase in temperature and inter-annual variability (IPCC 2014). Changes in these ecosystems are related to carbon stocks and pasture productivity, affecting the length of the grass growing season and hence the forage quality and quantity and indirectly through livestock disease increase (IPCC, 2014). Therefore, addressing the adaptation of grassland ecosystems to these changes in order to identify the species and dynamics that can better cope with these changes, to increase their resilience and improve the adaptation capacities seems crucial. However, socio-economic changes are expected to have a still greater effect on mitigation and adaptation potentials (Schmidhuber and Tubiello, 2007); among those, pressure on livestock production systems is expected to increase together with competition on grains between animal and human feeding (Ates *et al*., 2014).

2. Ecosystem services provided by Mediterranean grasslands

Applying the ES framework to grassland and livestock farming systems may help in considering regulating, supporting and cultural ES that these ecosystems provide to society and also to integrate those at the same level with provisioning ES, which is the dimension that so far seems to be more studied. This approach may also contribute to the assessment of the multiple trade-offs and synergies that exist between ES, allowing for a better integration of agricultural policies in other sectors (Rodríguez-Ortega *et al*., 2014).

Due to the fact that grassland management is quite diverse, the flow of ES may greatly differ from one system to the other, depending among other factors on the intensity of the production systems. Hence services and disservices can take place along a gradient of intensity in the use of the resources.

Grasslands are mainly acknowledged for their provisioning services; this is the most prominent service and has motivated their existence. Furthermore, provisioning services (i.e. grazing) produce a series of benefits, such as meat or milk, which are private market goods appraised by the famer. Many times, these farm products have special sensorial and nutritive qualities linked frequently to labels such as products with Denomination of Origin. Hence, beyond this provisioning dimension, these products may also raise values linked to cultural heritage for consumers (Zander *et al*., 2009). This cultural value may play an important role in connecting rural and urban populations.

Hence, the provisioning services provided by these spaces that have a clear private components in terms of benefits for the farmer that appraises them, are intertwined with cultural and heritage values linked to these traditional breeds and also with reduced impacts on regulating services. The later services have features of public and semi-public goods and hence are not appraised by the farmer and internalized in the system.

Managed grasslands are usually ecosystems with high species diversity (Ribeiro *et al*., 2014). Biodiversity as a whole is usually considered as a supporting service as it enables the ecosystem processes and functions needed to deliver non-supporting services. The ecosystem properties that underlie ecosystem services depend largely on biodiversity and especially on functional diversity (the presence or abundance of particular functional groups or functional traits) rather than on species number (Hooper *et al*., 2005; Le Roux *et al.,* 2008).

In particular, a growing knowledge on plant functional traits (e.g. leaf dry matter content, vegetative height and date of flowering onset) is making it possible to quantify ecosystem services based on responses of functional traits to environmental change and/or effects on ecosystem properties (Diaz *et al.* 2007; Lamarque *et al.* 2011).

Grasslands provide regulating services storing important carbon stocks. Cultivation and urbanization of grasslands, desertification or overgrazing can be significant sources of carbon emissions. Grasslands can act as carbon sinks, although under certain climate conditions (drought or heat waves) they can switch from carbon sinks to carbon sources (Freibauer *et al.,* 2004). Most of the soil organic carbon content of grasslands is not in the biomass, but in the soil as a large part of the grassland biomass production is located in the rot biomass, unlike many arable production system (Huyghe *et al.,* 2014). However, this has been little studied in Mediterranean grasslands and hence coordinated experiments in different Mediterranean regions are required to quantify the carbon sequestration contribution of natural and semi-natural grasslands, as well as the contributions of key pasture species (Porqueddu *et al.*, 2014).

Soil erosion is a severe problem in Mediterranean countries. Grasslands as a permanent soil cover reduce the likelihood of soil losses (Schnabel *et al*., 2009). On the other hand, overgrazing by livestock is considered one of the causes of soil erosion in the Mediterranean (Papanastasis, 1998) and hence a disservice.

Grasslands contribute to maintain the openness of Mediterranean cultural landscapes characterize by its mosaic-like configuration (Farina, 2008). Considering the contribution of these areas to cultural services such as improving the aesthetic experience and recreational opportunities is difficult to elucidate. Aesthetic preferences are highly subjective and incorporate social constructs (Rodríguez-Ortega *et al*., 2014). However, diverse studies show that people tend to prefer diverse landscapes, and openness is a key feature in determining their preferences (Sayadi *et al*., 2005). The maintenance of open spaces with low biomass content in the landscape is key for reducing fire risk at the landscape scale (Ruiz-Mirazo *et al.,* 2011) and increase suppression opportunities.

IV – Operationalizing the ES concept: socio-economic benefits

Some of the major barriers to effective resource planning arise because different stakeholder groups hold different preferences for services (Martín-López *et al*., 2012), and these differ in their spatial or temporal patterns of benefits and costs (Laterra *et al.,* 2012). Hence, we need a variety of tools derived from the social sciences to appraise social preferences, needs, values, norms, behaviors of stakeholders and individuals, institutions and organizations towards ES (Cowling *et al.,* 2008). Addressing the social dimension of ES means tackling the demand of ESs together with its supply, rather than focusing on the supply side alone (Termorshuizen and Opdam, 2009).

Methods in economics allow to identify issues on property rights and assess the willingness of stakeholders to trade-off ES from a quantitative perspective, attaching a monetary value to these trade-offs.

Typically pasture-based LFS suffer displacement by other economic activities. From a strictly financial dimension, abandonment or land use change seem as more profitable options rather than maintaining the flow of services these systems provide to society. When the biodiveristy/genetic resources conservation generates economic values that are not captured in the market place, it generates a distortion where the incentives are against genetic resources conservation and in favour of the economic activities that erode such resources (Pearce and Moran, 1994). In fact, failing to account for these non-market values (such as future option values or socio-cultural values) works against the sustainability of the system.

This situation is related to non-enforceable property rights related not to the property of the land, but uttermost to the property over the services and the benefits derived for people. Most environmental services fall under the economic category of pure public goods or open access/common goods. A distinctive characteristic of a pure public good is that consumers do not have the option for not consuming it, e.g. carbon sequestration. The reason for the under provision of public goods is that the owner or provider cannot appropriate the full benefits. These public goods as positive externalities derived from the management of these ecosystems. However, the market system fails to 'price' this interdependence, as a result of which the affected party is uncompensated. Reasons for it are a lack of or weak property rights. At the European level agricultural systems fail to provide the services that society as a whole is demanding (Cooper *et al.*, 2009).

In this context valuing these ES, that is quantifying them in monetary units, is emerging as a framework within which policies targeted to halting the degradation of the natural environment are developed. While ecological models define the relationships and trade-offs among services that represents an "efficiency frontier", these, together with methods in economics that combine preferences that define the willingness of stakeholders to trade off ecosystem services on the efficiency frontiers, illuminate desirable outcomes that meet human needs and secure sustainability of the system (Cavender-Bares *et al.*, 2015).

1. Quantitative methods to assess the demand (and supply) of ES

The Total Economic Value (TEV) is an analytical framework used in economic valuation to link ecosystem process and functions with the benefits it provides for society, which can be assigned monetary economic values. The concept of TEV has been developed as a guarantee that the benefits are considered systematically and comprehensively, without any double counting. The TEV is the sum of use values and non-use values. Use values are further broken down as follows: 1. Direct use value, includes interaction with the ecosystem through consumptive use such as the harvesting of crops, or may be non-consumptive such as recreational activities.2. Indirect use value, derived from ecosystem services, such as cleaner water to downstream users, carbon sequestration, and flood control or erosion prevention. 3. Option value: considers having the option of using the resource in the future, directly or indirectly. Among non-use values we can distinguish: 1. Altruistic value: is derived from the satisfaction of knowing that other people have access to benefits of the farming system provides. 2. Bequest value: arises from the interest in preserving a certain ecosystem or species for future generations. 3. Existence value: is derived from the knowledge of the existence of a particular ecosystem or species.

Despite that use values are prevailing in the agricultural sector and farming systems, there is an increasing social demand for non-use values provided by agricultural landscapes and precisely these give name to the so-called multifunctionality of agricultural systems. Typically use values are observable in the market (e.g. meat price) or in surrogate markets (e.g. recreational value of a landscape through the estimation of the expenses visitors incurred in). However, values linked to non-use components need of economic valuation methodologies to be estimated, usually through surveys where people's welfare from preserving or enhancing the grassland ecosystem, for example, is measured as their willingness-to-pay for such an enhancement.

The following are some examples of valuation studies conducted to assess the non-market values linked to agrarian ecosystems. Bernués *et al*. (2014) assess the TEV of key ES of a Mediterranean mountain agroecosystem, most of them extensive grazing ecosystems. They conducted a survey to assess preferences for landscape changes (towards abandonment or encroachment), threatened species (bearded vulture), occurrence of fire events and quality products linked to the territory. In their study they also show how local population holds different preferences when compared to these of regional citizens who will not be directly affected by landscape management measures with local population more concerned about the ES related directly to their farming activity and regular citizens showing a more general concern. Hasund *et al.* (2010) conducted a survey to estimate the willingness-to-pay of the population for different types of elements and other environmental qualities of the agricultural landscape. This CE is designed to estimate the marginal values of 8 grassland types, 10 types of field elements and 9 agri-environmental-quality attributes. The attributes were carefully selected to be applicable as criteria for agrienvironmental payments. Their results show that people are willing to pay to preserve field elements, and value elements having more biodiversity, visual or cultural heritage interest considerably higher than those lacking these attributes, being the oak-wooded grassland the most highly valued landscape. Varela *et al.* (2014) conducted a survey in southern Spain that shows that the population values positively the fuel break maintenance by controlled grazing over traditional heavy machinery methods to control biomass content. Zander *et al.* (2013) assess the TEV of conserving two local Italian cattle breeds. The non-market values accounted by the conservation of these species accrue around 80% of the value of these species, including landscape maintenance, existence and future option values; the positive direct use values (market values) account for 20% of this TEV and are linked to product markets. Finally, economic valuation methods can also be employed to assess the willingness to accept of farmers, that is the supply side, and hence provide with useful information for the set-up of payment schemes that consider additionality among other issues (e.g. Vedel *et al.,* 2015).

Despite a vast array of valuation studies have been conducted to assess the non-market benefits of agroecosystems, of which we have just mentioned some examples, using these methodologies from the ES perspective still is challenging. The ecosystem service perspective seeks to distinguish benefits provided by natural ecosystems from those provided by human capital, labor, and technology (Bateman *et al.,* 2011, Brown *et al.,* 2007, Johnston and Russell, 2011). However, such differentiation is not needed in non-market valuation.

2. Qualitative methods to assess the demand of ES

Despite the appeal of quantitative studies, these may not be so well suited to grasp some dimensions of ES, such as the cultural or heritage dimension. Qualitative methodologies may then be needed to assess these dimensions from a non-monetary perspective. Socio-cultural valuation methods to elucidate social benefits include consultative and participatory methods, and even deliberative participatory valuation (Christie *et al.,* 2012).

Rodríguez-Ortega *et al.* (2014) provides a thorough review of studies on ES assessment in grassland ecosystems. Examples of these studies include the surveys conducted by Deraka *et al.* (2014) and Palomo *et al*. (2013) to quantify cultural services and the importance stakeholders allocate to different services, employing multicriteria participatory assessment and participatory mapping respectively. Lamarque *et al*. (2011) used qualitative surveys to check the ES that stakeholders identify (which ES for whom), the relative rankings of these ecosystem services, and how stakeholders perceive the provision of these ecosystem services to be related to agricultural activities. They identified a common set of ecosystem services that were considered important by stakeholders across the three regions, including soil stability, water quantity and quality, forage quality, conservation of botanical diversity, aesthetics and recreation (for regional experts), and forage quantity and aesthetic (for local farmers). They also observed contrasting representations of the relationships between soil fertility and diversity. Similarly, Plieninger *et al.* (2012) showed how residential owners and producers are concerned with different bundles of ES. Understating the perception of the service providers/managers is essential in the development of efficient policies. Oteros-Rozas *et al*. (2012 and 2013) identified the ES related to transhumance in Spain by carrying out socio-cultural assessments. The most important ES for social well-being were fire prevention, air purification and livestock production. Bernués *et al.* (2013) used focus groups to quantify the importance that farmers and non-farmers attached to the ES delivered by mountain agriculture. They found that aesthetics (landscape/vegetation), gene pool protection (biodiversity maintenance) and natural hazard prevention (forest fires), together with opportunities for recreation and culture, were the most important ES delivered by mountain livestock systems. This information was used on a later stage to conduct a valuation survey.

Therefore, when the stakeholders are considered an extra level of complexity is added, as the simultaneous provision of ES and trade-offs when addressed rom the stakeholder perspective means that different stakeholders pursue different goals on a given landscape. As a consequence, they need to develop a common view on problem and collectively design solutions. Methodologies such as fuzzy cognitive mapping (Kok, 2009) may help in involving stakeholders in the exploration and design of common solutions. Some EU projects are incorporating stakeholders' views in land-use planning and adaptation, combining expert and stakeholder views to develop future land use and water management scenarios (e.g. Volante and Bewater projects).

3. Consideration of the farm level

Consideration of social demand and stakeholders preferences has to come hand in hand with viability assessment at farm level. On-farm working condition (FWC) plays a substantial role in dropping number of farm in pasture-based livestock farming systems (Bernuès *et al.,* 2011).This working conditions may relate to monetary and also non-monetary concerns.

The big challenge in this situation is how farmers achieve their personal household/survival/production goals while maintaining and improving the resource base upon which they depend and the wider functions of their grassland ecosystem that the world demands. In the open markets where livestock producers have to trade, they act typically as price takers, meaning that reducing input costs is the main mechanism they can use to remain financially viable. And this has to be made under a conservative approach to maintain the resilience of the farm as a system able to cope with global changes.

Payments to pastoralists may then be needed to support wider environmental goals, as otherwise the income generated through livestock breeding has to cover the cost of maintaining the benefits and also the ecosystem processes and functions that deliver non-marketed services.

4. Frameworks for integration

The integration of all these components in frameworks and modelling approaches is evolving. There is a growing array of tools for analyzing how alternative ecosystem management interventions generate trade-offs in the provision of different ecosystem services (e.g. Raudsepp-Hearne *et al.*,2010; Kareiva *et al.,* 2007).

Assessment of trade-offs among services and the implications these trade-offs have for social well-being have been based on assessing projected changes in land use/land cover (e.g., White *et al*., 2012) or combined land-use/land-cover and climate change (e.g., Bateman *et al.,* 2013).

However, it is still scarce the development of analytical frameworks that encompass ecological mechanisms underpinning ecosystem services, biophysical trade-offs, preferences of stakeholders and system dynamics to account for the evolution through time is growing; such assessments can be linked explicitly to spatial information on service supply to show who benefits and who bears costs with changes in the bundle of services (Cavender-Bares *et al.,* 2015).

5. Policy implications

In broad terms, socioecological systems able to deliver a multiplicity of services beyond provisioning marketed services largely coincide with low agricultural inputs, low stocking densities and often labor-intensive management practices. Particularly important are the small-scale farming systems that are responsible for creating and maintaining the species-rich semi-natural grasslands, which are often true hot spots for biodiversity (EU, 2008).

Overall, the focus of the agricultural policy design for these systems should not focus only on provisioning ES, because this can result in decisions that reduce the TEV of the system (Bateman *et al*., 2013).

However, European policies are contradictory with grassland socio-ecosystems; some parts of it acknowledge the relevance of these systems, but on the other hand such policy framework that hinder their existence (Beaufoy and Poux, 2014). Examples of it are the CAP payments versus the EU biodiversity targets that support the maintenance of many semi-natural permanent pastures in farmlands within the Natura 2000 European network of protected areas and the maintenance, enhancement and restoration of ecosystem services (EC, 2011).

The consideration of these socioecological systems as a whole, beyond segregated conservation strategies, including a full recognition of silvopastoral systems (Moreno *et al*., 2014) is crucial to set coherent policy frameworks. Unfortunately, in the 2014–2020 CAP, the definition of permanent grasslands is far too restrictive; CAP subsidies favor open pastures (based only on herbaceous forage plants) in preference to silvopastoral systems, in which the presence of trees and shrubs frequently produces a reduction in the subsidy received by farmers (Porqueddu *et al*., 2014). This policy could seriously compromise the long-term persistence of many European silvopastoral systems, such as the Iberian dehesas and other traditionally grazed woodland pastures and shrublands (future for wood pastures).

Finally, Agri-Environmental Schemes (AESs) are currently designed to deliver improved biodiversity, among other things, but are not explicitly linked with ecosystem services. While many of the reported relationships between ecosystem services and biodiversity are based on sensible predictions,. Although biodiversity can in itself provide a range of 'cultural' ecosystem services (most of which are likely to be hard to value economically), what is urgently needed is an evidence-base on which to move forward (Whittingham, 2001), bearing in mind that it is farmers who implement AESs, and so it is crucial that such stakeholders are included in the design of the schemes. One option is to design incentive schemes for bundles of multiple ecosystem services (Martín-López *et al*., 2012)

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