

Chapter 14

Rangeland Ecosystem Services: Nature's Supply and Humans' Demand

Oswaldo E. Sala, Laura Yahdjian, Kris Havstad, and Martín R. Aguiar

Abstract Ecosystem services are the benefits that society receives from nature, including the regulation of climate, the pollination of crops, the provisioning of intellectual inspiration and recreational environment, as well as many essential goods such as food, fiber, and wood. Rangeland ecosystem services are often valued differently by different stakeholders interested in livestock production, water quality and quantity, biodiversity conservation, or carbon sequestration. The supply of ecosystem services depends on biophysical conditions and land-use history, and their availability is assessed using surveys of soils, plants, and animals. The demand for ecosystem services depends on educational level, income, and location of residence of social beneficiaries. The demand can be assessed through stakeholder interviews, questionnaires, and surveys. Rangeland management affects the supply of different ecosystem services by producing interactions among them. Trade-offs result when an increase in one service is associated with a decline in another, and win–win situations occur when an increase in one service is associated with an increase in other services. This chapter provides a conceptual framework in which range management decisions are seen as a challenge of reconciling supply and demand of ecosystem services.

Keywords Provisioning • Regulation • Cultural • Trade-offs • Win–win • Stakeholders • Human well-being

O.E. Sala (✉)
School of Life Sciences and School of Sustainability, Arizona State University,
Tempe, AZ, USA
e-mail: Oswaldo.Sala@asu.edu

L. Yahdjian • M.R. Aguiar
Faculty of Agronomy, Institute for Agricultural Plant Physiology and Ecology (IFEVA),
University of Buenos Aires and CONICET, Buenos Aires, Argentina

K. Havstad
USDA-ARS, Jornada Experimental Range, New Mexico State University,
Las Cruces, NM, USA

14.1 Introduction

Ecosystem services are the benefits that society receives from nature (Daily 1997; MA 2005). They include the provisioning of food, wood and medicinal resources, and services that contribute to climate stability, control of agricultural pests, and purification of air and water (Fig. 14.1). Ecosystem services are broadly classified in four different categories: provisioning, regulating, cultural, and supporting (MA 2005). Provisioning ecosystem services include the contribution of essential goods such as food, fiber, and medicinal. Regulating ecosystem services include carbon sequestration, prevention of soil erosion, and natural flood control. Cultural ecosystem services include intellectual, inspirational, and recreational activities. The fourth category is supporting ecosystem services, which include services that are dependent on ecological processes such as primary production and nutrient cycling and that are intimately related to biological diversity.

Since its conceptualization, the focus of ecosystem services has changed from the description of the processes involved in delivery of a single service at a point in time (Daily 1997) to approaches for analyzing the capacity of nature to produce multiple ecosystem services. The next steps have been assessing multiple ecosystem services

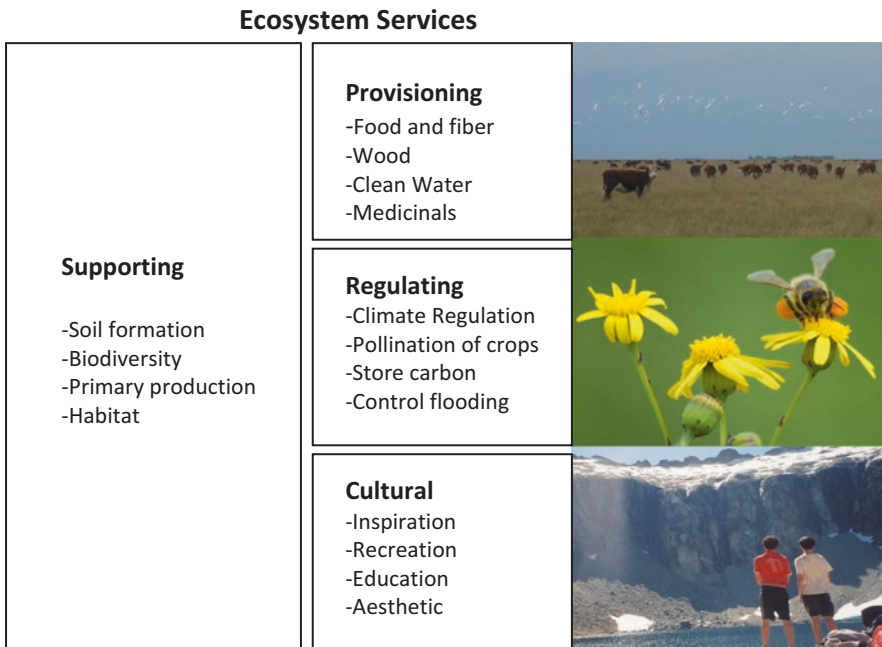


Fig. 14.1 Four categories of ecosystem services as classified by the Millennium Ecosystem Assessment (MA 2005). Photo credits (from top): Laura Yahdjian, Magdalena Druille, Felipe Cabrera

under alternative land-use regimes (Foley et al. 2005). Management aimed at increasing the supply of one specific ecosystem service may increase or decrease the supply of others creating trade-offs and win-win situations, respectively.

Rangelands, the land on which the potential native vegetation is predominately grasses, grasslike plants, forbs, or shrubs (Kauffman and Pyke 2001; Chap. 1, this volume), encompass hot and cold deserts, grasslands, savannas, and woodlands. They occupy approximately 54 % of terrestrial ecosystems, and they sustain 30 % of world population, including a myriad of stakeholders (Reynolds et al. 2007; Estell et al. 2012). Rangelands produce a great variety of ecosystem services but only few of them have market value (Sala and Paruelo 1997). For example, commodities produced by rangelands such as meat or wool have market value but other ecosystem services such as regulating, cultural, and supporting services mostly do not have a market value although it is possible to estimate it indirectly.

The science of ecosystem services has grown exponentially as indicated by the number of academic publications per year on this topic, from a few per year in the 1980s to a wealth of papers in the last decade (Fig. 14.2). The total number of peer-reviewed publications addressing ecosystem services exceeded 1200 and entire books and journals have been devoted to this topic. The Millennium Ecosystem Assessment was developed around this concept showing the enormous impact of the ecosystem

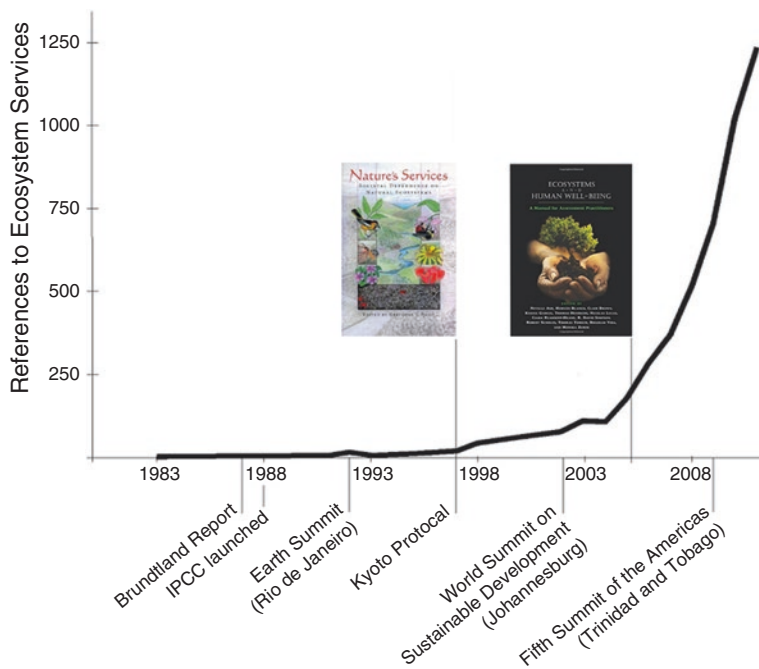


Fig. 14.2 Number of scientific publications emphasizing ecosystem services during the period 1963–2012 (modified from Rositano et al. 2012)

service conceptual framework (MA 2005). The most common approach to study ecosystem services has been related to the assessment of the capacity of ecosystems to deliver services, i.e., the supply of ecosystem services. Ecological production functions define how the spatial distribution of ecosystem structure and functioning determine the delivery of ecosystem services (Daily 1997). Even when ecosystem services have been defined as the outcome of ecosystem processes desired by people, the main focus of ecosystem service research has been to identify the potential of a region to produce ecosystem services independently of whether people are demanding them or not. In addition, the economic valuation of ecosystem services has also been a frequent target of research (Costanza et al. 1997; Gomez-Baggethun et al. 2010). In summary, the science of ecosystem services has developed rapidly in the past decades, but it has focused primarily on the supply of services and has largely overlooked the human demand for ecosystem services only until recently (Yahdjian et al. 2015).

Human demand represents the other side of an ecosystem service equation of supply and demand, which is related to the social beneficiaries. Human consumption of resources and utilization of services that are supplied by ecosystems depend upon both their capacity to produce them and the societal value and need placed on those resources and services (Tallis and Polasky 2011). Demand for ecosystem services changes among stakeholders or social beneficiaries, who are the individuals or groups of individuals who have an interest in ecosystem services because they get a profit from them and could have an active or passive influence on their delivery (Lamarque et al. 2011). Stakeholders not only exhibit different demands, but they also have different valuations of various ecosystem services. Indeed, an ecosystem service is not a universally applicable physical phenomenon, but one whose value is shaped by its users. Sustainable land management depends on reconciling supply and demand for ecosystem services by different stakeholders.

Rangelands are ideal for analyzing the balance between supply and demand for different types of services because of the variety of ecosystem services that they provide and the diverse suite of stakeholders interested in different services. In contrast, hyperarid ecosystems provide supporting, cultural, and regulating services but few provisioning services. Similarly humid ecosystems are generally transformed into crop- and wood-production systems, or are subject to human commercial, residential, and industrial development at the expense of cultural, provisioning, and regulating services. In addition, rangelands are broadly threatened by land degradation and climate change (Herrick et al. 2013). In general, rangelands produce abundant ecosystem services in quantity and variety, but the large value and threat of degradation contrast with the fact that humans usually assign small value to them, particularly when compared with tropical or temperate forests (Martin-Lopez et al. 2012).

The transformation of rangeland ecosystems into croplands is constrained by biophysical conditions and economic feasibility (Havstad et al. 2007). For example, mesic rangelands have been converted to agriculture land, while arid and semiarid rangelands continue to be used as grazing lands, with investments in domestic animals, veterinary and reproductive management, fences, and water points that in combination result in a significant increase in livestock production (Oosterheld et al. 1992).

In this chapter, we describe the (1) main ecosystem services provided by rangelands and the major categories of social beneficiaries and (2) most common methods used to estimate supply and demand of these services, and (3) analyze the determinants of supply and demand of ecosystem services and discuss the existence of trade-offs and win-win conditions in the provision of services. Finally, we provide a new conceptual framework for the management of rangelands that is based on reconciling supply and demand of ecosystem services. This framework is dependent on place, time, and the specific valuation that each stakeholder has of specific ecosystem service. The framework recognizes that both supply and demand of ecosystem services change in space and time and are strongly influenced by land management decisions.

14.2 Categories of Rangeland Ecosystem Services

In this section, we describe the main ecosystem services provided by rangelands in each of the four categories of ecosystem services as defined by the Millennium Assessment (MA 2005). We then analyze the balance between supply and demands for each type of ecosystem service.

Provisioning services are the products obtained from ecosystems that can be directly harvested, and, in general, have a market value such as food, fiber, fuel, and freshwater. The main goods produced in rangelands are freshwater for drinking and irrigation; forage to produce meat, milk, wool, and leather; and medicinal products (Sala and Paruelo 1997). What frequently drives the demand for provisioning services is the immediate need of humans for particular plant or animal species, including production of desirable forage species and the harvest of wild game (Perrings et al. 2011). The relationship between supply and demand for these products changes among regions and among the specific provisioning services. At a global scale, the demand for provisioning services in rangelands is higher than the supply, but at local scales, supply may exceed demands (Yahdjian et al. 2015). In the case of water for irrigation, water is required during specific periods when water is scarce, so supply and demand may be spatially or temporally disconnected, which is particularly important since most rangelands are water limited. As such, for provisioning services, the demand surpasses the supply, which is particularly evident for freshwater and food (Yahdjian et al. 2015). The supply of provisioning ecosystem services changes in time at different scales. The supply of meat and wool fluctuates with seasons and production systems, but also changes at decadal timescales as a result of land degradation and market fluctuations (Texeira and Paruelo 2006). Supply of provisioning ecosystem services changes in space over multiple scales, from differences among locations within a specific community to variation along regional precipitation gradients (Adler et al. 2005). Finally, the demand for provisioning services changes among beneficiaries depending on their income, education, and urban versus rural residence (Yahdjian et al. 2015).

Regulating services are the benefits humans receive from regulating ecosystem processes, such as climate regulation, air quality maintenance, water purification, and erosion control. Rangelands sequester large quantities of carbon, principally into the soil, and avoid carbon losses to the atmosphere that would occur if rangelands were to be transformed into croplands or severely degraded (Sala and Paruelo 1997). In the case of carbon sequestration, demand is higher than the supply because this process cannot offset actual carbon emissions from human activities (Tallis and Polasky 2011). Every unit of sequestered greenhouse gas emitted will allow us to minimize environmental and economic damage that would have occurred otherwise. The whole world benefits from a unit of carbon sequestration regardless of where it occurs because greenhouse gases thoroughly mix in the global atmosphere. Carbon sequestration in rangelands is important because of the area that rangelands occupy although per unit area carbon storage is lower than other ecosystems, such as wetlands and forests (Reynolds et al. 2007). Not only do rangelands account for a significant fraction of the global carbon cycle, but they also account for most of the interannual variability in the global carbon sink (Ahlström et al. 2015).

Cultural services are the nonmaterial benefits that humans obtain from ecosystems and they include cultural diversity, spiritual, and religious values, knowledge systems, and recreation. They involve consumptive and nonconsumptive services. Cultural services in rangelands are related to human experiences associated with activities such as wild game hunting, traditional lifestyles, and tourist ranching experiences. The demand for cultural services changes according to the region analyzed (Tallis and Polasky 2011) and has changed over time. For example, in the southwestern USA, the Bureau of Land Management, who administers a large fraction of federal lands in the region, reported an increase in the number of visitors to their lands from 20 to 45 M per year for the 2000–2010 period (Yahdjian et al. 2015). Similarly, the National Park Service reported for the same period an annual increase of 15 M visitors from 35 to 50 M per year.

Supporting services are those that are necessary for the production of all other ecosystem services such as processes that maintain biodiversity to produce goods or cycle nutrients (MA 2005). In rangelands, supporting services are primary production, nutrient cycling, conservation of soils, and biodiversity, which represent a large storehouse of genetic, species, and functional diversity. Rangelands represent the natural ecosystem where annual grasses and legumes are most abundant and from where a large fraction of domesticated species originated (Sala and Paruelo 1997). The key to sustaining biodiversity is harmonizing its protection with the delivery of as many other ecosystem services as possible. Land degradation, which in most cases results from overgrazing, weed invasions, energy extraction, and exurban development, directly affects the provision of supporting services. Arguably, rangeland degradation has a larger and more imminent impact than climate change on the ability of these systems to fulfill human needs (Herrick et al. 2013). At the global scale, the supply of supporting services is higher than the demand, but human use does not directly apply since, by definition, supporting services are not directly used by people, even when they influence the supply of provisioning, regulating, and cultural services.

14.3 Social Beneficiaries of Rangeland Ecosystem Services

In the same way that ecosystem services are classified, the social beneficiaries of services may be classified in categories according to the particular ecosystem services they use. Beneficiaries of ecosystem services are individuals, commercial entities, and the public sector and they may be distributed across local, regional, national, and global scales (Table 14.1). The demand for ecosystem services is complex and the classification of service beneficiaries, who often vary in their ecosystem-service preferences, can be a useful tool for identifying potential trade-offs and for balancing multiple, often conflicting, demands for services. If people's preferences for two or more services are known and they can be expressed accurately in the same units of value, then making the trade-off decision is (at least conceptually) straightforward and involves a simple cost-benefit calculation (Carpenter et al. 2009). Rangeland managers face the need to manage multiple ecosystem services and their interactions (Raudsepp-Hearne et al. 2010) and the demands of multiple beneficiaries (Yahdjian et al. 2015).

The supply and demand of ecosystem services occur at different spatial scales. Some ecosystem services are very local (pollination service, cultural services) whereas others are global (sequestration of greenhouse gases, air and water purification). The different scales involved in the provision of ecosystem services raise the possibility of a mismatch between supply and demand. Mismatches may also occur between those who control the provision of ecosystem services (supply) and those who benefit from them (users).

The main beneficiaries in rangelands are ranchers, land-owner producers, land tenants, service providers, recreational hunters, conservationists, landscape planners, passive and active nature tourists, and government and nongovernmental organizations (Scheffer et al. 2000; Castro et al. 2011; Yahdjian et al. 2015). While ranchers historically have demanded mainly provisioning services, their demands have broadened (Brown and McDonald 1995) while tourists and conservationists classically have demanded more supporting and cultural services. It is important here to further highlight that ranchers vary enormously in their demand

Table 14.1 Potential beneficiaries of ecosystem services across different spatial scales

Spatial scale			
Stakeholders	Local	National/regional	Global
Individual	Hunter/gatherer, subsistence farmers, and tourists	Tourists, consumers, educators, and students	Tourists, consumers, educators, and students
Commercial entity	Local entrepreneurs, farmers, traders, and artisans	Regional economic organizations	International enterprise including fishery and forestry industries
Public sector	Local government	National and regional government	International community

Rows represent stakeholders and columns represent various spatial scales at which stakeholders interact with rangeland ecosystems (modified from Newcome et al. (2005))

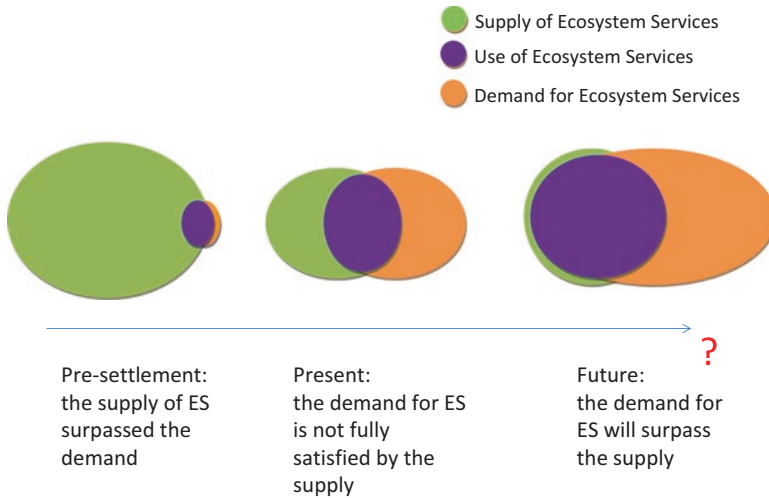


Fig. 14.3 Supply and demand for ecosystem services following European settlement (modified from data in Carpenter et al. (2009))

for ecosystem services. Similarly, people living in urban centers demand clean air and water that are provided by adjacent rural areas. The contrasting demands of different beneficiaries influence the analysis of land-management actions and their consequences on different ecosystem services.

The demand for ecosystem services in rangelands has diversified in the past decades, from mainly provisioning services to an increasing demand for more diverse services including regulating and cultural services (Yahdjian et al. 2015). The balance between supply and demand has also changed greatly from the time of European settlement in North America (Fig. 14.3). The ability of ecosystems to produce services is declining and the demand for them is increasing, with serious implications for both people and the environment. However, we have not developed sufficient knowledge to quantify and model the demand for ecosystem services as we have for their supply. So, the question remains, which category of ecosystem services will have greatest demand in the future? How provision of and demand for ecosystem services will be balanced in rangelands? Which trade-offs among ecosystems services will be most important in the future? Who will be responsible for making these decisions?

14.4 Methods for Estimating Supply and Demand

Different tools and models have been developed to assess the production of ecosystem services, including the valuation of market and nonmarket services, in both economic and noneconomic terms. The combination of ecological production

functions and economic valuation describes the monetary value of ecosystem services. Recently, the Natural Capital Project has developed a tool to integrate biophysical and economic information on ecosystem services (Tallis et al. 2011).

The demand for ecosystem services is related to the social beneficiaries and is usually described by the location, type, and intensity of people's demand for services. The demand has been evaluated focusing on the perception of ecosystem services by different stakeholders (De Chazal et al. 2008; Quétier et al. 2010; Martin-Lopez et al. 2012). Preferences have been assessed by compiling responses to questionnaires and interviews (Lamarque et al. 2011; Martin-Lopez et al. 2012). During social surveys, ecosystem services are identified spontaneously, and the more "visible" services, such as recreation, aesthetic, and natural hazard regulation, are commonly described. Other questionnaires request that people rank ecosystem services according to their preferences. During the ranking exercises more "invisible" services, such as pollination and soil fertility, often emerge (Lamarque et al. 2011; Martin-Lopez et al. 2012). Finally, the traditional surveys formally used to value nonmarket ecosystem services, such as the willingness to pay for conservation of certain resources or the existence value, may also be included in studies of demands.

The main drivers associated with people preferences for ecosystem services were monthly income, level of education (from traditional ecological knowledge to formal education), and place of residence (the rural–urban continuum; Yahdjian et al. 2015). In addition, other social variables like age, gender, culture, and geographical location were also associated with the interest that people have in ecosystem services (MA 2005).

The relationship between the supply of ecosystem services and the demand for them determines the actual use of ecosystem services by society (Tallis and Polasky 2011). Food production per hectare or the amount of clean water used for irrigation are examples of estimates of the use of provisioning ecosystem services. When global analyses are implemented, remote drivers and teleconnections, such as international trade practices and agreements, have to be taken into account. Trade patterns, which can be dynamic and quite nuanced, show how demand for certain services in one country leads to changes in the provisioning services in other countries.

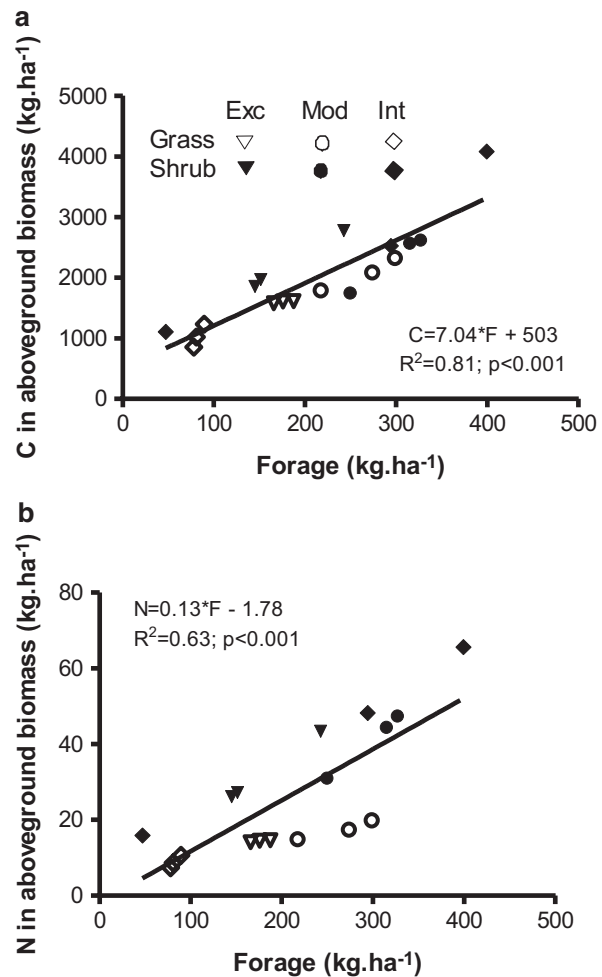
14.5 Trade-Offs and Win–Win Interactions

There are cases of synergistic and antagonistic interactions among different types of ecosystem services. Synergistic interactions, or win–win conditions, indicate that management leading to the increase of one type of ecosystem service may result in the increase of other ecosystem services. For example, some ecosystem services respond similarly to specific management practices and ecological conditions, such as those that may lead to increased carbon sequestration then resulting in increased water holding capacity, and, many of them, such as cultural services and biodiversity conservation, produce multiple intertwined values (Bennett et al. 2009).

Antagonistic interactions, or trade-offs, indicate that management practices or events that increase one type of ecosystem service may negatively affect other ecosystem services (Oñatibia et al. 2015). For example, land management practices that lead to increases in the provisioning of food may result in a reduction of clean water purification, creating trade-offs in the provisioning of ecosystem services (Raudsepp-Hearne et al. 2010). Planting trees to increase carbon sequestration or timber production may decrease stream flow in arid areas and represents a trade-off (Nosetto et al. 2008). In summary, ecosystem service research has advanced to identify nature as a complex provider of human benefits (MA 2005).

The rangelands of Patagonia provide an example of trade-offs and win-win relationships among ecosystem services depending on management. An example of win-win is the maximization of carbon, nitrogen, and forage availability at intermediate grazing intensities (Fig. 14.4). A critical provision service, such as forage biomass, is

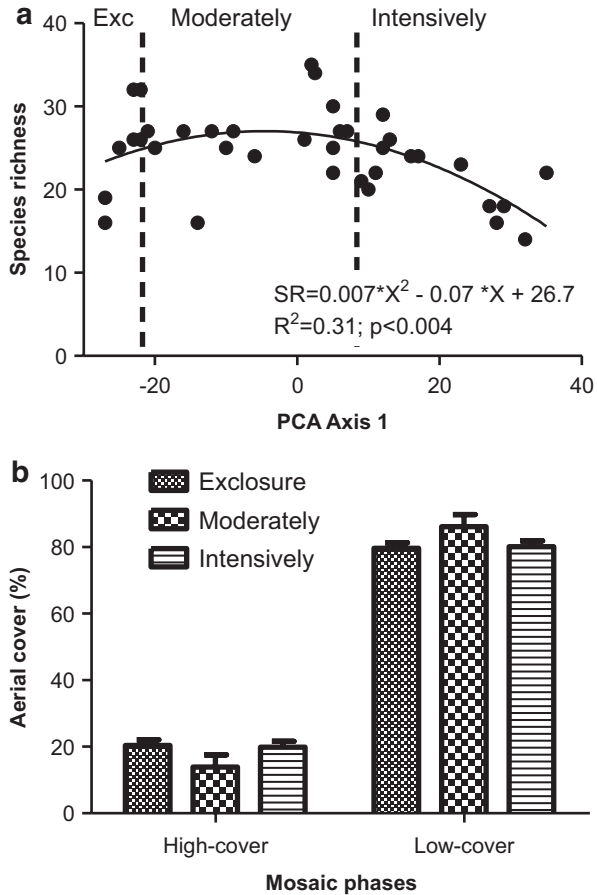
Fig. 14.4 Example of a win-win interaction between a supportive ecosystem service, carbon and nitrogen stocks, and a provisioning ecosystem service forage production as depicted by the complementary relationships between carbon (C) (a) and nitrogen (N) (b) in forage of a Patagonian rangeland. Paddocks are used with different stocking rates. Exclosure (Exc) includes fields without domestic animals for at least 27 years. Moderately (Mod) and intensively (Int) grazed paddocks had 0.2 and 0.4 sheep ha⁻¹ (redrawn from Oñatibia et al. (2015))



positively related to regulation services such as carbon and nitrogen sequestration at intermediate grazing intensities. Carbon and nitrogen stocks (C) in vegetation (above and belowground) were significantly higher in moderately grazed paddocks than in enclosure and intensively grazed ones (Oñatibia et al. 2015; Fig. 14.4). In this example, the relationship between forage biomass and carbon and nitrogen stocks had a positive linear relationship indicating that a trade-off did not occur (Fig. 14.4).

A trade-off between a supporting and a provisioning ecosystem service occurred in Patagonian rangelands. Grazing intensity shows a unimodal relationship species richness with a maximum value at moderate grazing intensities (Perelman et al. 1997; Fig. 14.5a). A decrease in richness is associated with intensive grazing because local extinction of forage species was not compensated by remaining non-palatable or weedy species. Patch diversity is another critical component of biodiversity in rangelands (Chap. 5, this volume). Abundance of different patch types shows a response similar to that of species richness. Under moderate grazing conditions, high-cover patches decrease whereas low-cover patches increase.

Fig. 14.5 Example of a trade-off between a supportive ecosystem service, biodiversity, and grazing intensity, which is an indicator of a provisioning ecosystem service, livestock production in the Patagonian rangelands, Argentina, under different management strategies. **a** Species richness along three grazing histories (*Exc* enclosures; moderate and intense grazing, separated by vertical dashed lines; redrawn from Perelman et al. (1997)), and **b** aerial percent cover of two patch types in low- and high-cover patches under three grazing regimes (mean ± ES), based on sampling the same paddocks as in Fig. 14.4 (redrawn from Cipriotti and Aguiar (2010))



14.6 Rangeland Management and Ecosystem Services: A Historical Perspective

For nearly a century the basic principles of rangeland management have been described and re-explained as the nature of goods and services derived from rangelands. Early in the twentieth century, Sampson (1923) outlined basic management principles and practices to support the continued provision of food and fiber via livestock grazing from rangelands. The need for these principles grew out of an era of resource overexploitation all over the world (Texeira and Paruelo 2006; Sayre et al. 2012). In that era, the provisioning services of food and fiber from rangelands were a central focus. There was either a lack of interest or a general unawareness of other goods and services from these “waste” lands at that time. This would be true for most rangeland environments on all continents in their early stages of settlement and development (Chap. 1, this volume). Management principles of the early twentieth century classically focused on requirements to control overgrazing and erosion through establishment of proper limits of the numbers of livestock, avoidance of grazing forage plants too early in their growth cycle, and effective distribution of livestock use across rangelands. These same principles have persisted to guide livestock grazing as detailed in subsequent texts on rangeland management into the early twenty-first century (Stoddart and Smith 1943; Vallentine 1989; Heitschmidt and Stuth 1991; Holechek et al. 2011).

These traditional principles for sustained provisioning of food and fiber goods have proven to be either inadequate or unappreciated, however, in guiding range management in the provisioning of multiple ecosystem goods and services in recent decades. Though these principles still have application in terms of recognizing limits to the supply of services and extraction of goods, the principles of rangeland management would be more appropriately portrayed as those of the science of managing trade-offs among ecosystem services and negotiating among stakeholders with competing interests. In reality, the management principles, which were articulated by Arthur Sampson nearly a century ago, are insufficient to manage landscapes in the twenty-first century. Currently, the provisioning of ecosystem services is dictated by dynamics of land-use fragmentation, ecological legacies of past management, oppressive constraints of antiquated infrastructure, inadequate social institutions, heterogeneous nuances of topography, fragilities of specific species, economic pressures of global demands for local goods and services, uncertainties of changing climates, political expediencies, and an array of cultural factors seldom acknowledged in the land management textbooks of the past. These complexities have been in play for decades and increasingly so with an expanding human population living on or adjacent to the world’s rangelands. The articulation of a more sophisticated set of management principles has not kept pace with these newer landscape realities (Chap. 1, this volume).

14.7 Rangeland Management and Ecosystem Services: Landscape, Time, and Human Interactions

Given these current realities, the provisioning of goods and services from rangelands is now more appropriately perceived as a function of landscape, time, and human gradients (Fig. 14.6). The landscape gradient is shaped by ecological constraints resulting from an array of ecological sites and their existing conditions. It is this landscape gradient that was classically addressed through the basic management principles developed during the twentieth century when rangelands were viewed to provide a more narrow set of goods and services than expected in the twenty-first century.

Governance and socioeconomic conditions represent a critical component of this new conceptual framework. Complex patterns of land ownership (both public and private) and multiple administrative jurisdictions underscore the importance of governance and stakeholder engagement in supplying ecosystem services (Petz et al. 2014). The importance of viewing rangeland landscapes as socio-ecological systems with diverse governing institutions engaged in planning and management has been well described for some landscapes (Huntsinger and Oviedo 2014). Of additional importance are the social and cultural characteristics of resident populations. For

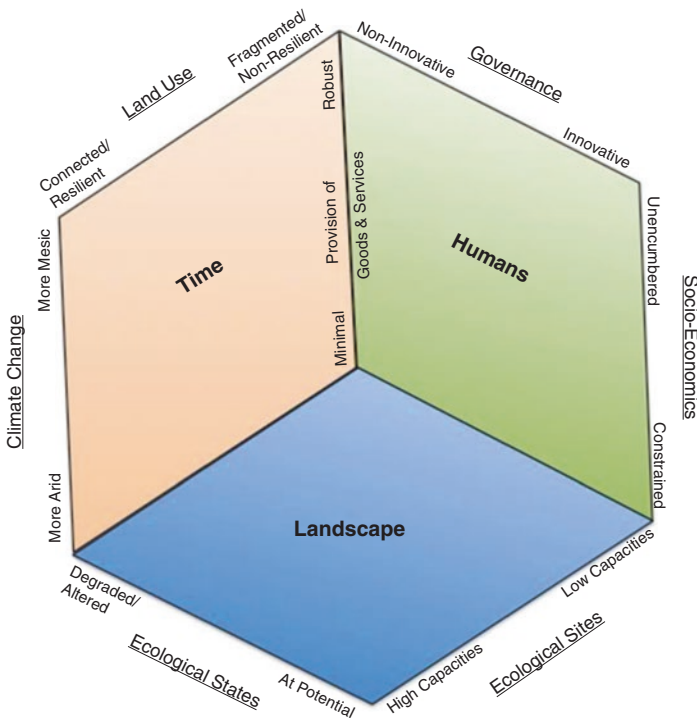


Fig. 14.6 Conceptual diagram of landscape, time, and human gradients that influence the provisioning of ecological goods and services from rangelands (adapted from Sayre et al. (2013))

example, level of education, household income, and place of residence were described as the main human aspects driving demand for ecosystem services (Yahdjian et al. 2015). The need to consider and incorporate these human-related drivers into our management models far exceeds the utilities of the basic principles of rangeland management as articulated frequently throughout the twentieth century.

Land use is a major driver of the array of goods and services that can be supplied, and it is a dynamic feature of landscapes around the world (Foley et al. 2005). Changes in land use can dramatically shift provisions of goods and services, and shifts may signify persistent alterations. Land-use gradients are further complicated by the uncertainties of climate variability and climate change. Though climate models are increasingly sophisticated and generating “near-term” projections (Taylor et al. 2012), their limitations and complexities are still problematic. However, recent statistical methods have provided tools to downscale global climate models to spatial scales that have application to land management (Abatzoglou 2013). What is now emerging is a more focused and applicable set of projections of climatic variables and their probabilities that would improve the ability of designing rangeland management that optimizes the provisioning of different ecosystem services under a changing climate. For example, interactive maps for various climate variables for the 2040–2060 period based on global climate models scaled down to the county level are now available for the USA (Fig. 14.7). Climatic projections at fine scales

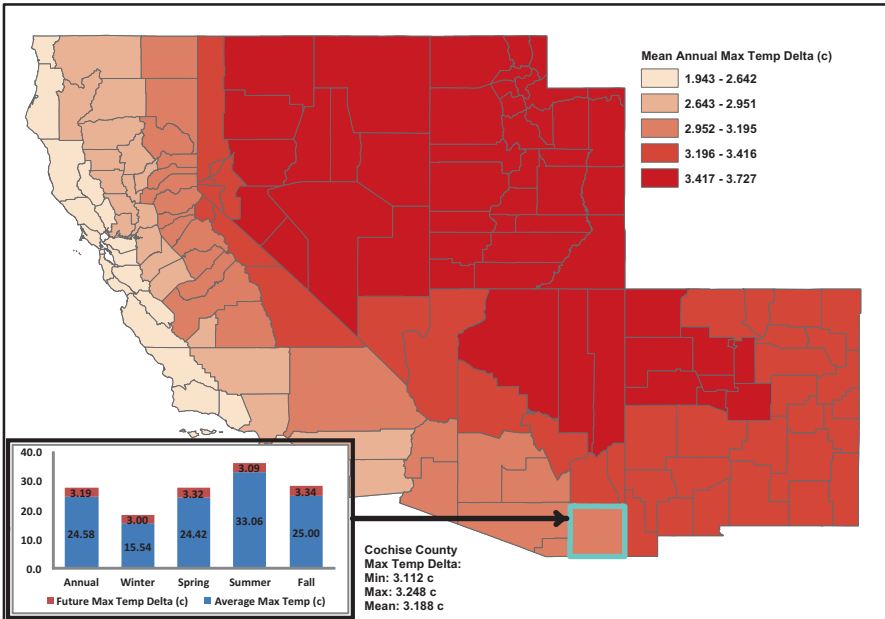


Fig. 14.7 Projected changes in mean annual temperature for the 2040–2060 period scaled to the county level for the five states: California, Nevada, Arizona, Utah, and New Mexico, calculated from Abatzoglou (2013) and adapted from Taylor et al. (2012). Details for Cochise County, AZ, are included

of time and space will be effective in creating more quantitative understandings of pending changes that will have direct bearing on either the benefits these landscapes can provide or the conditions or processes that lead to desired benefits, either of which are definitions of ecosystem goods and services (Bommarco et al. 2013; see also Chap. 7, this volume).

The utility of the conceptual framework including landscape, time, and human gradients for deriving ecosystem services is evident in recent case studies from specific landscapes around the world (Fig. 14.6). For example, in the Little Karoo of South Africa, four different scenarios were created to evaluate the impact of one service, biodiversity, on the resulting supply of livestock forage, carbon storage, and water recharge from this 19,730 km² landscape (Egoh et al. 2010). Mapped habitats (landscape gradient) were evaluated for capacities to supply these different services. Different realistic land-use potentials (time gradient) were evaluated, such as development of tourism or of carbon markets, in terms of resulting impacts on other ecosystem services. Resulting impacts on revenues and opportunity costs for landowners (human gradient) were also calculated. Scenarios where biodiversity conservation could be achieved with provision of other services were realistic if human factors, including incentives to develop markets for services and institutions to encourage the supply of nontraditional ecosystem services, were emphasized. The importance of understanding landscape capacities to supply various ecosystem services was essential, but consideration of time and human gradients was paramount to managing for ecosystem services over time. Other recent case studies further illustrate the importance of inherent ecological capacities and existing ecological conditions, linkages of ecosystem services to the presence or absence of adequate infrastructure, such as roads and management institutions, subsistence requirements for resident human populations, rates of land-use and land-cover changes that can enhance one set of services at the expense of others, and importance placed on biodiversity within landscapes (Zhao et al. 2004; Muñoz et al. 2013; Pan et al. 2014).

Increasingly, the guidelines for managing ecosystem goods and services materialize for any specific landscape out of quantitative efforts to describe and map their occurrence or potential at spatial scales of relevance to landscape, human, and time gradients (Crossman et al. 2013). Though specific principles guiding processes of description and mapping of ecosystem goods and services are incomplete and methods are diverse, the resulting benefits are tangible. For example, a study both mapped and valued ecosystem services occurring across the Ewaso Ng'iro watershed of northern Kenya (Ericksen et al. 2011). Ecosystem services included medicinal plants, crops, livestock, wildlife, tourism, marketed carbon, wood and fiber, drinking water, flood regulation, cultural identification, and open space. The supply of these services was highly dependent on land use, inherent landscape capacities, existence of supply and their market values, proximity to infrastructure (i.e., accessibility), temporal dynamics of supply, social and cultural values of specific services, and spatial arrangement of the watershed and its sub-catchments. In the end, the final services that could be valued for uses within the watershed were livestock, tourism, and crops, and they were heterogeneously distributed and mapped across the watershed. In this fashion, recommendations could be developed based on specific knowledge of landscapes,

demands, and values. For example, the market value of tourism within the watershed was minimal, but the value was highly dependent on infrastructure (national parks, roads). Decisions to further develop tourism could be then made with knowledge of costs both to develop the necessary infrastructure as well as evaluate resulting impacts on other benefits, such as livestock production.

In south-eastern Australia, one study evaluated and ranked existing land-cover types for six ecosystem services of forage production, biodiversity, water regulation, provision of water, carbon stock, and timber production (Baral et al. 2013). In this fashion, the provisioning of these services could be evaluated both spatially for the different cover types and land use and temporally given the known changes in land use and land cover over the past two centuries. In general, their analyses concluded that less modified landscapes resulted in a great supply and diversity of these services. However, specific land uses that modified cover, such as conversion of pasture to plantation, could result in a greater array of services, such as plantations providing timber production from regions within the basin. Their mapping processes provided a basis for evaluating possible services and the resulting trade-offs created by land-use decisions. Characterization and quantification of ecosystem services provide a bridge that can link our knowledge of ecological processes across landscapes, time, and human processes (Fu and Forsius 2015). Continued efforts to clearly establish the principles for these characterizations and quantifications are critical to both understanding linkages among human and ecological processes for a landscape and sustaining output of a demanded supply of goods and services.

Exploitation and utilization of natural resources pose the questions of property ownership and legitimate stakeholders (Latour 2013). Are natural resources the property of individuals, and if so which individuals, or are they the property of humankind? In some regions, provisioning ecosystem services belong to the owners of the land. At the same time, all other ecological services (supporting, regulating, and cultural) in general are not marketable and usually are not claimed or owned by a single individual (single or organized with economic purposes). The number, nature, and diversity of stakeholders have increased in the recent past as the importance of regulating and cultural services has increased relative to provisioning services (Yahdjian et al. 2015). Demand for cultural ecosystem services results in the creation of national parks or natural reserves as in the example for South American rangelands (Murdoch et al. 2010). This strategy raises ethics issue since inhabitants—mostly small ranchers and peasants—are forced to migrate and adopt alternative lifestyles. Additionally migration may also initiate other social conflicts in urban centers (Easdale et al. 2009). It is important to keep in mind as we analyze human-induced degradation of ecological services that there is a network of stakeholders, in addition to ranchers, and sociopolitical processes that are involved (Easdale and Domptail 2014). Many component processes have nonlinear dynamics that may potentially exhibit thresholds (Walker et al. 1981). More than seven decades since the rise of rangeland science the challenge of grazing management remains unresolved because of the changing demands for ecosystem services. Integration of human and biophysical dimensions through the study of ecological services may be a rewarding path.

14.8 Conceptual Framework for Ecosystem Services and Range Management

The concept of ecosystem services has emerged as a powerful tool for guiding management of rangelands in the twenty-first century. Ecosystem services serve as a way of clarifying what is that different stakeholders want from rangelands, ranging in scale from paddocks and counties to regions across national boundaries. Ecosystem services also serve to clarify what goods and services that land is able to supply. The optimal management strategy results from reconciling supply and demand of ecosystem services (Yahdjian et al. 2015) as described in the following equation. Land use is a function of:

$$\text{Land use} = f \left[\sum_{j=1}^n \left(\text{ES}_j \text{ supply} \right), \left(\sum_{i=1}^m \left(\text{ES}_j \text{ Demand}_{\text{stakeholder } i} * \text{Political Power}_{\text{stakeholder } i} \right) \right) \right] \quad (14.1)$$

Here, land use or, in our specific case, rangeland management practices depend on the sum of the supply of all the ecosystem services ES_j from 1 to n , and the sum of the demand for each ES_j from each stakeholder from i to m . Finally, the demand of each stakeholder is weighed by their political power.

For example, rangelands all over the world are being invaded by woody plants (Estell et al. 2012). This transformation from grasslands into shrublands and savannas affects the provisioning of ecosystem services. Woody-plant encroachment affects the provisioning of different ecosystem services from livestock production to maintenance of biodiversity and yielding of clean water (MacLeod and Johnston 1990; Turpie et al. 2008; Anadón et al. 2014). Equation (14.1) can be applied to the rangeland management issue of whether to remove or not woody plants. Rangelands can supply different services including clean water and livestock production, which are enhanced by woody-plant control. On the contrary, the ecosystem service erosion control may be diminished by removal of woody plants. Different stakeholders with different political power value these different services differently. The final solution to the management question of whether to remove woody plants will depend on both (1) the effect that woody plants have at each specific location on ecosystem services and (2) the valuation that each stakeholder has on each ecosystem service.

Similarly, Anadón et al. (2014) analyzed the impact of woody-plant encroachment on livestock production in US and Argentinean rangelands. These are two rangelands, which are similar from the biophysical standpoint and in their ability to supply ecosystem services. However, these rangelands have contrasting socioeconomic conditions that affect the demand for ecosystem services. In Argentina, livestock production is the most valued and primary ecosystem service of interest. In the USA, on the contrary, rangelands face multiple demands for ecosystem services, including biodiversity conservation and recreation, in addition to livestock production. Anadón et al. (2014) showed that the different

demand for ecosystem services modified the impact of woody-plant invasions on livestock production. In Argentina, woody-plant cover accounted for 50 % of the livestock production but, in the USA, it explained only half of the variability in livestock production.

14.9 Future Directions

Ecosystem services represent an important conceptual link between the biophysical constraints and human demand. However, our understanding of the supply and demand of ecosystem services is unbalanced. We know much more about the supply of ecosystem services than we know about the demand for different ecosystem services from different beneficiaries. It will be important to enhance our understanding of the demand from different groups of beneficiaries for specific ecosystem services within specific landscapes. In addition, it will be necessary to understand and quantify the determinants of the demand for ecosystem services.

In order to predict the future of rangelands and develop appropriate management strategies, we need to understand the future supply and the demand for ecosystem services. Our strategy to tackle this daunting task is to separate the effect of drivers to the response of supply and demand to their drivers. For example, the future supply of the ecosystem service forage production depends on climate change (the driver) and sensitivity of ecosystems to climate (the response to driver). Similarly, the demand for the ecosystem service recreation depends on the proportion of urban population (driver) and the sensitivity of recreation demand to urbanization (response to driver). This requires an interdisciplinary approach where land owners, land managers (public and/or private), ecologists, climatologists, and social scientists work in close collaboration.

14.10 Summary

Rangeland ecosystem services are the benefits that society receives from rangelands. They include the provisioning of food, wood and medicinal resources, and services that contribute to climate stability, control of agricultural pests, and purification of air and water. Rangeland ecosystem services are classified in four categories: provisioning, regulating, cultural, and supporting. Provisioning ecosystem services include the contribution of essential goods such as food, fiber, and medicinal. Regulating ecosystem services include carbon sequestration, prevention of soil erosion, and natural flood control. Cultural ecosystem services include intellectual, inspirational, and recreational activities. The fourth category is supporting ecosystem services, which include services that are dependent on ecological processes such as primary production and nutrient cycling and that are intimately related to biological diversity.

The supply of ecosystem services is mostly determined by biophysical factors such as climate, soils, as well as historical land use. Human demand represents the other side of the rangeland ecosystem service equation, which is related to the social beneficiaries. Human consumption of resources and utilization of services that are supplied by rangelands depend upon both their capacity to produce them and the societal value and need placed on those resources and services. Demand for ecosystem services changes among social beneficiaries, who are the individuals or groups of individuals who have an interest in ecosystem services. Different methods have been developed to assess the demand for specific ecosystem services, including collation of responses to questionnaires and interviews and classical economic tools such as willingness to pay.

There are cases of trade-offs and win-win interactions among different types of ecosystem services. Win-win conditions occur when management aimed at increasing one type of ecosystem service results in the increase of other ecosystem services. For example, grazing management that results in increased forage supply also increases carbon and nitrogen stocks in rangeland soils. Trade-offs occur when management results in the increase of one ecosystem service and the decrease in other. For example, increasing grazing intensity and livestock production may, in certain cases, decrease biodiversity.

Principles of range management developed in the twentieth century focused primarily on the biophysical components of the rangeland ecosystems and the requirements to control overgrazing and erosion through the establishment of general management principles, including proper limits on numbers of livestock, avoiding grazing of forage plants too early in their growth cycle, and effectively distributing livestock use across rangelands. In the twenty-first century, principles for managing rangelands need to be much broader. Currently, the provisioning of ecosystem services is dictated by dynamics of land-use fragmentation, ecological legacies of past management, infrastructure, fragilities of specific species, economic pressures of global demands for local goods and services, uncertainties of changing climates, political expediencies, and an array of cultural factors seldom acknowledged in the land management textbooks of the past.

Here, we propose a novel conceptual framework for rangeland management based on the premise that management always aims at reconciling supply and demand of ecosystem services. The supply of each ecosystem service is based mostly on biophysical characteristics and the use history that could have affected its potential. The demand for each ecosystem service is different for each group of beneficiaries or stakeholders. Finally, the demands of each group of beneficiaries do not have the same impact because of their differential capacity to influence decision making. Therefore, demands for ecosystem services here are weighed by the political power of each group of beneficiaries or stakeholders. In conclusion, there is not a universally optimal management strategy for a rangeland because demands for ecosystem services, power of each group of beneficiaries, and supply of ecosystem services change through time.

Acknowledgements Authors want to thank Courtney Currier for her support during the course of this project, which received financial support from the US National Science Foundation LTER DEB 1235828.

References

- Abatzoglou, J.T. 2013. Development of gridded surface meteorological data for ecological applications and modelling. *International Journal of Climatology* 33: 121–131.
- Adler, P.B., D.G. Milchunas, O.E. Sala, I.C. Burke, and W.K. Lauenroth. 2005. Plant traits and ecosystem grazing effects: Comparison of U.S. sagebrush steppe and Patagonian steppe. *Ecological Applications* 15: 774–792.
- Ahlström, A., M.R. Raupach, G. Schurgers, B. Smith, A. Arneeth, M. Jung, M. Reichstein, J.G. Canadell, P. Friedlingstein, and A.K. Jain. 2015. The dominant role of semi-arid ecosystems in the trend and variability of the land CO₂ sink. *Science* 348: 895–899.
- Anadón, J.D., O.E. Sala, B.L. Turner, and E.M. Bennett. 2014. The effect of woody-plant encroachment on livestock production in North and South America. *Proceedings of National Academy of Sciences* 111: 12948–12953.
- Baral, H., R.J. Keenan, J.C. Fox, N.E. Stork, and S. Kasel. 2013. Spatial assessment of ecosystem goods and services in complex production landscapes: A case study from south-eastern Australia. *Ecological Complexity* 13: 35–45.
- Bennett, E.M., G.D. Peterson, and L.J. Gordon. 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters* 12: 1394–1404.
- Bommarco, R., D. Kleijn, and S.G. Potts. 2013. Ecological intensification: Harnessing ecosystem services for food security. *Trends in Ecology and Evolution* 28: 230–238.
- Brown, J.H., and W. McDonald. 1995. Livestock grazing and conservation of southwestern rangelands. *Conservation Biology* 9: 1644–1647.
- Carpenter, S.R., H.A. Mooney, J. Agard, D. Capistrano, R.S. DeFries, S. Diaz, T. Dietz, A.K. Duraipappah, A. Oteng-Yeboah, H.M. Pereira, C. Perrings, W.V. Reid, J. Sarukhan, R.J. Scholes, and A. Whyte. 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences of the United States of America* 106: 1305–1312.
- Castro, A., B. Martín-López, M. García-Llorente, P. Aguilera, E. López, and J. Cabello. 2011. Social preferences regarding the delivery of ecosystem services in a semiarid Mediterranean region. *Journal of Arid Environments* 75: 1201–1208.
- Cipriotti, P., and M. Aguiar. 2010. Resource partitioning and interactions enable coexistence in a grass-shrub steppe. *Journal of Arid Environments* 74: 1111–1120.
- Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253–260.
- Crossman, N.D., B. Burkhard, S. Nedkov, L. Willemsen, K. Petz, I. Palomo, E.G. Drakou, B. Martín-Lopez, T. McPhearson, and K. Boyanova. 2013. A blueprint for mapping and modelling ecosystem services. *Ecosystem Services* 4: 4–14.
- Daily, G.C. (ed.). 1997. *Nature's services. Societal dependence on natural ecosystems*. Washington DC: Island Press.
- De Chazal, J., F. Quetier, S. Lavorel, and A. Van Doorn. 2008. Including multiple differing stakeholder values into vulnerability assessments of socio-ecological systems. *Global Environmental Change* 18: 508–520.
- Easdale, M., and S. Domptail. 2014. Fate can be changed! Arid rangelands in a globalizing world—a complementary co-evolutionary perspective on the current 'desert syndrome'. *Journal of Arid Environments* 100: 52–62.
- Easdale, M.H., M.R. Aguiar, M. Roman, and S. Villagra. 2009. Socio-economic comparison of two biophysical regions: Livestock production systems from Río Negro Province, Argentina. *Cuadernos De Desarrollo Rural* 6: 173–198.
- Egoh, B.N., B. Reyers, J. Carwardine, M. Bode, P.J. O'farrell, K.A. Wilson, H.P. Possingham, M. Rouget, W. De Lange, and D.M. Richardson. 2010. Safeguarding biodiversity and ecosystem services in the Little Karoo, South Africa. *Conservation Biology* 24: 1021–1030.

- Ericksen, P., M. Said, J.d. Leeuw, S. Silvestri, L. Zaibet, S. Kifugo, K. Sijmons, J. Kinoti, L. Nganga, and F. Landsberg. 2011. *Mapping and valuing ecosystem services in the Ewaso Ng'iro watershed. ILRI-WRI-Danida report*. Nairobi.
- Estell, R., K.M. Havstad, A. Cibils, D. Anderson, T. Schrader, and K. James. 2012. Increasing shrub use by livestock in a world with less grass. *Rangeland Ecology and Management* 65: 327–414.
- Foley, J.A., R. DeFries, G.P. Asner, C. Barford, G. Bonan, S.R. Carpenter, F.S. Chapin, M.T. Coe, G.C. Daily, H.K. Gibbs, J.H. Helkowski, T. Holloway, E.A. Howard, C.J. Kucharik, C. Monfreda, J.A. Patz, I.C. Prentice, N. Ramankutty, and P.K. Snyder. 2005. Global consequences of land use. *Science* 309: 570–574.
- Fu, B., and M. Forsius. 2015. Ecosystem services modeling in contrasting landscapes. *Landscape Ecology* 30: 375–379.
- Gomez-Baggethun, E., R. de Groot, P.L. Lomas, and C. Montes. 2010. The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics* 69: 1209–1218.
- Havstad, K.M., D.P.C. Peters, R. Skaggs, J. Brown, B.T. Bestelmeyer, E. Fedrickson, J.E. Herrick, and J. Wright. 2007. Ecological services to and from rangelands of the United States. *Ecological Economics* 64: 261–268.
- Heitschmidt, R., and J.P. Stuth. 1991. *Grazing management: An ecological perspective*. Portland: Timber Press.
- Herrick, J.E., O.E. Sala, and J.W. Karl. 2013. Land degradation and climate change: A sin of omission? *Frontiers in Ecology and the Environment* 11: 283–283.
- Holechek, J.L., R.D. Pieper, and C.H. Herbel. 2011. *Range management: Principles and practices*, 6th ed. New York: Pearson Education, Inc.
- Huntsinger, L., and J.L. Oviedo. 2014. Ecosystem services are social-ecological services in a traditional pastoral system: The case of California's Mediterranean rangelands. *Ecology and Society* 19: 8.
- Kauffman, J., and D. Pyke. 2001. Range ecology, global livestock influences. In *Encyclopedia of biodiversity*, ed. S. Levin, 33–52. San Diego: Academic.
- Lamarque, P., U. Tappeiner, C. Turner, M. Steinbacher, R.D. Bardgett, U. Szukics, M. Schermer, and S. Lavorel. 2011. Stakeholder perceptions of grassland ecosystem services in relation to knowledge on soil fertility and biodiversity. *Regional Environmental Change* 11: 791–804.
- Latour, B. 2013. *An inquiry into modes of existence*. Cambridge: Harvard University Press.
- MA. 2005. *Millennium Ecosystem Assessment (MA) synthesis report*. Washington, DC: Millennium Ecosystem Assessment.
- MacLeod, N., and B. Johnston. 1990. An economic framework for the evaluation of rangeland restoration projects. *The Rangeland Journal* 12: 40–53.
- Martin-Lopez, B., I. Iniesta-Arandia, M. Garcia-Llorente, I. Palomo, I. Casado-Arzuaga, D.G. Del Amo, E. Gomez-Baggethun, E. Oteros-Rozas, I. Palacios-Agundez, B. Willaarts, J.A. Gonzalez, F. Santos-Martin, M. Onaindia, C. Lopez-Santiago, and C. Montes. 2012. Uncovering ecosystem service bundles through social preferences. *PLoS One* 7: e38970.
- Muñoz, J.C., R. Aerts, K.W. Thijs, P.R. Stevenson, B. Muys, and C.H. Sekercioglu. 2013. Contribution of woody habitat islands to the conservation of birds and their potential ecosystem services in an extensive Colombian rangeland. *Agriculture, Ecosystems and Environment* 173: 13–19.
- Murdoch, W., J. Ranganathan, S. Polasky, and J. Regetz. 2010. Using return on investment to maximize conservation effectiveness in Argentine grasslands. *Proceedings of the National Academy of Sciences* 107: 20855–20862.
- Newcome, J., A. Provins, H. Johns, E. Ozdemiroglu, J. Ghazoul, D. Burgess, and K. Turner. 2005. *The economic, social and ecological value of ecosystem services: A literature review*. London: Economics for the Environment Consultancy (eftec).
- Nosetto, M., E. Jobbágy, T. Tóth, and R. Jackson. 2008. Regional patterns and controls of ecosystem salinization with grassland afforestation along a rainfall gradient. *Global Biogeochemical Cycles* 22: GB2015.
- Oosterheld, M., O.E. Sala, and S.J. McNaughton. 1992. Effect of animal husbandry on herbivore-carrying capacity at a regional scale. *Nature* 356: 234–236.

- Oñatibia, G.R., M.R. Aguiar, and M. Semmartin. 2015. Are there any trade-offs between forage provision and the ecosystem service of C and N storage in arid rangelands? *Ecological Engineering* 77: 26–32.
- Pan, Y., J. Wu, and Z. Xu. 2014. Analysis of the tradeoffs between provisioning and regulating services from the perspective of varied share of net primary production in an alpine grassland ecosystem. *Ecological Complexity* 17: 79–86.
- Perelman, S., R. León, and J. Bussaca. 1997. Floristic changes related to grazing intensity in a Patagonian shrub steppe. *Ecography* 20: 400–406.
- Perrings, C., S. Naeem, F.S. Ahrestani, D.E. Bunker, P. Burkill, G. Canziani, T. Elmqvist, J.A. Fuhrman, F.M. Jaksic, Z. Kawabata, A. Kinzig, G.M. Mace, H. Mooney, A.H. Prieur-Richard, J. Tschirhart, and W. Weisser. 2011. Ecosystem services, targets, and indicators for the conservation and sustainable use of biodiversity. *Frontiers in Ecology and the Environment* 9: 512–520.
- Petz, K., J. Glenday, and R. Alkemade. 2014. Land management implications for ecosystem services in a South African rangeland. *Ecological Indicators* 45: 692–703.
- Quétier, F., F. Rivoal, P. Marty, J. de Chazal, W. Thuiller, and S. Lavorel. 2010. Social representations of an alpine grassland landscape and socio-political discourses on rural development. *Regional Environmental Change* 10: 119–130.
- Raudsepp-Hearne, C., G.D. Peterson, and E.M. Bennett. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences of the United States of America* 107: 5242–5247.
- Reynolds, J.F., D.M.S. Smith, E.F. Lambin, B. Turner, M. Mortimore, S.P. Batterbury, T.E. Downing, H. Dowlatabadi, R.J. Fernández, and J.E. Herrick. 2007. Global desertification: Building a science for dryland development. *Science* 316: 847–851.
- Rositano, F., M. López, P. Benzi, and D.O. Ferraro. 2012. Servicios de los ecosistemas. un recorrido por los beneficios de la naturaleza. Ecosystem services. a travel through natural benefits. *Agronomía y ambiente. Revista de la Facultad de Agronomía de la Universidad de Buenos Aires* 32: 49–60.
- Sala, O., and J. Paruelo. 1997. Ecosystem services in grasslands. In *Nature's services: Societal dependence on natural ecosystems*, ed. G.C. Daily, 237–251. Washington, D.C.: Island Press.
- Sampson, A.W. 1923. *Range and pasture management*. New York: Wiley.
- Sayre, N.F., W. deBuys, B.T. Bestelmeyer, and K.M. Havstad. 2012. “The Range Problem” after a century of rangeland science: New research themes for altered landscapes. *Rangeland Ecology and Management* 65: 545–552.
- Sayre, N.F., R.R. McAllister, B.T. Bestelmeyer, M. Moritz, and M.D. Turner. 2013. Earth Stewardship of rangelands: Coping with ecological, economic, and political marginality. *Frontiers in Ecology and the Environment* 11: 348–354.
- Scheffer, M., W. Brock, and F. Westley. 2000. Socioeconomic mechanisms preventing optimum use of ecosystem services: An interdisciplinary theoretical analysis. *Ecosystems* 3: 451–471.
- Stoddart, L.A., and A.D. Smith. 1943. *Range management*, 547. NY: McGraw-Hill Book Co. Inc.
- Tallis, H., and S. Polasky. 2011. Assessing multiple ecosystem services: An integrated tool for the real world. *Natural capital. Theory and practice of mapping ecosystem services*, 34–52. Oxford: Oxford University Press.
- Tallis, H., T. Ricketts, A. Guerry, E. Nelson, D. Ennaanay, S. Wolny, N. Olwero, K. Vigerstol, D. Pennington, and G. Mendoza. 2011. *INVEST 2.1 beta user's guide*. Stanford: The Natural Capital Project.
- Taylor, K.E., R.J. Stouffer, and G.A. Meehl. 2012. An overview of CMIP5 and the experiment design. *Bulletin of the American Meteorological Society* 93: 485–498.
- Teixeira, M., and J.M. Paruelo. 2006. Demography, population dynamics and sustainability of the Patagonian sheep flocks. *Agricultural Systems* 87: 123–146.
- Turpie, J.K., C. Marais, and J.N. Blignaut. 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics* 65: 788–798.
- Vallentine, J.F. 1989. *Range development and improvements*. San Diego: Academic.

- Walker, B.H., D. Ludwig, C.S. Holling, and R.M. Peterman. 1981. Stability of semi-arid savanna grazing systems. *Journal of Ecology* 69: 473–498.
- Yahdjian, L., O.E. Sala, and K.M. Havstad. 2015. Rangeland ecosystem services: Shifting focus from supply to reconciling supply and demand. *Frontiers in Ecology and the Environment* 13: 44–51.
- Zhao, B., U. Kreuter, B. Li, Z. Ma, J. Chen, and N. Nakagoshi. 2004. An ecosystem service value assessment of land-use change on Chongming Island, China. *Land Use Policy* 21: 139–148.

Open Access This chapter is distributed under the terms of the Creative Commons Attribution-Noncommercial 2.5 License (<http://creativecommons.org/licenses/by-nc/2.5/>) which permits any noncommercial use, distribution, and reproduction in any medium, provided the original author(s) and source are credited.

The images or other third party material in this chapter are included in the work's Creative Commons license, unless indicated otherwise in the credit line; if such material is not included in the work's Creative Commons license and the respective action is not permitted by statutory regulation, users will need to obtain permission from the license holder to duplicate, adapt or reproduce the material.

