

A site-based approach to delivering rangeland ecosystem services

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Abstract. Rangeland ecosystems are capable of providing an array of ecosystem services important to the wellbeing of society. Some of these services (e.g. meat, fibre) are transported to markets and their quantity, quality and value are established via a set of widely accepted standards. Other services (e.g. climate mitigation, water quality, wildlife habitat) do not leave the land, but are, in fact, most valuable when they remain *in situ*. Determining their quantity, quality and value presents a challenge that must be met if there is to be a credible, accessible ecosystem services market for rangelands. In this paper we describe some of the ecosystem services that may be extracted from rangelands, discuss their unique ecological nature and relate those unique ecological properties to soil and vegetation attributes that can serve as a basis for measurement, both quality and quantity. We suggest the use of a soil/vegetation-based system in which similar climate, geomorphology and edaphic properties are grouped into ecological sites based on their response to disturbance. Within each ecological site, a unique state and transition model describes the dynamics of vegetation and soil surface properties, provides state indicators (vegetation structure, soil properties), predicts ecosystem services that may be derived at multiple scales, and organises information related to management to achieve ecosystem service objectives, including sustainability.

Additional keywords: ecological site descriptions, state-and-transition models, environmental markets.

Introduction

Ecosystem services are the benefits people gain from natural ecosystems (Millennium Ecosystem Assessment 2005). While the extraction (and appreciation of its importance) of food and fibre from land is as old as human societies, the notion of a much broader array of services, necessary for human existence and for enhancing human welfare, from ecological systems is gaining increasing attention (e.g. Daily 1997; Daily *et al.* 1997). As we have gained an increased understanding of the importance and complexities of ecosystem functions, the explicit links between these functions and the wellbeing of humans and their institutions has spurred interest in defining, measuring and valuing ecosystem services as a means of assigning at least partial value to nature.

Interest in rangelands has increased due to: (1) an expanded array of valuable goods and services provided by rangeland ecosystems; and (2) the understanding of the important link between the actions of rangeland managers and the mix of ecosystem services provided (Havstad *et al.* 2007). Because most rangelands are dominated by native vegetation and are managed largely without the homogenising effects of intensive inputs, many of the ecosystem services they provide are unique and would be very expensive, if not impossible, to replicate in other types of land-use and management systems.

In this paper we will examine some of the background and necessary framework for delivering an increasingly broad array of rangeland ecosystem services to complex markets. This delivery requires a credible alignment of measurement technologies, estimation across broad scales and valuation. Although the focus of our paper is primarily how to cost-effectively reconcile the relatively precise small-scale measurements with the need for delivery of large-scale attributes, we also include a discussion of approaches to valuation (setting a price) as context for evaluating how we might deliver ecosystem services to emerging markets.

An overview of ecosystem services

Human societies benefit from a wide range of products and processes that result from the functioning of natural ecosystems. Over the last decade, scientists, managers and policy makers have made considerable progress in defining ecosystem services and communicating their importance to the general public (e.g. Costanza *et al.* 1997; Balmford *et al.* 2002; Cork *et al.* 2002), including the important fact that conservation of natural ecosystems may provide services to society that are considerably cheaper than man-made alternatives (e.g. Kroeger and Casey 2007). The Millennium Ecosystem Assessment (MA) (2005) was a critical point in the effort to define both the array and the

importance of ecosystem services. The MA defined ecosystem services in four broad categories: provisioning, regulating, cultural and supporting. Provisioning services are generally the most widely recognised because of their role as directly consumable agricultural commodities (food and fibre). The three remaining: regulating, cultural and supporting ecosystem services are inherently more difficult to value and measure, but are increasingly recognised to be important in contributing to the total value of rangeland ecosystems. In the following overview of ecosystem services we select some examples of the different types of ecosystem-sourced goods and services and discuss the challenges for developing appropriate measurements.

Throughout the world, a large proportion of livestock and wildlife production for both food and fibre is directly dependent upon use of rangelands (FAO 2006). As a result, the provisioning ecosystem service most commonly associated with rangeland ecosystems is livestock production. However, while forage production from rangelands remains an important part of livestock production systems, an increasing use of managed pastures (including exotic species, use of fertiliser and/or irrigation) and supplemental forages has reduced the reliance on native rangelands and improved managers' abilities to respond to variability in environments and markets. Although global demand for beef is increasing, higher fossil fuel prices and a consumer preference trend towards less intensive livestock production (i.e. grass-fed beef) will maintain livestock production as an important, if not primary rangeland ecosystem service (FAO 2006).

Two additional and important provisioning services are genetic resources and energy production. Rangeland genetic resources (especially plant materials) are the basis for many restoration efforts in a wide variety of land-use systems (Whisenant 1999). Rangelands are frequently the source for plant genetic materials intended for use in conservation and restoration of degraded ecosystems. In addition, many rangeland plant species are highly desirable as ornamentals for gardening and landscaping. As more plants and animals are included on threatened and endangered species lists and subject to preservation and recovery plans, the value of rangelands as a source of genetic material is increasing in importance (West 1993; Woinarski and Fisher 2003).

Often overlooked as an ecosystem service provided by rangelands, energy has taken on an increased importance as emphasis on renewable sources has emerged as a land use. In particular, the development of solar and wind farms and their associated infrastructure has had an intensive, albeit spatially limited impact on rangeland ecosystems. Rangelands have long been affected by the infrastructure associated with traditional fossil energy (oil, gas, coal) extraction, but the increased intensity of road networks, well pads and storage and transport facilities has greatly altered some rangeland ecosystems (e.g. Sustainable Rangeland Roundtable 2008; Doherty *et al.* 2011). The networks supporting the extraction and delivery of rangeland provisioning services are relatively well developed and the markets for these services have well defined and well regulated standards. However, the effects of energy extraction infrastructure networks on other ecosystem services are poorly described and represent an important challenge for rangeland management. These energy extraction networks differ from more traditional rangeland

ecosystem services primarily in their intensity and in the mechanism for delivery, rather than in their fine-scale impacts.

Although pastoralism and related agricultural activities are likely to remain the dominant part of the economic, social and ecological fabric of rangeland landscapes, their relationship to other ecosystem services will inevitably change. For example, the lure of an agrarian lifestyle, even a simulated one, appears to be increasing as a motivating factor for people interested in acquiring rangeland (Barr *et al.* 2005; Tanaka *et al.* 2005). This demand for land, especially proximate to 'amenity-rich' urban centres, rural areas of significant scenic or recreation attraction, and transportation corridors is increasingly driving rural land prices higher and reducing the size of properties (Torell *et al.* 2005). While livestock production may remain the most economically important ecosystem service, the emergence of other services (e.g. climate regulation, water, genetic resources) and their interactions will drive many management decisions in the future. Provisioning services may also interact with other ecosystem services which are often non-monetary in nature and include tradition, family, and lifestyle considerations. In a survey of ranchers grazing public lands in the western United States, Gentner and Tanaka (2002) identified two primary groups of ranchers: hobbyists and professionals, each group comprising ~50% of the total number of survey respondents. For both groups of ranchers, lifestyle motives outranked profit motives as a principal reason to own land and livestock. Based on the trend towards valuing multiple ecosystem services as a basis for rangeland ownership and management, we feel it is safe to say that the measurement of regulating, cultural and supporting systems will be an important element of future rangeland values and an increasingly important focus for rangeland professionals.

Regulating ecosystem services, such as the sequestration of carbon (C) by plants and soil to contribute to climate stabilisation and pollution control, are gaining an increasing interest from land managers, markets and policy makers. Because C sequestration in rangeland soils to offset the accumulation of CO₂ in the atmosphere is currently the most developed application of a regulating ecosystem service, we will focus our discussion on the opportunities and challenges there.

Rangelands contain a substantial portion of the global terrestrial C pool, both in soils and vegetation. Estimates are that 3.7 billion ha of rangeland and grassland globally contain 306–330 Pg (Pg = 10¹⁵ g = 1 000 000 000 t) of organic C and 470–550 Pg of inorganic C, which is 20–25% of the global terrestrial C (Kimble *et al.* 2001) with the potential to increase sequestration as much as 0.3 Pg C/year (Lal 2004). While these estimates for rangeland sequestration potential assume that soil C levels on any particular site will come into equilibrium ~25–30 years after treatment, the national or global potential will be realised on a time scale approaching 50–60 years given the variable rates in technology uptake. Although these cumulative estimates are exceptionally large, most measured rates of C flux in intact rangeland ecosystems do not exceed 0.10 t C/ha.year (Svejcar *et al.* 2008). However, changes in land uses that are commonly observed in rangelands can have a more substantial impact on C stored within soils than changes in land management *per se*. For example, conversion of cultivated soils to perennial grass cover can increase soil C >1 t C/ha.year (Lal *et al.* 1998) over periods exceeding 20 years. While the conversion of

cultivated land to perennial rangeland cover may not be common in developing areas, land retirement programs such as the Conservation Reserve Program in the United States (~14 million ha) and the changes projected as a consequence of implementing the Murray–Darling Basin Plan (MDBA 2010) make the management and accounting for land conversion an important part of national greenhouse gas management planning, policies and accounting. On the negative side, C losses due to conversion of rangelands to cultivation may exceed 20 t C/ha at the time of conversion (Lal *et al.* 1998).

Although the release of C is much slower, the negative effects of desertification in arid areas can contribute significant amounts of C to the atmosphere. Losses of organic and inorganic C in arid lands can exceed 1 t C/ha.year (Monger *et al.* 2009). In general, if a rangeland ecosystem is degraded due to chronic overgrazing, soil C is lost to the atmosphere through accelerated wind and water erosion, which can result in the loss of organic C at a rate up to 1 t C/ha.year over 20–25 years (Schuman *et al.* 2002). Evaluating the dynamics of C sequestration as a result of invasive and/or exotic woody plants is complicated. Woody plants can access sources of water and nutrients otherwise inaccessible to grasses, which may stimulate productivity, increasing levels of ecosystem C (below- and aboveground) even though ecological processes are altered and other valuable ecosystem services are degraded (Asner *et al.* 2004).

Achieving the substantial potential of rangelands to provide regulating ecosystem services, using C sequestration as a test case, requires simultaneously achieving three objectives: managing intact systems to increase C at relatively low rates, avoiding large and significant losses of C to land-use conversion and landscape degradation, and restoring depleted and degraded rangelands to some level of functionality. Accounting for regulating ecosystem services will require a system that can predict and detect relatively small changes per unit area, track changes in land use and management and adjust for changes in driving variables.

Cultural ecosystem services are those benefits that people largely obtain through spiritual enrichment, cognitive development and aesthetics, perhaps best summed up as ‘wellbeing’. This particular class of ecosystem services is perhaps the most difficult to quantify because it is extremely subjective. On the other hand, cultural services can be the most ardently pursued by various rangeland stakeholders and also can be the source of considerable controversy and conflict. For example, both the United States and Australia have strong indigenous people’s movements that have identified cultural ecosystem services in their claims to ownership or access to rangeland resources and landscapes. Further confounding the measurement and valuation of cultural ecosystem services are measurable, but inconsistent, external attributes linked to history and proximity (Tomblin 2009). In many cases, currently accepted standards for monitoring rangeland health (e.g. biotic integrity, soil surface stability, hydrologic function) are of secondary importance in determining the value or amount of cultural ecosystem services that a given set of landscape resources can provide (BLM 2005). Nevertheless, such measures of rangeland health or ecological processes may, in many cases, usefully serve as a baseline or departure point for ecosystem management that also meets cultural objectives (Banzhaf and Boyd 2005). Because of the

highly variable and subjective nature of defining cultural ecosystem services, measuring and valuing them will be inherently difficult and likely to be beyond the expertise of the majority of rangeland managers. However, the link between the continued access to and appreciation of these services and the conservation and maintenance of ecosystems within which these services are contained means that rangeland management will be important and will be a necessary part of any cultural service delivery.

Because most rangelands are located in relatively arid areas, the supporting ecosystem service of water cycling is usually the most immediate link to the human populations in these areas (Havstad *et al.* 2007). The partitioning of the water that moves through rangeland ecosystems, and the ultimate delivery of that water to competing consumers, is a consequence of the interactions of many relatively static factors (climate, topography, soils and geology) and dynamic factors (land management). While the static influences are relatively fixed, soil and vegetation management practices can have significant effects on hydrologic processes. For example, attributes such as soil surface structure and vegetation cover (spatial pattern and species composition) can respond rapidly to management and have significant effects on catena and watershed-scale hydrology (Thurow 1991). In highly modified rangeland landscapes (e.g. energy development, hobby pastoralism, low-density residential development), the impacts of roads on hydrology can be significant because the road networks alter drainage patterns and concentrate flow.

Attributes of soils and vegetation may also affect how water that infiltrates the soil is partitioned. Soil water can be used for plant production, lost to evaporation from the soil or enter groundwater (Wilcox and Thurow 2006). Most rangeland management guidelines are built around assumptions of the desirable increase in herbaceous vegetation at the expense of woody vegetation, with the outcome of an ‘improved’ hydrologic cycle consisting of increased infiltration, greater water use for grass (forage) growth and increased groundwater recharge. An important tenet of these guidelines is that water accessed and evaporated by the deep-rooted shrubs and trees is lost from the system in an undesirable manner. While this pattern holds reasonably well for more mesic rangelands, the effects of soil × vegetation interactions in arid and semiarid systems are highly variable. Thus, projections of the benefits (especially increased water yields) from rangeland management for increased herbaceous dominance may be seriously flawed (Wilcox and Thurow 2006). The provision of water as a supporting service from rangeland watersheds needs to be evaluated at and across multiple spatial scales before we can compare alternative management scenarios, develop policies and design programs for implementation.

One particularly cogent example of the importance of these types of ecosystem service tradeoffs and the relative uncertainty of the link between management actions and objectives is the proposal, by the Australian Government via the Murray–Darling Basin Authority, to reallocate a substantial portion of the water in the basin from agricultural production (provisioning service) to environmental stabilisation and restoration (supporting services). Regardless of the outcome of this decision-making process and its subsequent implementation, the effects on human populations both within and beyond the basin will be significant. Although it is

highly unlikely that rangeland professionals can precisely predict the outcomes of specific decisions and actions, it is imperative that the science behind those decisions be transparent and well communicated.

Theoretically, rangeland value is determined by the net sum of the value of all ecosystem services derived from that piece of land. Of course, some ecosystem services (e.g. livestock production), are relatively easy to measure (i.e. number of animals sold and their weight at auction) and value (i.e. livestock prices); some (e.g. C sequestration) are difficult to measure, but easy to value (i.e. global market prices); some (e.g. crop genetic diversity) are easy to measure, but difficult to value (i.e. cultural significance) and some (e.g. water yields) are both difficult to measure and value. The challenge to the rangeland profession in the coming decades is to develop transparent systems for measuring rangeland ecosystem services and communicating those measures to the public, policy makers and individual land owners and other residents. In some cases, the value of even precisely measured ecosystem services will be transparent, but for many others the value will always remain in the eye of the beholder and will change over time. Moreover, some services are intermediate services (e.g. nutrient cycling, habitat provision) to the provision of more tangible environmental products and services (e.g. crops, hunting opportunities) and problems of double counting of services values can be significant (Boyd and Banzhaf 2006; Kroeger and Casey 2007).

Delivering ecosystem services

There is a general saying in management circles that 'if it cannot be measured, it will not get done', and this is certainly true of the challenge for conserving or expanding the provision of ecosystem services from rangelands. The general concept of ecosystem services was largely advanced by ecologists to draw attention to the benefits of conserving natural ecosystems and ecological processes, particularly in the face of rapid economic development and human population growth that was seen to be putting many of these systems and processes at risk (Foley *et al.* 2005; Millennium Ecosystem Assessment 2005). However, as noted previously, common definitions of ecosystem services typically include a wide array of natural processes, environmental products and human benefits that are frequently lumped together and, therefore, hard to specifically quantify, let alone attribute economic value to (e.g. Kroeger and Casey 2007). But without actively seeking to ensure such quantification or valuation the likelihood of optimal levels of management actions being taken to conserve or expand the level of ecosystem services generated within rangelands is necessarily limited.

Quantification and valuation of ecosystem services both present unique challenges and a growing literature now exists on both the theoretical and practical nature of these challenges and options, especially for creating viable commercial markets for services (e.g. Murtough *et al.* 2002; Whitten and Shelton 2005; Kroeger and Casey 2007). It is beyond the scope of this paper to address the complexities of the many issues associated with setting a value for the ecosystem services that are necessarily canvassed within this literature. We will, however, discuss some of the approaches to valuing services that are relevant to

rangelands to illustrate the economic framework in which ecosystem services must be delivered.

Putting a value on ecosystem services

Considering first the scope for valuation of ecosystem services, a broad challenge lies in these services providing a mix of 'market' and 'non-market' benefits which may be further apportioned into 'use' and 'passive' values (Kroeger and Casey 2007). Related to the possession of 'market' values is also the issue of many ecosystem services having attributes of 'public goods' and the ability of individual range managers to capture economic benefits for services provided (e.g. Murtough *et al.* 2002).

Market and use values are more likely to be able to be captured in a formal market than non-market (by definition) and passive values for which markets generally do not exist or cannot be created without detailed monitoring protocol or high transaction costs. An example that captures each of these elements in one ecological asset may be ground cover provided by perennial grasses (Kroeger and Casey 2007), these grasses provide direct market and use values through grazing, agistment or sale of hay, but they also provide non-market values through aesthetic appeal, habitat for ground-dwelling fauna, and simple stewardship or existence values of knowing that the natural pasture is there as opposed to bare ground or an exotic weed. The ground cover may also provide a filtering service to purify water entering watercourses or recreational amenity which may or may not have direct use or passive values that may be valued with varying degrees of difficulty through damage mitigation estimates (e.g. Greig and Devonshire 1981), willingness to pay valuation (e.g. Mitchell and Carson 1989; Rolfe and Prayaga 2007) or stated preference valuation techniques (e.g. Sinden and Worrell 1979; Rolfe *et al.* 2008). Critically, not all environmental resources provide both market and non-market benefits and, regardless of joint possession of these benefit types, the relative scale of market and non-market benefits from particular tracts of rangelands will vary according to a variety of factors, including the size and richness of the local resource endowment, ecological health of the resources *in situ*, local land uses, adjacent land uses and opportunities for substitutes to provide similar services. Of note, while non-market benefits from natural ecosystems do present challenges for valuation, the limited available studies of broad-acre agricultural landscapes, do suggest that these benefits may be substantial (e.g. Lockwood *et al.* 2000) and the continued pursuit of appropriate valuation techniques worthwhile to promote optimal rangeland resource use.

While efficient markets can yield values for basic commodities (e.g. livestock) that reflect their private and public benefits, this is often predicated on their being reasonably fungible (i.e. readily substituted with like commodities across time and space). Many ecosystem services are non-fungible and do vary considerably in value with time and between locations, meaning that their values are highly context-dependent. For example, the presence of local habitat that supports pest-controlling biological agents will vary between land uses (e.g. native pastures, sown pastures or crops), location of the service (within, adjacent to or at a distance from the beneficiary land use) and timing of the service (e.g. fallow, plant emergence, flowering or senescence), and the availability and cost of substitutes (e.g. fire, mechanical or chemical control), and

yield potential of the different land uses. As a consequence, the empirical estimates of ecosystem service values yielded so far necessarily relate to specific discrete sites and contexts and cannot be readily transferred to other sites and contexts without thoughtful qualification (e.g. Kroeger and Casey 2007). Serious attention is being applied to refining techniques for 'benefits-transfer' between sites and contexts (e.g. Brouwer 2000; Kumar 2005) although the methods remain largely experimental and pragmatic, ranging for example, from constant single-point estimates, marginal-point estimates, multi-attribute benefit transfer functions, to meta-value analyses which pool the results of several different valuation studies (e.g. Rolfe 2006).

Ecosystem processes commonly operate over large spatial scales and, as a result, many ecosystem services other than the more immediate use services (e.g. livestock) possess the general characteristics of 'public' goods (e.g. Murtough *et al.* 2002). Such goods once provided can be enjoyed by many parties most of whom cannot be expected to contribute to their provision with the effect that markets, and hence values, are difficult to create. Moreover, because consumption of public good ecosystem services is essentially non-exclusive, not only is there limited incentive for individuals to conserve or promote such services, the incentive to maintain quality levels in the services will also be limited, further weakening the scope to establish viable markets without public intervention backed possibly with mandated minimum standards (e.g. Kroeger and Casey 2007). Care, however, needs to be taken in setting service values based on mandated standards because these typically become the upper bound on service quality (e.g. Boyd and Banzhaf 2006) and the issue remains of uniform standards or values failing to acknowledge the spatial heterogeneity of the service-generating potential of large rangeland landscapes.

To conclude, market based-valuation of some specific ecosystems services is both desirable and occurring. This is presently the case for the more functional provisioning services associated with the production of commodities such as livestock, timber products, extracted gravel and other marketable produce, but also for some supporting services such as vegetation type, soil fertility, shade, and availability of water for stock or irrigation, which can typically be captured through the private market value of rangeland. For harder to measure and poorly defined services, and especially those strongly characteristic of public goods with non-exclusive consumption potential and poorly defined property rights and those that lack uniform quality or have high prospective market transactions costs, the prospects for establishing value reward systems based on competitive markets and direct trading schemes remain poor. The alternative of some form of publicly created and regulated markets or compensating payment schemes would seem to be more likely to successfully encourage private rangeland managers to undertake actions to generate a broader range of desirable ecosystem services than might otherwise be the case. However, the effective management of such publicly supported compensation systems still requires measurement, monitoring and enforcement, but good initial design and implementation is critical (Kroeger and Casey 2007). Where the level and quality of a particular ecosystem service or suite of services is hard to measure (and hence monitor), is spatially heterogeneous, or is reasonably related to the type, level and quality of management inputs then a performance-based

incentive scheme centred on range managers undertaking specific activities would seem to offer genuine scope for conservation and expansion of ecosystem services.

Estimating ecosystem services

Quantification of the ecosystem services generated from rangelands requires a credible system for estimating the changes in a wide variety of ecosystem service types, both to provide market viability and to provide some incentive for adoption (Whitten *et al.* 2004; Boyd and Banzhaf 2006). In general, provisioning services leave the landscape and are concentrated in a well defined market place. Livestock or livestock products (wool, meat, hides, genetic resources, etc.) are typically transported and concentrated in regional markets centres where buyers, sellers and commission agents have *a priori* agreed to well understood criteria for defining the quality and quantity of the commodity that have evolved over long timeframes. Even in situations such as livestock markets where live animals are intended for slaughter and the exact outcomes (e.g. meat yield, carcass grade) of the process are unknown, there is usually a clear understanding that any risk is assumed by either the buyer or seller on pre-agreed terms and, importantly, can be managed. However, the products that are associated with the majority of regulating, cultural and supporting services rarely share the same level of definition and understanding of trading rules and risks.

An important assumption in measuring most ecosystems services derived from rangelands, particularly regulating, cultural and supporting services, is necessarily that direct measurement of the services is not a viable option. The inherent high spatial variability, relatively slow rate of change, extensive nature and low productivity of most rangeland ecosystems and management units precludes the implementation of a direct measurement-based protocol (Brown and Sampson 2009). Even when relatively inexpensive and accurate technologies for ground-based direct measurement are available to estimate site attributes that can be used to predict the presence and level of certain ecosystem services, it will seldom be cost-effective to deploy those technologies at the scale of a whole property or project.

Even though the commodity of land itself is relatively well defined, the variety of goods and services extracted from land for human uses are much less consistently appraised. The American Congress on Surveying and Mapping defines surveying as the 'science and art of making all essential measurements to determine the relative position of points and/or physical and cultural details above, on, or beneath the surface of the Earth, and to depict them in a usable form, or to establish the position of points and/or details' (ACSM 1994). Even protocols for describing land components (i.e. soils, vegetation, improvements) are relatively well established and have legal standing. A systematic approach for describing the dynamics of ecosystem components and the accompanying changes in ecosystem services is necessary to optimally define and usefully exploit the wide array of rangeland ecosystem services. In particular, an approach that ensures at least a minimum level of ecosystem resilience (sustainability) while providing an optimal, but highly variable, mix of standardised goods and services to a variety of customers and markets starts with the measurement of

ecosystem services and their link to ecological processes (Boyd and Banzhaf 2006).

Fortunately, rangeland ecology and management research and practice have provided the basis for developing transparent (if complex) links among various ecosystem components, management actions and the generation of ecosystem services (Fig. 1). The ecological processes that are associated with the five key elements of rangeland landscapes – environmental drivers (e.g. climate), soils/geology, resource redistribution, transport vectors, and historical legacies – interact to determine vegetation structure and dynamics with resulting effects on various ecosystem goods and services (Havstad *et al.* 2007). Rangeland management practices can be used to mediate those interactions and change the vegetation structure and dynamics within the limits that are imposed by these key elements. Those management actions and their effects on vegetation structure and dynamics will, in turn, influence the provision of ecosystem goods and services from a particular site. While this framework does provide a useful explanation of the effects of ecological processes and management on ecosystem services, it does little to provide a systematic basis for the estimation of ecosystem services that is so necessary. In the following section we offer a suggestion for addressing this deficiency.

Ecological site descriptions

For more than a century, range resource professionals have used the site concept to classify landscape components, describe ecological dynamics and interpret the scope for management options (Brown 2010). Although the site as a unique landscape component has a long history of employment in resource management, a current approach describing those interactions employs Ecological Site Descriptions (ESD) as a tool for informing land management decision making (USDA-NRCS 2003). Looking across any rangeland landscape it is apparent

that some parts are different to other parts in terms of the types and amounts of vegetation present. To capture this natural variation across landscapes the different parts are classified into units known as ecological sites, which are described as ‘a distinctive kind of land with specific characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation in response to natural disturbance and management actions’ (Bestelmeyer and Brown 2010). An ESD is a document that describes the ecological site and is composed of four elements:

- (1) Site context – a description of the biophysical elements of the site (climate, geology, soil properties, topography, hydrology) and their interactions as controlling factors (Duniway *et al.* 2010);
- (2) Plant communities – a state and transition model (Fig. 2) that describes the vegetation structure dynamics of the landscape subunit, including all the potential plant communities that may occupy the site including the indicators and attributes of each state (Bestelmeyer *et al.* 2010);
- (3) Site interpretations – interpretive management information describing the ecosystem services associated with each state and the management actions necessary to maintain or achieve each state (Gilgert and Zack 2010; Moseley *et al.* 2010); and
- (4) Supporting information – a literature covering sources of information and technical data utilised in developing the site description and the relationship of the site to other technical sites.

The first three components of an ESD reflect the components of the ecological process model presented in Fig. 1 and organised systematically into a state and transition model framework. Using an infrastructure based on ESD to support an ecosystem service approach to rangeland management would seem to offer several advantages.

First, the ESD approach allows for a unique set of ecosystem services or ecological processes to be assigned to specific

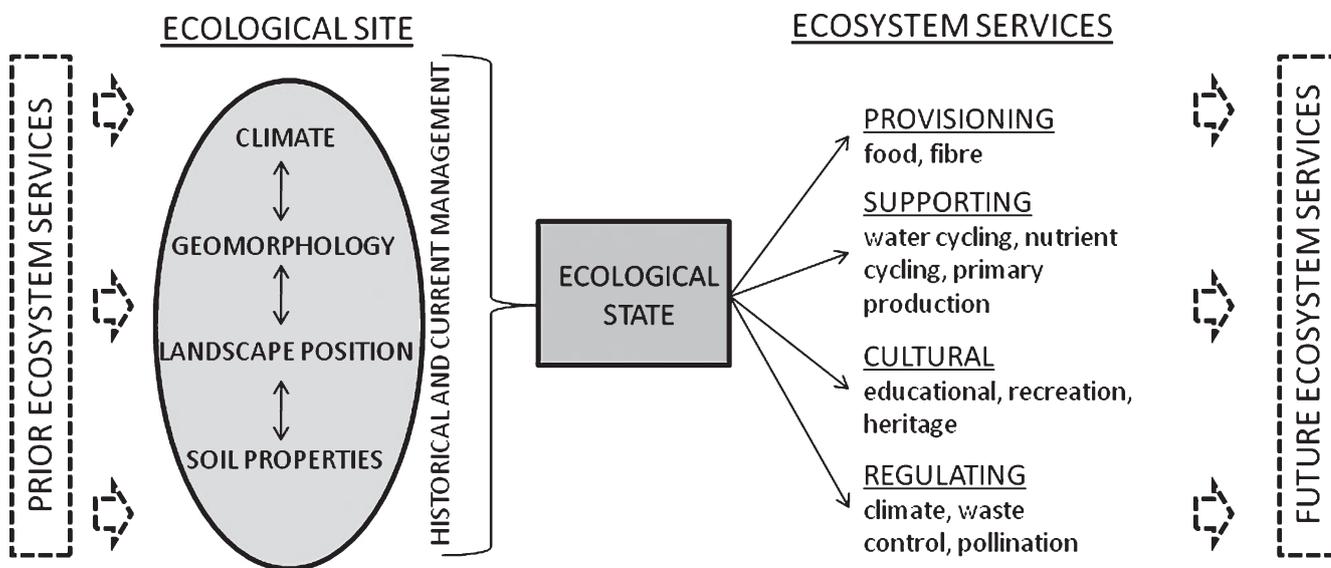


Fig. 1. Ecosystem services in rangeland landscapes are determined by the interactions of soils, geomorphology, climate and landscape interactions, modified by historical and current management effects on the ecological state (soil × vegetation interactions). Solid lines illustrate extant links, dotted lines illustrate prior and future alternatives (after Peters *et al.* 2006).

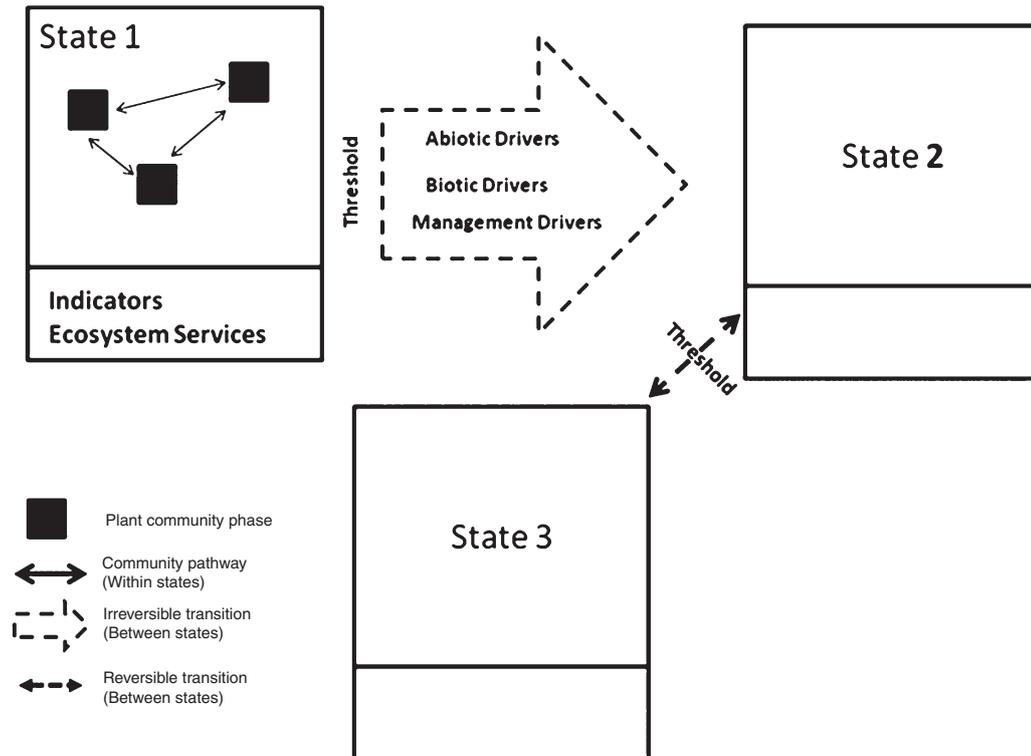


Fig. 2. A diagrammatic representation of a state and transition model. States are defined as recognisable, relatively resistant and resilient complexes with attributes that include a characteristic climate, the soil resource including soil biota, and the associated aboveground plant communities. A transition is the trajectory of system change between states that will not cease before the establishment of a new state. Different vegetative assemblages within states are referred to as plant communities and the change between these communities as community pathways (after Stringham *et al.* 2003).

rangeland units. The unique set of ecological processes associated with the soil : vegetation structure of each state can be used to infer a set of ecosystem services. In many cases, there are sufficient empirical field measurements to provide a solid basis for estimation. For other locations, the use of mathematical models can be used to estimate the amount of an ecosystem service and the uncertainty associated with the estimate (Brown *et al.* 2010). Regardless of the accuracy of the estimate of an ecosystem service by site/state, state and transition models provide a transparent means of making the estimate and allows for quantification and communication of risk.

For types of ecosystem services that do not require detailed knowledge of large-scale interactions, but merely to accumulate attributes on a per-unit area basis (e.g. forage production, C sequestration), a relatively simple approach is to aggregate the results of all the sites within the area of interest. For more complex ecosystem services that do require knowledge of the interactions of various landscape components (spatially explicit interactions), basic community-scale information can be used to parameterise models as a means of estimating ecological process change and the effects on ecosystem services at larger scales.

Second, the state and transition models (Fig. 2) within an ESD can be used for a variety of interpretations related to quantifying ecosystem services on a site. States within ecological sites are typically represented by changes in the structure of vegetation (i.e. grass : shrub ratio). These relationships are detectable using

a variety of protocols (including remote sensing) that reflect the indicators for each state (Fig. 2) of each site/state (Karl and Herrick 2010), which can dramatically reduce monitoring costs. If soil and vegetation attributes are assigned to sites/states as part of the ecological site development process (Bestelmeyer *et al.* 2010), those attributes can be linked to a variety of remotely sensed attributes and serve as a means of detecting changes in state over time across quite large areas. The drivers described in the transitions can be used to predict potential state changes in response to changes in climate and management (Fig. 2). If land managers enter into contracts with either private or public sector buyers to increase the amount of ecosystem services provided by a defined rangeland unit by moving from one state to another, the management plans in the contractual agreements can contain the specific practices that must be employed, guidelines for planning and protocols for verification (Chicago Climate Exchange 2009).

Third, using an ecological site system as a basis for defining and estimating the amounts of ecosystem services provides an explicit link between management and research. ESD can be used to provide a set of explicit hypotheses about the effects of management on ecological processes and attributes. These hypotheses can be tested and refined with field measurements in controlled experiments. The challenge is to examine the range of hypotheses and develop networks of sites and experiments that will be most cost-effective in resolving unknowns. This approach

will require a hierarchical organisation and analysis to ensure that limited resources and time are used to greatest effect (Brown 2010).

ESD were conceived and are largely developed as planning and decision-making aids, giving rangeland managers, advisors, verifiers and markets the same information on which to base decisions. While there may be disagreements about any particular aspect of an ESD and the ecosystem services that flow from it, a transparent and repeatable framework such as ESD can be used in structuring contracts, guiding management, calculating changes and verifying compliance. If there are disagreements, the resolution of those challenges can be accomplished via accepted experimental research approaches.

An ESD-based approach is not without challenges. First and foremost is the need for a credible map of soil properties/ecological sites at a scale sufficient for planning, decision making and implementation of rangeland management practices. Even if the maps are relatively coarse-scaled, the underlying principles of mapping must be elucidated to provide a basis for correlation and verification. An ESD without distribution rules has little use to planners, policy makers or land managers. While the current applications of ecological sites have been largely confined to areas where there is a correlated soil survey, emerging (and proven) technologies such as digital soil mapping (Sanchez *et al.* 2009) can be employed to create accurate, relatively fine-scale maps of soil properties that can be correlated with ecological sites using well developed, transferrable protocols (Bestelmeyer *et al.* 2010). Even though the employment of such advanced technologies is not without significant costs and expertise requirements, the cost:benefit ratios are likely to be very favourable. Another challenge is the substantial requirement for an accessible information system to house attribute data that will allow access to a variety of users (Talbot *et al.* 2010). If an ESD-based ecosystem services approach is implemented, all parties to any transaction must have access to the same information on a timely basis. The challenges will require a coordinated approach across state and national boundaries with a commitment of time and resources. The challenge of managing rangelands in a changing climate also extends to the delivery of ecosystem services. Using an ecological site-based system which defines the climatic variables assumed to be drivers of both landscape organisation and dynamics will allow for an improved prediction of the effects of climate change in the delivery of ecosystem services.

Conclusion

An ecological site-based system, used as the basis for a systematic approach for assigning ecosystem services to specific units of rangelands, solves many of the problems associated with the estimation of attributes of large and highly heterogeneous ecosystems. Using such an approach provides a solid scientific foundation, linking ecological processes (and drivers) to ecosystem outputs. Agreement upon a system by all parties involved in a transaction provides a transparent basis for markets, allowing buyers and sellers to evaluate and manage risk. A systematic approach can also assure that markets are in the public interest. However, the success of an ecological site approach as a basis for informing and supporting ecosystem services markets

will be entirely dependent upon the underlying support of a systematic soil and vegetation mapping effort, a research and development network to provide basic information and a network of science and management organisations to provide interpretations in response to changes in the underlying assumptions about driving variables and their relationship to ecosystem services.

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