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What can ecological science tell us about opportunities for carbon sequestration on arid rangelands in the United States?

Kayje Booker [a], Lynn Huntsinger [a,*], James W. Bartolome [a], Nathan F. Sayre [b], William Stewart [a]

[a] Department of Environmental Science, Policy, and Management, 133 Mulford Hall MC 3110, University of California, Berkeley, CA 94720-3110, United States
[b] Department of Geography, 507 McCone Hall MC 4740, University of California, Berkeley, CA 94720-4740, United States

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ABSTRACT

Scientific interest in carbon sequestration on rangelands is largely driven by their extent, while the interest of ranchers in the United States centers on opportunities to enhance revenue streams. Rangelands cover approximately 30% of the earth's ice-free land surface and hold an equivalent amount of the world's terrestrial carbon. Rangelands are grasslands, shrublands, and savannas and cover 312 million hectares in the United States. On the arid and semi-arid sites typical of rangelands annual fluxes are small and unpredictable over time and space, varying primarily with precipitation, but also with soils and vegetation. There is broad scientific consensus that non-equilibrium ecological models better explain the dynamics of such rangelands than equilibrium models, yet current and proposed carbon sequestration policies and associated grazing management recommendations in the United States often do not incorporate this developing scientific understanding of rangeland dynamics. Carbon uptake on arid and semi-arid rangelands is most often controlled by abiotic factors not easily changed by management of grazing or vegetation. Additionality may be impossible to achieve consistently through management on rangelands near the more xeric end of a rangeland climatic gradient. This point is illustrated by a preliminary examination of efforts to develop voluntary cap and trade markets for carbon credits in the United States, and options including payment for ecosystem services or avoided conversion, and carbon taxation. A preliminary analysis focusing on cap and trade and payment for avoided conversion or ecosystem services illustrates the misalignment between policies targeting vegetation management for enhanced carbon uptake and non-equilibrium carbon dynamics on arid United States rangelands. It is possible that current proposed carbon policy as exemplified by carbon credit exchange or offsets will result in a net increase in emissions, as well as investment in failed management. Rather than focusing on annual fluxes, policy and management initiatives should seek long-term protection of rangelands and rangeland soils to conserve carbon, and a broader range of environmental and social benefits.

1. Introduction

Rangelands are one of the most widely distributed landscapes in the world. Found at the more arid end of the earth's climates, approximately 30% of the ice-free global land surface can be considered rangeland (FAO, 2009), although estimates vary widely depending on the particular definition used (Lund, 2007). In turn, rangelands are thought to have as much as 30% of terrestrial carbon stocks (Schuman et al., 2002; FAO, 2009). Debates about the impacts of livestock grazing, climate change, and cultivation on rangelands now include concerns about their effects on carbon cycling. Interest in increasing carbon flux from the atmosphere into the soils and vegetation of rangelands in the United States has led to a number of national policies and market-based projects designed to encourage management that enhances this flux (McCarl and Sands, 2007). It is now commonplace to use the rationale of increasing carbon sequestration to argue for changes in grazing management. Focusing on the U.S., we argue that, given recent developments in the scientific understanding of rangeland ecological dynamics, grazing management strategies and associated management practices cannot lead to reliably increased capture of carbon on many arid rangelands. For this reason, policies for such rangelands that are based on additionality are unlikely to be effective, and may even lead to increased emissions.

Proposals for managing rangelands for climate change mitigation are gaining attention at state and federal levels in the United States. Primarily because they are so extensive, the 312 million ha of U.S. rangelands (USFS, 1989), defined here as grassland,

* Corresponding author. Tel.: +1 510 685 1884; fax: +1 510 643 5098.
E-mail addresses: kayje@berkeley.edu (K. Booker), huntsinger@berkeley.edu (L. Huntsinger), jwbart@berkeley.edu (J.W. Bartolome), nsayre@berkeley.edu (N.F. Sayre), billstewart@berkeley.edu (W. Stewart).

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shrubland, and savanna, contain significant carbon stocks. Traditional land use, largely grazing, does not involve tillage, potentially resulting in less soil carbon loss than that connected to cultivation (Uri and Bloodworth, 2000). It has been estimated that grazing lands contribute about 15% of U.S. soil carbon sequestration potential (Lal et al., 2003). U.S. rangeland livestock producers, generally operating with low and variable financial returns, continue to express considerable interest in diversifying income streams to include payments related to carbon sequestration (Diaz et al., 2009). Land management and conservation organizations also seek to promote management for increased carbon sequestration on private and public rangelands (Audubon California, 2012). As the U.S. failed to ratify the Kyoto treaty, the voluntary markets for trading carbon credits have thus far been the main thrust of initiatives for incentivizing management for carbon sequestration domestically.

While the specific applications are still contested, there is broad scientific consensus that non-equilibrium models better explain the ecological dynamics of arid rangelands, in the U.S. and throughout the world, than equilibrium models (Briske et al., 2005; Vetter, 2005). The ecological behavior of rangeland systems has been much debated and researched in the last twenty years, but it is not clear that what has been learned through investigation, experimentation, and theoretical development has been integrated into carbon sequestration initiatives and management recommendations. Further, a lack of information has led to over-generalized applications of scientific and traditional ecological knowledge despite the fact that such knowledge is linked to locales of specific environmental characteristics within rangeland systems. Just as different definitions of the term "rangeland" can lead to vastly different estimates of how much rangeland there is, over-generalization of ecological knowledge to areas of differing environmental parameters can lead to incorrect assumptions about potential management outcomes. Site specificity is important because rangelands are so widespread, temporally and spatially diverse, and diverse in structure and function.

Because synthesis of information about rangelands has suffered from poorly defined terms and variable usage, this paper begins with a definition of rangeland and a review of the development of explanatory rangeland vegetation change models and their linkage to ecological sites. Next, the interaction of rangeland ecological dynamics and management for carbon sequestration is analyzed. Finally, the implications of this science for carbon sequestration management and policy initiatives are presented and discussed, and we offer recommendations for rangeland carbon policies that accommodate recent developments in rangeland ecological science.

2. Rangelands and rangeland ecosystem dynamics

Rangelands have been defined as a type of vegetation, a land use, or what is left when other types are excluded. Definitions of rangeland that include specific uses, usually livestock grazing (NRCS, 1997; Holechek et al., 2010), are not a good basis for stable descriptions of extent or processes. Defining rangelands as "land not permanently ice and snow, urban, cropland, or forest" (Stoddard et al., 1975) does not identify what rangelands actually are. Defining rangelands as grasslands, shrublands, and savanna (Heady and Child, 1994) incorporates a wide range of communities from arid to semi-arid and can be distinguished from other more productive systems like woodlands, forests, wetlands, and croplands. These distinctions are essential for predicting and measuring carbon at the landscape scale. Included within this definition are what have been defined as "grazing lands" (NRCS, 1997; Follett and Reed, 2010) to emphasize the importance of large herbivore grazing, and intensively managed lands used for grazing that have been termed "pasturelands" (Holechek et al., 2010). Rangelands can be temporally transient, especially at the margins with forest, wetlands, and croplands (Heady and Child, 1994). Rangelands with sufficient rainfall, or suitable for irrigation, may be temporarily or permanently converted to cropland or forest. U.S. arid and semi-rangelands generally fall to the west of the 100th Meridian.

For nearly a century, the management of U.S. rangelands has been informed by predictive models for vegetation change linked to geographic areas known first as "range sites" and now as "ecological sites" (Brown, 2010). Early in the twentieth century, Sampson (1917) adapted the then new concepts of Clementsian plant succession into a model relating grazing pressure to vegetation change away from and towards an equilibrial "climax" of ideal plant species composition. This linear, deterministic model was used in developing a general framework for evaluating progress in sustainable livestock grazing and rehabilitation of deteriorated rangeland. The utility of this approach was greatly enhanced by the development of what was called the "quantitative range condition" model (Dyksterhuis, 1949), which measured range condition as the difference between the current species composition and productivity and the ideal climax state. What operationalized this approach was combining Sampson's ideas about species composition with newer theories of an edaphic climax to identify what were termed *range sites*, defined as rangeland areas with a similar potential climax state (SCS, 1976). This formed the basis for evaluating the "health" of rangelands and for informing grazing management.

The term *ecological site* replaced range site by the early 1990s (NRCS, 1994). This was more than just an alteration in terminology, as the change reflected significant advances since the 1980s in models describing succession. It has been found that non-equilibrium models better explain ecological dynamics then do equilibrium-based models, particularly when rangeland is at the arid end of a gradient from dry to mesic conditions (Briske et al., 2005; Vetter, 2005). Non-equilibrium or disequilibrium models posit that abiotic factors such as weather, soil structure, erosion, and water table depth are the dominant drivers of rangeland productivity and species composition (Ho, 2001), and that the relationship with livestock grazing is often non-linear (Westoby et al., 1989; Ellis and Swift, 1988). On arid rangelands spatial and temporal variation in water and forage resources is high, annual production is as unpredictable as rainfall and temperature patterns, and extremes of precipitation or temperature are not uncommon. Non-equilibrium models also posit the existence of multiple stable (within a management timeframe) vegetation states maintained largely by abiotic factors, rather than a single endpoint climax or stable equilibrium state (Westoby et al., 1989; Stringham et al., 2003) created mostly by biotic interactions, including grazing pressure. As a result, an ecological site is described more by climate, topography, and soils, than reference to a climax vegetation (Brown, 2010). Assessments of range condition have been largely decoupled from the use of linear distance to climax.

Westoby et al. (1989) provided an alternative approach to describing the dynamics of managed ecological sites using state and transition models which accommodate non-linear, non-equilibrium ecology and varied management objectives. Current use of ecological site by federal agencies emphasizes concepts of stable states and thresholds and utilizes recent advances in available soil information and Geographic Information System (GIS) technology (Brown, 2010). The more traditional goals of sustainable grazing management and enhanced forage production have been joined by the need to evaluate and anticipate response of rangelands to global change and the potential for carbon sequestration.

3. Rangeland ecosystem dynamics and carbon sequestration

Effective climate change mitigation strategies for rangelands must be based on knowledge of how management and land use interact with carbon cycling. Carbon sequestration is the movement of carbon stock, or stored carbon, from the atmospheric pool to other carbon pools, including soils and vegetation. In general, the movement of carbon between pools, usually measured and reported on an annual basis, is described as carbon flux, and management can influence both flux and stocks of carbon. Incentivizing management that increases the net flux of carbon from the atmosphere into soils and vegetation was a goal of the carbon credit trading market of the Chicago Carbon Exchange, and developing grazing management programs to create additional carbon sequestration is a common subject of research and speculation. Less attention has been given to the protection of carbon stocks already in soils.

Much of the political and scientific discussion on terrestrial carbon sequestration has centered on forests, where annual net flux can be significantly increased through management of tree growth and is easily measured. Yet, in terms of long-term carbon storage, rangelands can be superior to forests because relatively more of the total site carbon is stored in the soil (White et al., 2000; Paruelo et al., 2010) where it is usually better protected from atmospheric release than carbon stored in vegetation. However, carbon inputs in rangelands, i.e. net carbon flows from the atmosphere, are comparatively small, and soil carbon is more difficult to measure than carbon in trees.

Soil carbon stocks are dynamic and sequestration may be described as a function of inputs, losses, and storage time (Silver, pers. com). For example, plant residues and organic amendments add carbon, while carbon is lost through organic matter decomposition (Subak, 2000). Especially on more arid ecological sites, net primary productivity on rangelands is relatively low, limiting carbon input. Due to the rapid turnover of grasses and forbs, most carbon flux from the atmosphere into vegetation through photosynthesis is quickly lost to the atmosphere through decay. A small part of the carbon in the vegetation goes into the soil and is stored longer as soil organic matter. Increased exposure of soil to rainfall and the atmosphere by disturbance or tillage can enhance flux to the atmosphere under some conditions.

Although the relationship between soil carbon and plant production is not universal, low production, and, thus, low carbon inputs to the soil, necessarily limit annual carbon sequestration. Vegetation production in rangelands varies widely, in part because rangeland as a land classification masks major differences in climate and vegetation. In the United States rangelands include montane meadows, blue oak woodlands, valley grasslands, basin sagebrush, bluebunch wheatgrass prairie, bluestem and grama grass prairie, and warm desert. The wettest of these rangelands are capable of producing more than 3500 kg/ha per year, while the driest yield on average only 200–300 kg/ha (Huntsinger and Starrs, 2006). These variations are linked to the climate and geomorphology that define ecological sites (Parton et al., 1994). As might be expected, given this variability, estimates of the average carbon uptake on temperate rangelands vary widely.

4. Implications for management

Because rangeland vegetation mediates and constrains the carbon flux from the atmosphere into soils and plants, three major non-exclusive carbon management principles can be identified when rangeland ecological dynamics are considered. First, in rangeland ecosystems carbon flux into plants and soils is low, highly spatially and temporally variable, strongly influenced by stochastic events like weather, and largely outside the control of management. Second, in some rangeland environments, because of limited and slow plant growth, and significant storage of carbon in mineral form close to the surface, management that causes soil loss can significantly increase carbon flux to the atmosphere. Finally, carbon flows and pool sizes may be less variable and more amenable to enhancement through management at the less arid end of the rangeland climate gradient. These principles largely determine the outcome of carbon sequestration strategies in rangelands, and must be considered in assessing the ability to mitigate climate change through rangeland management.

In the United States, as in many parts of the world, the most productive rangeland ecological sites, such as tallgrass prairie, have largely been converted to crops; extant rangelands tend to be at the less productive and more arid end of the spectrum (Fig. 1). In these rangelands, the annual carbon flux into the soil is low, and even, at times, negative, especially during drought (Zhang et al.,

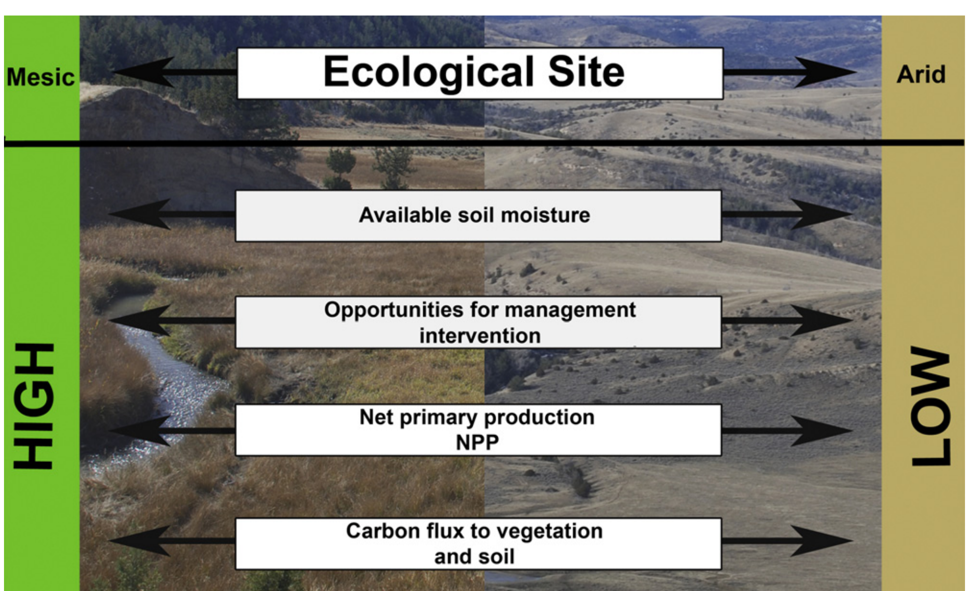

Fig. 1. Characteristics of ecological sites pertaining to carbon sequestration as influenced by aridity.

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2010; Svejcar et al., 2008; Wylie et al., 2007) (Fig. 1). Because the impacts of biotic interactions such as plant competition are overshadowed by abiotic influences such as the timing and amount of rainfall, temperature variations, and soil type (Westoby et al., 1989; Parton et al., 1994; Briske et al., 2003), the most important factors influencing carbon sequestration are not amenable to management (Westoby et al., 1989; Briske et al., 2003, 2005) Building management strategies for such rangelands based on equilibrium type community models may exaggerate the role of grazing and potential management impacts (Briske et al., 2005; Huntsinger and Bartolome, 2004; Bartolome et al., 2008).

In contrast, more mesic and productive rangeland systems have a greater likelihood of responding to grazing and other types of management (Vetter, 2005). In rangeland ecosystems where environmental conditions support plant growth sufficient for plant competition and other biotic interactions to play a major role in vegetation development, grazing management that leads to increased soil carbon storage by plants, and increased woody and perennial vegetation with extensive root systems, can positively influence carbon sequestration, in scenarios similar to those of other mesic ecosystems. In fact, most information documenting carbon response to grazing is from less arid rangelands (Gilmanov et al., 2010; Conant and Paustian, 2002), and the highest estimates of potential rangeland carbon sequestration (Conant and Paustian, 2002; Ogle et al., 2004; Morgan et al., 2010) include management activities such as fertilization and sowing of legumes that have little to no effect where productive potential is limited by aridity (Table 1).

Over-generalizing results from one rangeland type to another can lead to false expectations. Much of the estimated potential for increasing carbon uptake is based on the goal of shifting rangeland communities either among or within vegetation states (Westoby et al., 1989) or to other land uses like irrigated pasture, cropland, or forestland. Yet such opportunities are severely constrained by rangeland ecological dynamics specific to each ecological site. Opportunities to increase carbon sequestration on rangelands are highly variable and best predicted at a coarse scale by the position of the ecological site along an aridity gradient. The magnitude of carbon sequestration and management influences diminishes with decreasing rainfall (Fig. 1). At a broad regional scale, precipitation can be used as an initial filter to tailor interventions to the possibilities of the ecological site.

5. What can rangeland management accomplish?

Management actions should be linked to ecological site and aim to conserve the existing stocks of carbon in soils, as well as increase flux from the atmosphere and length of storage time while minimizing loss (FAO, 2009). Manipulation of rangeland vegetation and above-ground carbon stocks must consider both long and short term effects on wildfire, erosion, wildlife, and water dynamics. Below are discussed the major ways that management might either prevent release of carbon or gradually increase carbon stocks, especially in previously degraded areas (Table 1). However, when considering the effects of these management actions on carbon stocks, it must be remembered that abiotic factors will constrain and at times overwhelm the results of any management action. As the primary economic use of rangelands throughout the world is livestock grazing, special attention is directed to grazing management.

5.1. Grazing management

Although grazing management is the focus of many proposals for and studies of rangeland carbon sequestration, grazing by wild and domestic herbivores has been shown to have mixed and unpredictable effects on carbon cycling in rangelands (Derner and Schuman, 2007; Ingram et al., 2008; Diaz et al., 2009; Piñeiro et al., 2010; Briske et al., 2011). Processes on arid rangelands are dominated more by abiotic factors than biotic interactions like grazing (Bartolome et al., 2008). Much of the variability in results and interpretations of grazing effects stem from inadequate

Table 1
Estimates of potential annual carbon sequestration on US rangelands.

Author	Tonnes C/ha	Tonnes CO2e/ha	Million TCO2e US Range-lands	%US CO2e emissions[d] (%)	Management scenario
Morgan et al. (2010) (low)[a]	0.07	0.26	80	1.37	"Best management practices," citing Schuman et al. (2002) and Derner and Schuman (2007)
Morgan et al. (2010) (high)[a]	0.3	1.10	343	5.88	
IPCC (2007)[a]	0.03	0.11	34	0.59	"Cool dry" and "warm dry" types, using grazing, fire, fertilization
Schuman et al. (2002)[b]	0.34	1.243	388	6.64	Considers both mitigation gains and avoided losses using conversion from croplands to grasslands, proper grazing, and keeping lands in CRP
Lal et al. (2007) (High)[c]	n.a.	n.a.	257	4.40	Includes pasture and semi-arid rangelands using fertilization, manure, and planting of improved species
Lal et al. (2007) (Low)[c]	n.a.	n.a	48	0.82	
Conant et al. (2001) (average)[a]	0.54	1.98	618	10.59	"Most common types of improvement were fertilization, improved grazing management, and conversion to pasture from native and cultivated lands, accounting for >90% of all studies"
Ogle et al. (2004) (low)[a]	0.4	1.47	458	7.84	Assumes "most managed grasslands are in a nominal condition" and a single management change including sowing legumes, planting more productive varieties, irrigating, and applying fertilizer
Ogle et al. (2004) (high)[a]	0.9	3.30	1031	17.65	Assumes 67% of rangelands currently degraded and multiple of the above listed interventions

[a] Reported sequestration potential in tonnes of carbon per hectare per year. These were converted to tonnes in US rangelands assuming 312 million hectares of rangelands in US based on USFS (1989)

[b] Reported sequestration potential as total MMT carbon, assuming 183 Mha rangelands in U.S. This was converted to tonnes of carbon per hectare then multiplied by 312 million so as to be consistent with other cited sources.

[c] Reported sequestration for grazing and pasture lands entire US but did not give either a per hectare number nor a total number of grazing and pasture hectares in US based on USFS (1989).

[d] Based on US emissions of 5839.3 MMT CO_2 in EIA (2008).

attention to differences among ecological sites. Arid rangelands are underrepresented in grazing studies, yet within the bounds of grazing intensities that can sustain livestock production, manipulating grazing systems, for example using rotational, delayed, or season-long grazing systems, is unlikely to make a significant difference in carbon stocks or flows in the abiotically-controlled non-equilibrium arid systems of most extant rangelands in the US today (Briske et al., 2008).

The major potential impacts of grazing management on carbon stocks are in driving or preventing vegetation state transitions, such as increasing or decreasing woody plants, and in preventing soil disturbance and exposure. On croplands, tillage increases carbon loss by mixing and exposing soil and organic matter to the atmosphere and weather, increasing decomposition and both organic and inorganic carbon loss (Uri and Bloodworth, 2000), while leaving more durable residue leads to a higher level of carbon retention in the soil (Subak, 2000). Minimizing soil disturbance, and leaving adequate residue, is also likely to prove useful in retaining carbon stored in rangelands.

5.2. Grazing and woody vegetation

If grazing on a particular ecological site influences the abundance of woody plants, the effect on carbon dynamics can be large. Growth of shrubs sequesters carbon in woody material that persists longer than herbaceous matter. The effect of grazing and its relation to the growth of shrubs depends on the grazing system itself (number and type of animals, duration, frequency, and spatial distribution) (Huntsinger, 1996) and on the ecological site. In drier areas of the southwestern US and much of Mexico, heavy grazing can contribute to a transition from grasses and forbs to woody shrub cover (Laycock, 1991). Conversely, in wetter areas, such as coastal California, decreased grazing can lead to woody shrub invasion in which grasses are choked out by dense shrubs (Russell and McBride, 2003). Grazing of cheatgrass in Great Basin sagebrush vegetation may reduce the chance of wildfire that eliminates native woody shrubs (Daubenmire, 1970).

Research examining the carbon effects of these vegetation changes shows that results are also linked closely to ecological site (Jackson et al., 2005). Recent work suggests that drier sites are more likely to retain organic carbon and wetter sites more likely to lose it through increased oxidation rates with increases in woody vegetation (Jackson et al., 2002). Any gains and losses in soil carbon must then be compared with concomitant gains and losses in aboveground carbon to assess the net carbon gain or loss. In desert grasslands, woody cover increases, but as grasses decline due to shading and root competition, there is more exposed soil. While it seems likely that such a situation would increase carbon sequestration over time due to the increase in woody plants (Silver et al., 2010), the bare ground and lack of grasses may mean a loss in soil carbon stocks from exposure and decay (Subak, 2000). Dense shrub stands also leave little room for grasses that slow runoff and increase percolation, potentially reducing soil moisture and net primary production. Soil carbon may be sequestered for a longer period of time and be less vulnerable to fire and drought than aboveground woody biomass, meaning that continued grass dominance may prove to be more valuable for climate change mitigation in the long term. In California coastal grasslands, shrub increases almost certainly increase carbon stocks. However, the shrubs may increase fire potential and the possibility that much of the stored carbon will be released to the atmosphere within a few decades (Russell and McBride, 2003).

In sum, manipulating the amount of woody vegetation through grazing, where feasible, remains an intervention opportunity that is manageable, tractable, and likely has a significant effect on carbon stocks. However, more needs to be known about the effects of these vegetation state changes on carbon, especially soil carbon, in different ecological sites, and how to balance increases in above ground carbon stores with possibly higher fire probabilities. The consequences of altered disturbance regimes also must be evaluated, or short-term gains may result in long term loss. Finally, other considerations, such as wildlife habitat, viewsheds, and water cycling, must be taken into account before recommending actions to cause or prevent major vegetation change.

5.3. Reforestation and afforestation

Afforestation or reforestation are assumed to increase carbon stocks on rangelands (Morgan et al., 2010; Jackson et al., 2005), but again outcomes are linked to ecological site. Afforestation is the planting of trees in areas that are not currently forests, while reforestation is the replanting of trees that have been removed through fire or cutting. On some sites, protection of trees from browsing or other influences that suppress them is another approach to increasing trees. Although many arid rangeland sites are too dry for trees, some more mesic rangelands were once, and in some cases are still, home to a variety of broadleaf and coniferous trees. Unlike increases in woody shrubs, the introduction, or re-introduction, of broadleaf trees appears to have clear, positive effects on rangeland carbon sequestration (O'Halloran et al., 2009; Baldocchi et al., 2010; Morgan et al., 2010; Jackson et al., 2005). Besides the carbon stored in the aboveground woody material, broadleaf trees typically have large root systems, and, in some rangeland systems, actually increase production of grasses and forbs beneath the canopy (Frost and McDougald, 1989).

Reforestation of previously forested rangelands, or increasing the tree cover on sparsely forested rangelands, can have positive carbon impacts, but environmental and social tradeoffs arise and can also be linked to ecological site. Trees are heavy water users and affect the hydrological cycle both in their immediate area and downstream (Jackson et al., 2005, 2007), reducing water availability for other vegetation and other uses where water is limiting. On arid lands, water is often the major limitation on tree growth, and there is a tradeoff between water use and carbon fixation, with trees transpiring as much as 500 kg of water for each kg of carbon fixed on an annual basis. In addition, roughly 60% of this carbon has been found to return to the atmosphere by respiration. In short, plant loss of water vapor through stomata is far more than 1000 times the net carbon gain (Sabaté and Gracia, 2011), which may make for an unfortunate tradeoff in arid areas. In Mediterranean oak (*Quercus ilex*) forests, respiration of up to 90% of annual rainfall has been recorded (Gracia et al., 2011). Additionally, while planting trees at lower densities can actually increase understory grass and forb production on some ecological sites, in other cases thick tree cover may choke out undergrowth and may not be compatible with grazing use or habitat for grassland and shrubland wildlife. Concerns have been raised that carbon policies could promote monocultural tree plantations on rangelands that may be more susceptible to epidemics of disease and pests. Management that balances promotion of broader biodiversity goals with carbon sequestration is often better aligned with visual and cultural preferences (Caparrós et al., 2010).

As with woody vegetation in general, drought- and fire-prone ecological sites raise additional concerns when it comes to afforestation and reforestation. Plantings of trees or other woody vegetation on sites where they are vulnerable to drought or likely to burn may result in a net carbon loss. Attention to the density of woody vegetation, the characteristics of surrounding vegetation, and the resulting fuel structure is essential. Some widespread fast growing plantation trees, such as eucalyptus, suppress understory vegetation through allelopathy, shading, and heavy duff, and are highly vulnerable to fire.

5.4. Erosion

Soil erosion, which may result not only from poor grazing practices but also from cultivation, road construction, or other management actions and natural processes, may cause a net decrease of soil carbon stocks and increased carbon flux to the atmosphere. Lal et al. (1998) estimate that 15 MMT of soil carbon is emitted to the atmosphere each year due to water erosion of US soils. Eroded soils may have reduced production potential, and reduced carbon storage capacity (Rhoton and Tyler, 1990). However, the net carbon effect of erosion is not entirely clear as recent work focusing primarily on croplands suggests that erosion can function as a carbon sink in some circumstances (Van Oost et al., 2007; Harden et al., 2008).

The degree of soil erosion that takes place depends primarily on vegetation cover, vegetation residue on the soil (Holechek et al., 1998), soil type, slope, and rainfall amount and seasonal distribution. On rangelands, vegetation cover and the accumulation of vegetation residue, also known as litter, mulch or residual dry matter, can be successfully manipulated by managing grazing intensity (Bartolome et al., 2006). A number of natural resource protection programs currently exist that use residual dry matter as an indicator for the appropriate amount of livestock impact. On croplands, residue has been identified as a key component in retaining soil carbon (Subak, 2000).

5.5. Restoration

Restoring woody vegetation to sites that were cleared, restoring croplands by reconverting them to rangelands, restoring vegetation to bare soils, and restoring soil stability can all increase carbon sequestration as well as storage. Plants are the primary vehicle for adding carbon to terrestrial environments and planting grass may help to increase soil carbon storage (Diaz et al., 2009). However most of the research investigating revegetation by planting grass and vegetation manipulation has taken place in the context of converting marginal agricultural lands to grasslands, rather than restoring lands that were already classified as rangeland (Derner et al., 2005). Studies examining the restoration of agricultural fields to grasslands have shown that doing so increases carbon storage by replenishing some of the carbon lost from years of soil disturbance through tillage and erosion (Diaz et al., 2009). Converting rangelands that have been cultivated back to rangeland can change the soil from a source to a sink and restore carbon storage levels to that of native range (Uri and Bloodworth, 2000).

In some cases removal of invasive species may enhance soil and plant storage by reducing fire hazard. For example, cheatgrass (*Bromus tectorum*), an invasive grass, is associated with increased fire frequency (Bradley et al., 2006). Organic amendments, such as manure and compost, have shown promise for increasing soil carbon storage (Conant et al., 2001; Follett et al., 2000; Moffet et al., 2005; Paschke et al., 2005) and are being investigated for their potential to increase sequestration (Marin Carbon Project, 2012). It is possible that organic amendments do not actually enhance carbon sequestration following their application, but increase soil carbon storage due to remnant carbon from the organic materials themselves (Conant et al., 2001). These efforts have not been studied at the more arid end of the spectrum, and treatment of extensive areas may not be practical, especially if transport results in an increase in emissions.

5.6. Fire

While rangeland fires result in an immediate pulse of carbon to the atmosphere, their effect on carbon stocks over the longer term is not entirely clear, perhaps because research results may not be linked to specific ecological sites, fire characteristics vary, carbon stores are largely below ground, and weather is highly variable. In the tallgrass prairie, for example, Bremer and Ham (2010) found moderate soil carbon loss from annual burning, whereas Fornara and Tilman (2008) found no decrease in soil carbon accumulation. Fire may maintain a specific vegetation state, as when frequent burning prevents shrub encroachment into grassland, or revitalize a chaparral stand. Conversely, frequent fire may also lead to a removal of woody vegetation and a transition to grassland, for example when sagebrush is eliminated and replaced by cheatgrass in Great Basin sagebrush. Grazing is sometimes used to reduce the risk of fire through elimination of fine fuels, preventing shrub encroachment, or suppressing shrub regrowth after a fire or clearing (Huntsinger et al., 2007). It is important to consider ecological site and the fuel conditions that will result from management actions.

6. Implications for policies for carbon sequestration

Carbon sequestered as a result of policy implementation is considered *additional*, meaning that it would not have been sequestered, or would have been released to the atmosphere, in the absence of the policy. Carbon policies that exchange sequestration and emission credits should ensure that participants sequester additional carbon as a result of a policy, yet the non-equilibrium dynamics at the arid end of the climate spectrum makes reliably attaining additionality through management difficult if not impossible. How can policies to sequester additional rangeland carbon accommodate ecological findings that the net flux of carbon on arid rangelands is low, highly variable, and largely controlled by abiotic factors?

6.1. Flux-based policies for carbon sequestration

Flux-based policies for carbon sequestration offer incentives for increasing average flux from the atmosphere to soils, usually on an annual basis, through changes in management. These policies include most proposals for cap-and-trade as well as payments for environmental services. They face a variety of obstacles in arid rangeland systems including the size and variability of rangeland flows, transaction costs associated with measurement, and, particularly, the assumption that changes in management can reliably increase terrestrial carbon sequestration.

The small and variable net carbon flux on arid rangelands means that annual payments based on average flux measurements will not be large on an areal basis, and in some cases may even be negative. It has been argued that, though small, these flows and the rewards based upon them may still "add up." That is to say that some rangeland landowners, including the U.S. government, own such large acreages that small rewards per unit area would still be meaningful once aggregated, as long as the net flux is positive. A small increase in income could be meaningful for an enterprise where returns from rangeland grazing itself are also small and variable, and vulnerable to unpredictable drought.

On the flip side, small carbon losses driven by drought could also add up to a meaningful loss for which the landowners would be responsible. Aggregation may work for some landowners under some types of policies for carbon sequestration. However, under any flow-based policy that attempts to account accurately for carbon sequestered, transaction costs posed by measurement and monitoring are likely to diminish any yearly reward paid to landowners to the point that few would find it worthwhile to participate (Subak, 2000).

Table 2 provides examples of the potential range of annual carbon payments landowners might expect before transaction costs are accounted for, showing average, best, and worst-case

Table 2
Average, best, and worst-case scenarios from annual carbon payments per hectare for northern mixed prairie, assuming zero transaction costs, based on carbon flux data from Svejcar et al. (2008). Prices were taken from the Ecosystem Marketplace website in August of 2010.

Type of carbon payment program	Average $/ha/yr	Best $/ha/yr	Worst $/ha/yr
Certified ($10.24/tonne)[a]	$19.86	$44.64	−$10.14
Voluntary ($3.03/tonne)[b]	$5.88	$13.20	−$3.00

[a] Clean development mechanism.
[b] Chicago climate exchange.

scenarios using carbon flux data from Svejcar et al. (2008) in 2010 carbon offset prices from both the regulated (European Emissions Trading Scheme, Clean Development Mechanism) and voluntary (the now-defunct Chicago Climate Exchange) markets (Table 2). While neither of these markets currently makes payments for U.S. rangeland carbon, they offer evidence of the range of carbon prices that might be expected in certified and voluntary carbon trading. Data on sequestration were derived from the northern mixed prairie (Svejcar et al., 2008) and chosen for analysis because they were from a site that most consistently sequestered (rather than emitted) carbon, with sequestration occurring six of seven years. Notably, every site from the rangeland data set was a carbon emitter at least one year out of seven. The worst case scenario occurs when a site is a net emitter, a case that is dealt with differently depending on the carbon policy. It is likely that the possibilities for income augmentation increase along the gradient from arid to mesic sites (Fig. 1). While even small income augmentations may be motivating for ranchers, pinpointing where along this gradient net flux shifts to consistently positive over the long term is a challenge in itself, and will vary by ecological site. Those benefiting from payments based on additionality are expected to also take responsibility for negative fluxes.

The spatial and temporal heterogeneity of rangelands raises transaction costs, which are likely to heavily undercut potential payments. The transaction costs of quantification of flows depend to some extent on the degree of accuracy that policymakers and the public demand, or inaccuracy that they are willing to accept (Conant and Paustian, 2002). Current methods of directly measuring soil carbon can be accurate but, given high inter- and possibly intra- site variability, if used for monitoring individual projects, would need to be replicated in such large numbers as to be cost prohibitive (Laca et al., 2010; Conant and Paustian, 2002). Although there are ongoing attempts to create models to substitute for direct measurement, current techniques do not provide acceptable levels of certainty on arid and semiarid rangelands (Brown et al., 2010), and data for arid shrublands are almost entirely lacking (Morgan et al., 2010). In more productive rangelands such as tallgrass prairie, Bremer and Ham (2010) report that carbon inputs and losses in the system are both so large that small errors in measurement or modeling may affect results to such an extent that it becomes difficult to determine whether ecosystems are sources or sinks, and multiple authors cite the low ratio of flows to stock as a complicating factor in detecting a signal (Morgan et al., 2010; Conant and Paustian, 2002). The relatively high levels of below-ground storage complicate the use of more efficient and less costly measurement methods such as remote sensing (Gonzalez, 2008).

If landowners representing a large area of rangelands were to participate in some type of coordinated carbon scheme, then it might be possible to estimate sequestration on the aggregated rangelands with reasonable reliability, although the total amount is still likely to be quite low. The Chicago Climate Exchange, for example, used averages ranging from 0.12 to 0.32 tCO_2 per acre per year (0.3–0.8 tCO_2 per hectare per year). Perhaps the average could be used as a conservative benchmark, and landowners could request higher rates on the condition that they pay for their own measurement and monitoring to demonstrate their achievements. However, at current values, research shows that such broad-based rangeland landowner participation is not likely unless prices rise significantly and other forms of support are offered to help landowners change their practices (Gosnell et al., 2011).

While landowner incentives and transaction costs raise questions regarding the implementation of flux-based carbon sequestration policies, the more fundamental issue is whether such policies will result in additional rangeland carbon sequestration. These policies inherently assume that actions taken by rangeland managers can reliably and predictably drive additional carbon sequestration. However, as discussed above, rangeland science suggests that in arid rangelands, carbon flows, and potential additionality, are largely beyond the control of rangeland managers (De Steiguer et al., 2008; Brown et al., 2010; Westoby et al., 1989). This means that most additional carbon sequestered after a policy is implemented is likely not the result of management changes, and would have been sequestered even in the absence of the policy. Even if the problems previously mentioned – high variability and irregular losses – can be solved through broad-based averaging schemes, the carbon sequestered is still not truly additional unless it increases sequestration over previous levels due to changes made as a result of the policy.

Such management-based additionality is undermined where management actions lack predictable, consistent, and measurable effects on carbon sequestration, as they do in all but the most mesic rangeland ecological sites. For those management interventions that do have an effect, the response follows a precipitation gradient, with larger and more predictable effects in more mesic rangelands and smaller, less predictable effects as aridity increases. This gradient of effects must be kept in mind when considering carbon sequestration policies. Policies that depend on carbon sequestration as a result of management changes may be appropriate for more mesic rangeland sites that are responsive to management interventions (Fig. 1), but they are inconsistent with the findings of ecological science on arid rangelands, where management effects are largely unpredictable and heavily constrained by abiotic factors.

If participants are not sequestering more carbon than non-participants, or increasing sequestration rates above those prior to program implementation, then such policies have little use in climate change mitigation. Moreover, offering incentives based on the idea that increased uptake compensates for emissions elsewhere, when uptake cannot be verified to be increasing, is fraudulent and potentially harmful. If the incentives are financed by additional fossil fuel based emissions, as with private carbon offset markets, then any gap between promised and actual achievement will involve a real addition of greenhouse gases to the atmosphere. Therefore, there is a high likelihood that policies that allow others to emit carbon using credits or offsets based on assumed but difficult to measure increased rangeland carbon sequestration will result in an overall increase in emissions.

6.2. *Policy principles for carbon sequestration in arid U.S. rangelands*

Based on the discussion above, we posit four principles for policies aimed at carbon sequestration in arid U.S. rangelands to make them consistent with modern rangeland ecology:

1. Policies should not require short-term carbon accounting. Based on the difficulties of measuring and monitoring soil carbon in heterogeneous rangelands as well as the low and variable flows of carbon, this type of accounting currently incurs transaction

costs that divert much of the revenue away from landowners. Moreover, short-term variations in carbon are unlikely to be the result of management changes and are thus not additional.

2. Policies should not assume that changes in management always function as the primary basis for additional carbon storage. As shown above, especially at the more xeric end of the spectrum, which includes most extant U.S. rangelands, management actions are constrained and often overwhelmed by abiotic factors to such an extent that carbon sequestration or release cannot be attributed to management.
3. Credits from rangeland carbon sequestration based on management should not be considered to offset emissions. Because the amount of carbon sequestered on rangelands resulting from changes in management is usually very small, and is effectively impossible to determine relative to the role of abiotic factors, the carbon that can be considered truly additional is both negligible and undefined. Supporting the many other co-benefits of ranching is not sufficient justification for carbon sequestration payments or credits—only sequestration that is additional should be used to justify payments or credits that offset emissions. Until and unless arid rangeland sequestration can meet the additionality standard reliably, policies should not allow emissions elsewhere to be offset by such sequestration, as they would thereby risk an overall increase in emissions.
4. Policies should seek to conserve rangelands and encourage restoration through conversion of marginal or degraded agricultural lands back to rangelands. Although the carbon sequestered on a short-term basis is low, variable, and not reliably enhanced through management, over the long-term most rangelands, even on the arid end of the spectrum, function as significant sinks, whereas more intensive land uses are likely to be a source. Rangelands that have been previously used for cropland have an especially high capacity for sequestration upon conversion back to rangeland because they are removed as a high soil carbon emission source. If avoidance of rangeland conversion or reconversion to rangeland results directly from policy, the carbon sequestered over time can be considered additional. However, to be consistent with Principles 1 and 3, conservation or reconversion of these rangelands should not require explicit short-term carbon accounting or be used to offset other emissions.

7. Applying policy principles based on ecological science to cap and trade, payment, and taxation options for creating sequestration incentives

The above policy principles can be used as a basis to assess whether or to what extent policies are consistent with the ecological dynamics of rangelands. Two general types of policies for carbon sequestration are highlighted here: (1) those that are flux-based, as described above, including cap and trade and payment for ecosystem services and (2) those that focus on long-term rangeland conservation and restoration through payments for avoided conversion or restoration. In addition, we consider the potential impacts of another type of carbon policy, a carbon tax, which, if implemented, could have an indirect effect on rangelands. These policies represent the most widely proposed and considered current options for rangelands in the United States. Examining them in the light of rangeland ecological dynamics enables a general assessment of their potential to influence carbon cycling. The global policy situation for carbon sequestration is quite fluid, as across the globe organizations and states explore ways of overcoming social and ecological challenges to successfully encouraging carbon storage and sequestration. At the current time the U.S. policy environment is generally hostile to non-market regulatory mechanisms, and policy options are further limited by poor links between economic decision-making criteria and ecological science (Norgaard, 2009).

7.1. Flux-based policies for carbon sequestration

These types of incentives reward additionality.

7.1.1. Cap and trade

Cap and trade schemes for rangelands, which functioned through the Chicago Climate Exchange (CCX) until December 2010, have gotten the most attention from U.S. policy makers and analysts and are still under consideration for future domestic climate change mitigation efforts. However, many supporters of cap and trade systems for rangeland carbon have expressed doubts about their potential to actually increase carbon sequestration (Laca et al., 2010). Instead, some proponents have focused on the co-benefits supported by carbon credit income that can contribute to rangeland conservation. Supporters have cited biodiversity, preservation of ranching culture and economies, improved management and adoption of new practices, and prevention of sprawl as some of the benefits to be expected from cap-and-trade (Kroeger et al., 2009).

In analyzing cap-and-trade programs in general and the results of the CCX in particular, it is clear that they are inconsistent with the policy principles for rangeland carbon sequestration stated earlier. One of the critical flaws of the CCX offset program, from a climate change perspective, was that it assumed changes in management would lead to additionality. Yet there is no assurance that carbon sequestered under the CCX would not have been sequestered in the absence of such a scheme. This is both because many participants did not actually change management practices (Diaz et al., 2009; Gosnell, 2009) and because of the predominance of abiotic drivers in these systems. Carbon cannot be reliably increased on arid rangelands purely through changes in management.

The lack of management changes may have been peculiar to the design and enforcement of the CCX, which, as a voluntary trading scheme, was not subject to some of the same concerns and oversight as some other certified programs. But the lack of additionality from management changes is a fundamental contradiction between any cap-and-trade or flux-based policies and the ecological science of rangelands. If the target of the policy is reducing net emissions, then achieving additionality is essential. In a cap and trade system, the magnitude and additionality of the sequestration generating the offset credit must be certain to compensate for the emissions of the buyer of the credit, or net emissions will not decrease.

To better assure additionality by screening out background abiotic factors, trading schemes could require baseline flux measurements and only pay managers for the excess annual carbon sequestered above the baseline. However, additional carbon resulting from management action would likely be negligible, and flows are inherently low. Once the transaction costs of measuring and monitoring are taken to account, it seems unlikely that such a system would result in net income for landowners. Moreover, although baseline flux measurements provide some information on typical carbon flows independent of management changes, determining how much of the carbon flux to attribute to management (i.e. the portion of carbon that is truly additional), rather than due to changes in precipitation or other abiotic factors, would be impossible.

Another way around the additionality problem is for rangelands to be considered under the cap instead of outside the system as offsets. In such a scheme, yearly flows would be measured and landowners would be allotted a credit (either purchased or free, depending on the specifics of the policy) based on the historical emissions of their entire operation. As the cap was lowered, landowners would have to decrease their emissions commensurate with the lowering cap or purchase credits from others in order to maintain their level of emissions. If they were able to lower their emissions below their credit allotment, they could sell the extra credits. This is the basic cap and trade scheme as it applies to all covered industries. However, to cover rangelands under such a system requires the expense of measuring, monitoring, and regulating emissions on all rangelands in the United States with some accuracy, and political opposition would likely be ferocious as landowners would be lumped with true polluters such as the coal industry. Moreover, as most landowners would probably be unable to decrease annual emissions, over time the system would penalize landowners for something out of their control, possibly creating perverse incentives for rangeland conversion.

Thus, either as offsets or under a cap, most extant U.S. rangelands are not suited to a cap-and-trade system based on additionality of net flux. The co-benefits of payments to landowners from a carbon trading scheme must not obscure the fact that carbon emissions reduction is the policy goal, and such reductions are unlikely to result reliably from changes in management.

7.1.2. Payments for ecosystem services

Direct payments for ecosystem services (PES) can include payments for increased carbon sequestration. However, direct payments, whether from state or private entities, cannot overcome the problems of achieving additionality on arid rangelands through management. In arid environments, payments for management that protects the soil, already part of responsible grazing management, might contribute to protecting carbon stocks, but this response will be constrained by abiotic factors.

To the extent that PES relies on changes in management to increase carbon sequestration, PES suffers from the same lack of additionality issues as other flux and management-based policies such as cap-and trade. The PES framework, however, is preferable to cap-and-trade in that, in a PES framework, any carbon that is sequestered cannot be used to offset increased emissions elsewhere. In this way payments for ecosystem services are more consistent with our policy principles because net emissions are not increased as a result of implementation. Instead, potential emissions are prevented.

Management that protects the soil is also valuable for protecting watershed, wildlife, recreational, and scenic values, as well as the long term productivity of the land for grazing. The ecosystem service of preventing carbon loss can be seen as bundled within a diversity of potential ecosystem services that together might justify a payment that would alter the management and long term use of the land. If, together, these payments resulted in preventing the conversion of rangelands into intensive agricultural or other uses, the policy itself, though not the changes in management, could prevent additional carbon emissions and would be consistent with our suggested policy principles. The issue of avoided conversion is more fully discussed below.

7.2. Long-term conservation: payments for avoided conversion or restoration

Payments for avoided conversion or loss of forests have attracted growing international interest and might be considered for U.S. rangelands (Verified Carbon Standard, 2012). If correctly designed, payments for avoided rangeland conversion or restoring marginal croplands to rangelands could yield additional rangeland carbon storage and are consistent with our suggested policy principles. Although the process is very slow, rangelands do accumulate carbon over time, and land uses associated with them are much less carbon-intensive than the common alternative of industrial agriculture. In addition to the various greenhouse gas emissions associated with crop inputs such as fertilizer, croplands have been found to be net carbon emitters when consumption and emissions of crops is considered, while rangelands are generally either carbon neutral or small sinks (Perez-Quezada et al., 2010; Uri and Bloodworth, 2000). Moreover, conversion from grassland to annual crops can lead to a 60% loss of soil carbon stocks and a 95% loss of above ground carbon (FAO, 2009). Therefore, keeping lands as rangelands, with associated land uses such as extensive grazing, can at least prevent increases in emissions, and restoring croplands to rangelands can yield increased sequestration.

To a certain extent, such a system already exists and has been successful on a carbon basis–the Conservation Reserve Program of the U.S. Department of Agriculture (Uri and Bloodworth, 2000). By paying farmers to hold land in reserve, the CRP caused farmers to restore some of their land to rangelands. Studies of recovering rangelands generally and CRP land in particular show that they quickly accumulate and store soil carbon (IPCC, 2007; Gebhart et al., 1994; Derner and Schuman, 2007). Similar policies might avoid the transaction costs of cap-and-trade as long as the payment system is not tied to annual flows of carbon. It also does not create problems of additionality or environmental robustness as long as the carbon sequestered does not count as a credit against other emissions.

A policy not based on payment for actual carbon flux may be hard to justify politically since payments and sequestration services would not be directly linked. There would be no mechanism to determine how much carbon a single landowner was storing, or how much flux to the atmosphere was prevented, in return for the payment. One option might be payments based on an algorithm that takes into account risk of conversion and loss of future flows at a time-discounted rate, again using modeling or proxies for estimation and verification. Another option might be a voluntary, market-based scheme, along the lines of conservation easements, in which individuals or organizations would pay landholders either not to convert extant rangelands or to restore degraded croplands to rangelands. Although the carbon sequestration benefits themselves might not generate the necessary interest, they could usefully complement other conservation values that are currently driving such programs.

Preserving rangelands over the long run or converting degraded croplands to rangeland is consistent with our policy principles and would result in more carbon sequestered by playing to the strengths of rangelands as carbon sinks–they are nationally extensive and associated with low intensity land uses. However, the great extensiveness of rangelands may also prove to be a problem if this policy is widely pursued. Given that there are so many acres of rangelands in the United States, the size of payments, if paid to all owners of rangelands, may have to be small per unit area. If too small, the size of payment could limit the effectiveness of the policy.

7.3. Carbon tax

The final policy type to be considered is a national carbon tax. Although it is not generally seen to affect rangelands directly because rangelands per se would not be taxed, a strong carbon tax can indirectly affect rangelands through impacts to intensive agriculture, although the impacts of such a policy are difficult to predict.

Currently ranchers relying on extensive rangeland grazing are of less economic importance relative to intensive beef producers whose cattle gain weight more quickly and predictably. The unpredictability of forage production on arid rangelands is one of the factors that make returns low and variable. In 2009, less than 3% of the beef market in the United States came from production systems other than conventional grain-finished beef (Mathews and Johnson, 2010). A carbon tax could tip the balance of beef production more towards extensive grazing by increasing the cost of corn and other feeds, thereby increasing total costs of feedlot beef production.

Rangeland beef production is based on extensive grazing requiring few energy, crop, and labor inputs. The production system is not without greenhouse gas impacts: ranchers use gasoline for their vehicles, grass-fed cattle have been shown to release more methane per pound of gain than feedlot cattle (Harper et al., 1999; Peters et al., 2010), and many ranchers fertilize and irrigate some small pasture areas to produce supplemental feed for their herd or move their herds to pastures in different biomes seasonally in an effort to buffer highly variable rangeland production. Yet, range cattle feed primarily on native or naturalized grass grown without fertilization or irrigation, waste is returned to the soil to fertilize future grass production and contribute to soil organic carbon, and density of animals is low. Feedlots, in contrast, rely primarily on industrially produced corn, which may be the most carbon intensive conventional crop in the US, as well as other feedstuffs produced with inputs that are large sources of GHG emissions, and they support a high density of livestock with consequently high emissions (Harrington and Lu, 2002; West and Marland, 2002).

From an overall climate change standpoint, grass-fed beef is likely preferable because of the broader long-term context of production. For example, if corn prices were to increase enough to make rangeland beef cost-competitive, causing the industry to shift back to more range-fed beef, it would diminish the largest market for commodity corn and reduce the number of cattle produced. This, in turn, might encourage more Midwestern farms to convert from corn to other crops that are less energy intensive, perhaps, on more marginal lands, even to grazing. On the more mesic former rangelands where corn predominates, carbon sequestration from such a shift would be both larger and more certain to occur than on drier rangeland types. It is also likely that fewer cattle could be produced overall, potentially reducing total methane emissions.

Indirectly, then, a carbon tax might encourage conversion from cultivation back to rangelands, a move that would vastly increase the sequestration of carbon, both through avoided emissions and the high soil sequestration rates seen on restored agricultural lands (Derner and Schuman, 2007; Morgan et al., 2010).

However, predicting the effects of a national carbon tax is complicated by the global nature of beef production. If the result of a carbon tax is higher beef prices, as predicted, then there could be unfavorable land use effects on a global, or even national scale, as the higher beef prices entice more people to raise beef and either raise grain to feed those cattle or expand grazing area at the expense of carbon rich forests. These indirect international land use effects and equilibrium price dynamics, which have been hypothesized for biofuel production (Searchinger et al., 2008), must be taken into account when considering whether a carbon tax would be the best policy to promote carbon sequestration on rangelands.

8. Conclusions

Attention to ecological site and the link to "best fit" ecological models is essential to assessing carbon sequestration potential on rangelands as well as other types of natural landscapes (Norgaard, 2009). Reviewing what is known of rangeland ecological dynamics and the distribution and extent of ecological sites suggests that management cannot reliably increase carbon uptake on most rangelands. Carbon uptake on the relatively small remaining areas of more mesic and productive rangelands may be more responsive to management, but care must be taken to differentiate these rangelands and their potential from typical arid and semiarid rangelands. On the other hand, over the great extent of rangeland, protection of carbon stocks present in soils or conversion to rangelands from more intensive uses would make a significant contribution to global carbon balance. Management strategies to accomplish this should be developed and implemented.

We have described four policy principles that will ensure that policies for rangeland carbon sequestration are consistent with the ecological science of rangelands: policies should (1) not require short-term explicit carbon accounting (2) not assume that changes in management can create additional carbon sequestration (3) not use arid rangeland sequestration that is not consistent and verified to offset emissions; and (4) should focus on conserving rangelands or reverting degraded agricultural lands to rangeland.

Applying these principles to proposed rangeland carbon sequestration policies shows that cap and trade is problematic in terms of environmental integrity, transaction costs, and additionality. Particularly troubling is a common assertion that absent the ability to measure or assure that additional carbon is sequestered as a result of management practices, carbon payments support an array of co-benefits from rangeland ecosystems. While this is no doubt true, under cap and trade, credits allowing increased carbon emissions are granted to others under the assumption that they are compensated for by increased uptake in the rangeland system.

Payment for ecosystem services may be consistent with the ecological dynamics of rangelands when they reward carbon storage and management practices that protect the carbon in soils, rather than increased sequestration. Payments for avoided conversion ("avoided de-rangification") or reconversion of marginal agricultural lands to rangelands are also consistent with the science of carbon dynamics on rangelands, but low prices may limit the effectiveness of direct payments. Perhaps through bundling them with payments for other co-benefits of extensive production and land conservation, payments could reach a level allowing them be a viable strategy. This type of policy could support enhancement of soil storage capacity, by conserving rangelands so that they can continue to store carbon over the long-term. It would also preserve the contribution of rangelands to the global food supply. A national carbon tax, if high enough and properly designed, could have a strong, though indirect, impact on rangeland carbon sequestration if it encouraged conversion from cropping to rangeland grazing.

Carbon sequestration incentives will be a disappointing source of income for ranchers on arid lands unless they are part of a realistic policy program that can provide consistent, even if small, income augmentation. Ranchers are receptive to diversifying income streams as long as such income is consistent with their personal and stewardship goals, is voluntary, and avoids an intrusive government role (Ma and Coppock, 2012; Cheatum et al., 2011; Gosnell et al., 2011). Research has also shown that ranchers in general are not particularly familiar with the details of carbon sequestration (Cheatum et al., 2011; Ma and Coppock, 2012) and that their response to the likely financial benefits alone as an incentive is "tepid" (Ma and Coppock, 2012). Ranchers reported that increasing forage production, improving soil quality, increasing water retention and infiltration and enhancing drought resistance were more compelling reasons to change management practices (Ma and Coppock, 2012). Bundling of carbon sequestration with an array of other co-benefits (Cheatum et al., 2011) as part of a payment for ecosystem services or avoided conversion program may be the best received type of approach for the ranching community, as well as being more suited to the ecology of arid ecosystems.

At the international level, a diverse array of programs have attempted to cope with the ecological constraints of rangelands. However, many of the dynamics underlying rangeland carbon fluxes and management are not well understood and warrant further study. In particular, modeling and remote sensing methods to reduce transaction costs of soil carbon measurement and monitoring are promising. In general, more information about carbon dynamics along the gradient from arid to more mesic rangeland ecosystems is needed. Research that increases the data on the ecological dynamics of rangeland ecological sites, and how they pertain to carbon sequestration, is also needed. Other topics in need of further inquiry include a full systems-level carbon comparisons of feedlot and grass-fed beef.

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